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Polish River Basins and Lakes – Part II

Biological Status and Water Management

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Aims and Scope

Since 1980, *The Handbook of Environmental Chemistry* has provided sound and solid knowledge about environmental topics from a chemical perspective. Presenting a wide spectrum of viewpoints and approaches, the series now covers topics such as local and global changes of natural environment and climate; anthropogenic impact on the environment; water, air and soil pollution; remediation and waste characterization; environmental contaminants; biogeochemistry; geoecology; chemical reactions and processes; chemical and biological transformations as well as physical transport of chemicals in the environment; or environmental modeling. A particular focus of the series lies on methodological advances in environmental analytical chemistry.

Series Preface

With remarkable vision, Prof. Otto Hutzinger initiated *The Handbook of Environmental Chemistry* in 1980 and became the founding Editor-in-Chief. At that time, environmental chemistry was an emerging field, aiming at a complete description of the Earth's environment, encompassing the physical, chemical, biological, and geological transformations of chemical substances occurring on a local as well as a global scale. Environmental chemistry was intended to provide an account of the impact of man's activities on the natural environment by describing observed changes.

While a considerable amount of knowledge has been accumulated over the last four decades, as reflected in the more than 150 volumes of *The Handbook of Environmental Chemistry*, there are still many scientific and policy challenges ahead due to the complexity and interdisciplinary nature of the field. The series will therefore continue to provide compilations of current knowledge. Contributions are written by leading experts with practical experience in their fields. *The Handbook of Environmental Chemistry* grows with the increases in our scientific understanding, and provides a valuable source not only for scientists but also for environmental managers and decision-makers. Today, the series covers a broad range of environmental topics from a chemical perspective, including methodological advances in environmental analytical chemistry.

In recent years, there has been a growing tendency to include subject matter of societal relevance in the broad view of environmental chemistry. Topics include life cycle analysis, environmental management, sustainable development, and socio-economic, legal and even political problems, among others. While these topics are of great importance for the development and acceptance of *The Handbook of Environmental Chemistry*, the publisher and Editors-in-Chief have decided to keep the handbook essentially a source of information on "hard sciences" with a particular emphasis on chemistry, but also covering biology, geology, hydrology and engineering as applied to environmental sciences.

The volumes of the series are written at an advanced level, addressing the needs of both researchers and graduate students, as well as of people outside the field of "pure" chemistry, including those in industry, business, government, research establishments, and public interest groups. It would be very satisfying to see these volumes used as a basis for graduate courses in environmental chemistry. With its high standards of scientific quality and clarity, *The Handbook of Environmental Chemistry* provides a solid basis from which scientists can share their knowledge on the different aspects of environmental problems, presenting a wide spectrum of viewpoints and approaches.

The Handbook of Environmental Chemistry is available both in print and online via www.springerlink.com/content/110354/. Articles are published online as soon as they have been approved for publication. Authors, Volume Editors and Editors-in-Chief are rewarded by the broad acceptance of *The Handbook of Environmental Chemistry* by the scientific community, from whom suggestions for new topics to the Editors-in-Chief are always very welcome.

Damià Barceló
Andrey G. Kostianoy
Editors-in-Chief

Preface

The book *Polish River Basins and Lakes* is based on the scientific developments and results obtained by Polish scientists within many years of research related to the management of catchment areas of lakes and river basins in the context of global change. It consists of two volumes: Part I: *Hydrology and Hydrochemistry* and Part II: *Biological Status and Water Management*. Complementing each other, the volumes constitute the first such comprehensive study on changes in the chemical as well as the biological status of Polish surface waters. Presented by almost 100 Polish researchers, the environmental topics cover a wide range of disciplines and several main study areas, e.g. chemistry, hydrology, hydrochemistry, biology, ecology, microbiology, ichthyology and water management.

The two parts of the book contain 35 chapters. The first volume refers to Polish river basins and Polish lakes' catchments, anthropogenic pollution sources and chemical pollution of water and sediments, the evaluation of chemical dynamics and the impact on climate change projections. The second volume deals with the assessment of biological status of numerous Polish rivers and lakes. All of the quality elements associated with aquatic ecosystems including the macrophytes, phytoplankton, zooplankton, macroinvertebrates, fish and, as a matter of growing concern, invasive alien aquatic species are evaluated in this volume. Also presented are the general state of biodiversity of Polish surface waters, a set of conservation and restoration practices as well as the review of protected sites within the basins and catchments areas.

The authors hope that the book content will be of interest to environmental chemists, geologists, hydrologist, biologists, students and surface water managers as well as the general public.

We would like to thank the authors of this book for their valuable contribution and efforts to create its chapters. We would also like to underline the importance of the suggestions and the recommendations given by Prof. Bogusław Zdanowski during the process of the realization of the book.

Finally, we would like to extend our sincere thanks to Prof. Damià Barceló who invited us and inspired to create this book.

Olsztyn, Poland
November 2018

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Contents

Periphyton Inhabiting Reeds in Polish Water Ecosystems

Martyna Bąkowska, Natalia Mrozińska, Monika Szymańska,
Nikol Kolárová, and Krystian Obolewski

Contents

Abstract This study presents the results of a long-term study of periphyton inhabiting submerged parts of shoots of *Phragites australis*, carried out in the ecosystems of northern Poland (lakes, rivers, oxbows and a dam reservoir). The development of epiphytic organisms representing the level of producers and consumers was found on the substrate formed by the reed in each of the studied aquatic ecosystems. The coastal lake was characterized by the highest taxonomic diversity and the smallest oxbow lake undergoing restoration. In the autotrophic fraction, Bacillariophyta predominated in the studied ecosystems and were accompanied by very high amounts of chlorophytes and *Cyanobacteria*. Phytoperiphyton had the highest abundance in lakes (~40 mln cells m^{-2}) and the lowest in rivers

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(~7 mln cells m^{-2}). Among the heterotrophic fraction, there were 14 taxa in the lakes and only 9 taxa in the dam reservoir. Zooperiphyton reached the highest abundance in lakes (~140 thousand indiv. m^{-2}) and were the lowest in the dam reservoir (~7 thousand indiv. m^{-2}). The largest share in the microperiphyton was reached by Protista (~60%) and Rotifera (~20%), while the lowest abundance were representatives of Cladocera (~2%) and Oligochaeta (~1%). Macrozooperiphyton were represented primarily by Chironomidae larvae (~75%). The structure and abundance of periphyton may indicate the trophic state of individual aquatic ecosystems, which is important in determining their ecological condition according to the Water Framework Directive.

Keywords Aquatic ecosystems · Epiphytic organisms · Poland · Reed

1 Introduction

Among the many ecological formations inhabiting water reservoirs, periphyton is one of the most interesting and least known. As an aggregation of bacteria, fungi, algae and animals, various surfaces grow, immersed in non-sterile water [1]. Although this formation occurs in all aquatic ecosystems on both biotic and abiotic grounds, it has not yet been thoroughly investigated [2]. Due to its qualitative and quantitative diversity, it has a positive impact on the functioning of water ecosystems, especially in the littoral zone. In this area, the water reservoir and macrophytes form a buffer zone [3–5].

In aquatic ecosystems, macrophytes, including the dominant species *Phragmites australis* (Cav.) Trin. ex. Stuedel fulfil a very important role in the functioning of aquatic ecosystems [6–12]. As a result of their habitation, macrophytes affect the quality of water by cleaning it, reducing the content of biogenic salts and determining the trophic status [2]. They are both a biogeochemical barrier to pollution (buffer zone) and a place with high biodiversity. Studies of aquatic ecosystems constantly provide new information that can be used in various fields of knowledge. They form the basis, not only for understanding the dynamics of changes occurring within biocenoses, but also for modelling transformations covering entire ecosystems [13–17]. Among the many factors determining the structure of the periphyton, one of the most important is the level of water movement, which makes it difficult to settle the substrates immersed in water. The role of this factor is most easily observed in lotic and lenitic ecosystems such as rivers, lakes and oxbow lakes.

The aim of the study was to determine and compare the qualitative and quantitative structure of periphyton that inhabit a selected reed (*P. australis*) in different aquatic ecosystems in northern Poland.

2 Material and Methods

Examination of the periphyton was carried out in 2000–2014 on the substrate formed by common reed in various types of Polish aquatic ecosystems. The reed is a cosmopolitan species widespread throughout the globe [18]. At the same time, it is one of the largest herbaceous plants in the central European flora with a height of up to 4 m. The common reed is a typical swamp and aquatic plant characterized by a wide range of ecological tolerance. It grows on various types of substrates from gravel and sandy to peat soils of various types of gyttos and mules [18]. It can be found not only in water reservoirs but also on their banks, in peat bogs, in wet meadows and in riverside bushes [19]. Reeds tolerate both water ripples and permanent flooding. They grow on the banks of relatively clean stagnant and flowing waters and on waters that are heavily polluted with municipal sewage. At the same time, they fulfil an important ecological role, because the submerged fragments of reed shoots are covered with periphyton, for which they are a convenient habitat [20].

3 Study Areas

During the long-term research on reed periphyton, biocenotic analyses were made of many lotic and lentic ecosystems in the northern part of Poland (Fig. 1).

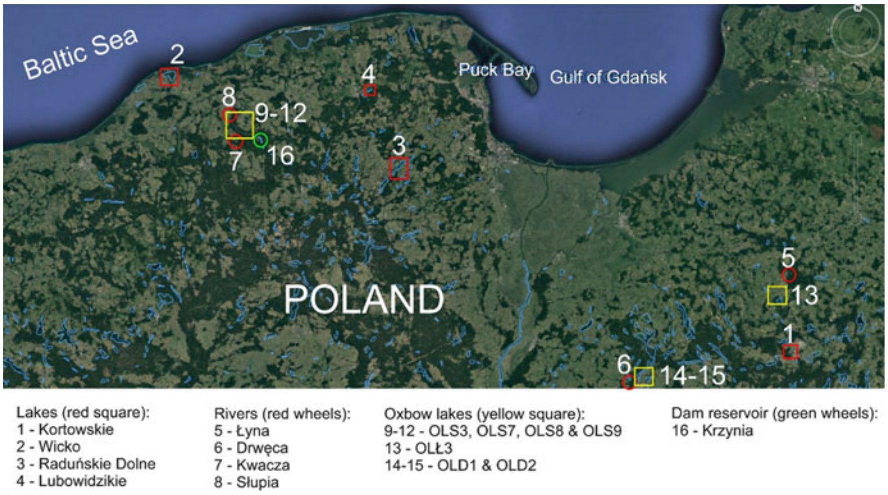

Fig. 1 Location of sampling sites in northern Poland

3.1 Lakes

3.1.1 Lake Kortowskie (1)

Lake Kortowskie belongs to a complex of reservoirs located near the city of Olsztyn (53°45′43″N 20°26′44″E). The surface of the lake is approx. 94 ha and its maximum depth is 5.9 m, which ranks it in the second category of susceptibility to degradation. The lake belongs to several flow reservoirs and is contaminated by inflows that carry humic substances, minerals and sewage [21].

3.1.2 Lake Wicko Przymorskie (2)

This lake is located on the Polish coast of the Baltic Sea, formed as a result of the closing of the bay headland produced during littoral transgression. It lies along the coastline and belongs to the coastal lakes (54°32′22″N 16°37′08″E). It is shallow with an average depth of 2.7 m. Wicko is a typically eutrophic lake with a well-developed littoral zone. There is a great abundance and diversity of plant species. The water body is characterized by low transparency and a significant amount of mineral salts dissolved in water [22, 23].

3.1.3 Lake Raduńskie Dolne (3)

The lake is a large gutter reservoir in the Kashubian Lake District (54°17′46.3″N 18°03′03,9″E) with a significant depth (35.4 m) and area (737.2 ha). The lake is located in the most attractive area of the Kashubian Landscape Park, known as the Kashubian Switzerland. Until the construction of the artificial barrier in Łączyna, it was one of the elements, together with Lake Raduńskie Górne, named “Lake Raduńskie (German: Radaunen-See)”. The Radunia River flows along the entire length of the lake gutter, constituting a part of the Raduńskie Kółko waterway. It is poorly overgrown with rush vegetation due to its bottom shape and fairly rapid increase in depth [24, 25]. This lake is subject to high eutrophication, with waters of II class of purity and II class of susceptibility to degradation [26].

3.1.4 Lake Lubowidzkie (4)

This is a postglacial lake in the area of the Reda-Łeba glacial valley in the Lębork district of the Pomeranian Voivodship (54°33′22″N 17°49′46″E). It is characterized by a developed and afforested shoreline passing into the upper terraces. Lake Lubowidzkie is a small (158 ha^2) and quite deep (up to 16 m) Pomeranian water reservoir, located in the catchment area of the Łeba River, supplying the coastal Lake Łebsko. For this reason, the water flowing out of Lake Lubowidzkie influences the level of fertility of the protected area of the Słowiński National Park to a certain extent. This lake is highly eutrophic, which contributes to the reduction of water transparency associated with the development of phytoplankton [24].

3.2 Rivers

3.2.1 Łyna River (5)

This is a river in north-eastern Poland (Warmia and Masuria Voivodship) and Russia (Kaliningrad Oblast) which constitutes the left tributary of Pregoła. It flows through the Olsztyńskie Lake District and the Sępopolska Lowland. Its length is 263.7 km (190 km in Poland), and the basin area is 71,256 km^2 (5,777 km^2 in Poland). The sources of the river are located within the landscape reserve (53°26′28″N 20°24′49″E). There is a deep and indented gorge created by the source of the river, which is an example of back erosion. The Łyna River flows through a series of gutter lakes (Brzeźno Duże, Kiernoz Mały, Kiernoz Wielki, Lake Łańskie, Ustrych), among which the largest and the deepest is Lake Łańskie. The Łyna changes its direction several times and crosses several layers of front moraines, which affects the diverse nature of particular sections of the valley [27, 28]. The sampling sites were located near the city of Dobre Miasto.

3.2.2 Drwęca River (6)

This is a river in northern Poland in the Masurian Lake District and the Chełmińsko-Dobrzyński Lake District, which is the right tributary of the lower Vistula (52°59′55″N 18°41′27″E). Its length is 250 km [29] and the basin covers 5,343.5 km^2. It can be classified as a lowland-mountainous river, thanks to the differences between the levels on the sections connecting it with the tributaries [30]. The sources of the river are located within the water reserve, which covers the entire Drwęca, some of its tributaries and flow lakes. Since 1961, the entire length of the river has been the longest ichthyological nature reserve in Poland, covered by Natura 2000 program as a special protection area of the Drwęca River Valley (PLH280001), and the Drwęca River Marshy Valley (PLB040002) special bird protection area habitats [27]. The sampling sites were located near the village of Bratian (53°27′22″N 19°36′17″E).

3.2.3 Kwacza River (7)

This is a small river, which is the left-bank tributary of Słupia (54°23′28″N 17°02′00″E), about 21 km long [31]. At the beginning of the twentieth century, it was strongly transformed by melioration works. The alternate arrangement of permeable and impermeable sediments and discontinuities and their diversification of thickness are the reasons for the significant supply of the river with groundwater [1]. The flora of this area is quite well known. Its basin is located within the “Dolina Słupi” Landscape Park, and in 2004 it was included in the Natura 2000 area. The sampling sites were located near the village of Kwakowo.

3.2.4 Słupia River (8)

The Słupia River is a coastal river (54°19′24,4″N 17°54′45,1″E), and its basin is located in the area of the Pomeranian Voivodship. The length of the river is 138.6 km [29], and the catchment area covers 1,310 km^2, and it is classified as a small river. The Słupia River is characterized by numerous meanders in the lower reaches below the Krzynia reservoir and a large decline which is often characteristic of mountain rivers. Due to anthropopression in the last century, the length of the river decreased by about 30 km as a result of straightening the river bed and cutting off about 50 oxbow lakes [32, 33]. The sampling sites were located near the city of Słupsk.

3.3 Oxbow Lakes

Oxbow lakes are eutrophic reservoirs with very varied vegetation. They form a completely different type of water biotope in relation to the river bed. These are usually non-flow floodplain lakes, but there are also those that are permanently connected to the river bed. They can be supplied by surface or underground water. The growing number of cultivated fields causes that the basin of oxbow lakes is usually an area undergoing anthropopression, which causes rapid eutrophication of the reservoir and mass blooms of phytoplankton. Biogenic elements, such as nitrogen and phosphorus, concentrate in these waters, which is why they are characterized by a fairly high water trophic state and are clearly warmer than channel water. These factors cause aquatic communities and hygrophilous vegetation, with a significant biomass production, to develop very rapidly in the oxbows, which often leads to a very rapid process of shallowing and overgrowth. The youngest oxbow lakes have cut-off meanders, are connected on one side along their entire width with the river bed and are almost devoid of coastal rushes. The oldest lakes are completely overgrown with reed beds and even swampy alder forests. Most of them are oxbow lakes that are only partly overgrown, which have both a water surface and a wide zone of rushes [25, 34].

3.3.1 Słupia Oxbow Lakes (9–12)

OLS3 (54°23′26,9″N 17°02′00,7″E)

The oxbow lake is located on the right bank of the Kwacza River, near its mouth to the Słupia river, and its length is about 167.8 m. In 2007, the oxbow lake was connected to the Kwacza River to improve the environmental conditions (semi-lotic). The oxbow lake is covered by an agricultural forest area [35]. The sampling sites were located near the village of Kwakowo.

OLS7 (54°25′35,6″N 17°01′18,5″E)

This area is about 0.1 ha. It is a right bank flow reservoir (semi-lotic) connected to the river bed by one small channel for most of the year. This reservoir was created as a result of human activities, but at the moment it is not subject to direct anthropogenic influence or human interference. Natural succession and slow overgrowth are observed here [35]. The sampling sites were located near the village of Łosino.

OLS8 (54°25′34,0″N 17°01′37,7″E)

The oxbow lake is located on the left side of the Słupia River, lying on a 41-km-long river with an area of about 0.1 ha, completely cut off from the river bed (lentic), and only during the flood surges does the oxbow lake have contact with river waters. The reservoir is surrounded by a few trees and its bottom is covered with muddy sediments [35]. The sampling sites were located near the village of Łosino.

OLS9 (54°25′42,6″N 17°01′17,0″E)

The lentic oxbow lake is also located on the left side of Słupia, with an area of about 0.5 ha with a strongly developed shoreline. It is characterized by a strong development of reed beds and spot alders shading the water surface. The water of the reservoir is cloudy, and the bottom is covered with a silty sediment [35]. The sampling sites were located near the village of Łosino.

3.3.2 Łyna Oxbow Lake (13)

OLŁ3 (54°01′03,9″N 20°24′11,9″E)

The maximum length of the oxbow is about 503 m and the maximum depth is about 2.6 m. The oxbow lake was created artificially; its one arm is connected to the river (semi-lotic) which enables water exchange and migration of hydrobionts. Its vegetation is mainly narrow reed strips and sedges. The shore of the oxbow is quite steep and the silty bottom has a high content of detritus [36]. The sampling sites were located near the city of Dobre Miasto.

3.3.3 Drwęca Oxbow Lakes (14–15)

OLD1 and OLD2 (53°28′33,5″N 19°35′48,0″E and 53°28′38,2″N 19°35′51,8″E)

The SD1 and SD2 oxbow lakes are located next to each other and are not connected to the river (lentic), which is one of the reasons for their overgrowth. There are quite narrow belts of rush plants here. Their length is OLD1 about 267 m and OLD2 about 447 m. The water in these oxbow lakes is rather cloudy and has a green-brown colour. The sampling sites were located near the village of Bratian.

3.4 Dam Reservoir

The Krzynia reservoir (16) located in the Słupia catchment on the "Dolina Słupia" Landscape Park (54°20′05″N 17°13′22″E) is a highly shallow reservoir, acting as a settling tank for pollutants carried by river waters. In the lower part, there is a hydroelectric power plant regulating the volume of water in the reservoir as well as the floodplain area below it [37]. Reeds are found in dense fields unevenly distributed throughout the entire reservoir [38].

4 Results

On the biotic substrate, the presence of periphyton representatives from all trophic levels was found. The density of microalgae was the highest in lakes and the lowest on the reed inhabiting river banks (Table 1). Irrespective of the type of aquatic ecosystems, epiphytic Bacillariophyta were responsible for 80–90% of this fraction (Fig. 2).

The level of biodiversity (Shannon index) was comparable and fluctuated in the narrow range of 0.8–0.9.

As far as small consumers (microzooperiphyton) are concerned, their highest concentrations were also observed in lakes, which were 20 times greater than those found in the dam reservoir and 15 times greater than in the rivers (Fig. 3). The main component of the microzooperiphyton in the studied aquatic ecosystems was Protista, whose share reached 78% (lakes), while Rotifera (~40%) was only recorded in the oxbow lakes in comparable quantities (Fig. 3).

Table 1 Densities of microzooperiphyton (thousand indiv. m^{-2}), macrozooperiphyton (indiv. m^{-2}) and phytoperiphyton (mln cells m^{-2}) in the studied Polish water ecosystems

	Lakes				Rivers				Oxbow Lakes							Dam reservoir Krzynia [38]
	Kortowskie	Wicko Przymorskie [22, 23]	Raduńskie Dolne	Lubowidzkie [24, 49]	Łyna River [25, 36]	Drwęca	Kwacza [31]	Słupia [32, 37]	OLŁ3	OLD1	OLD2	OLS3 [33, 35]	OLS7 [33, 35]	OLS8 [33, 35]	OLS9 [33, 35]	
Microzooperiphyton	352.3	153.1	5.3	63.6	5.5	13.5	9.6	8.9	17.2	12.9	11.3	6.1	290.4	24.8	82.8	6.7
Protista	306.5	122.1	1.9	16.9	3.9	10.6	5.3		8.0	9.3	9.1	4.6	111.6	9.4	51.3	4.2
Ciliata-libera	0	0	0.1	2.7	0.2	1.2	0.7	0	0.6	1.3	0.9	0.5	0	0	0	0.5
Testacea	0	26.5	0.3	0.4	0	0.1	0.1	0.5	0	0	0	0	0	0	0	1.3
Peritricha[a]	306.5	94.9	1.5	13.8	3.7	9.4	4.6	1.7	7.4	7.9	8.3	4.1	0	0	0	2.4
Tardigrada	0	0.7	>0.1	0.4	0	0	0	0.1	0	0	0	0	0.9	0.6	0	0
Rotifera	2.0	8.6	0.6	5.5	1.0	0.1	2.4	2.4	1.7	1.4	0.3	0.9	153.2	6.2	22.9	0.9
Nematoda	41.1	12.7	2.1	36.4	0.2	0.4	1.8	0.9	2.0	1.1	1.1	0.5	12.7	3.8	3.8	1.5
Oligochaeta	1.2	1.0	0.3	1.3	0	1.3	0.1	0.2	2.4	0.2	0.4	0.1	0.2	0.2	0.2	>0.1
Chaetogaster sp.	0.4	0.1	0.2	0.7	0	1.2	0.1	>0.1	2.4	0.1	0.3	0	0	0	0	2
Nais sp.	0.8	0.9	>0.1	0	0	0.1	0	0.2	0	0.1	0.1	0.1	0	0	0	0
Stvlaria sp.	0	0	>0.1	0.6	0	0	0	0	0	0	0	0	0	0	0	>0.1
Copepoda	1.4	2.7	0.3	2.2	0.2	0.6	0	2.2	2.5	0.4	0.3	0	8.1	2.2	2.2	0
Cyclopoida	0	0.2	>0.1	0.2	0	0.6	0	0	0	0	0	0	0	0	0	0
Calanoida	0	0.1	0	0.2	0.2	0	0	0	0	0	0	0	0	0	0	0
Harpacticoida	1.1	1.2	0.1	1.2	0	0	0	1.6	0.7	0.3	0.3	0	0	0	0	0
Nauplius	0.3	1.2	0.1	0.6	0	0	0	0.6	1.8	0.1	0	0	0	0	0	0
Cladocera	>0.1	5.4	>0.1	0.9	0.2	0.5	0	0.9	0.5	0.4	0.1	0	3.7	2.4	2.4	0.1
Bosminia sp.	0	0.2	>0.1	0.6	0.2	0	0	0	0	0.1	0	0	0	0	0	>0.1
Chvdorus sp.	>0.1	4.1	0	0.3	0	0.5	0	0.9	0.5	0.3	0.1	0	0	0	0	0.1
Daphnia sp.	0	1.2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Arachnidae	0	0	0	0	0	0	0	0	0.1	0.1	0	0	0	0	0	0
Macrozooperiphyton	223	21,961	159	5,165	300	400	80	279	65	150	2	65	1,300	100	1,800	609
Chironomidae larvae	223	13,057	159	5,165	100	400	30	108	59	120	1	59	1,000	63	900	581
Hydra vulgaris	0	294	0	0	0	0	0	0	0	0	0	0	0	0	0	28
Trichoptera	0	0	0	0	200	0	0	0	0	0	0	0	0	0	0	0
Plecoptera	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hirudinea	0	64	0	0	0	0	50	0	0	30	0	0	0	0	0	0
Asellus aquaticus	0	0	0	0	0	0	0	163	6	0	1	6	100	22	0	0

(continued)

Table 1 (continued)

	Lakes				Rivers				Oxbow Lakes							Dam reservoir Krzynia [38]
	Kortowskie	Wicko Przymorskie [22, 23]	Raduńskie Dolne	Lubowidzkie [24, 49]	Łyna River [25, 36]	Drwęca	Kwacza [31]	Słupia [32, 37]	OLŁ3	OLD1	OLD2	OLS3 [33, 35]	OLS7 [33, 35]	OLS8 [33, 35]	OLS9 [33, 35]	
Other	0	8.6	0	0	0	0	0	8	0	0	0	0	200	15	900	0
Phytoperiphyton	89.8	19.4	37.3	23.4	2.1	5.1	1.9	15.2	7.4	6.9	8.2	15.0	34.3	9.4	18.0	29.7
Bacillariophyta	82.6	13.7	36.3	23.3	2.1	5.1	1.9	14.2	7.4	6.9	8.2	15.0	31.2	6.3	15.3	27.1
Chlorophyta	7.1	4.4	1.0	>0.1	0	0	>0.1	0.8	0	0	0	0	3.1	3.1	2.7	1.5
Cyanobacteria	0.1	1.3	0	>0.1	0	0	0	0.2	0	0	0	0	0	0	0	1.1

[a] zooids m^{-2}

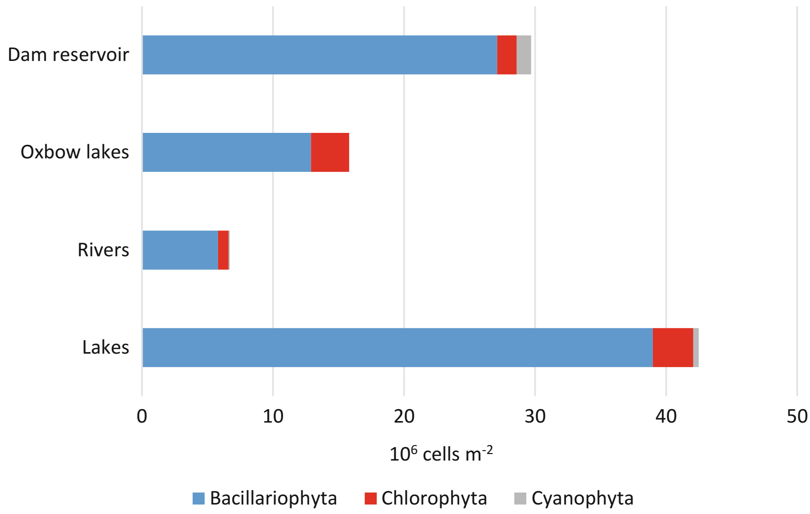

Fig. 2 Comparison of algae density (mln cells m^{-2}) inhabiting reeds in the studied Polish aquatic ecosystems

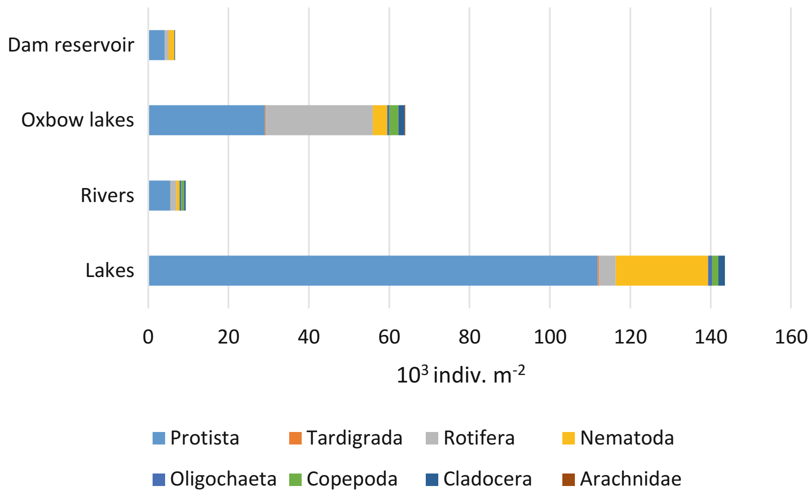

Fig. 3 Comparison of the density of periphyton microfauna (thousand indiv. m^{-2}) on reed substrate in the studied Polish aquatic ecosystems

The level of biodiversity underwent significant changes and reached the maximum value of 1.69 in rivers and the lowest in lakes 0.96.

Larger consumers (macrozooperiphyton) achieved the highest density values in lakes, while in other types of ecosystems, their abundance was lower and comparable (Fig. 4). The main component of macrozooperiphyton in all the studied water ecosystems was Chironomidae larvae, whose participation reached the level of 95%

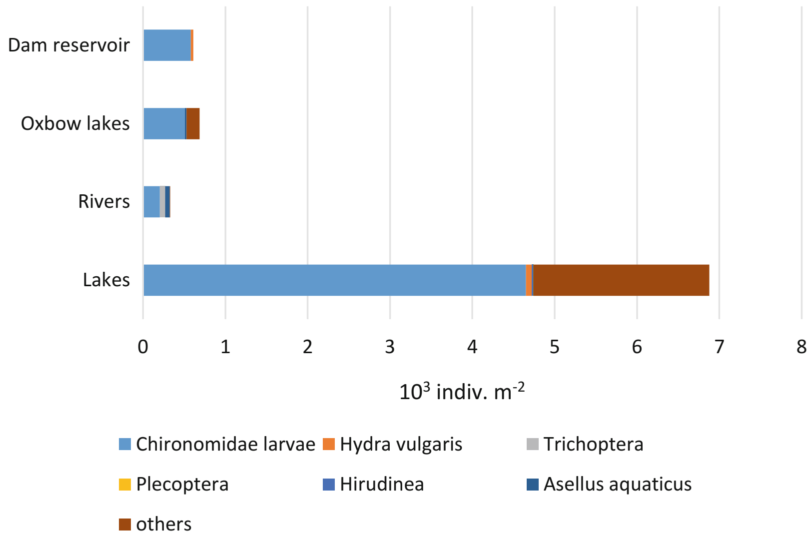

Fig. 4 Comparison of the density of periphyton macrofauna (indiv. m^{-2}) on reed substrate in the studied Polish aquatic ecosystems (Table 1). Densities of microzooperiphyton (thousand indiv. m^{-2}), macrozooperiphyton (indiv. m^{-2}) and phytoperiphyton (mln cells m^{-2}) in the studied Polish water ecosystems

(dam reservoir) and was the lowest in rivers (~60%). The level of biodiversity underwent significant changes in the studied Polish water ecosystems and had the maximum value of 0.96 in rivers and 0.19 in the dam reservoir.

4.1 Lakes

4.1.1 Lake Kortowskie

Phytoperiphyton P. australis were inhabited most abundantly by algae ($\bar{x} = 89.8$ mln m^{-2}) among the lakes studied (Table 1). Bacillariophyta were predominant, accounting for 92% of phytoperiphyton. The concentration of diatoms on the reed ranged from 8.9 mln to 21.6 mln cells per m^{-2} ($\bar{x} = 82.6$ mln m^{-2}). They were accompanied by Chlorophyta representatives, accounting for 8% of algae, and their density ranged from 0.8 to 42.3 mln cells per m^{-2} ($\bar{x} = 0.7$ mln cells m^{-2}). The density of *Cyanobacteria* ranged from 0 to 0.4 mln cells m^{-2} ($\bar{x} = 0.05$ mln cells m^{-2}).

Zooperiphyton The microzooperiphyton of Lake Kortowskie was represented by nine taxa, whose density reached 352.3 thousand indiv. m^{-2}, the highest recorded in the studied Polish ecosystems. Among these invertebrates, Peritricha (Protista), Nematoda and Copepoda achieved the largest numbers. Peritrich's quantitative share in the microfauna was 87% and was accompanied by Nematoda, accounting for 12% of the microzooperiphyton. In the macrofauna, only one taxon was observed (Chironomidae larvae), whose average density in the lake was 223 indiv. m^{-2} (Table 1).

4.1.2 Lake Wicko Przymorskie

Phytoperiphyton The Lake Wicko Przymorskie reeds were most inhabited by algae ($\bar{x}$ = 19.4 mln m^{-2}) (Table 1). Among the epiphytic algae, Bacillariophyta predominated, constituting 70% of phytoperiphyton. The concentration of diatoms on the reed ranged from 2.9 mln to 17.6 mln cells m^{-2} ($\bar{x}$ = 13.7 mln cells m^{-2}). Chlorophyta representatives were accompanied by 22% of epiphytic algae, and their density ranged from 0.8 to 6.3 mln cells m^{-2} ($\bar{x}$ = 4.4 mln cells m^{-2}). *Cyanobacteria* also had a significant share in Lake Wicko, with a density of 0 to 2.1 mln cells m^{-2} ($\bar{x}$ = 1.4 mln cells m^{-2}).

Zooperiphyton The study of periphyton colonizing the biotic substrate formed by *P. australis* involved 14 taxa of epiphytic microfauna, whose average density was 153 thousand indiv. m^{-2}. Peritricha and Nematoda were commonly found in this fraction. Rotifera and Testacea also had a very high presence. Peritricha (95,000 indiv. m^{-2}) dominated in terms of density in the microzooperiphyton and accounted for 54% of the community, and they were accompanied by shelled amoeba – 15% of the community (26,000 indiv. m^{-2}). Nematoda reached a significant abundance – 7% (13,000 indiv. m^{-2}). The density of the remaining microinvertebrates ranged from 0.1 to 8 thousand indiv. m^{-2} (0.1–4.9%). Macrofauna inhabiting *P. australis* were represented by five taxa with a density of 22.1 thousand indiv. m^{-2}. Among them, the highest density was recorded for Chironomidae larvae, 8% (over 13,000 m^{-2}), and bryozoa (over 5.6 thousand m^{-2} – 3%). The abundance of other taxa ranged from 0.06 to 3 thousand m^{-2}.

4.1.3 Lake Raduńskie Dolne

Phytoperiphyton Periphyton colonizing reeds in this lake were numerously represented ($\bar{x}$ = 37.3 mln cells m^{-2}) and dominated by Bacillariophyta, which accounted for 98% of phythoperiphyton. The concentration of diatoms on reeds ranged from 0.4 to 76.3 mln cells m^{-2} ($\bar{x}$ = 36.3 mln cells m^{-2}) (Table 1). They were accompanied by Chlorophyta, which accounted for 2.5% of the algae. Their density ranged from 0 to 1.8 million cells m^{-2} (mean density of about 1 mln cells m^{-2}). Among the green algae, the genus *Spirogyra* and *Cladophora* dominated.

Zooperiphyton There were 13 taxa of microfauna on reeds in Lake Raduńskie Dolne, with an average density of 5.3 thousand indiv. m^{-2}. Among microperiphyton in terms of quantity, Nematoda reached the highest concentration (41%), and the subdominants were sedentary Peritricha ciliates (28%). Rotifera (11%) also reached significant density, while Testacea, *Chaetogaster* sp., Harpacticoid, Cladocera and

Copepoda larvae (nauplii) grew poorly in reeds, and their percentage share in periphyton microfauna ranged from 3% to 6%. The macrofauna on the reeds was represented only by Chirodominae larvae, whose average density was 159 indiv. m^{-2}, which was the lowest among the lakes studied (Table 1).

4.1.4 Lake Lubowidzkie

Phytoperiphyton In the reed phytoperiphyton of Lake Lubowidzkie, representatives of Bacillariophyta, Chlorophyta and *Cyanobacteria*, with a total concentration of 23.4 mln cells m^{-2}, were recorded (Table 1). Among the phytoperiphyton, the predominant taxa were diatoms, which accounted for almost 100% of this fraction. Chlorophyta and *Cyanobacteria* were present in a small density, only >1% of phytoperiphyton.

Zooperiphyton On the reeds in Lake Lubowidzkie there were ten taxa of microfauna, with an average density of 63.6 thousand indiv. m^{-2}. Among microperiphyton in terms of quantity, Nematoda reached the highest density (57%), and the subdominants were sedimentation Peritricha and Ciliata libera (22%, respectively). Cladocera appeared on the reeds to a lesser extent. The only representatives of the macrofauna were Chironomidae larvae, which constituted 100% of the macrozooperyphyton density, and their average density was 5.2 thousand indiv. m^{-2} (Table 1).

4.2 Rivers

4.2.1 Łyna River

Phytoperiphyton On the tested substrate in the Łyna River, the only representatives of producers were diatoms – Bacillariophyta (Table 1). Their average density was 2.1 mln cells m^{-2}. Single forms and colonies were distinguished between them. A more numerous group turned out to be single forms, whose average density was 1.6 mln indiv. m^{-2} with a percentage share of nearly 80%.

Zooperiphyton The microzooperiphyton in the Łyna River were represented by Protista, Rotifera, Nematoda, Copepoda and Cladocera, whose total density amounted to 5.5 thousand indiv. m^{-2} (Table 1). The largest share in the microfauna was reached by Protista 70% ($\bar{x} = 3.9$ thousand indiv. m^{-2}), among which the most abundant were Peritricha. The Protista were accompanied by Rotifera, constituting approx. 20% of the total number of microfauna. Macrozooperiphyton were represented by Chironomidae and Trichoptera larvae. On the reeds in this river (which is an astonishing situation), the Trichoptera larvae dominated, with a share of 74% ($\bar{x} = 750$ indiv. m^{-2}). Typical representatives of macrofauna periphyton – Chironomidae larvae – accounted for only 26% with a concentration of 240 indiv. m^{-2} (Table 1).

4.2.2 Drwęca River

Phytoperiphyton In the Drwęca River, algae were represented exclusively by diatoms (Bacillariphyta), whose density was 5.1 mln m^{-2}. Among them, both single and colony forms were identified. Single forms accounted for 98% of phytoperiphyton.

Zooperiphyton The representatives of the periphyton microfauna included Protista, Rotifera, Nematoda, Oligochaeta, Copepoda and Cladocera, whose total concentration amounted to 13.5 thousand indiv. m^{-2} (Table 1). The main component of the microfauna were Protista with 11 thousand indiv. m^{-2}, and the least numerous (which is a rare situation) were Rotifera – 81 thousand indiv. m^{-2}. Protista accounted for 80% of microfauna and were classified as the dominant group, while the rest were taxonomic sub-groups. The macrofauna were represented by Chironomidae, Plecoptera and *Asellus aquaticus* larvae, whose total density was 400 indiv. m^{-2}. Throughout the study period, the main component of macrofauna were Chironomidae larvae, which accounted for 96% of the periphyton macrofauna.

4.2.3 Kwacza River

Phytoperiphyton The representatives of the producers were Bacillariophyta and Chlorophyta, whose density was the lowest among the analysed rivers ($\bar{x}$ = 1.9 mln indiv. m^{-2}). Diatoms accounted for 96% of phytoperiphyton and their average density was 1.9 mln cells m^{-2}. Chlorophyta accounted for only 2.3% of the total phytoperiphyton. Their highest density was 0.1 mln indiv. m^{-2} (Table 1).

Zooperiphyton The density of the microfauna fraction was high compared to other rivers studied ($\bar{x}$ = 9.6 thousand indiv. m^{-2}). The most abundant group of microfauna was Protista (55%), and the least abundant was Oligochaeta (1%). The predominant protists were accompanied by Rotifera, which constituted 25% of the microzooperiphyton ($\bar{x}$ = 2.4 thousand indiv. m^{-2}). Oligochaeta was represented only by predators *Chaetogaster* sp. ($\bar{x}$ = 0.3 thousand indiv. m^{-2}). Macrofauna was represented by Chironomidae, Hirudinea and *A. aquaticus* larvae, which occurred in particular seasons. Hirudinea accounted for the largest share, reaching almost 50% of the entire macrozooperiphyton. Their density was 50 indiv. m^{-2}. The density of the larvae Chironomidae was 29% ($\bar{x}$ = 29.4 indiv. m^{-2}).

4.2.4 Słupia River

Phytoperiphyton Phytoperiphyton in the Słupia River were represented by Bacillariophyta, Chlorophyta and *Cyanobacteria* ($\bar{x}$ = 15.2 mln cells m^{-2}), (Table 1). Among the representatives of epiphytic producers, diatoms prevailed, which accounted for over 93% of the total phytoperiphyton, and were accompanied by Chlorophyta (accounting for 6%) and *Cyanobacteria* (accounting for 1%).

Zooperiphyton In the Słupia River, there was a taxonomically diverse formation of microzooperiphyton, in which representatives of Protista, Rotifera, Tadrigrada, Nematoda, Oligochaeta, Copepoda and Cladocera were observed with a total density of 8.9 thousand indiv. m^{-2} (Table 1). The largest share among them was achieved by Rotifera, which accounted for nearly 27% of this formation ($\bar{x}$ = 2.4 thousand indiv. m^{-2}). They were accompanied by Protista ($\bar{x}$ = 2.2 thousand indiv. m^{-2}), which found favourable conditions for development and constituted 25% of the microfauna. The microzooperiphyton were very resistant to environmentally sensitive types of invertebrates such as Tardigrada (they constituted only 1%). Oligochaeta (2%) also revealed a small density. Macrozooperiphyton in the Słupia River were represented by three taxa: Chironomidae, Hirudinea and Crustacea (*A. aquaticus*) larvae. The dominant species, with 60% of the macrofauna, was *A. aquaticus* ($\bar{x}$ = 13 indiv. m^{-2}) accompanied by Chironomidae larvae. Hirudinea appeared in a negligible amount on the reed substrate (less than 1%).

4.3 Oxbow Lakes

4.3.1 Słupia River (OLS3)

Phytoperiphyton Algae colonizing the reed in the OLS3 reached a significant density ($\bar{x}$ = 15 mln cells m^{-2}). They were represented only by Bacillariophyta (Table 1). This taxon occurred throughout the study period and was mainly represented by single forms – 77%.

Zooperiphyton The microfauna was represented by Protista, Rotifera, Nematoda, Oligochaeta and Cladocera with a total density of 6.1 thousand indiv. m^{-2} (Table 1). The most numerous representatives of Protista were 81% ($\bar{x}$ = 4.6 thousand indiv. m^{-2}). The second most important component of this animal fraction were Rotifera ($\bar{x}$ = 0.9 thousand indiv. m^{-2}). A small percentage of the entire microzooperiphyton was made up of Oligochaeta (3%) and Copepoda (3%). The periphyton macrofauna was represented by only two taxa: Chironomidae larvae and *A. aquaticus* with very low density ($\bar{x}$ = 65 indiv. m^{-2}). Chironomidae larvae were significantly dominant, accounting for 90% of the total macrozooperyphyton, whereas the concentration of *A. aquaticus* was very low.

4.3.2 Słupia River (OLS7)

Phytoperiphyton The reed substrate was inhabited by algae dominated by diatoms ($\bar{x}$ = 34.3 mln cells m^{-2}), among which single-celled forms dominated. The average density of diatoms was 31.3 mln cells m^{-2} and green algae 3.1 mln cells m^{-2} (Table 1). In addition, there were also representatives of Chlorophyta, which accounted for 17% of algae.

Microzooperiphyton The biotic substrate created by the reeds in this water reservoir was the development site of microfauna represented by Protista, Rotifera, Nematoda, Oligochaeta, Cladocera, Copepoda and Tardigrada, whose density was the highest ($\bar{x}$ = 290 thousand indiv. m^{-2}) among the studied oxbow lakes (Table 1). In terms of quantity in the microzooperiphyton, Rotifera had the highest average density (53%), followed by Protista (38%). Macrofauna were represented by Chironomidae and Odonata larvae and representatives of Hirudine, Gastropoda, and Crustace (*A. aqaticus*). In terms of quantity, Chironomidae larvae (58%) and Hirudinea (19%) predominated.

4.3.3 Słupia River (OLS8)

Phytoperiphyton The algae colonizing reeds in this oxbow lake were the least abundant in the Słupia catchment (9.4 mln cells m^{-2}) and were characterized by the high dominance of Bacillariophyta, whose percentage share was 68% ($\bar{x}$ = 6.3 mln cells m^{-2}). There were also representatives of Chlorophyta, but no *Cyanobacteria* were present (Table 1).

Zooperiphyton The substrate created by *P. australis* was a place for the development of microfauna represented by the same species as in the OLS7 oxbow lake (Table 1). However, the abundance of algae was almost four times lower here. In terms of quantity, as in the case of the OLS7 oxbow lake, the highest density was achieved by Protista (38%); their average density was 9.4 thousand indiv. m^{-2}. They were accompanied by Rotifera (25%), with an average density of 6.2 thousand indiv. m^{-2}. The macrozooperiphyton represented a low abundance group of epiphytic organisms ($\bar{x}$ = 100 indiv. m^{-2}). The presence of Chironomidae larvae was observed, which constituted 63% of the macrozooperiphyton. A small density was achieved by *A. aqaticus* and Gastropoda (Table 1).

4.3.4 Słupia River (OLS9)

Phytoperiphyton The reed substrate was inhabited by algae ($\bar{x}$ = 17 mln cells m^{-2}) dominated by Bacillariophyta, which accounted for 85% of the microphytoperiphyton. The concentration of diatoms ranged from 6.4 mln m^{-2} to 24.6 mln cells m^{-2} (Table 1). Apart from them, Chlorophyta representatives appeared with a density of 5.6 mln cells m^{-2}.

Zooperiphyton The group consisted of the same taxa as in OLS7 and OLS8, with a density of 82.8 thousand indiv. m^{-2} (Table 1). The highest density was achieved by Protista, constituting 52% of microfauna, Rotifera (23%) and Nematoda (14%). Macrozooperiphyton was the most abundant among the oxbow lakes. The dominant taxa in this fraction were Gastropoda (49%) and Chironomidae larvae (50%). Hirudinea had the smallest density, with a percentage share of 1% (Table 1).

4.3.5 Łyna River (OLŁ3)

Phytoperiphyton Algae inhabiting the stems of reeds in the Łyna oxbow lake were much more abundant than in the Słupia River oxbow lakes. The representatives of the producers observed in the OLŁ3 floodplain were Bacillariophyta, which accounted for 100% of the phytoperiphyton. The average density of diatoms was 15 mln cells m^{-2} (Table 1).

Zooperiphyton The microfauna fraction reached a density of 17.2 thousand indiv. m^{-2}, which was a lower value than in Słupia's old riverbanks. Protista, Rotifera, Nematoda, Oligochaeta, Copepoda, Cladocera and Hydrachnellae were identified among them (Table 1). The highest percentage participation in the microperiphyton was achieved by protists, which accounted for more than 75% of microfauna ($\bar{x} = 8$ thousand indiv. m^{-2}). In this group, Peritricha dominated, which accounted for 50% of protists. The macrozooperiphyton in the OLŁ3 oxbow lake were represented by Chironomidae larvae and *A. aquaticus*. The average density of Chironomidae larvae was 58.6 thousand indiv. m^{-2}, thus representing 90% of the macrozooperiphyton.

4.3.6 Drwęca River (OLD1)

Phytoperiphyton On the common reed, the dominant taxa (100% phytoperiphyton) were Bacillariophyta. Diatoms were present throughout the whole study period, reaching a density of 6.9 mln cells m^{-2}, which was the lowest among the tested oxbow lakes. The single forms constituted 85% of the phytoperiphyton, similar to the Łyna oxbow lakes, where single diatoms were the dominant form.

Zooperiphyton In OLD1, the microfauna representatives were Protista, Rotifera, Nematoda, Oligochaeta, Copepoda, Cladocera and Arachnidae ($\bar{x} = 12.9$ thousand indiv. m^{-2}) (Table 1). The dominant taxa were Protista, which accounted for 73% of the microzooperiphyton at a density of 9.1 thousand indiv. m^{-2}. They were accompanied by Rotifera (11%), and Nematoda (9%). The biotic substrate *P. australis* was settled by large invertebrate animals: Chironomidae larvae (80%) and Hirudinea, which accounted for 20% of the total macrozooperiphyton, whose total density amounted to 150 indiv. m^{-2}.

4.3.7 Drwęca River (OLD2)

Phytoperiphyton On the common reed, the dominant taxa (100% phytoperiphyton) were Bacillariophyta. Diatoms were present throughout the research period, reaching a density of 8 mln cells m^{-2}. Single forms accounted for 72% of the phytoperiphyton (as in in OLD1), where single diatoms were also the dominant form.

Zooperiphyton The fauna in OLD2 was very similar in terms of quality and quantity to OLD1. The dominant taxa were Protista, which accounted for 73% of the microzooperiphyton at a density of 8.3 thousand indiv. m^{-2}. They were accompanied by Rotifera (10%) and Nematoda (9%). The biotic substrate (*P. australis*) was inhabited by large invertebrate animals: Chironomidae larvae (85%) and *A. aquaticus*, which constituted 15% of the total macrozooperiphyton, whose total density was 120 indiv. m^{-2}.

4.4 Dam Reservoir (Lake Krzynia)

Phytoperiphyton The abundance of algae was relatively high and amounted to 29.7 mln cells m^{-2}, slightly lower than in lakes (Table 1). Representatives of the producers in the Krzynia reservoir included Bacilariophyta, which constituted about 99% of the total phytoperiphyton. Chlorophyta and *Cyanobacteria* were found in small amounts, which accounted for about 1% of the total phythoperiphyton.

Zooperiphyton The density of microfauna was low ($\bar{x} = 6.7$ thousand indiv. m^{-2}). The highest share in the microzooperiphyton was Protista (63%), and the dominant group was Peritricha. The average density of protists was 4.2 thousand indiv. m^{-2}. Nematoda accounted for 22% of the total microperiphyton. A comparatively small density compared to other ecosystems was achieved by Rotifera, which constituted only 14% of microfauna. The macrozooperiphyton in the reservoir of Krzynia was represented by Chironomidae larvae and *Hydra vulgaris*. Chironomidae larvae constituted 95% of the macrofauna, and their average density was 0.6 thousand m^{-2}. *H. vulgaris* was 5% of macrozooperiphyton.

5 Discussion

Epiphytic organisms are one of ecological formations, which are present in all types of water bodies, regardless of their trophic status [39], and influence on dynamics of enviromental conditions in water ecosystems [40–42]. Knowledge in recent years about the periphyton and its impact on life in water reservoirs is increasing. For Polish aquatic ecosystems, the level of recognition of this formation is still negligible [1]. The role of periphyton is quite important, because it primarily takes part in the process of self-purification of water. The substrates are overgrown by producers, consumers and reducers belonging to the brush formation, which eliminate mineral salts from the water and break down compounds and organic substances. They also eliminate seston in its mineral form and pelagic bacteria, thus affecting the clarity of waters. Periphyton are group of organisms characterized by high resistance to factors that have a negative impact on the environment and are therefore present in distorted ecosystems [43]. The occurrence and development of the periphyton depend on the influence of various and variable factors shaping the conditions in the environments from which the samples were taken. In these and other studies, differences in the abundance of periphyton are generally attributed to environmental factors, which include nutrients or salinity [44–47]. Depending on the environment, its pollution, eutrophication and connection with the river or sea, grassland syndromes develop, which differ to a certain degree in the species composition and the number of individual taxa.

Qualitative and quantitative research of periphyton point to the fact that this formation plays a significant ecological role in water reservoirs contributing to the increase of biodiversity, water purification and deeutrophisation, creating a food base for other animals, including economically useful fish species. According to Piesik [43, 48], a series of zooperiphyton feeds on algae, which stimulates their development. Such a situation was observed in Lake Kortowskie, in which high abundance of zooperiphyton is accompanied by numerous algae. At the same time, the factor that strongly affects the amount of phytoplankton is quantity of nutrients. This is the case, among others, in Lake Raduńskie Dolny, which is a reservoir subject to high eutrophication (the II class of water quality) and II class of susceptibility to degradation [26]. Eutrophication contributes to reducing the transparency of water, which affects the development of phytoplankton. The physico-chemical conditions observed in this lake are similar to those observed in Lake Lubowidzkie [49]. It is a small water reservoir, to which large amounts of phosphorus are introduced from the fish ponds, which largely determine the level of trophics and the associated increase in primary production. It also seems that water pollution is one of the factors affecting the quality and quantitative development of periphyton organisms in the coastal Lake Wicko. It has no free exchange of water with the sea and freshwater inflow is dispresed.

Possible contaminants entering the lake may come from periodically supplying rivers and drainage ditches that supply water from fields and meadows. For the studied rivers, since only one large urban agglomeration is located in their catchments, its impact on the condition of the lotic ecosystem can be demonstrated. Rivers have constant contact with anthropogenic pollution, which produces, among others, an increase in the content of nitrogen and phosphorus ions [50]. Heavy metal ions appear in small amounts in surface waters from municipal sewage, as well as area runoffs. A very important factor is the movement of water, which “washes away” the epiphyte and greatly hinders the construction of periphyton structures [51]. As a result, the abundance of the periphyton is the lowest observed in these ecosystems. In the oxbow lake, the action of this factor is severely weakened, which is why the growth of periphyton is stronger on the reed shoots [10]. A characteristic feature of the periphyton growing in common reed in the oxbow lakes was the quite rich species composition on this plant. The rivers of northern Poland are characterized by a rather straight line of the trough in the lower section. In the nineteenth and twentieth centuries, drainage processes were carried out, as a consequence of which entire floodplains are now covered with numerous oxbow lakes. These phenomena caused many adverse changes in the river ecosystems, limiting the retention capacity and their overgrowth with rush vegetation. This process is limited by the full opening of oxbow lakes to the inflow of river waters [52]. This promotes the migration of organisms and the growth of biodiversity [8] but also limits the development of reed areas. The consequence of this phenomenon is the limitation of the possibility of developing the periphyton as it is in OLS3. In the closed oxbow lakes (OLD1, OLD2, OLS8 and OLS9) and semiopen (OLŁ3 and OLS7), the abundance of periphyton is much higher, which can be explained by moderate or negligible water movement, accessibility of the substrate and ease of access to nutrients [10].

In phytoperiphyton, the highest density was achieved by Bacillariophyta, especially single-celled forms. The domination of diatoms in the periphyton is a typical phenomenon of Polish aquatic ecosystems. Among microzooperiphyton, there were 8–14 taxa in the case of lakes, 6–10 taxa in rivers, 5–11 in oxbow lakes and nine taxa in the dam reservoir. The analysis of the collected material indicated that the highest density in lakes was achieved mainly by Protista; the exception was Lake Lubowidzkie, in which the greatest density was reached by herbivorous Nematoda. As in the lakes, in the examined rivers and oxbow lakes, Protista were dominant. The exception was the Słupia River and its OLS7 and OLS9 oxbow lakes, where Rotifera prevailed. Both Rotifera and Protista (Peritricha) are organism sestons [7, 53]. Pertiricha are also regarded as pioneer organisms, which in terms of abundance usually gain dominance among microfauna [48, 54, 55]. These organisms can have great importance in the process of eliminating pollutant loads from water. With increased pollution, it is possible to observe increasing density of Rotifera, Nematoda, Copepoda and Tartigrada [56].

Among the macrofauna on the substrate formed by common reed in various types of water bodies, Chironomidae larvae predominated. The number and development of Chironomidae larvae is dependent on trophic dependencies and seasonal changes [48, 57]. The pronounced dominance of the larvae indicates higher eutrophication of water bodies [58]. An example can be Lubowidzkie Lake, which is a highly eutrophicated reservoir, which is confirmed by the results of studies in which Oligochaeta and Chironomidae larvae gain significant dominance, with the extreme poverty of other macrofauna taxa [59]. The presence of a dam on the river and the creation of a dam reservoir entail an intensification of the accumulation of sediments carried by the river [38]. This causes shallowing and silting of the dam reservoir which is Lake Krzyńskie and the expansion of rush macrophytes as a potential site for the development of periphyton [60]. These changes had a huge impact, primarily on the bottom fauna, whose main representatives and permanent inhabitants of the retention reservoir were Chironomidae and Oligochaeta larvae. A similar situation was observed during research conducted in the 1980s [61, 62]. The greatest taxonomic diversity was recorded in Lake Wicko. This may be since there is a much higher amount of nitrogen and phosphorus in Pomeranian lakes, which promotes the growth of algae, which are food for first and second order consumers. They are also strongly eutrophicated lakes with frequent lack of oxygen in the bottom zone, which causes the migration of invertebrates to vertical substrates, including reed shoots.

In the structures of the periphyton based on antagonistic and nonantagonistic interactions, important here are algae-eating, predatory and omnivorous forms that are characteristic for this hydrobionts group. Therefore, the periphyton is used to test the water microcosm [61]. Periphyton plays a very important role in the concept of revitalizing water reservoirs, because it can develop on various types of biotic and abiotic substratum. It can act as an indicative group, purifying water and creating a food base for fish [2]. Periphyton have a significant impact on the environment, constituting an essential element of all aquatic ecosystems. They can settle on various types of substrates (artificial, natural). Algae and zooperiphyton should be subject to more detailed taxonomic and quantitative research to show their role in water ecosystems. Taxa, which are indicator organisms, are very useful in determining the state of water quality, because their presence or absence can show the conditions that prevail in a given aquatic ecosystem.

6 Concluding Remarks

The growth of organisms representing all trophic levels (producers, consumers, reducers) was found on the substrate formed by *P. australis* in each of the examined water reservoirs. The greatest abundance of periphyton was recorded in lakes and the lowest in rivers. In the case of biodiversity, its highest values were observed in rivers and the lowest in lakes. Among phytoperiphyton, unicellular Bacillariophyta dominated in the studied water reservoirs. In all ecosystems, the main components of the microfauna were Protista and Rotifera (sedimentators) and, in the case of macrofauna, Chironomidae larvae. The analysis of periphyton diversity indicated that it was the highest in the coastal lake (Lake Wicko) and the lowest in the reopening oxbow lake (OLS3).

References

1. Obolewski K (2009) Charakterystyka przyrodnicza doliny rzeki Kwaczy [W:]. In: Obolewski K (red.) Krótkoterminowe ekologiczne efekty renaturyzacji małych rzek nizinnych na przykładzie rzeki Kwaczy. Słupsk, pp 65–71
2. Piesik Z, Wawrzyniak-Wydrowska B (2003) Organizmy porośłowe (perifiton) zasiedlające trzcinę (Phragmites australis) w ujściowym odcinku rzek Odry i Gunicy. Człowiek i środowisko przyrodnicze Pomorza Zachodniego. Uniwersytet Szczeciński, Szczecin, pp 161–172

3. Kühl H, Kohl JG (1992) Nitrogen accumulation, productivity and stability of reed stands (*Phragmites australis (Cav.) Trin. Ex Steudel*) at different lakes and sites of the lake districts of Uckermark and Mark Brandenburg. Int Revue Ges Hydrobiol 77:85–107
4. du Laing G, van Ryckegem G, Tack FMG, Verloo MG (2006) Metal accumulation in intertidal litter through decomposing leaf blades, sheaths and stems of *Phragmites australis*. Chemosphere 63:1815–1823
5. Obolewski K, Skorbiłowicz E, Skorbiłowicz M, Osadowski Z (2010) Influence of chemical elements contained in reed *Phragmites australis (Cav.) Trin.* on epiphytic algae in riparian buffer zone. Fresenius Environ Bull 19(2a):348–353
6. Lakatos G, Grigorszky I, Bíró P (1998) Reed-periphyton complex in the littoral of shallow lakes. Verh Int Ver Theor Angew Limnol 26:1852–1856
7. Kiss MK, Lakatos G, Borics G, Gid'o Z, De'ak C (2003) Littoral macrophyte–periphyton complexes in two Hungarian shallow waters. Hydrobiologia 506–509:541–548
8. Roberts E, Kroker J, Körner S, Nicklisch A (2003) The role of periphyton during the recolonization of a shallow lake with submerged macrophytes. Hydrobiologia 506–509:525–530
9. Jayawardana JMCK, Westbrooke M, Wilson M, Hurst C (2006) Macroinvertebrate communities in *Phragmites australis* (Cav.) Trin. ex Steud. reed beds and open bank habitats in central victorian streams in Australia. Hydrobiologia 568:169–185
10. Obolewski K (2011) Composition and density of plant-associated invertebrates in relation to environmental gradients and hydrological connectivity of wetlands. Oceanol Hydrobiol St 40(4):52–63
11. Karosiene J, Paskauskas R (2012) Spatial variability of epiphyton communities structure in a temperate estuarine lagoon. Estuar Coast Shelf Sci 114:100–104
12. Casartelli MR, Ferragut C (2015) Influence of seasonality and rooted aquatic macrophyte on periphytic algal community on artificial substratum in a shallow tropical reservoir. Int Rev Hydrobiol 100:158–168
13. Scheffer M, Carpenter SR, Foley JA, Folke C, Walker B (2001) Catastrophic shifts in ecosystems. Nature 413:591–596
14. Jeppesen E, Soendergaard M, Pedersen AR, Jürgens K, Strzelczak A, Lauridsen TL, Johansson LS (2007) Salinity induced regime shift in shallow brackish lagoons. Ecosystems 10:48–58
15. Telesh IV (2004) Plankton of the Baltic estuarine ecosystems with emphasis on Neva Estuary: a review of present knowledge and research perspectives. Mar Pollut Bull 49:206–219
16. Cieśliński R (2011) Geograficzne uwarunkowania zmienności hydrochemicznej jezior wybrzeża południowego Bałtyku. Wydawnictwo Uniwersytety Gdańskiego, Gdańsk
17. Obolewski K, Bąkowska M (2017) Epiphytic invertebrate patterns in coastal lakes along a gradient of salinity and water exchange with the sea. Estuar Coast Shelf S 197:150–158
18. Podbielkowski Z (2002) Fitogeografia części świata, t. 2. Wydawnictwo Naukowe PWN, Warszawa
19. Wolfgang E (1998) Przewodnik flora i fauna wód śródlądowych. Wydawnictwo Multico Oficyna Wydawnicza, Warszawa
20. Piesik Z, Obolewski K (2000) Epiphytic organisms (peryphyton) inhabiting reed, Phragmites australis and artificial substrates in Lake Kopań. Baltic Coastal Zone 4:73–86
21. Lossow K, Gawrońska H, Mientki C, Łopata M, Wiśniewski G (2005) Jeziora Olsztyna. Stan troficzny, zagrożenia. MAZUR PRINT, Olsztyn
22. Piesik Z, Obolewski K (2003) Fauling fauna (zooperipfyton) in habiting *reed Phragmites australis* (CAV.) Trin. ex STEUD. in lake Wicko Przymorskie. Baltic Coastal Zone 8:81–94
23. Obolewski K (2007) Macrozoobenthos in a shallow lake Wicko Przymorskie. Limnol Rev 7(2):71–78
24. Obolewski K (2002) Organizmy poroślowe (perifiton) zasiedlające trzcinę *Phragmites australis* i pałkę *Typha latifolia* oraz sztuczne podłoże w pomorskim jeziorze Lubowidzkie – badania wstępne. Słupskie Prace Przyrodnicze, seria Limnologia 2:71–82

25. Obolewski K, Glińska-Lewczuk K (2005) Influence of the hydrological conditions on the distribution of zoobenthos in the Łyna river headwater. Inżynieria Ekologiczna 13:136–142
26. Raport o stanie środowiska województwa pomorskiego (1999) WIOŚ, Gdańsk, Bibl. Monitoringu Środowiska
27. Mileska M (1992) Słownika geograficzno – krajoznawczy Polski. Wydawnictwo Naukowe PWN, Warszawa, p 491
28. Kondracki J (2002) Geografia regionalna Polski. Wydawnictwo Naukowe PWN, Warszawa
29. Rusiecki J (1981) Szlakiem Drwęcy. Informator turystyczny. Wydawnictwo Krajowa Agencja Wydawnicza, Gdańsk
30. Glińska-Lewczuk K (2009) Cechy środowiska przyrodniczego dorzecza Słupi [W:]. In: Obolewski K (red.) Krótkoterminowe ekologiczne efekty renaturyzacji małych rzek nizinnych na przykładzie rzeki Kwaczy, Słupsk, pp 9–31
31. Obolewski K (2009) Zmiany struktury jakościowo-ilościowej makrozoobentosu rzeki Kwaczy pod wpływem zabiegów renaturyzacji. In: Obolewski K (red.) Krótkoterminowe efekty ekologiczne renaturyzacji niewielkich rzek nizinnych na przykładzie rzeki Kwaczy. Park Krajobrazowy "Dolina Słupi" i AP, Słupsk, pp 201–222
32. Obolewski K (2010) Ocena jakości wód powierzchniowych na obszarach zurbanizowanych z wykorzystaniem makrobezkręgowców na przykładzie rzeki Słupi. Ochrona Środowiska 32(2):35–42
33. Obolewski K, Strzelczak A (2008) An attempt to structure periphyton dynamics with artificial neural networks in the example of oxbow lake re-opening study (Słupia river, Northern Poland). Ecol Quest 10:31–40
34. Bobrowicz G (2003) Starorzecza i inne naturalne eutroficzne zbiorniki wodne. Przyroda Polska nr 3/2003
35. Obolewski K, Glińska-Lewczuk K (2011) The adaptive response of benthic macroinvertebrates to the reconnection of floodplain lakes: a case study of Osokowy Staw in the Słupia river floodplain. Clean (Weinh) 39(9):853–862
36. Skrzypczak M (2006) Chruściki (Trichoptera) środkowego odcinka rzeki Łyny w okolicach Smolajn (k. Dobrego Miasta). UWM, Olsztyn
37. Jarosiewicz A, Obolewski K (2013) Influence of small hydropower plants on the fluctuation of surface and ground water level, discharge and temperature – a case study of the Słupia River. Sci Rev Eng Environ Sci 62:363–373
38. Obolewski K (2008) Rozmieszczenie i struktura makrozoobentosu w zbiorniku Krzynia na terenie Parku Krajobrazowego "Dolina Słupi". Parki Narodowe i Rezerwaty Przyrody 27(1):81–97
39. Lakatos G (1989) Composition of reed periphyton (biotecton) in the Hungarian part of lake Fertö. BFB-Bericht 71:125–134
40. Heino J (2000) Lentic macroinvertebrate assemblage structure along gradients in spatial heterogeneity, habitat size and water chemistry. Hydrobiologia 418:229–242
41. White J, Irvine K (2003) The use of littoral mesohabitats and their macroinvertebrate assemblages in the ecological assessment of lakes. Aquat Conserv 13(4):331–351
42. Trigal C, García-Criado F, Fernández-Aláez C (2006) Among-habitat and temporal variability of selected macroinvertebrate based metrics in a Mediterranean shallow lake (NW Spain). Hydrobiologia 563(1):371–384
43. Piesik Z (1992) Bilogia i ekologiczna rola organizmów porośłowych (peryfiton) zasiedlających sztuczne podłoża w różnych typach wód. Rozprawy i studia T. (CXCVI), Uniwersytet Szczeciński, Szczecin, p 122
44. Arnold SL, Ormerod SJ (1997) Aquatic macroinvertebrates and environmental gradients in Phragmites reeds swamps: implications for conservation. Aquat Conserv 7:153–163
45. Sandilands KA, Hann BJ, Goldsborough LG (2000) The impact of nutrients and submersed macrophytes on invertebrates in a prairie wetland, Delta Marsh, Manitoba. Arch Fur Hydrobiol 148:441–459

46. Sahuquillo M, Poquet JM, Rueda J, Miracle MR (2007) Macroinvertebrate communities in sediment and plants in coastal Mediterranean water bodies (Central Iberian Peninsula). Ann Limnol Int J Lim 43:117–130
47. Anton-Pardo M, Armengol X (2014) Aquatic invertebrate assemblages in ponds from coastal Mediterranean wetlands. Ann Limnol Int J Lim 50:217–230
48. Piesik Z (1978) Rola barier utworzonych z sieci stylonowych w oczyszczaniu ścieków i sestonu i rozpuszczonych postaci miogenów. Maszynopis, Biblioteka Główna Akademii Rolniczej w Szczecinie
49. Obolewski K (2006) Periphyton inhabiting reed, Phragmites australis and artificial substrate in the eutrophicated Lubowidzkie Lake. Arch Environ Prot 32(3):67–82
50. Obolewski K (2009) Wpływ odczynu oraz koncentracji jonów chlorkowych i tlenu w wodzie rowów melioracyjnych na obfitość perifitonu zasiedlającego trzcinę Phragmites australis Trin EX. Steud. Zeszyty Problemowe Postępów Nauk Rolniczych 540:77–87
51. Sand-Jensen K, Madsen TV (1989) Invertebrates graze submerged rooted macrophytes in lowland streams. Oikos 55:420–423
52. Obolewski K, Glińska-Lewczuk K, Ożgo M, Astel A (2016) Connectivity restoration of floodplain lakes: an assessment based on macroinvertebrate communities. Hydrobiologia 774:23–37
53. Tyszka-Mackiewicz J (1983) Bioseston i peryfiton rzeki Wisły na odcinku od Puław do Warszawy. Poznańskie Towarzystwo Przyjaciół Nauk. Prace Komisji Biologicznej 8L:1–114
54. Relini G, Montanari VS, Pisano E (1976) Prime fasi di insedimento su substrati duri immersi in acque del Porto di Genova a diverso grado di inquinamento. Archo Oceanogr Limnol 18(3):113–140
55. Zhang L, Liu J, Li Y (2015) Comparison of the spatial and temporal variability of macroinvertebrate and periphyton-based metrics in a macrophyte-dominated shallow lake. Front Earth Sci 9(1):137–151
56. Piesik Z, Poleszczuk G (2005) Próba określenia zależności równowartości chlorkowej (Cl%) wód Zalewu Szczecińskiego od wielkości napływu wód Odry i od poziomu wód Zatoki Pomorskiej. Słupskie Prace Biol 2:63–74
57. Löhlein B (1996) Seasonal dynamics of aufwuchs Naididae (Oligochaeta) on *Phragmites australis* in a eutrophic lake. Hydrobiologia 334:115–123
58. Périllon C, Hilt S (2016) Groundwater influence differentially affects periphyton and macrophyte production in lakes. Hydrobiologia 778:91–103
59. Raport o stanie środowiska województwa pomorskiego (1983) WIOŚ, Gdańsk, Bibl. Monitoringu Środowiska
60. Raport o stanie środowiska województwa pomorskiego (1988) WIOŚ, Gdańsk, Bibl. Monitoringu Środowiska
61. Su J, Kang D, Xiang W, Wu CX (2017) Periphyton biofilm development and its role in nutrient cycling in paddy microcosms. J Soils Sediments 17:810–819
62. Russo AR (1987) Role of habitat complexity in mediating predation by the gray damelfish Abudefduf sordidus on epiphytal amphipods. Mar Ecol Progr Ser 36:101–105

Trends in the Phytoplankton Variability of the Selected Polish Lakes

Agnieszka Napiórkowska-Krzebietke and Elżbieta Zębek

Contents

Abstract The directional changes of phytoplankton in Polish lakes were presented to show some tendency concerning the total biomass, composition, and biodiversity as related to environmental variables. The selected lakes were analyzed concerning various types of antropopressure, e.g., relatively low, medium, and huge human impact including sewage inflow history, different hydrological regime nature as natural phenomenon of flow-through lakes and as a consequence of artificial including into water-cooling system with short retention time, and different restoration actions (biomanipulation and artificial aeration).

Smaller biomass and more varied structure of phytoplankton (co-dominated by Bacillariophyta, Cryptophyta, Miozoa, and Cyanobacteria) were typical of the PEG Model for mesotrophic or even oligotrophic temperate lakes with lower trophy level.

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On the contrary, a large biomass with Cyanobacteria domination in summer was typical in more eutrophied lakes. The prominent dominants were primarily chroococcalean *Microcystis aeruginosa* and *M. wesenbergii* and filamentous *Planktolyngbya limnetica*, *Pseudanabaena limnetica*, *Limnothrix redekei*, and *Planktothrix agardhii*. Species richness and values of biodiversity index were usually higher in more eutrophied lakes than in mesotrophic lakes. The overall relationships of phytoplankton groups with environmental variables indicated that water transparency in less eutrophied lakes while water temperature and nutrient concentrations in more eutrophied lakes induced phytoplankton growth.

Keywords Biodiversity · Cyanobacteria · Phytoplankton biomass · Plankton Ecology Group Model · Species richness

1 Introduction

Climate changes can accelerate the eutrophication of water bodies at a global scale. As negative consequence, the quality of different water bodies tends to worsen with more frequent events of persistent algal blooms. Some evidences confirm that recent changes in climate have had a significant effect on the lake functioning primarily in the case of an increase in the cyanobacterial bloom abundance, frequency, duration, and global distribution [1, 2]. Global warming can also effectively promote the changes in the growth and structure of phytoplankton including a dominance shift toward cyanobacteria which altogether are the symptoms of a progressive eutrophication in the lakes [3–5]. Water eutrophication is additionally accelerated by nutrient enriched inflows, industrial waste, and municipal sewage, together with the other substances which are filtered from the catchment area into the water bodies [6, 7]. The excessive growth of phytoplankton, including primarily cyanobacteria of genus *Microcystis* (Fig. 1), has become a global health hazard and a large economic problem affecting the resources of recreational and potable waters. Under favorable conditions of light, temperature, and nutrient, cyanobacteria can cause persistent and harmful blooms (known as harmful algal blooms, HABs) with a high possibility to release of cyanotoxins into the water column posing, in turn, a special threat to people, pets, and wild animals bathing in or drinking such waters. Recently, the frequency, intensity, and timing of cyanobacterial blooms have been strictly dependent on eutrophication level enhanced by climate warming. However, it is also described as a natural phenomenon [8]. Some findings from Polish lakes clearly confirm that cyanobacterial blooms usually refer to nuisance problems of almost all aquatic ecosystems [9–12]. Therefore, the aim of this article was to describe the directional changes of phytoplankton in the selected Polish lakes and to show the tendency including primarily total biomass, composition, and biodiversity as related to environmental variables.

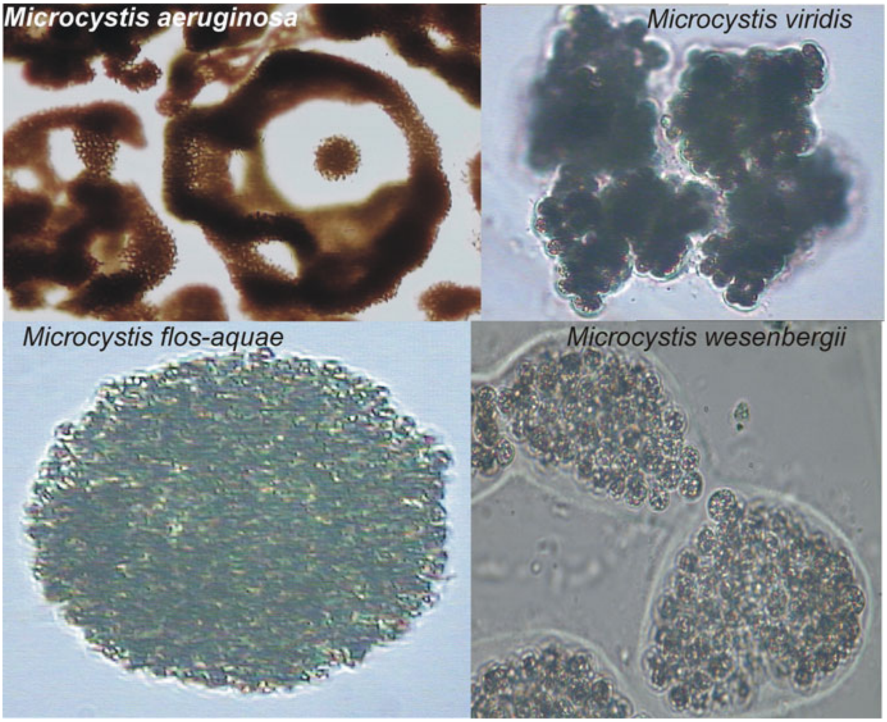

Fig. 1 Cyanobacteria of the genus *Microcystis* in phytoplankton

2 Trends of Phytoplankton Changes in Polish Lakes

2.1 Methodological Approach

The selected Polish lakes (Fig. 2) were analyzed taking into account various types of antropopressure, e.g., relatively low, medium, and huge human impact, different hydrological regime nature: (1) as natural phenomenon of flow-through lakes and (2) as a consequence of artificial including into water-cooling system with short retention time and as well as different restoration actions (biomanipulation and artificial aeration). Trends in changes of phytoplankton biomass and structure were, then, analyzed in the lakes differed in morphometry, trophy, and mictic type. The lakes are both stratified and nonstratified and have surface area from 7.8 ha to 2,600.0 ha and maximum depth from 1.9 m to 108.5 m (Table 1). The trophy parameters which are primarily the concentrations of phosphorus, nitrogen, and chlorophyll *a* as well as water transparency also decided about the lakes' differentiation (Table 2).

The average seasonal concentrations of total nitrogen varied between 0.63 mg dm^{-3} and 3.00 mg dm^{-3}, total phosphorus between 0.048 mg dm^{-3} and 0.850 mg dm^{-3}, and chlorophyll *a* between 3.0 μg dm^{-3} and 92.7 μg dm^{-3}. The

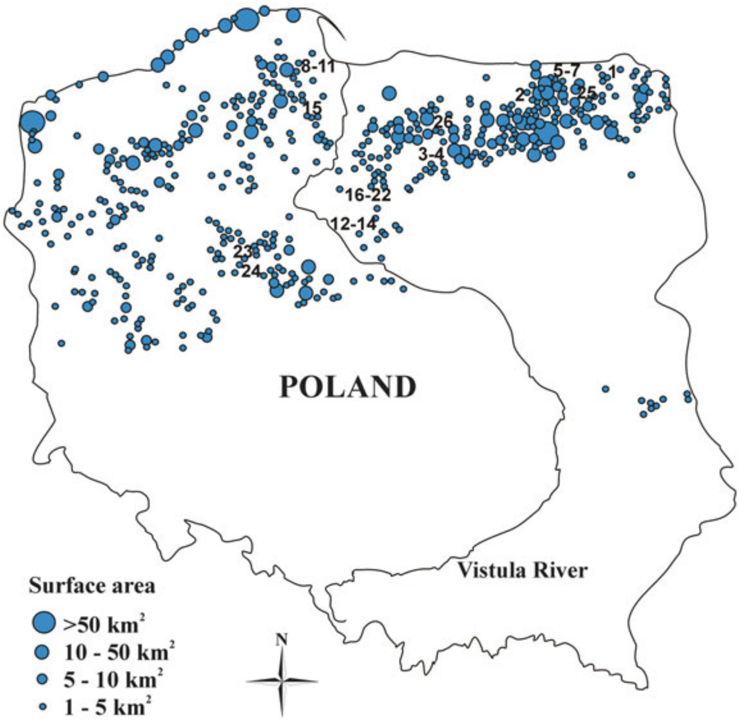

Fig. 2 The location of selected Polish lakes, 1–26 – the numbers of lakes given in Table 1

average water transparency expressed as Secchi disk depth ranged from 0.3 m to 5.4 m. The simplest method to express trophic state of lakes is the trophic state index TSI given by Carlson [13]. Originally, TSI consists of three components, i.e., TSI_{SD}, $TSI_{TP,}$ and TSI_{Chl} calculated from Secchi disk depth and concentrations of total phosphorus and chlorophyll *a*, respectively. All measurements should be primarily done during the summer period. Total nitrogen was additionally included into TSI as TSI_{TN} [14].

The calculated values of TSI were chosen in this article as a multiparametric key factor to express the changes of water quality. The total biomass and structure of phytoplankton were analyzed according to standard methods. Generally, they are subjected to the considerable fluctuations and directional changes connected strictly with the lake trophic state. The seasonal variability of phytoplankton growth in the lakes was also compared with the PEG (plankton ecology group) Models for temperate lakes [16–19]. Therefore, the recent data (from 2000 up to date) of total phytoplankton biomass were transformed and presented as the optimal trend line using the polynomial method to obtain the best fit to data points.

The biomass of main phytoplankton groups, i.e., Cyanobacteria, Bacillariophyta, Chlorophyta, Cryptophyta, and Miozoa (previously Dinophyta), was also correlated with physical and chemical water parameters using the canonical correspondence

Table 1 Morphometric characteristics of the lakes

No	Lakes	Surface area (ha)	Max depth (m)	Mean depth (m)	Mictic type
1	Hańcza	311.4	108.5	38.7	S
2	Dejguny	765.3	45.0	12.0	S
3	Pluszne	903.3	52.0	14.9	S
4	Łańskie	1,042.3	53.0	16.0	S
5	Mamry Północne	2,504.0	43.8	11.7	S
6	Kirsajty	207.0	5.8	3.2	NS
7	Niegocin	2,600.0	39.7	9.9	S
8	Klasztorne Duże	57.5	8.5	4.8	S
9	Klasztorne Małe	13.7	20.0	8.1	S
10	Mielenko	7.8	1.9	1.3	NS
11	Karczemne	40.4	3.2	2.0	NS
12	Święte	32.1	2.1	1.5	NS
13	Skępskie Wielkie	120.0	4.2	2.8	NS
14	Skępskie Małe	15.8	2.0	1.0	NS
15	Wierzysko	57.5	7.6	4.4	NS
16	Dąbrowa Wielka	615.1	34.7	8.2	S
17	Dąbrowa Mała	173.4	34.5	10.0	S
18	Hartowieckie	69.6	5.2	2.9	NS
19	Zarybinek	73.8	7.0	2.4	NS
20	Grądy	112.7	9.1	4.7	NS
21	Tarczyńskie	163.8	9.2	3.8	NS
22	Zwiniarz	50.0	5.8	3.0	NS
23	Licheńskie	147.6	12.6	4.6	NS
24	Ślesińskie	152.3	24.5	7.2	S
25	Warniak	38.4	3.7	1.2	NS
26	Jeziorak Mały	26.0	6.4	3.4	NS

S stratified, *NS* nonstratified

analysis (CCA). To reduce the number of variables a forward selection procedure using the Monte Carlo test with 999 permutations was applied. These relationships were presented on a biplots graph using Canoco for Windows 4.5 software. The currently accepted taxonomic names of phyla and species were given according to AlgaeBase [20].

2.2 Lakes with a Relatively Low or Medium Human Impact

The lakes with a mesotrophic and meso-eutrophic state (TSI values 47–59) are represented primarily by the deepest Lake Hańcza in Poland and relatively large Lakes Dejguny, Pluszne, and Łańskie (Fig. 3a) as well as Mamry Północne, Niegocin, and small Lake Kirsajty (Fig. 4a). Lake Hańcza was characterized by

Table 2 Average seasonal values of trophy parameters and trophic state index (TSI)

Lake	TN (mg dm^{-3})	TP (mg dm^{-3})	SD (m)	Chl *a* (μg dm^{-3})	TSI[a]
Hańcza	0.68	0.086	5.4	3.5	49
Dejguny	0.93	0.070	3.7	6.7	51
Pluszne	0.90	0.113	2.3	11.1	59
Łańskie	0.76	0.131	2.9	10.2	57
Mamry Północne	1.43	0.056	5.1	3.0	47
Kirsajty	n.d.	0.067	3.1	4.5	49
Niegocin	1.48	0.119	2.7	11.9	58
Klasztorne Duże	1.77	0.292	0.9	43.1	69
Klasztorne Małe	2.02	0.446	0.6	55.0	73
Mielenko	1.99	0.156	0.6	32.8	72
Karczemne	3.00	0.850	0.3	84.3	86
Święte	1.39	0.181	1.6	7.8	64
Skępskie Wielkie	2.47	0.133	0.5	46.4	70
Skępskie Małe	2.13	0.190	0.7	26.6	74
Wierzysko	1.98	0.473	0.8	92.7	78
Dąbrowa Wielka	1.03	0.048	2.7	12.5	53
Dąbrowa Mała	1.01	0.054	1.7	19.2	57
Hartowieckie	1.14	0.068	1.2	26.7	60
Zarybinek	1.07	0.076	1.1	35.5	63
Grądy	1.28	0.084	0.8	56.7	64
Tarczyńskie	1.28	0.088	0.8	55.3	64
Zwiniarz	1.40	0.117	0.8	68.9	69
Licheńskie	0.92	0.124	1.8	23.2	61
Ślesińskie	0.80	0.122	2.3	18.3	57
Warniak (clear-water)	1.29	0.080	2.5	8.1	56
Warniak (turbid-water)	1.15	0.092	1.9	12.4	59
Jeziorak Mały (clear-water)	1.90	n.d.	0.9	n.d.	63
Jeziorak Mały (turbid-water)	2.40	n.d.	0.7	n.d.	66

TN total nitrogen, *TP* total phosphorus, *SD* Secchi disk, *Chl a* chlorophyll *a*
[a]TSI based on summer data, *TSI 40–50* mesotrophy, *TSI 50–60* meso-eutrophy, *TSI 60–70* eutrophy, *TSI > 70* hypereutrophy

the lowest phytoplankton biomass while Lake Dejguny with a little higher biomass. The distinct similarity in both lakes concerned a growth pattern throughout the growth season with two biomass peaks, i.e., the first lower peak in spring or early summer and the second higher peak in autumn, which are typical of mesotrophic lakes constantly with the PEG Model [16, 18, 19]. A distinctly different phytoplankton growth pattern with the second biomass peak in late summer was typical of Lakes Łańskie and Pluszne with higher trophic levels. High phytoplankton biomass with maximum of 23 mg dm^{-3} was noted in Lake Pluszne in 2007–2008; however, the maximum reached only ca. 3 mg dm^{-3} in recent years. Furthermore, Lakes

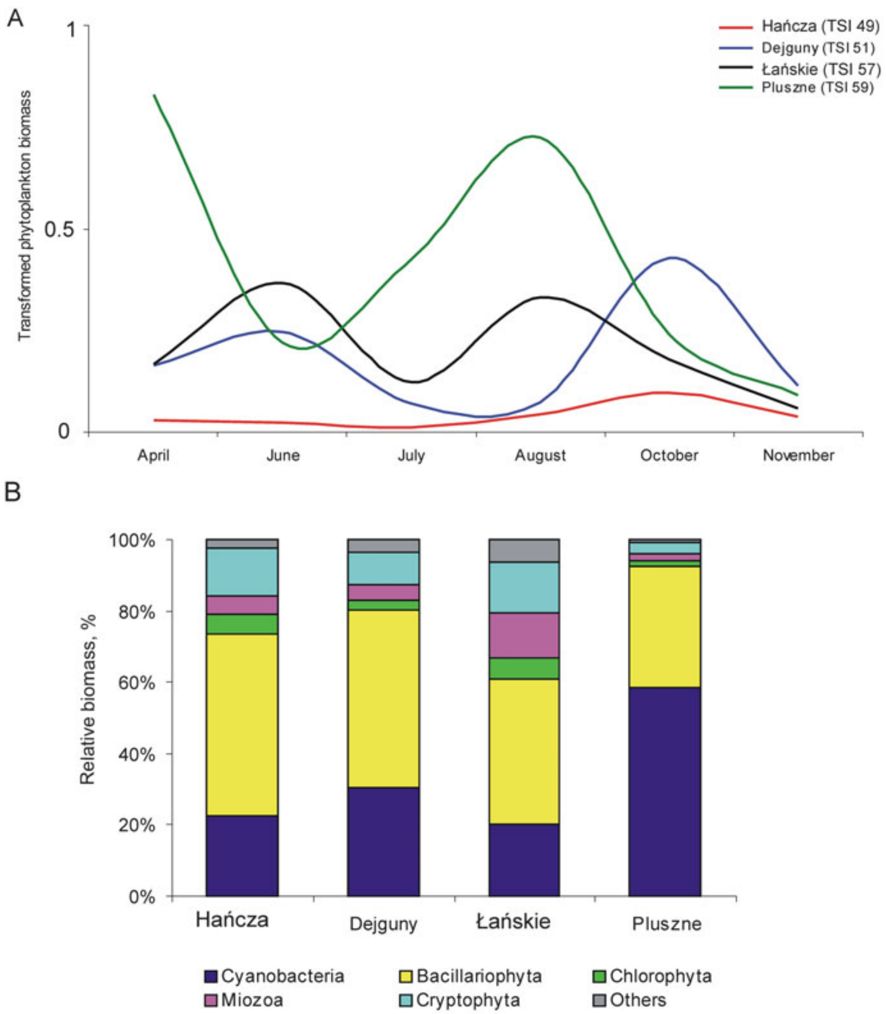

Fig. 3 Seasonal phytoplankton growth pattern (**a**) and an averaged phytoplankton structure (**b**) in the lakes with trophy index TSI 49–59 and $TB_{max} = 23$ mg dm^{-3} (data according to Napiórkowska-Krzebietke and Hutorowicz [22, 24] and Napiórkowska-Krzebietke et al. [23])

Łańskie and Pluszne are included into river-lake system where the coprecipitation process (binding of bicarbonates and phosphorus on calcite) could balance a directly available to phytoplankton phosphorus content in the epilimnion [21].

Distinct domination of Bacillariophyta (ca. 50% of total biomass) in phytoplankton assemblages throughout the growth season was recorded in three of these lakes with the lowest and relatively low biomass (Fig. 3b). Cyanobacteria at 20–30% was the second abundant group. The share above 10% had only Cryptophyta. The common feature for Lakes Hańcza and Dejguny was also the tendency of

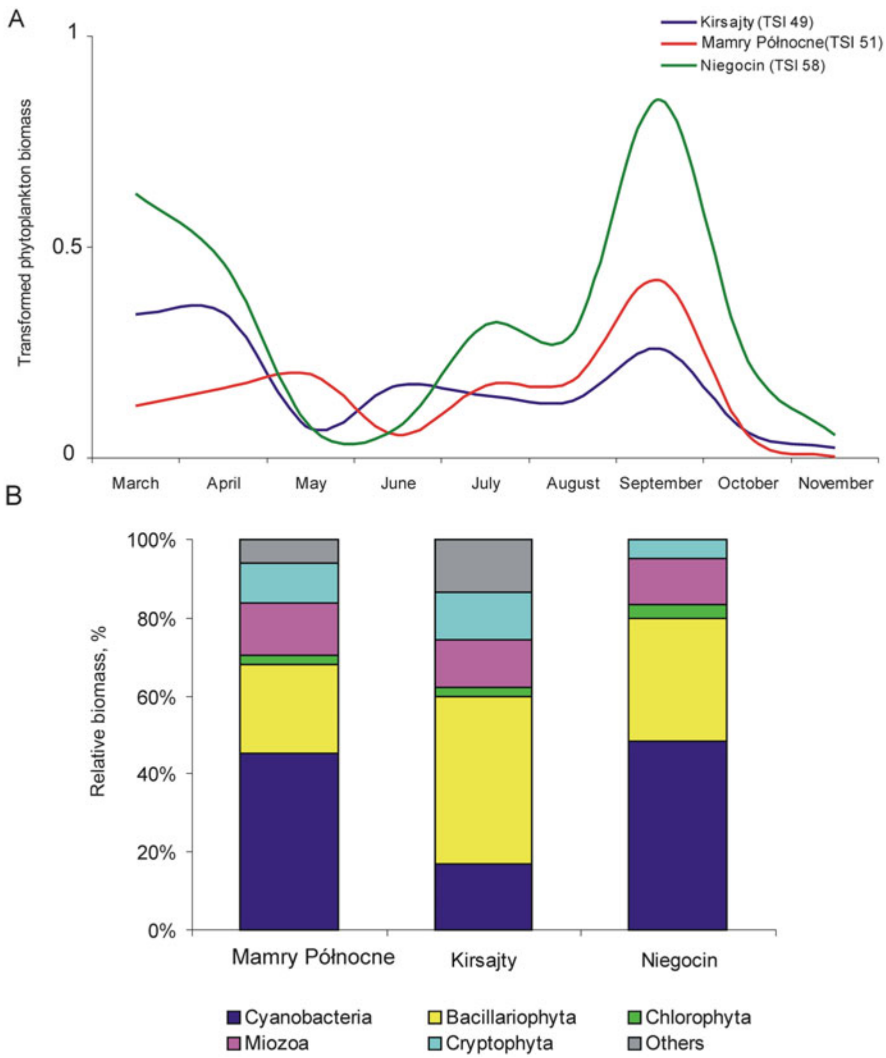

Fig. 4 Seasonal phytoplankton growth pattern (**a**) and an average seasonal composition (**b**) in the selected lakes of Great Masurian Lakes in 2000–2001 with trophy index TSI 49–58 and $TB_{max} = 9$ mg dm^{-3} (data according to Napiórkowska-Krzebietke and Hutorowicz [25, 26, 28], modified)

establishing a "metalimnetic niche" where diatoms of the genus *Cyclotella* in Lake Hańcza and cyanobacteria of the genus *Planktothrix* in Lake Dejguny had favorable conditions for their growth [22, 23]. Relatively low total biomasses and domination of diatom species, *Cyclotella* sp. and *Tabellaria flocculosa* (Roth) Kütz., or chrysophyte species, *Dinobryon sociale* Ehr. and *D. divergens* Imh., with low shares of cyanobacteria in both lakes can indicate the reference conditions and the least good ecological status, as it was previously mentioned for Lake Dejguny [24]. These dominants are typical of deep, well-mixed waters with tolerance for nutrient or/and light deficiency. However, the predominance of filamentous species *Planktothrix agardhii* (Gom.) Anagn. et Kom. and *Planktolyngbya limnetica* (Lemm.) Kom.-Legn. et Cronb. indicated also a clear tendency toward more eutrophic conditions and possibility to occur the cyanotoxins in Lake Dejguny. The similar average phytoplankton structure with dominants of the genera *Cyclotella*, *Fragilaria*, *Asterionella*, and *Dinobryon* was also in Lake Łańskie having a good ecological status supported additionally by intensive water flows [21]. The different phytoplankton assemblages developed in Lake Pluszne. Cyanobacteria attached approximately 60% and diatoms above 30% of total biomass indicated also its more advanced trophy level. Cyanobacteria dominants were represented by the genera of *Aphanizomenon*, *Limnothrix*, *Planktolyngbya*, and *Pseudanabaena* well-known as toxin-producing ones [10]. Cyanobacteria domination or co-domination accompanied by diatoms was characteristic for this lake. However, biomass of these groups varied significantly from year to year.

However, maximum biomass in having additional sewage inflow history Lake Niegocin distinctly suggests higher trophy level. The overall phytoplankton structure in both deep and stratified Lakes Mamry Północne and Niegocin was similar, i.e., with a domination of cyanobacteria and relatively high share of diatoms (Fig. 4b).

On the contrary, phytoplankton in a shallow polymictic Lake Kirsajty was dominated primarily by diatoms but with a relatively high share of cyanobacteria. Comparing to earlier studies [25, 26], biomass in Lakes Mamry Północne and Kirsajty significantly increased at the beginning of twenty-first century, and structure shifted from diatoms, cryptophytes, and dinoflagellates toward a distinct cyanobacteria domination in summer. This suggested a progressive eutrophication of both lakes in 2000–2001. Toxin-producing species, such as *Microcystis aeruginosa* (Kütz.) Kütz., *Aphanizomenon flos-aquae* Ralfs ex Born. et Flah., *Limnothrix redekei* (Goor) Meff., and *Gloeotrichia echinulata* P.G.Richt. began to dominate in the summer periods. All this was a result of stronger anthropopresure, including primarily tourism and recreation development and nutrients flows from the agricultural catchment areas. In Lake Niegocin cyanobacteria-dominated phytoplankton developed under conditions of heavy pollutions until the modernization of the wastewater treatment plant in 1995 [27]. Then, the reduced sewage inflow contributed to significant but not stable decrease in phytoplankton biomass and structure shift toward dinoflagellates (*Ceratium hirundinella* (O.F.Müll.) Duj. and *Peridinium* sp.) domination up to 2000 [28, 29]. After 2000, *Aphanizomenon flos-aquae*, *Planktolyngbya limnetica*, and *Microcystis aeruginosa* were dominants in summer phytoplankton which are able to produce the toxins. According to Jakubowska et al. [30], the concentrations of microcystins (MC-LR, MC-RR, and MC-YR) up to 0.159 μg dm^{-3} were recorded in this lake.

2.3 Urban Lakes

The urban stratified Lakes Klasztorne Duże and Klasztorne Małe and nonstratified Lakes Święte, Skępskie Wielkie, Skępskie Małe, Mielenko, Wierzysko, and Karczemne can be also determine as flow-through lakes differing in a retention time but with an exposition to many point, nonpoint, or diffused sources of pollution, since the 1950s up to date which influenced strongly the water quality [31]. The data set from these lakes, which were characterized by a good, poor, or bad ecological status, includes the total phytoplankton biomasses in the range of 1.5–181.3 mg dm^{-3} [31, 32]. The optimal trend lines present the seasonal growth patterns of phytoplankton along with increasing TSI values from 64 to 86 for Lakes Święte, Klasztorne Duże, Skępskie Wielkie, Mielenko, Klasztorne Małe, Skępskie Małe, Wierzysko, and Karczemne, respectively (Fig. 5a). These TSI values are characteristic of eutrophic and hypereutrophic lakes [13, 15]. The common feature in the seasonal phytoplankton growth was primarily one main biomass peak occurring during the late summer in the lakes at TSI $>$ 70. It is consistent with an original PEG Model typical of eutrophic [16, 18, 19] or nutrient-rich [17] freshwaters. The lakes with trophic index of TSI $<$ 70, i.e., hypereutrophic, were characterized by very high biomass during spring and summer including the occurrence of persistent phytoplankton blooms with a high risk of cyanotoxins [12]. Urban lakes used usually for recreation or even water supply are also treated as a special concern or even "worthy of the best care" waters [33].

An average seasonal structure of phytoplankton in the urban lakes was presented in Fig. 5b. In Lake Święte which was characterized by the lowest TSI and phytoplankton biomass, primarily diatoms of the genera *Navicula*, *Cyclotella*, and *Ulnaria* and cyanobacteria of the genera *Oscillatoria* and *Woronichinia* dominated the assemblages contributing to on average of 30% in the total biomass. Chlorophytes, cryptophytes, and chrysophytes contributed each in 10%. In two other eutrophic lakes, with TSI $\leq$ 70, the share of cyanobacteria was 30 and 80%. In Lake Klasztorne Duże, dinophytes (37%), primarily represented by *Ceratium furcoides* (Lev.) Langh., and cyanobacteria (34%) – *Cuspidothrix issatschenkoi* (Usač) Rajan, *Planktothrix agardhii* (Gom.) Anagn. and Kom., and *Limnothrix redekei* – co-dominated the phytoplankton. The same species of filamentous cyanobacteria with additionally *Planktolyngbya* and *Pseudanabaena* were the most abundant only in Lake Skępskie Wielkie. The prominent dominants of chroococcalean cyanobacteria were, in turn, in hypereutrophic Lake Karczemne, Lake Klasztorne Małe, and Lake Skępskie Małe. *Microcystis aeruginosa* Kütz. and *Microcystis wesenbergii* (Kom.) Kom. developed mostly abundant in these lakes at maximum of total biomass 180 mg dm^{-3}, 100 mg dm^{-3}, and 80 mg dm^{-3}, respectively. In Lake Skępskie Małe, diatoms of the genera *Aulacoseira*, *Cyclotella*, and *Stephanodiscus* had also a significant contribution in the total biomass. The phytoplankton structure in Lake Mielenko was generally similar to Lake Święte. The proportion of co-dominant chlorophytes (primarily of the genus *Pediastrum*), cyanobacteria, cryptophytes, diatoms, and chrysophytes in Lake Mielenko varied

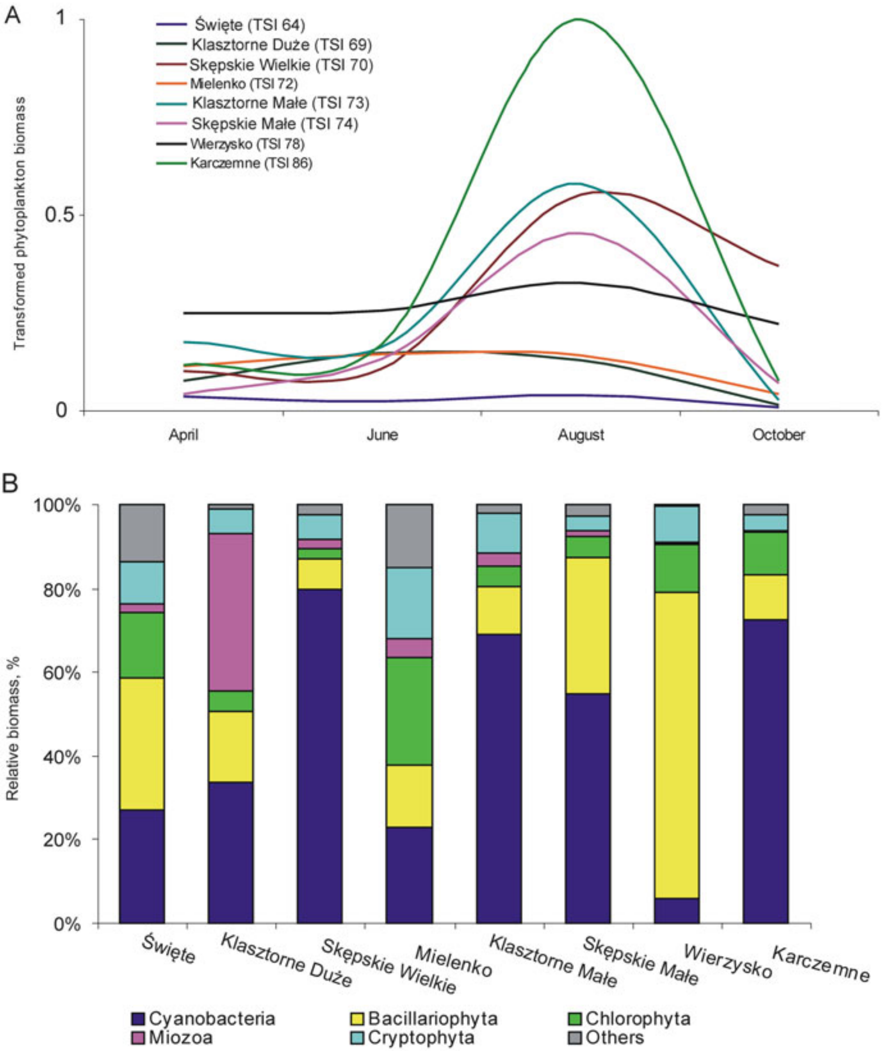

Fig. 5 Seasonal phytoplankton growth pattern (**a**) and an average seasonal composition (**b**) in urban lakes with trophic state index TSI 64–86, TB_{max} = 181 mg dm^{-3} (according to Napiórkowska-Krzebietke Dunalska [32], Napiórkowska-Krzebietke et al. [31], modified)

with declining trend from 26% to 15%. The diatom domination throughout the season with distinct *Pantocsekiella* and *Cyclotella* blooms in summer can be specified as a very rare phenomenon in Lake Wierzysko. These small-sized and fast-growing diatoms were significantly related to high content of ammonium, nitrates, and phosphorus [31] in analogy to findings of Saros et al. [34]. Furthermore, well-mixed water column and a very short retention time could favor the growth of small-sized centric diatoms in a polymictic lake.

2.4 Lakes in the Wel River Catchment

The phytoplankton of the naturally flow-through lakes situated in the Wel River catchment was generally typical of moderately or strongly eutrophic waters or from high to bad ecological status [35, 36]. However, the generalized seasonal growth pattern was different among lakes studied primarily due to various mictic type and trophy level (Fig. 6a). Only Lakes Dąbrowa Wielka and Dąbrowa Mała are stratified and meso-eutrophic. Biomass peak in spring followed by a distinct decline during a "clear-water phase" in May was typical of almost all lakes. The other two peaks occurred in June and August or in July and September in more or less eutrophied nonstratified lakes, respectively. Exceptionally in Lake Zwiniarz, the phytoplankton growth pattern with one summer peak was typical of very nutrient-rich waters [17] or eutrophic temperate lakes [16, 18, 19].

Concerning the average seasonal structure, diatoms (27–49%), and cyanobacteria (23–44%) co-dominated the phytoplankton assemblages in almost all lakes (Fig. 6b). Lower shares of approximately 11, 8, and 5% had Cryptophyta, Chlorophyta, and Miozoa, respectively. The other phytoplankton groups, including planktonic Charophyta, Euglenophyta, and Ochrophyta, contributed only 3% in the total biomass.

The highest biomass formed diatom species of the genera *Cyclotella*, *Stephanodiscus*, and *Asterionella*. Cyanobacteria *Planktolyngbya limnetica*, *Pseudanabaena* sp., *Aphanizomenon gracile* (Lemm.) Lemm., *Cuspidothrix issatschenkoi*, and *Gloeotrichia echinulata* and dinoflagellates *Ceratium hirundinella* dominated primarily in summer in the lakes with TSI $<$ 60, i.e., Lakes Dąbrowa Wielka, Dąbrowa Mała, and Hartowieckie. The shares of cyanobacteria with similar dominant species and additionally *Dolichospermum sigmoideum* (Nyg.) P.Wackl. Hoffm. et Kom. and *D. flos-aquae* (Bréb. ex Born. et Flah.) P.Wackl. Hoffm. et Kom. distinctly increased in summer phytoplankton of more eutrophied Lake Zarybinek. The nostocalean cyanobacteria had lower contribution to the total biomass, whereas the share of cryptophyte *Cryptomonas erosa* Ehr. and cyanobacteria *Aphanocapsa* and *Aphanothece* distinctly increased in Lakes Grądy and Tarczyńskie. Only in Lake Zwiniarz (TSI 69) cyanobacteria formed the largest biomass at average 60% of total biomass. *Aphanizomenon gracile*, *Planktothrix agardhii*, *Microcystis smithii* Kom. et Anagn., and *Microcystis aeruginosa* were dominants and dinoflagellate *Ceratium furcoides* formed a relatively high biomass only in this lake.

2.5 Lakes Artificially Included into Water-Cooling System

Lakes Licheńskie and Ślesińskie are artificially included into water-cooling system and have a short retention time. Since 1958, Lake Licheńskie has been heated throughout the year by receiving ca. half of the post-cooling waters from the

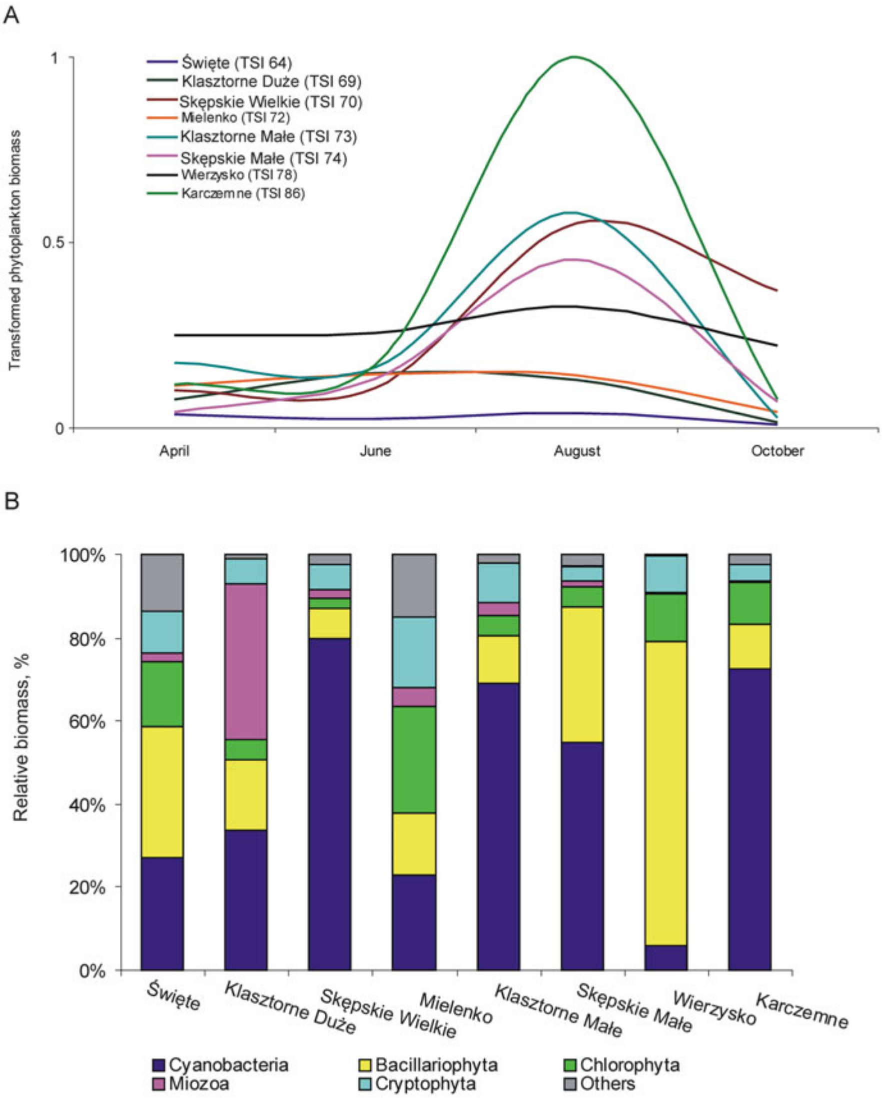

Fig. 6 Seasonal phytoplankton growth pattern (**a**) and an average seasonal composition (**b**) in the lakes of the Wel River catchment with trophy index TSI 53–69 and TB_{max} = 61 mg dm^{-3} (according to Napiórkowska-Krzebietke et al. [35] and Hutorowicz et al. [36], modified)

Konin Electric Power Plant and the Pątnów Electric Power Plant. Lake Ślesińskie is included into this cooling system only in summer, and it receives the heated waters from Lake Licheńskie. The water exchange time differed in both lakes, and it was ca. 2–9 days (longer in summer) in Lake Licheńskie whereas in Lake Ślesińskie on average 11 days in summer and maximum of 36 days in 1995–2010 [37, 38]. Lower retention time in 2004–2005 in Lake Licheńskie influenced the phytoplankton

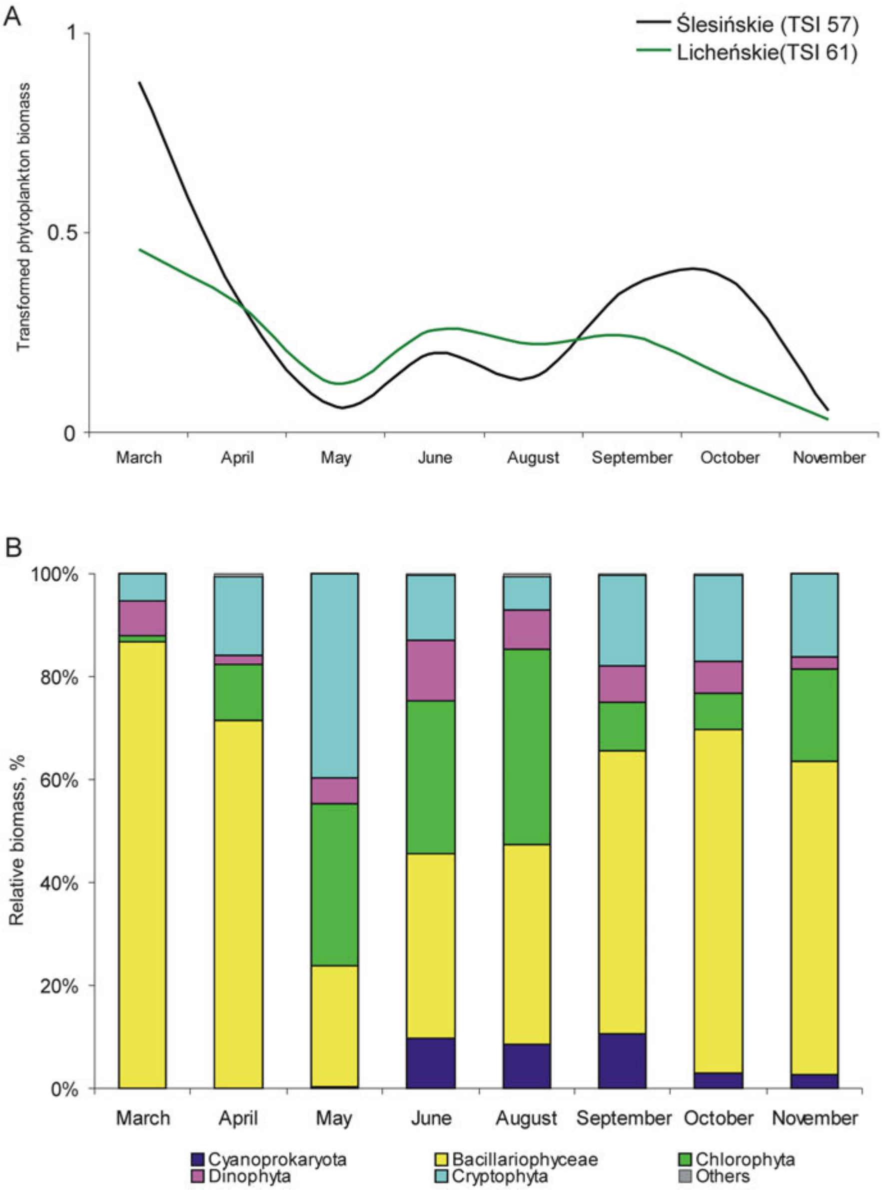

Fig. 7 Seasonal phytoplankton growth pattern in the selected lakes (**a**) and an average structure for both lakes (**b**) included into water-cooling system with trophy index TSI 57–61 and $TB_{max} = 13 \text{ mg dm}^{-3}$ (data according to Napiórkowska-Krzebietke [41], modified)

growth pattern which was similar to Lake Ślesińskie (Fig. 7a). However, some differences concerned higher biomass in March and the occurrence of biomass peak in October only in Lake Ślesińskie. The previous studies suggested that both seasonal phytoplankton dynamic and structure were shaped primarily by higher water temperature and water exchange rate in comparison to other Polish lakes and as well as a relatively high productivity [39, 40].

The phytoplankton structure in both lakes was more or less similar throughout the growth seasons of 2004–2005 (Fig. 7b). Diatoms represented primarily by *Stephanodiscus hantzschii* Grun., *S. neoastrea* Håk. et Hick., *Ulnaria ulna* (Nitzsch) Comp., *U. capitata* (Ehr.) Comp., *Cyclotella* sp., *Nitzschia pusilla* Grun., and *N. palea* (Kütz.) W. Smith were the most abundant in spring and autumn [41]. Cryptophytes (*Cryptomonas curvata* Ehr., *C. erosa*) with chlorophytes (*Oocystis marssonii* Lemm., *Scenedesmus quadricauda* (Turpin) Bréb., *Binuclearia lauterbornii* (Schmidle) Proschk.-Lavr.) and chlorophytes with diatoms co-dominated phytoplankton in May and in summer, respectively. However, cyanobacteria, dinoflagellates, and cryptophytes reached also the shares of 10% in summer. Both phytoplankton biomass and structure were generally linked with an intensive mixing of waters.

2.6 Lakes with Restoration Actions

A small, polymictic pond-type Lake Warniak is characterized primarily by the longest in Poland restoration action history. Since 1967, this lake was subjected to different biomanipulation methods including the introduction of non-native benthophagous common carp (*Cyprinus carpio* L.), herbivorous grass carp (*Ctenopharyngodon idella* (Val.)), seston-feeding silver carp (*Hypophthalmichthys molitrix* (Val.)), and bighead carp (*Hypophthalmichthys nobilis* (Rich.)) [42]. Both consecutively and simultaneously, introduction was to control the development of macrophytes and phytoplankton. Historically, the state of Lake Warniak shifted from a clear-water state in 1967–1974 into a turbid-water state in 1975–1994 [42]. Finally, the attempts to remove the introduced fishes and reintroduction of submerged macrophytes have been made as the next step of restoration. As a satisfactory effect of changes, the return into a clear-water state from 1995 lasting to 2004 was recorded.

Recent (from 2000 up to date) studies of phytoplankton include the period of pilotage stockings with predatory fish, primarily pike (*Esox lucius* L.) and an occurrence of two alternative stable states of a shallow lake, i.e., a clear-water state lasting up to 2004 and a turbid-water state from 2005 up to date. Two similar peaks of biomass, one in spring and the other in summer, recorded up to 2004 were consistent with an original PEG Model typical of seasonal growth pattern in oligotrophic temperate lakes [16, 18, 19] or in moderately nutrient-rich waters [17] (Fig. 8). This state was supported by abundant occurrence of *Chara* species (primarily *Chara rudis* (A.Braun) Leonh., *C. globularis* Thuill., *C. contraria* A. Braun ex Kütz., and *C. tomentosa* L.). Since 2005, the changes of phytoplankton biomass with a distinct one biomass peak occurring in summer have resembled the patterns typical of eutrophic or nutrient-rich lakes.

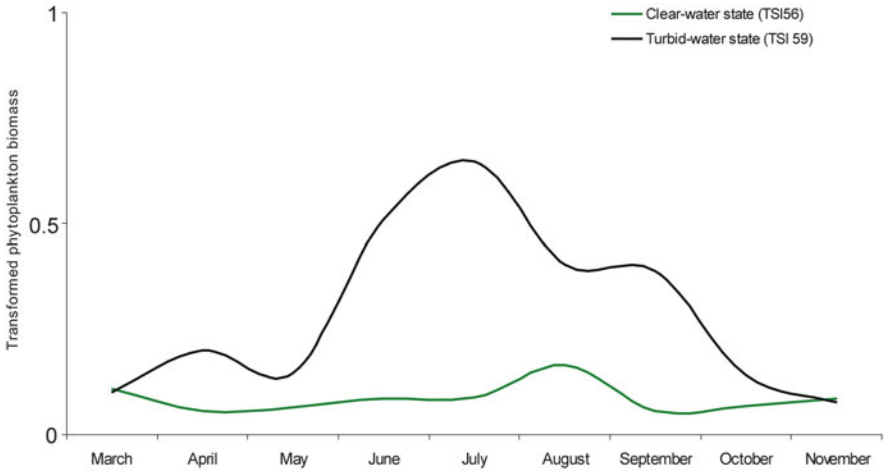

Fig. 8 Seasonal phytoplankton growth pattern in Lake Warniak with the trophy index TSI 56–59 and $TB_{max} = 20$ mg dm^{-3} (data according to Napiórkowska-Krzebietke [42], modified)

In 2000–2004, cryptophytes dominated the phytoplankton assemblages throughout the growth season (Fig. 9a). Dominants were *Cryptomonas curvata*, *C. marssonii* Skuja, *Plagioselmis nannoplanctica* (H.Skuja) G.Nov., I.A.N.Luc. et S.Morr., and *Rhodomonas lens* Pasch. et Ruttn. Diatoms and chlorophytes had also relatively high shares of 39 and 27% in spring and summer, respectively. In 2005–2014, i.e., during the turbid-water state, diatoms, cryptophytes, and chrysophytes co-dominated spring and autumn phytoplankton (Fig. 9b). *Ulnaria acus* (Kütz.) Aboal, *Fragilaria crotonensis* Kitt., *Pantocsekiella comensis* (Grun.) K.T.Kiss et E.Ács, *Asterionella formosa* Hass., *Dinobryon divergens* O.E.Imh., *Uroglenopsis americana* (G.N.Calk.) Lemm., and *Mallomonas lychenensis* W.Conr. were then the most abundant. Summer phytoplankton, in turn, was co-dominated by cyanobacteria with dominant picoplanktonic colonial *Aphanocapsa incerta* (Lemm.) G.Cronb. et Kom. and dinoflagellates primarily *Peridinium willei* Huitf.-Kaas and *Ceratium hirundinella*.

The detailed studies [42] confirmed that the period after Charales disappearance in Lake Warniak was characterized by rapid and high amplitude changes in biomass with only a slight change in trophy level, i.e., TSI values from 56 to 59. The unstable phytoplankton structure was manifested then by a few shifts from generally cyanobacteria-dominated through dinoflagellate-dominated and cryptophyte-dominated to diatom-dominated assemblages. Furthermore, dominant phytoplankton species were typical either of nutrient-rich or nutrient-poor lakes. The biomanipulation stage with predatory pike stockings in 2000–2014 supported the clear-water state in Lake Warniak but only together with removal of herbivorous and seston-feeding fish and Charales re-domination. Adverse changes in water quality appeared after Charales disappearance in 2004 despite continued stockings.

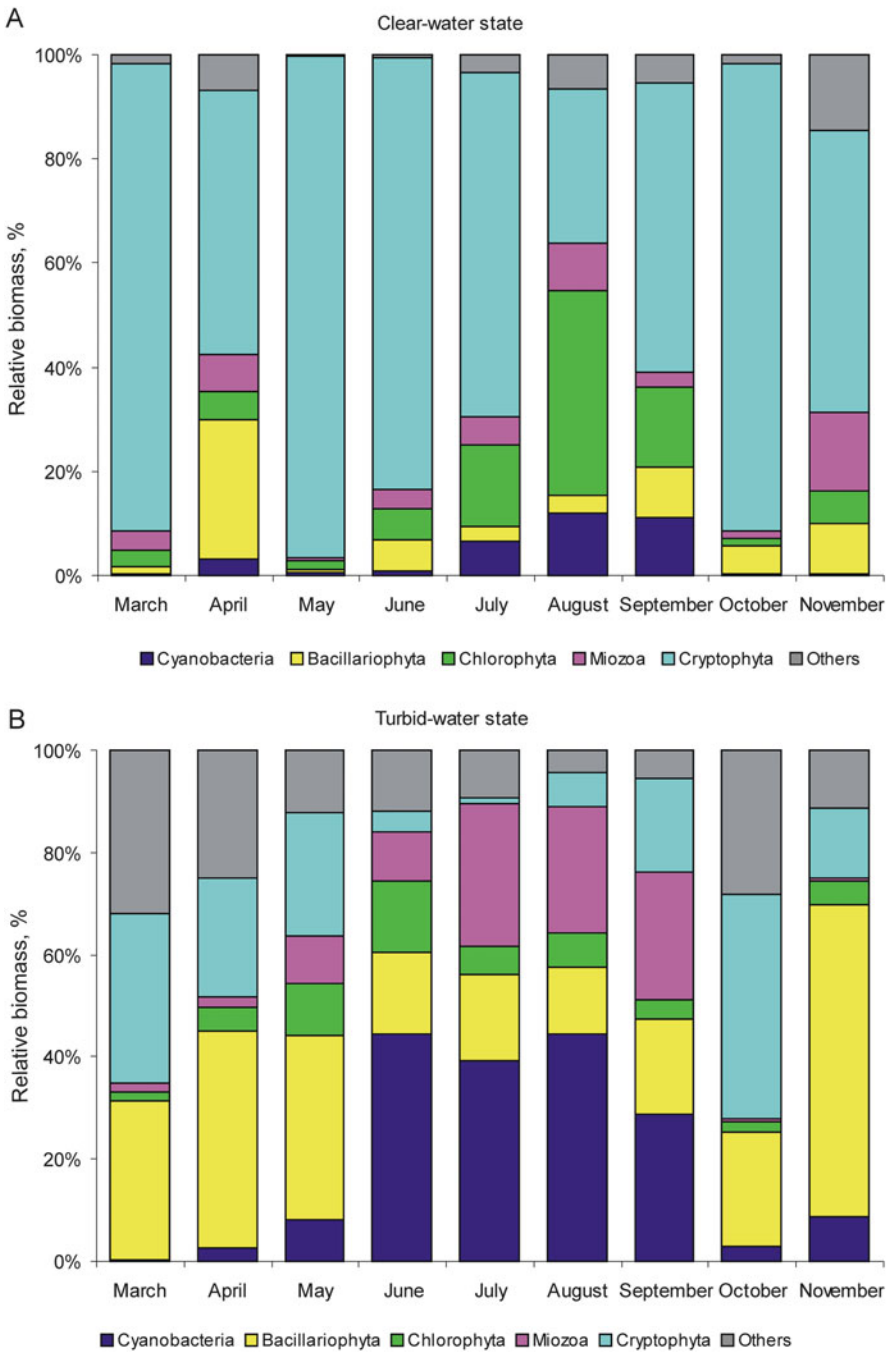

Fig. 9 Seasonal phytoplankton composition in Lake Warniak during the clear-water state (**a**) and turbid-water state (**b**) (data according to Napiórkowska-Krzebietke [42], modified)

Eutrophication processes can be effectively decelerated by reducing the inflow of nutrients, especially phosphorus, through lake basin restoration methods [43–45]. Lake Jeziorak Mały has been restored with the installation of separators for storm water pretreatment and fountain-based aeration. In this lake, the maximum of phytoplankton biomass at 41 mg dm^{-3} was recorded in June at turbid-state water, while two peaks (in April and June) were found at clear-state water conditions (Fig. 10). The first state of water quality was characterized by lower water transparency at the mean SD of 0.7 m ($TSI_{SD} = 64.3$) and higher mean TN of 2.4 mg dm^{-3} ($TSI_{TN} = 67.1$) than at the second water state (SD of 0.9 m – $TSI_{SD} = 61.5$, 1.9 N mg dm^{-3} – $TSI_{TN} = 63.7$).

In Lake Jeziorak Mały, phytoplankton was dominated by diatoms in spring (April), with higher proportion (75%) together with cyanobacteria (22%) at the clear- than turbid-water state (Fig. 11a), while higher proportion of cryptomonads at 20% and chlorophytes at 6% were noted at the second water state (Fig. 11b). Cyanobacteria dominated in both water states in hotter months (from May to September, A, and May to August, B) with the maximum of 91% and 70% in June, respectively. In the studied state periods, the exchange in dominant cyanobacteria species from *Limnothrix redekei* to *Planktolyngbya brevicellularis* Cronb. et Kom was recorded [46]. In July dinoflagellates (Miozoa) with *Ceratium hirundinella* and *Parvodinium inconspicuum* (Lemm.) S. Carty as dominant species, and chrysophytes (as others with *Dinobryon* sp.) shared significant percentages of the total phytoplankton biomass (34% at the clear-water and 23% at the turbid-water conditions, respectively). Whereas, at the second water state, the maximum of Miozoa (24%) was recorded only in September together with significant proportion of diatoms (36%) and cryptomonads (21%). Diatoms were dominated by the species

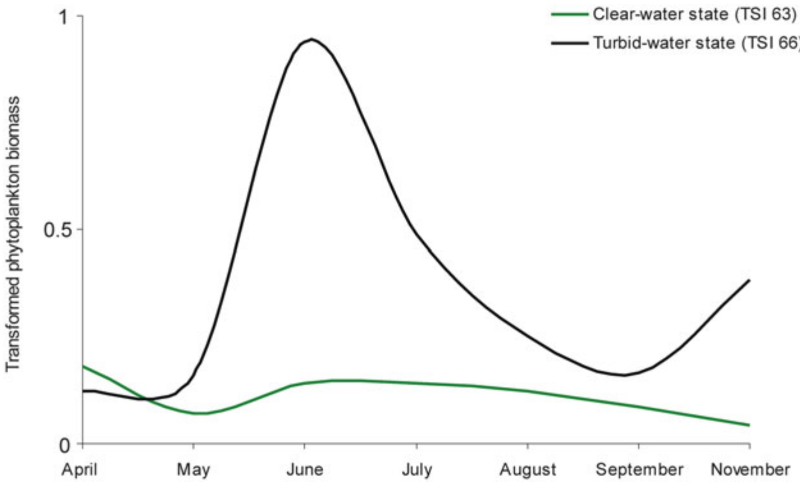

Fig. 10 Seasonal phytoplankton growth pattern in Lake Jeziorak Mały with the trophy index TSI 63–66 and $TB_{max} = 41$ mg dm^{-3} (data according to Zębek and Napiórkowska-Krzebietke [46], modified)

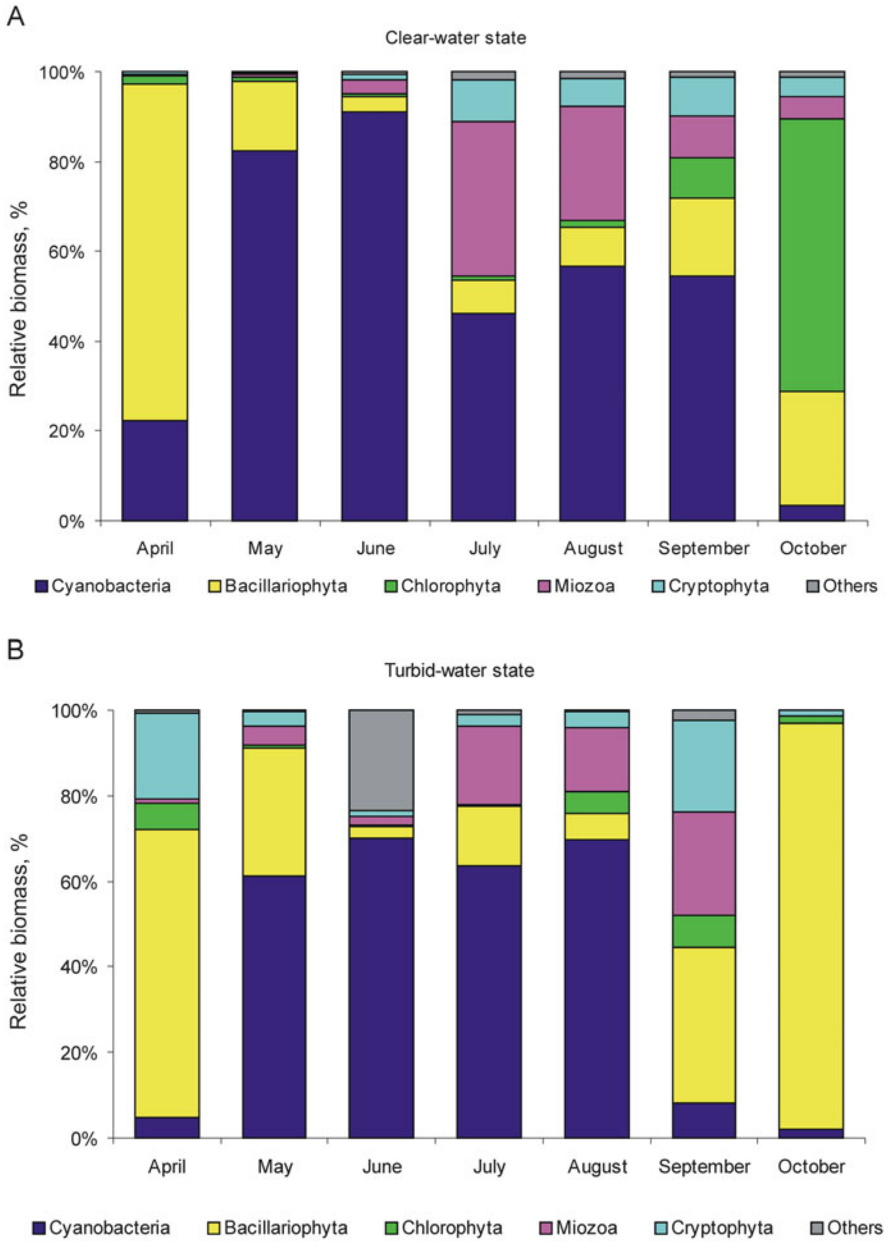

Fig. 11 Seasonal phytoplankton composition in Lake Jeziorak Mały during the clear-water state (**a**) and turbid-water state (**b**), data according to Zębek and Napiórkowska-Krzebietke [46], modified)

Asterionella formosa, *Fragilaria crotonensis*, and *Ulnaria delicatissima* (W.Smith) Aboal et P.C.Silva, while the highest proportion in the cryptomonad biomass attached *Cryptomonas erosa*. However, chlorophytes (61%) dominated at clear-water state and diatoms (95%) at turbid-water state in October (Fig. 11a, b). Chlorophyte dominants were *Chlamydomonas* spp., *Desmodesmus communis* E.Heg., *Monoraphidium griffithii* (Berk.) Kom.-Legn., *Tetraedron minimum* (A.Braun) Hansg., and *Golenkinia radiata* Chod. The differentiations in phytoplankton biomass structure between clear- and turbid-water states were caused by variation of physicochemical water parameters (SD, TN, P-PO_4) and the sum of monthly rainfalls (maximum in July) [46]. The lower values of total nitrogen and orthophosphates were registered in years at clear-water state. The growth of cyanobacteria was induced also by a value of rain precipitation. The minimum of cyanobacteria biomass was noted at precipitation above 100 mm as extreme developmental conditions for them in years at turbid-water state [47, 48]. Additionally, the structure of phytoplankton biomass especially in summer months was changed by the fountain operations. These induced a disturbance in the cyanobacteria and stimulated growth of other phytoplankton groups at decreasing water temperature, oxygenation, and nutrient concentrations [49]. The analysis demonstrated that the restorative procedures in Lake Jeziorak included the installation of separators for storm water pretreatment, and the fountain-based aeration induced changes in the annual biomass structure of phytoplankton. It was proved by the decrease of cyanobacteria proportion and the increase in other phytoplankton groups, especially diatoms, dinoflagellates, and chrysophytes at the turbid-water state conditions.

3 Phytoplankton Biodiversity in the Lakes

Generally, lake trophic conditions determine also the species biodiversity. Differences at the trophic level are also manifested by taxonomic and functioning biodiversity, and it can be emphasized by ecosystem processes including productivity and nutrient inflow. Here, large and rapid changes in ecosystem processes can also be due to anthropogenic causes [50]. In selected Polish lakes, phytoplankton assemblages were formed by taxa belonging to eight phyla: Cyanobacteria, Bacillariophyta, Chlorophyta, Miozoa (previously Dinophyta), Cryptophyta, Ochrophyta (including primarily chrysophytes), planktonic Charophyta (previously included into Chlorophyta), and Euglenophyta. From 12 to 114 species were usually recorded per one sampling term (Table 3). In mesotrophic lakes, averages of 34–39 and maximum of 71 species were noted. Higher species richness (SR) was, in turn, in more eutrophied lakes, i.e., meso-eutrophic (30–81 and 109, respectively), eutrophic (34–90 and 114, respectively), and hypereutrophic (62–87 and 101, respectively). Generally, the SR values increased along with the increase of a lake trophy level. Based on SR average data, the lakes can be given in the ascending order: mesotrophic < meso-eutrophic < eutrophic < hypereutrophic; however, the maximum

Table 3 Biodiversity of phytoplankton along with trophy level gradient in the selected Polish lakes

Lake, code	Trophy[a]	Species richness	Shannon-Weaver Index
Mamry Północne, MP	m	34 (15–59)	2.14 (1.10–3.08)
Hańcza, Ha	m	37 (24–52)	1.93 (0.91–2.56)
Kirsajty, Ki	m	39 (23–71)	2.04 (0.51–3.14)
Dejguny, De	m-e	43 (31–56)	1.81 (0.63–2.50)
Dąbrowa Wielka, DW	m-e	51 (35–69)	2.14 (1.21–2.83)
Warniak (clear-water), Wa1	m-e	43 (12–81)	1.78 (0.50–2.86)
Łańskie, La	m-e	42 (25–58)	2.00 (0.84–2.79)
Ślesińskie, Sl	m-e	81 (43–109)	2.18 (1.24–3.38)
Dąbrowa Mała, DM	m-e	58 (38–88)	2.17 (0.93–2.98)
Niegocin, Ni	m-e	43 (23–64)	1.87 (0.50–2.52)
Pluszne, Pl	m-e	30 (16–43)	1.50 (0.81–2.44)
Warniak (turbid-water), Wa2	m-e	55 (16–87)	2.03 (0.36–3.13)
Hartowieckie, Hr	e	69 (61–80)	2.53 (1.81–3.16)
Licheńskie, Li	e	72 (18–105)	2.58 (0.94–3.37)
Zarybinek, Za	e	90 (60–114)	2.79 (1.74–3.29)
Jeziorak Mały (clear-water), JM1	e	38 (23–72)	2.98 (2.74–3.18)
Grądy, Gr	e	85 (61–111)	2.67 (1.87–3.42)
Tarczyńskie, Ta	e	89 (66–102)	2.40 (1.79–2.83)
Święte, Sw	e	89 (77–110)	2.69 (2.12–3.16)
Jeziorak Mały (turbid-water), JM2	e	34 (15–49)	3.00 (0.84–3.87)
Zwiniarz, Zw	e	74 (60–85)	2.56 (2.09–2.95)
Klasztorne Duże, KD	e	53 (45–57)	1.64 (0.55–2.49)
Skępskie Wielkie, SW	h	79 (71–88)	1.89 (1.61–2.17)
Mielenko, Mi	h	84 (79–88)	2.99 (2.59–3.30)
Klasztorne Małe, KM	h	71 (53–85)	2.27 (0.95–3.04)
Skępskie Małe, SM	h	87 (71–101)	2.17 (1.90–2.39)
Wierzysko, Wi	h	62 (55–78)	1.55 (0.86–2.43)
Karczemne, Ka	h	69 (62–74)	2.14 (1.52–2.87)

S stratified, *NS* nonstratified
[a]Based on TSI, *m* mesotrophy (TSI 40–50), *m-e* meso-eutrophy (TSI 50–60), *e* eutrophy (TSI 60–70), *h* hypereutrophy (TSI > 70)

occurred in eutrophic lakes. Similarly, an average value of biodiversity index was the highest in eutrophic lakes, whereas the lowest in meso-eutrophic lakes.

The differences in both species richness and biodiversity index values were additionally resulted from the other factors. For example, the degree of heating the waters and water exchange time in the lakes included into water-cooling system, i.e., Lake Licheńskie and Ślesińskie, influenced the species richness and their frequency [41]. More species of diatoms, euglenoids, and chlorophytes were generally recorded in Lake Licheńskie with the warmer waters, faster water exchange, and frequent lack of an ice cover. Such conditions were generally advantageous for a higher phytoplankton biodiversity (Shannon-Weaver Index 2.58) in this lake.

In Lake Warniak, higher species richness and biodiversity (55 species and S-WI 2.03 on average) were generally recorded during the turbid-water state than the clear-water state. Strong competition and allelopathic effect of *Chara* species significantly helped to stabilize better water quality but negatively influenced the biodiversity [42]. On the contrary, the values of SR and S-WI in Lake Jeziorak Mały were usually higher during phase with lower phytoplankton biomass.

4 Relationships Between Phytoplankton and Environmental Variables

To present the general tendency of phytoplankton-environment relationships between the studied lakes, correlated phytoplankton groups (Cyanobacteria, Bacillariophyta, Chlorophyta, Miozoa, and Cryptophyta) and the chosen most important physicochemical parameters (water temperature; SD, water transparency; TN; and $P\text{-}PO_4$) were significant for the model. Table 4 shows the mean, marginal, standard deviations, and λ values of analyzed variables at $p < 0.05$. Standard deviations did not exceed the twice value of means, thus indicating that the data was statistically representative. The variance of analyzed variables is characterized by the individual λ values in the range from 0.03 for water temperature to 0.65 for total biomass of phytoplankton.

The Monte Carlo test revealed that the correlation between biomass of phytoplankton groups and physicochemical variables from the canonical correspondence analysis was significant both for Axis 1 and Axis 2 (eigenvalue 0.0126, 0.060, 0.032, and 0.016). CCA phytoplankton group analysis identified significant relationships between biomass and physicochemical variables in the studied lakes (Fig. 12). This employed a data set of 28 samples, 5 main phytoplankton groups, and 4 environmental variables, with the first axis accounting for 49.8% of the total phytoplankton group variation and 52.74% of physicochemical water parameters. CCA analysis

Table 4 The basic values to statistical analysis of phytoplankton groups and physicochemical variables at $p < 0.05$

Parameter	Mean	Minimum	Maximum	Standard deviation	λ
Total biomass	24.1	0.03	61.8	18.5	0.65
Cyanobacteria	12.5	0.02	44.9	14.2	0.47
Bacillariophyta	6.7	0.02	35.1	8.1	0.28
Chlorophyta	1.9	0.02	6.3	1.9	0.44
Miozoa	0.98	0.02	6.2	1.4	0.11
Cryptophyta	1.85	0.04	4.1	1.26	0.57
Temperature	15.2	12.2	20.6	1.42	0.03
Secchi disk	1.21	0.3	5.4	0.94	0.45
Total nitrogen	1.6	0.6	3.0	0.6	0.31
Phosphates	0.10	0.004	0.56	0.15	0.07

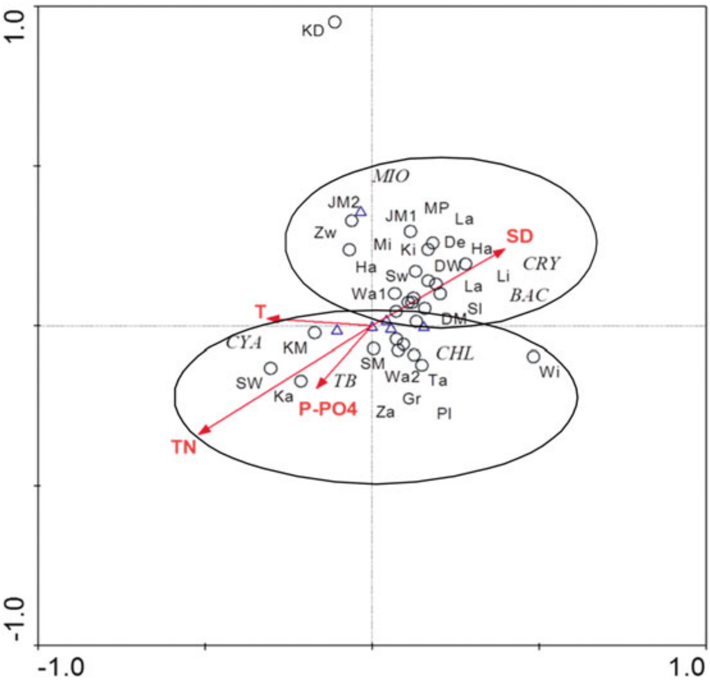

Fig. 12 Biplots ordination diagram of canonical correspondence analysis showing the relationships between total biomass (*TB*) and taxonomic phytoplankton groups (*CYA* Cyanobacteria, *BAC* Bacillariophyta, *CHL* Chlorophyta, *MIO* Miozoa, *CRY* Cryptophyta) and physicochemical water parameters (*SD* water transparency, *T* water temperature, *TN* total nitrogen, *P-PO$_4$* phosphates) in the lakes; the explanation of lakes' codes was given in Table 3

divided two general groups of lakes with varied phytoplankton assemblages. In the first group, diatoms, cryptomonads, and dinoflagellates (Miozoa) correlated with water transparency (SD) in the restored shallow lakes (Jeziorak Mały both two states and Warniak at the clear-water state) and also in both stratified and nonstratified lakes with generally lower trophy level, i.e., Hańcza, Dejguny, Łańskie, Kirsajty, Mamry Północne, Niegocin, Dąbrowa Wielka, Dąbrowa Mała, Ślesińskie, Licheńskie, Święte, and Mielenko. However, in the second group, the total biomass of phytoplankton and separately cyanobacteria and chlorophytes biomasses were stimulated by water temperature, TN, and P-PO$_4$ in the lakes with generally higher trophy level and both mictic types, i.e., Skępskie Wielkie, Klasztorne Małe, Karczemne, Skępskie Małe, Warniak at turbid-water state, Grądy, Zarybinek, Tarczyńskie, and Pluszne.

The phytoplankton-environment relations were also checked for the classification of lakes studied using PCA. The first two PC factors explained approximately 80% of the total variability (Fig. 13). The PCA plot ordination concerning PC1 (57.53% of the total variability) showed a general grouping of the lakes with lower trophy level (primarily mesotrophic and meso-eutrophic) and higher trophy level (primarily

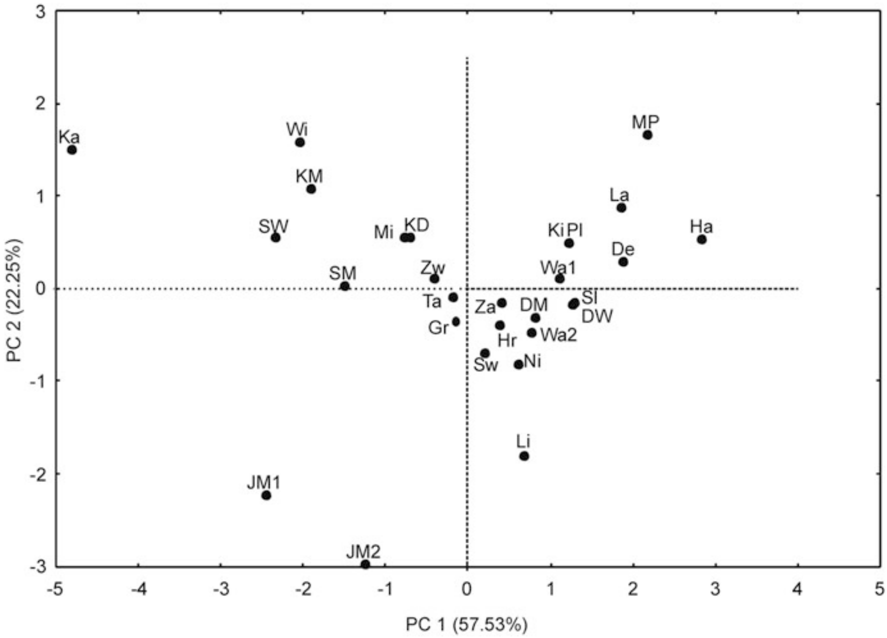

Fig. 13 Plot ordination diagram of principal component analysis based on phytoplankton-environment relations showing grouping of the lakes, the explanation of lakes' codes were given in Table 3

eutrophic and hypereutrophic) consistent with Table 3. Lake Karczemne with the highest phytoplankton biomass and nutrient content and Lake Jeziorak Mały were distinctly out from these groups.

5 Conclusions

The phytoplankton biomass and structure in Polish lakes are generally subjected to the significant fluctuations and directional changes connected primarily with the trophic state. However other factors, e.g., different mictic type (stratified and nonstratified lakes) or water-flow regimes deciding about retention time, play also an important role. Generally, phytoplankton in the lakes with lower trophy level was characterized by smaller biomass and varied structure including primarily co-domination of Bacillariophyta, Cryptophyta, Miozoa, and Cyanobacteria throughout the growth season. A growth pattern was usually typical of mesotrophic or even oligotrophic temperate lakes according to the PEG Models. The seasonal changes in the phytoplankton with a large biomass and domination primarily by chroococcalean cyanobacteria of the genus *Microcystis* or filamentous cyanobacteria of the genera *Planktolyngbya, Pseudanabaena*, *Limnothrix*, and *Planktothrix* in summer periods were, in turn, typical of eutrophic lakes. Higher species richness and biodiversity index were generally characteristic of more eutrophied lakes. The phytoplankton response to environmental variables confirmed a strict correlation with water transparency in the lakes with lower trophic state whereas with water temperature and nutrients in the lakes with higher trophic state.

The results of changes of phytoplankton biomass and structure at the clear-water and turbid-water states may have a great importance for appropriate lake restoration technologies applied to achieve efficient shallow lake management. However, each lake should be treated separately.

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References

1. Paerl HW, Hall NS, Calandrino ES (2011) Controlling harmful cyanobacterial blooms in a world experiencing anthropogenic and climatic-induced change. Sci Total Environ 409:1739–1745
2. Carey CC, Ibelings BW, Hoffmann EP, Hamilton DP, Brookes JD (2012) Eco-physiological adaptations that favour freshwater cyanobacteria in a changing climate. Water Res 46:1394–1407. https://doi.org/10.1016/j.watres.2011.12.016
3. Elliot JA (2012) Is the future blue-green? A review of the current model predictions of how climate change could affect pelagic freshwater cyanobacteria. Water Res 46(5):1364–1371. https://doi.org/10.1016/j.watres.2011.12.018
4. Jeppesen E, Meerhoff M, Davidson TA, Trolle D, Søndergaard M, Lauridsen TL, Bekliğlu M, Brucet S, Volta P, González-Bergonzoni I, Nielsen A (2014) Climate change impacts on lakes: an integrated ecological perspective based on a multi-faced approach with special focus on shallow lakes. J Limnol 73(S1):88–111
5. Rigosi A, Carey CC, Ibelings BW, Brookes JD (2014) The interaction between climate warnings and eutrophication to promote cyanobacteria is dependent on trophic state and varies among taxa. Limnol Oceanogr 59(1):99–114
6. Bernhardt H (1987) Strategies of lake sanitation. Shweiz Z Hydrol 49(2):202–219
7. Reynolds CS (2003) The development of perceptions of aquatic eutrophication and its control. Ecohydrol Hydrobiol 3(2):149–163
8. Anderson DM, Glibert PM, Burkholder JM (2002) Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. Estuaries 25(4b):704–726
9. Mazur-Marzec H, Spoof L, Kobos J, Pliński M, Meriluoto J (2008) Cyanobacterial hepatotoxins, microcystins and nodularins, in fresh and brackish waters of Pomeranian Province, northern Poland. Oceanol Hydrobiol Stud 37(4):1–19. https://doi.org/10.2478/v10009-008-0014-0
10. Kobos J, Błaszczyk A, Hohlfeld N, Toruńska-Sitarz A, Krakowiak A, Hebel A, Stryk K, Grabowska M, Toporowska M, Kokociński M, Messyasz B, Rybak A, Napiórkowska-Krzebietke A, Nawrocka L, Pełechata A, Budzyńska A, Zagajewski P, Mazur-Marzec H (2013) Cyanobacteria and cyanotoxins in Polish freshwater bodies. Oceanol Hydrobiol Stud 42(4):358–378. https://doi.org/10.2478/s13545-013-0093-8
11. Napiórkowska-Krzebietke A (2015) Cyanobacterial bloom intensity in the ecologically relevant state of lakes – an approach to water framework directive implementation. Oceanol Hydrobiol Stud 44(1):97–108. https://doi.org/10.1515/ohst-2015-0010

12. Napiórkowska-Krzebietke A, Dunalska J, Grochowska J, Łopata M, Brzozowska R (2015) Intensity and thresholds of cyanobacterial blooms – an approach to determine the necessity to restore urban lakes. Carpath J Earth Environ 10(2):123–132
13. Carlson RE (1977) A trophic state index for lakes. Limnol Oceanogr 22:361–369
14. Kratzer CR, Brezonik PL (1981) A Carlson type trophic state index for nitrogen in Florida lakes. Water Res Bull 17:713–715
15. Carlson RE, Simpson J (1996) A coordinator's guide to volunteer lake monitoring methods. North American Lake Management Society, Madison, pp 1–96
16. Sommer U, Gliwicz ZM, Lampert W, Duncan A (1986) The PEG-model of seasonal succession of planktonic events in freshwaters. Arch Hydrobiol 106:433–471
17. Oleksowicz AS (1988) The dynamics of algal communities in the Kashubian Lakes of different trophy. Rozprawy UMK, Torun, p 84. (in Polish)
18. de Senerpont Domis LN, Elser JJ, Gsell AS, Huszar VL, Ibelings BW, Jeppesen E, Kosten S, Mooij WM, Roland F, Sommer U, van Donk E, Winder M, Lürling M (2013) Plankton dynamics under different climatic conditions in space and time. Freshw Biol 58:463–482. https://doi.org/10.1111/fwb.12053
19. Brentrup JA, Williamson CE, Colom-Montero W, Eckert W, de Eyto E, Grossart HP, Huot Y, Isles PDF, Knoll LB, Leach TH, McBride CG, Pierson D, Pomati F, Read JS, Rose KC, Samal NR, Staehr PA, Winslow LA (2016) The potential of high-frequency profiling to assess vertical and seasonal patterns of phytoplankton dynamics in lakes: an extension of the Plankton Ecology Group (PEG) model. Inland Waters 6(4):565–580. https://doi.org/10.5268/IW-6.4.890
20. Guiry MD, Guiry GM (2018) AlgaeBase. World-Wide Electronic Publication, National University of Ireland, Galway. http://www.algaebase.org; searched on 06 August 2018
21. Napiórkowska-Krzebietke A, Stawecki K, Pyka JP, Zdanowski B, Zębek E (2016) Phytoplankton and the physicochemical background in an assessment of the ecological and trophic conditions in vendace-type lakes. J Elem 21(1):159–172. https://doi.org/10.5601/jelem.2015.20.2.891
22. Napiórkowska-Krzebietke A, Hutorowicz A (2013) A comparison of epilimnetic versus metalimnetic phytoplankton assemblages in two mesotrophic lakes. Oceanol Hydrobiol Stud 42(1):89–98
23. Napiórkowska-Krzebietke A, Stawecki K, Pyka JP, Hutorowicz J, Zdanowski B (2013) Phytoplankton in relation to water quality of a mesotrophic lake. Pol J Environ Stud 22(3):793–800
24. Napiórkowska-Krzebietke A, Hutorowicz A (2014) Phytoplankton in an ecological status assessment of the vendace-type Lake Dejguny (northeastern Poland). Arch Pol Fish 22:29–40. https://doi.org/10.2478/aopf-2014-0004
25. Napiórkowska-Krzebietke A, Hutorowicz A (2005) Long-term changes of phytoplankton in Lake Mamry Północne. Oceanol Hydrobiol Stud 34(3):217–228
26. Napiórkowska-Krzebietke A, Hutorowicz A (2007) Long-term changes in the biomass and composition of phytoplankton in a shallow, flow-through lake Kirsajty (Masurian Lakeland, Poland). Pol J Nat Sci 22:512–524
27. Napiórkowska-Krzebietke A, Wierzchowska M, Błocka B, Hutorowicz J, Hutorowicz A, Zdanowski B (2007) Changes in the trophic state of Lake Niegocin after the modernization of a local wastewater treatment plant. Limnol Rev 7(3):153–159
28. Napiórkowska-Krzebietke A, Hutorowicz A (2006) Long-term changes of phytoplankton in Lake Niegocin, in the Masurian Lake Region, Poland. Oceanol Hydrobiol Stud 35(3):209–226
29. Hutorowicz A, Napiórkowska-Krzebietke A (2007) Changes in phytoplankton assemblages after the reduction of sewage discharge into Lake Niegocin (Mazurian Lake District, Poland). Oceanol Hydrobiol Stud 36:137–145
30. Jakubowska N, Zagajewski P, Gołdyn R (2013) Water blooms and cyanobacterial toxins in lakes. Pol J Environ Stud 22(4):1077–1082
31. Napiórkowska-Krzebietke A, Dunalska JA, Zębek E (2017) Taxa-specific eco-sensitivity in relation to phytoplankton bloom stability and ecologically relevant lake state. Acta Oecol 81:10–21. https://doi.org/10.1016/j.actao.2017.04.002

32. Napiórkowska-Krzebietke A, Dunalska J (2015) Phytoplankton-based recovery requirement for urban lakes in the implementation of the Water Framework Directive's ecological targets. Oceanol Hydrobiol Stud 44(1):109–119. https://doi.org/10.1515/ohst-2015-0000
33. Naselli-Flores L (2008) Urban lakes: ecosystems at risk, worthy of the best care. In: Sengupta M, Dalwani R (eds) Proceedings of Taal2007: the 12th world lake conference, pp 1333–1337
34. Saros JE, Strock KE, Mccue J, Hogan E, Anderson NJ (2014) Response of Cyclotella species to nutrients and incubation depth in Arctic lakes. J Plankton Res 36(2):450–460. https://doi.org/10.1093/plankt/fbt126
35. Napiórkowska-Krzebietke A, Pasztaleniec A, Hutorowicz A (2009) Phytoplankton – the element in the ecological status assessment for lakes of the Wel river catchment area. Teka Kom Ochr Kszt Środ Przyr – OL PAN 6:200–205
36. Hutorowicz A, Napiórkowska-Krzebietke A, Pasztaleniec A, Hutorowicz J, Lyche Solheim A, Skjelbred B (2011) Phytoplankton. In: Soszka H (ed) Ecological status assessment of the waters in the Wel River catchment. Guidelines for integrated assessment of ecological status of rivers and lakes to support river basin management plans. IFI, Olsztyn, pp 143–168. (in Polish)
37. Pyka JP, Zdanowski B, Stawecki K (2013) Long-term trends in changes of the chemical composition of waters in lakes heated by electric power plants. Arch Pol Fish 21:343–355
38. Stawecki K, Zdanowski B, Pyka JP (2013) Long-term changes in post-cooling water loads from power plants and thermal and oxygen conditions in stratified lakes. Arch Pol Fish 21:331–342
39. Socha D (1997) Spatial and seasonal phytoplankton diversity in Licheńskie and Ślesińskie lakes, Konińskie District, in 1991–1993. Arch Pol Fish 5(1):117–136
40. Socha D, Hutorowicz A (2009) Changes in the quantitative relations of the phytoplankton in heated lakes. Arch Pol Fish 17(4):239–251
41. Napiórkowska-Krzebietke A (2009) Diversity and dynamics of phytoplankton in lakes Lichenskie and Ślesinskie in 2004–2005. Arch Pol Fish 17:253–265
42. Napiórkowska-Krzebietke A (2017) Phytoplankton response to fish-induced environmental changes in a temperate shallow pond-type lake. Arch Pol Fish 25(4):211–264. https://doi.org/10.1515/aopf-2017-0020
43. Jeppesen E, Sondergaard M, Meerhoff M, Lauridesen TL, Jensen JP (2007) Shallow lake restoration by nutrient loading reduction – some recent findings and challenge ahead. Hydrobiologia 584:239–252. https://doi.org/10.1007/s10750-007-0596-7
44. Wichelen J, Declerck S, Muylaert K, Hoste I, Greenens V, Vandekerkhove J, Michels E, Pauwn N, Hoffmann M, Meester L, Vyverman W (2007) The importance of drawdown and sediment removal for the restoration of the eutrophied shallow Lake Kraenepoel (Belgium). Hydrobiologia 584:291–303. https://doi.org/10.1007/s10750-007-0611-z
45. Mcdonald KE, Lehman JT (2013) Dynamics of Aphanizomenon and Microcystis (cyanobacteria) during experimental manipulation of an urban impoundment. Lake Reservoir Manage 29(2):103–115. https://doi.org/10.1080/10402381.2013.800172
46. Zębek E, Napiórkowska-Krzebietke A (2016) Response of phytoplankton to protective-restoration treatments enhancing water quality in a shallow urban lake. Environ Monit Assess 188:623. https://doi.org/10.1007/s10661-016-5633-4
47. Zębek E, Szwejkowska M (2014) Influence evaluation of pretreated storm water on analysis of cyanobacteria numbers in Jeziorak Mały urban lake at various precipitation rates. Ochrona Środowiska 36(1):27–31. (in Polish)
48. Zębek E (2015a) Response of planktonic cyanobacteria and periphyton assemblages to physicochemical properties of stormwater in a shallow urban lake. J Elem 20(1):231–245. https://doi.org/10.5601/jelem.2014.19.2.679
49. Zębek E (2015b) Effect of the fountain-based water aeration system on phytoplankton growth in a urban lake. Teka Kom Ochr Kszt Środ Przyr – OL PAN 12:128–137
50. Thebault E, Loreau M (2006) The relationship between biodiversity and ecosystem functioning in food webs. Ecol Res 21:17–25. https://doi.org/10.1007/s11284-005-0127-9

Biodiversity of Zooplankton in Polish Small Water Bodies

Natalia Kuczyńska-Kippen

Contents

Abstract Although ponds located in a low-transformed landscape harbour higher biodiversity than ponds in areas with a large impact of anthropopression, both types of water bodies can contribute to the enrichment of fauna on local and regional scales. This review presents aspects of pond zooplankton diversity with reference to the occurrence of species, common and rare, and significant drivers of their distribution. The results of various studies carried out on small water bodies in Poland revealed a great level of zooplankton diversity, which points directly to a high variation of the origin of types of ponds. Land use within the direct catchment area influences the creation of zooplankton diversity, although a greater impact is connected with various habitats, particularly the open water zone and macrophyte-dominated areas. The complex architecture of elodeids is responsible for the highest zooplankton diversity with many rare species, offering a great number of available

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ecological niches for littoral animals and profitable anti-predator conditions for planktonic species. Therefore, one should strive to maintain or even increase the complexity of aquatic vegetation within even small-surfaced ponds. The generally high share of rare species found in ponds underlines their high ecological value but, at the same time, a lack of thorough studies. The most common rotifers found in Polish ponds were *Keratella cochlearis*, *Anuraeopsis fissa*, *Polyarthra vulgaris* and *Keratella quadrata* as well as *Chydorus sphaericus*, *Bosmina longirostris*, *Ceriodaphnia quadrangula* and *Eubosmina coregoni* among crustaceans. This reflects the wide ecological valence of these species and suggests that most ponds are eutrophic.

Keywords Aquatic diversity · Catchment type · Human impact · Macrophytes · Microcrustaceans · Origin type · Ponds · Rotifers

1 Introduction

1.1 Ponds as a Valuable Habitat for Maintaining Biodiversity

Small water bodies are among freshwater ecosystems which are critical for maintaining high biodiversity. Their large abundance in many parts of the world and greater total area than lakes contribute to an extremely high level of global diversity [1]. However, due to their small size and depth, they can be exposed to severe human disturbances [2, 3], and freshwater biodiversity may decline in response to human land use, eutrophication and habitat destruction while increasing in the presence of natural drivers such as area and habitat heterogeneity [4]. Despite being prone to human-originated stressors in the direct catchment area (e.g. agricultural or urban areas), they often remain undamaged and thus create a refuge for diverse organisms, including zooplankton, which may have disappeared from more polluted aquatic systems [3]. From the point of view of ecology, ponds play a very important ecotone role, being a transitional system between various biocoenoses and aquatic ecosystems and also creating an interface between terrestrial and aquatic environments. In this way they build a bridge that connects various wetlands, favouring the migration of many species. Small water bodies, along with boughs, mid-field woodlots and also ditches, small streams, oxbows or wetlands, are an element important for the preservation and enrichment of biodiversity, both in the biological, habitat and landscape aspects of a certain area.

Despite having a multiplicity of extremely useful functions, small water bodies have never been a central object of interest for the scientific community and have not been included in proper and long-term monitoring studies for centuries [3, 5]. Neither has their proper classification been developed even though many recent studies have shown that distinct differences in their functioning, compared to lake ecosystems, exist (e.g. [6]). The same concerns the lack of information on their functioning and inhabiting organisms, including zooplankton. Most organisms found in ponds are generally common species of a wide range of tolerance to environmental factors and can be found also in other types of freshwaters. However, due to the occurrence of more specialised species, the communities of plankton organisms can be quite specific and are called heleoplankton. Characteristics of biocoenoses of small water bodies require taking into account a number of different environmental factors, such as origin, pond morphology, specificity of the location in a diverse landscape, hydrological relations, abiotic parameters of their waters as well as various types of habitats. These diverse habitat elements will be responsible for the development of diverse but also specific plankton communities.

2 Zooplankton Diversity in Ponds

2.1 Taxonomic Diversity of Zooplankton

Small water bodies contribute greatly to preserving but also enriching the biodiversity both on a local and regional scale, making up an optimal habitat for many groups of organisms. Natural ponds in various landscapes are usually expected to host higher species diversity than nutrient-rich degraded ponds, such as fishponds, urban ponds or those located within an agricultural landscape. However, if the biodiversity of such an individual pond remains rather low, it can contribute to much higher diversity at the regional level [7]. This is why to ensure the maximum possible biodiversity, conservation practices should consider the landscape-scale organisation of ecological communities [8].

The diversity of zooplankton communities greatly depends on a number of environmental factors. However, the potential diversity reflects study regularity and the amount of studied water bodies and/or sites. Recognition of the taxonomic structure of zooplankton also refers to the recognition of their diverse habitats. According to [9], as a result of thorough seasonal analysis of zooplankton, each temperate water body will contain ca. 150 rotifer species. Looking at the examination intensity as well as the number of studied ponds, the same pattern of species richness was obtained for ponds in central-western Poland (Table 1). The highest diversity was attributed to the greatest amount of water bodies taken into account, where almost 300 ponds were examined in the Wielkopolska province. A similar species number was obtained for a group of only six ponds, studied on regular basis, 13 times in 1 year, compared to a much larger group of ponds (55), examined on only one occasion.

The research on 54 ponds, with 28 pastoral and 26 forest ponds, revealed a high richness with 265 zooplankton species in total [10]. The most diverse genus was *Lecane* (Fig. 1) with 26 species, which constitutes almost half of the lecanid structure in Poland. This genus is very diverse, inhabiting predominately benthic and littoral environments, both dominated by floating-leaved plants, submerged macrophytes or helophytes [11]. The greatest diversity occurs in the standing or slowly flowing waters of tropical and subtropical climatic zones, where up to 40 species can be found in one water body. Some *Lecane* species (e.g. *Lecane bulla, Lecane luna*,

Table 1 The total number of species with the mean number of species and SD as well as minimum (min) and maximum (max) values of Rotifera, Cladocera and Copepoda in the open water area of small water bodies in four provinces in Poland

No of study	N samples	N species Rotifera		N species Cladocera		N species Copepoda	
		Total (mean ± SD)	Min–max	Total (mean ± SD)	Min–max	Total (mean ± SD)	Min–max
1.	278	258 (21 ± 10)	0–57	63 (5 ± 3)	0–18	34 (2 ± 2)	0–9
2.	74	122 (20 ± 5)	7–35	30 (4 ± 3)	0–14	19 (2 1 1)	1–5
3.	55	132 (18 ± 7)	5–41	32 (5 ± 4)	0–14	22 (2 ± 2)	0–6
4.	42	66 (18 ± 6)	5–29	24 (7 ± 3)	1–13	10 (2 ± 2)	0–6
5.	27	93 (19 ± 8)	4–36	24 (5 ± 3)	1–10	13 (2 ± 2)	0–5
6.	15	61 (15 ± 3)	7–20	22 (4 ± 3)	1–10	5 (1 ± 1)	0–3

Total number of samples in each study is given
1. 278 ponds; sampled once (2006–2015); region: Wielkopolskie province
2. 6 ponds; sampled 13 times in 1 year (2009); region: Kujawsko-Pomorskie province
3. 55 ponds; sampled once (2006–2013); region: Kujawsko-Pomorskie province
4. 6 ponds; sampled 7 times in 1 year (2009); region: Poznań agglomeration, Wielkopolskie province
5. 27 ponds; sampled once (2010); region: Dolnośląskie province
6. 15 ponds; sampled once (2010); region: Lubuskie province

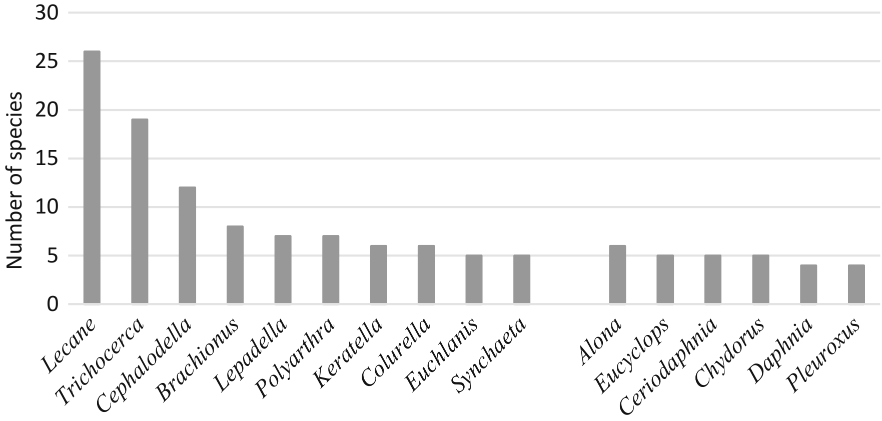

Fig. 1 The number of zooplankton species representing certain genus in small water bodies

Lecane closterocerca or *Lecane lunaris*) are considered to be eurytopic, frequently occurring in Polish ponds (Table 2). *Trichocerca*, the second rich genus in terms of the number of species, is mainly characteristic of vegetated zones. Its presence in the pelagic zone is usually accidental, a result of being washed away from among macrophytes. Thus, only some species, e.g. *Trichocerca capucina* or *Trichocerca pusilla*, occur in the open water. Altogether 19 species were identified, out of 37 trichocercids in Poland [11], with *Trichocerca similis*, *Trichocerca pusilla* and *Trichocerca rattus* being most frequent in ponds. The third genus of significant importance for Rotifera species richness was *Cephalodella*, which has 41 species in

Table 2 The most frequent species (with the level of frequency in %) – ≥5%, representing Rotifera, Cladocera and Copepoda in the open water area of a set of 410 small water bodies located within the central-western part of Poland

Rotifera	%
Keratella cochlearis (Gosse)	76
Lecane closterocerca (Schmarda)	65
Bdelloidea	65
Anuraeopsis fissa (Gosse)	59
Polyarthra vulgaris Carlin	57
Keratella quadrata (O.F. Müller)	50
Lepadella patella (O.F. Müller)	47
Brachionus angularis Gosse	46
Colurella uncinata (O.F. Müller)	45
Keratella cochlearis f. *tecta* (Lauterborn)	44
Polyarthra remata (Skorikov)	39
Trichocerca similis (Wierzejski)	38
Filinia longiseta (Ehrenberg)	38
Lepadella ovalis (O.F. Müller)	35
Testudinella patina (Hermann)	34
Brachionus quadridentatus (Hermann)	32
Mytilina ventralis (Ehrenberg)	29
Mytilina mucronata (O.F. Müller)	27
Trichocerca pusilla Lauterborn	27
Colurella adriatica Ehrenberg	26
Lepadella quadricarinata (Stenroos)	26
Lecane hamata (Stoces)	26
Synchaeta pectinata Ehrenberg	26
Lecane bulla (Gosse)	25
Euchlanis dilatata Ehrenberg	24
Asplanchna priodonta Gosse	23
Lecane luna (O.F. Müller)	23
Cephalodella catellina (O.F. Müller)	21
Trichocerca rattus (O.F. Müller)	21
Cephalodella gibba (Ehrenberg)	20
Keratella testudo (Ehrenberg)	20
Lecane lunaris (Ehrenberg)	20
Brachionus calyciflorus Pallas	20
Colurella obtusa (Gosse)	16
Pompholyx complanata Gosse	16
Lepadella rhomboides (Gosse)	19
Pompholyx sulcata (Hudson)	15
Cephalodella ventripes Dixon-Nuttall	14
Trichocerca weberi (Jennings)	14
Ascomorpha ecaudis (Perty)	13
Cephalodella auriculata (O.F. Müller)	12

(continued)

Table 2 (continued)

Polyarthra major Burckhardt	11
Squatinella rostrum (Schmarda)	11
Brachionus rubens Ehrenberg	10
Lepadella acuminata (Ehrenberg)	10
Lepadella triptera Ehrenberg	10
Lophocharis oxysternon (Gosse)	10
Platyias quadricornis (Ehrenberg)	10
Lecane quadridentata (Ehrenberg)	9
Trichotria pocillum (O.F. Müller)	9
Trichocerca dixon-nuttalli (Jennings)	9
Keratella ticinensis (Callerio)	8
Dissotrocha aculeata (Ehrenberg)	8
Lecane furcata (Murray)	8
Polyarthra dolichoptera Idelson	17
Brachionus budapestinensis (Daday)	7
Ascomorpha saltans Bartsch	7
Filinia brachiata (Rousselet)	7
Brachionus diversicornis (Daday)	6
Lophocharis salpina Ehrenberg	6
Trichocerca brachyura (Gosse)	6
Gastropus hyptopns (Ehrenberg)	6
Testudinella mucronata (Gosse)	6
Trichocerca capucina Wierzejski & Zacharias	6
Colurella colurus (Ehrenberg)	5
Scaridium longicaudum (O.F. Müller)	5
Kellicottia longispina (Kellicott)	5
Lecane elsa Hauer	5
Beauchampiella eudactylota (Gosse)	5
Lecane flexilis (Gosse)	5
Cephalodella tenuior (Gosse)	5
Trichocerca vernalis Hauer	5
Filinia terminalis (Plate)	5
Hexarthra mira (Hudson)	5
Lecane nana (Murray)	5
Trichocerca musculus (Hauer)	5
Cladocera	%
Chydorus sphaericus (O.F. Müller)	59
Bosmina longirostris (O.F. Müller)	45
Ceriodaphnia quadrangula (O.F. Müller)	28
Eubosmina coregoni Baird	26
Scapholeberis mucronata (O.F. Müller)	20
Simocephalus exspinosus(Koch)	16
Ceriodaphnia pulchella Sars	16

(continued)

Table 2 (continued)

Alonella excisa (Fischer)	15
Pleuroxus aduncus (Jurine)	15
Daphnia pulex (De Geer)	14
Alona rectangula Sars	13
Daphnia cucullata Sars	11
Diaphanosoma brachyurum (Lievin)	10
Daphnia longispina O.F. Müller	10
Acroperus harpae (Baird)	10
Ceriodaphnia reticulata (Jurine)	10
Daphnia galeata Sars	8
Alonella exigua (Lilljeborg)	8
Graptoleberis testudinaria (Fischer)	6
Simocephalus vetulus (O.F. Müller)	6
Peracantha truncata (O.F. Müller)	6
Ceriodaphnia laticaudata P.E. Müller	5
Tretocephala ambigua (Lilljeborg)	5
Copepoda	%
Thermocyclops oithonoides (Sars)	21
Mesocyclops leuckarti (Claus)	15
Megacyclops viridis (Jurine)	13
Harpacticoidae	11
Eudiaptomus gracilis (Sars)	10
Eucyclops serrulatus (Fischer)	8
Cyclops vicinus (Sars)	7
Acanthocyclops vernalis (Fischer)	6
Eucyclops macruroides (Lilljeborg)	5
Eudiaptomus graciloides (Lilljeborg)	5

Poland [11]. In the analysed ponds, 12 species were found, mainly inhabitants of periphytic environments with *Cephalodella catellina*, *Cephalodella gibba* and *Cephalodella ventripes* occurring as the most common. In turn, the *Brachionus* genus is considered to favour environments with large areas of open water. In macrophytes, brachionids, such as *Brachionus quadridentatus*, choose loosely arranged habitats where they can freely swim between individual plants. In the analysed ponds, as many as eight *Brachionus* species were found, which constitutes half of the *Brachionus* fauna in Poland. Among them *Brachionus angularis*, *Brachionus quadridentatus* and *Brachionus calyciflorus* were most frequent. The majority of brachionids are indicators of high trophic state of water [12]. Two more genera, *Lepadella* and *Polyarthra*, were represented by seven species each, constituting 30% and 100% of species identified from Poland. *Lepadella* is a typically littoral-associated genus, while *Polyarthra* prefers open water environments. *Polyarthra major* prefers low-trophy waters, while *Polyarthra longiremis* is associated with eutrophic waters [11]. Generally *Polyarthra vulgaris*, *Polyarthra remata* and *Polyarthra major* as well as *Lepadella patella*, *Lepadella ovalis* and *Lepadella quadricarinata* were common in Polish ponds. Moreover, six species were found among the *Colurella* and *Keratella* genera. All colurellids are periphytic organisms, while *Keratella* is a limnetic genus. One of the most frequently occurring species was *Keratella cochlearis*. *Keratella cochlearis* f. *tecta* and *Keratella quadrata*, both frequently occurring in the analysed material, are indicators of eutrophic waters [12], so their mass appearance corresponds to a high trophy of ponds. Among colurellids, *Colurella uncinata*, *Colurella adriatica* and *Colurella obtusa* were most frequent. Other genera were represented by five or less species (Table 2).

From among crustaceans, the *Alona* genus, a typical macrophyte-rich inhabitant [13], was the most diverse with five species, *Alona affinis*, *Alona costata*, *Alona guttata*, *Alona rectangula* and *Alona rustica*, out of ten recorded from the Polish fauna. Three genera, *Eucyclops*, *Ceriodaphnia* and *Chydorus*, were represented by five species each. The genus *Eucyclops*, usually described as epibenthic, is represented in the Polish fauna by five species [13], and all of them were also found in the studied ponds. Ceriodaphnids are encountered in both pelagic and littoral environments [14]. Out of eight Polish species, five – *Ceriodaphnia laticaudata*, *Ceriodaphnia megops*, *Ceriodaphnia pulchella*, *Ceriodaphnia quadrangula* and *Ceriodaphnia reticulata* – were found in the studied ponds. The first two species preferably inhabit ponds, while the three remaining are typical of various types of water bodies. *Ceriodaphnia quadrangula* belonged to the most frequent ceriodaphnids (Table 2). *Chydorus*, with five species found in ponds, is known to inhabit a variety of freshwater environments, often preferring macrophyte-dominated habitats.

2.2 Most Frequent Zooplankton Species

There were 48 rotifer species found in more than 10% of over 400 studied ponds from central-western Poland. Despite analysing only the open water, the most frequent species were both of pelagic and littoral origin. However, limnetic species, such as *Keratella cochlearis*, *Anuraeopsis fissa*, *Polyarthra vulgaris* and *Keratella quadrata*, were of the highest frequency (Table 2).

Keratella cochlearis, a representative of the family Brachionidae, is known for being widely distributed, also in diverse Polish waters [15–17]. In ponds, *Keratella cochlearis* was recorded in 216 sites out of 254 investigated sites (85%), which confirms its cosmopolitan range. Its size and spine length are smaller in eutrophic and hypertrophic waters than in mesotrophic and oligotrophic [18]. Moreover, its morphology may also be associated with the presence of fish. Therefore, this common species *Keratella cochlearis* also plays an indicative role. *Lecane closterocerca*, recorded from 65% of ponds, is a periphytic species. Its common presence might result from a generally littoral character of ponds which are often overgrown by macrophytes. However, it is also very common in the open water patches of ponds, often being a dominating species [16, 19]. Moreover, bdelloids as a group reached a high frequency. This reflects the high variability of conditions prevailing in ponds as bdelloids inhabit specific freshwater environments among which are moist soils, mosses, bogs [20], psammic habitats [21] or extremely cold habitats [22]. But they are also typical for lakes, streams and springs [23] as well as for small water bodies [24] which can be prone to serious changes in abiotic parameters, specifically changes in the water level [7]. Another three species, which reached a very high frequency in Polish ponds, *Anuraeopsis fissa*, *Polyarthra vulgaris* and *Keratella quadrata* (Table 2), are typically limnetic species, commonly occurring in various types of water bodies all over the world.

From among crustaceans, 16 cladoceran and 5 copepod species occurred with $\geq$10% frequency (Table 2). *Chydorus sphaericus*, *Bosmina longirostris*, *Ceriodaphnia quadrangula* and *Eubosmina coregoni* were the most common cladocerans in the central-western part of Poland. The first two species are indicators of eutrophy in lakes, suggesting the generally high trophic state of the majority of ponds [25]. *Chydorus* sphaericus is one of the most widespread species from among freshwater crustaceans [14, 26], often belonging to dominating species [27, 28]. It inhabits a range of plant-associated habitats but is also known for its adaptation to lead a pelagic lifestyle due to the fact that it uses filamentous algae as both a substrate and a food source [29]. *Bosmina longirostris*, with a high adaptive ability to changing conditions, is also a worldwide distributed species [30], often occurring in Polish ponds [31].

2.3 Rare Zooplankton

Ponds in Poland are often a source of species of a high conservation value, including zooplankton [27]. A comparison made between five types of water bodies (lakes, ponds, ditches, rivers, streams) within an agricultural area in lowland England proved that ponds supported the highest biodiversity in respect to the number of species and the highest index of species rarity across the studied area [32]. The occurrence of rare species in freshwaters depends on many factors, where the degree of recognition of a particular type of ecosystem is very important. Many faunistic studies, usually carried out in the previous century, contributed valuable information on the occurrence of such species (e.g. [33–35]). Ponds located within a natural landscape of low anthropogenic transformation may have a high ecological value, being a rich source of rare or threatened species, but ponds located in agricultural areas also contribute to the enrichment of regional diversity due to the presence of their own unique species. Therefore, even if it is presumed that intensified anthropopressure will reduce the chance of rare species occurring, the maintenance of habitat complexity referring to various macrophyte cover increases overall biodiversity and that of species that are infrequent in the national fauna.

A detailed analysis of 54 ponds [10] revealed the presence of 39 species of zooplankton that are rare or are described as infrequently occurring in the Polish fauna [11, 14], mainly from among rotifers – 29 species representing 15 genera.

Among cladocerans, nine such species from among seven genera were found and only one copepod species (Table 3). Contrary to expectation, field ponds had more rare species (31) than forest ponds (23). Most of these species were littoral-associated, which is a signal for the need to conduct research not only in the open water but also in vegetated zones, despite the small area of ponds. Finding such a large number of rare species also likely resulted from the lack of interest in conducting basic research and thus a small degree of recognition of this ecosystem. In most cases, a positive relationship between the number of rare species with the Shannon-Weaver diversity index and zooplankton richness was found, while a negative relation existed with abundance. Among rare species from Wielkopolska, *Brachionus polyacanthus* deserves special attention as this is the second report regarding this species from Poland [10]. In another study carried out on 65 ponds differing in regard to catchment type, origin, depth, size, macrophyte cover, the presence of fish and level of shading, out of 197 taxa identified, 32 were classified as rare, endangered or new to Polish fauna [36]. The comprehensive study, conducted over 5 years, on microcrustaceans within different types of aquatic environments in the Upper Narew Valley presented 74 species in 559 samples, with a new species for the Polish fauna (*Metacyclops planus*) from oxbows [37]. This indicates a necessity for conducting detailed analyses encompassing various ecosystems. Moreover, a study of 53 ponds in south-eastern Poland (the Central Roztocze Upland) revealed the presence of 54 cladoceran species with several rare species: *Ceriodaphnia dubia*, *C. rotunda*, *C. setosa*, *Bunops serricaudatus*, *Ilyocryptus agilis*, *Lathonura rectirostris*, *Macrothrix laticornis*, *M. rosea*, *Streblocerus serricaudatus*, *Chydorus ovalis* and *Rhynchotalona falcata* [38]. The author suggests that most of these species belonged to the Macrothricidae family, mainly consisting of benthic organisms. This is why they are often overlooked during typical studies, such as in the open water area. Therefore their rarity may be misleading.

Furthermore, some rare species may be found in small water bodies of unique type, such as mining subsidence pools. A study in the Silesian Uplands (southern part of Poland) revealed the presence of a rare halophilous species *Notholca salina*, observed in hypo- and mesosaline ponds [39]. It was underlined that identification of rare species is often restricted due to the low frequency of conducted studies. *Notholca salina*, like other *Notholca* species, occurs in cold seasons. This is why in studies carried out in the optimum summer season, such species do not appear, and the overall biodiversity may be underestimated. Another type, meteor craters, a very rare type of pond worldwide, can also be a source of rare species. A study conducted on a group of such ponds, located in the forest near the city of Poznań, revealed rare zooplankton species such as *Keratella paludosa*, *Lecane elsa* and *Tretocephala ambigua* [40, 41]. However, the origin of ponds was not decisive; rather the specificity of environmental conditions favoured the occurrence of diverse communities along with rare species. Other specific habitats are artificial ponds located in botanical gardens or palm houses, such as the Poznań Palm House where the presence of rare species for Poland, *Asplanchna herricki*, *Colurella sulcata* and *Gastropus minor*, were detected [42]. It is not only in specific types of pond that rare species can be found; they can also be found in ponds used for fish

Table 3 The frequency (in %) of rare species in a group of 54 ponds, located in field (28) and forest (26) ponds[a]

Rotifera	%
Asplanchna sieboldi (Leydig)	2
Brachionus polyacanthus (Ehrenberg)	2
Cephalodella gibboides Wulf	11
Cephalodella gigantea Remane	2
Cephalodella mus Wulfert	21
Cephalodella tenuiseta Burn	2
Colurella sulcata (Stenroos)	15
Colurella tesselata (Glascott)	2
Euchlanis triquetra Ehrenberg	8
Lecane aculeata (Jakubski)	11
Lecane bifurca (Bryce)	2
Lecane clara (Bryce)	2
Lecane inermis (Bryce)	4
Lecane nana (Murray)	13
Lecane pyriformis (Daday)	12
Lecane stenroosi (Meissner)	2
Lepadella cristata (Rousselet)	2
Lepadella elliptica Wulfert	2
Lepadella triba Myers	2
Microcodon clavus Ehrenberg	2
Mytilina trigona (Gosse)	2
Notommata glyphura Wulfert	8
Plationus patulus (O.F. Müller)	4
Ptygura furcillata (Kellicott)	2
Resticula gelida Herring et Myers	2
Testudinella incisa (Ternetz)	2
Trichocerca bidens (Lucks)	2
Trichocerca iernis (Gosse)	6
Trichocerca vernalis Hauer	6
Cladocera	%
Alona karelica Stenroos	10
Alona rustica Scott	24
Chydorus gibbus Sars	2
Chydorus ovalis Kurz	4
Dunhevedia crassa King	6
Leydigia acanthocercoides (Fischer)	4
Moina brachiata (Jurine)	2
Scapholeberis kingi Sars	2
Tretocephala ambigua (Lilljeborg)	4
Copepoda	%
Paracyclops affinis (Sars)	2

[a]Material taken from [10], changed

farming, such as the rotifer *Filinia opoliensis* recorded by [43]. The occurrence of rare species for the Polish fauna is also connected with climate change. This supports the arrival of new tropical or subtropical species [44], which can be primarily found in heated lakes [45, 46] and then start their march into other aquatic ecosystems, including ponds. The determination of the optimum conditions for the occurrence of rare species will help to provide a basis of helpful knowledge for the management of small water bodies and thereby assist in promoting their rank on the landscape scale.

3 Drivers of Zooplankton Diversity in Ponds

There are variety of environmental factors that are responsible for structuring zooplankton and its diversity in small water bodies. Among them, biological predation from both planktivorous fish and invertebrates as well as competition among certain groups of zooplankton is some of the most important [47, 48]. In the case of abiotic factors, both physical and chemical variables impact zooplankton diversity [49]. Specifically, the water quality features have a pronounced effect on the evolution of distinct communities and particularly the diversity of both rotifers and microcrustaceans [25]. Moreover, the origin of ponds and the type of direct catchment are known to structure zooplankton assemblages in ponds, and finally the habitat type, belonging to the most significant predictors.

3.1 Zooplankton in Ponds of Various Trophic States

Zooplankton diversity can be used as a valuable tool for the assessment of water quality in ponds. Rotifers and cladocerans segregate with respect to water trophy. Eutrophic conditions are often associated with the highest diversity of rotifers, while mesotrophic conditions favour high diversity of crustaceans. This was demonstrated in the study conducted in various habitats (open water, helophytes, elodeids) of 274 pastoral ponds; in the central-western Poland, it was demonstrated that in each trophic state, the biomass of macrophytes was a key predictor of zooplankton diversity [25]. Specifically, a shift was recorded from the high preponderancy of elodeids (e.g. plant beds with *Myriophyllum* spp. or *Ceratophyllum demersum*) that were responsible for a rise in zooplankton diversity in mesotrophic waters to helophytes (*Typha angustifolia* or *Phragmites australis*) in hypereutrophic ponds. However, hypereutrophic conditions, caused by nutrient overloading, were unfavourable for zooplankton diversity.

Poor quality of water in ponds can also be detected by analysing the share of eutrophic species (e.g. *Anuraeopsis fissa*, *Brachionus angularis*, *Keratella cochlearis* f. *tecta*, *Keratella quadrata*, *Bosmina longirostris* and *Chydorus sphaericus*), which were among most frequent in Polish ponds. It was also noticed that the percentage of eutrophic species differed significantly between certain microhabitats. In forest ponds the open water area along with helophytes possessed the highest, while elodeids harboured the lowest share of eutrophic species [50].

3.2 Zooplankton in Ponds of Various Origins

Small water bodies represent different origin types, which in turn may have an impact on abiotic characteristics of water and also the life conditions for the inhabiting organisms. The basic division may refer to natural ponds such as kettle holes, oxbows and meteorite craters or to artificial ponds – man-made ponds, e.g. clay, gravel and sandpits – or mining subsidence reservoirs. Human activity contributes to the artificial creation of many ponds for industrial, agricultural or recreational purposes but also for fish production or fishing, or for purely aesthetic reasons, enriching the beauty of the landscape, such as park or ornamental ponds. Ponds can also be created for the needs of the ecosystem (e.g. fire protection, flood protection, depositing nutrients, etc.) but currently also as an educational tool and for experimental research.

Man-made small water bodies often contain a high diversity and can also host ecologically valuable species. A study conducted on peatbogs of the Łeczyńsko-Włodawskie Lakeland has shown that planktonic rotifer communities had high species diversity as well as the presence of rare species [51, 52]. Also, peatbog pools located in the western part of Poland (Wielkopolski National Park) created a valuable habitat for the occurrence of specific communities of zooplankton [53], where among 88 identified zooplankton species, taxa typical of astatic and/or acidic waters, e.g. *Lecane elsa*, *Lecane mira* or *Mytilina bisulcata*, occurred. A study carried out on two small peat pits near Turew in the Wielkopolska region showed that even though these ponds were neighbouring, out of 80 identified species, less than 50% were common for both water bodies [54]. This underlines the fact of the high value of ponds in maintaining various communities of organisms even in a restricted area.

Oxbows, which are old river beds, were defined by [55] as small water bodies located in river valleys, connected either permanently or only periodically or even completely separated from proper riverbeds. Detailed characteristics of oxbows in Poland were defined by [56], who gave evidence that this type of aquatic ecosystem creates ecological corridors and a refuge for various organisms. They also refer to the necessity for the restoration of the natural character of river valleys so as to conserve their natural ecological condition and biodiversity. The specificity of oxbows significantly contributes to the formation of high biodiversity. This is probably due to the generally low level of anthropogenic transformation in the surroundings of this type of small water body. Even though they can be found in pastoral landscape, oxbows are often located in protected areas with limited human activities, such as the studied ponds in the Rogalin Warta Valley or in the Nadwarciański Landscape Park [57]. The results of the study conducted on various types of aquatic ecosystems in the Upper Narew Valley over 5 years (almost 600 samples) showed that out of

74 microcrustacean species identified in total, more than 80% were attributed to oxbows, increasing the regional biodiversity of the whole floodplain [37]. In a study conducted on 55 ponds within the Wielkopolska region [58], a high distinction between rotifer communities inhabiting different origin types was detected with oxbows having the highest rotifer diversity compared to postglacial kettle holes and artificial ponds. Two types of rotifer assemblages were distinguished: (1) lower species diversity with relatively more pelagic species and a higher share of eutrophic fraction in anthropogenically modified postglacial kettle holes and (2) greater species diversity and a greater occurrence of littoral rotifers in oxbows and artificial ponds.

Analyses on another origin type of ponds – the meteor craters – located in the Morasko Meteorite Nature Reserve near Poznań also gave an opportunity to find diverse zooplankton communities. Most of the identified species, with many rare for the Polish fauna, were characteristic of small temporary water bodies [40, 41]. The astatic character of these ecosystems, reflected by their fishless character and strong fluctuations in abiotic parameters, determined structure but also size of zooplankton.

3.3 Zooplankton in Permanent vs. Temporary Ponds

Due to the period of water filling, two types of ponds are distinguished: (1) astatic – seasonal ponds (e.g. vernal pools, puddles, rock pools) with irregular fluctuations in their water level, usually fishless, characterised by a huge variability of environmental conditions, which requires the development of specific adaptations of organisms to survive periods of pond disappearance and (2) permanent ponds, naturally filled with water at all times, regardless of the season or the variability of environmental conditions, and often have fish. Division into these two types of ponds is also reflected in the zooplankton community structure. As ascertained by [7], certain zooplankton species may be associated with gradients in hydroperiod and fish predation level. Temporary ponds support various zooplankton communities, both of littoral character (e.g. *Lepadella ovalis*, *Chydorus sphaericus*) and large-bodied pelagic taxa (e.g. *Eudiaptomus gracilis*), while permanent ponds with fish presence had lower diversity, with typically eutrophic taxa such as *Brachionus angularis*, *Keratella cochlearis* f. *tecta* or *Trichocerca pusilla* occurring with a high frequency. However, even though fish ponds are less diverse, they were found to be a source of rare species such as *Brachionus falcatus*, *Lecane tenuiseta* or *Ceriodaphnia dubia* that were exclusively found in these ponds compared to natural fish-free ponds. High biological diversity in ephemeral water bodies, including species important for conservation value, was also confirmed by [59, 60]. Moreover, [61] divided periodic ponds into two "faunistic types," extracting (1) rich rotifer fauna in association with less muddy ponds, not surrounded by trees, and (2) poor rotifer fauna associated with ponds surrounded by bushes and trees, with a thick layer of sediments and decaying leaves.

3.4 Zooplankton in Ponds of Various Types of Catchment Area

The type of land use (forest, field or urban areas) and degree of anthropogenic transformation in the ponds' vicinity are known to be of great importance in influencing the water quality, and consequently it may determine the diversity of organisms, including zooplankton (e.g. [62]). In the case of small water bodies, which have a smaller catchment area compared to other types of aquatic ecosystems such as lakes, ditches, rivers and streams [32], only the direct surroundings can be taken into account. [63], who examined over 100 small water bodies in Belgium, showed that ponds within forested areas were characterised by significantly better water quality compared to ponds located in agricultural areas, which can be reflected in higher biodiversity.

In the study conducted in the Wielkopolska region [10], it was noticed that forest ponds were characterised by higher zooplankton diversity than field ponds. In both types of ponds, genera such as *Lecane*, *Trichocerca* and *Cephalodella* among rotifers and *Alona* and *Ceriodaphnia* among crustaceans were the most frequent. Most species of these genera are typically littoral organisms, although they were also frequently met in the open water areas, which is related to the specificity of ponds creating a mosaic of habitats.

The type of water body (forest vs. field) can be a significant predictor of zooplankton species distribution. In a study conducted on a group of 12 ponds, 6 each within field and forest surroundings, 2 groups of zooplankton species were distinguished: (1) forest-associated species (representatives of genera *Cephalodella*, *Lepadella*, *Lecane* or *Trichocerca*) and (2) field-associated species (representatives of genera *Keratella*, *Bosmina* or *Ceriodaphnia*) [49]. Moreover, even within one type of pond, a variation in the zooplankton structure can be observed. In a set of 21 urban ponds of different sizes and locations studied along the urbanisation gradient, it was found that the distance from the city centre, number of plant species and pH belonged to the most important parameters determining cladoceran fauna in urban ponds of the city of Łódź [64]. In another Polish city, Poznań, almost 20 various small water bodies (natural and artificial ponds, claypits and pools) were examined, and 114 species, representing ca. 25% of all rotifers from Poland, were recorded with *Brachionus angularis*, *Keratella cochlearis*, *Colurella uncinata*, *Lecane closterocerca* and *Lepadella patella* being the most common [65].

The impact of catchment conditions can also be reflected in the size of the animal's body. Cladoceran *Chydorus sphaericus* was slightly larger in fertile nutrient-rich field ponds [66]. In turn, the size of the rotifer *Filinia longiseta* was significantly lower in field ponds, which is explained by the stronger fish pressure in this type of water body [67].

3.5 Zooplankton in Various Habitats

Ponds, small and shallow ecosystems, were usually treated as a single landscape unit. Therefore, the open water was the only zone that was analysed. However, the small area of a pond can be divided into a number of microhabitats created by patches of macrophytes, creating different conditions for the inhabiting organisms. The distinctiveness of the microhabitats, related to the specific morphology and spatial structure of plants, called architecture, can be responsible for the formation of diverse groups of organisms [68]. This enables cohabitation of species representing different functional groups with different preferences for habitat or food. Macrophyte beds offer pelagic zooplankton a refuge from both planktivorous fish and invertebrate predation, but in the case of littoral species, they create a multiplicity of ecological niches enabling various organisms to co-occur and increase total biodiversity. In the study carried out on the impact of environmental factors that structured the zooplankton assemblages of 55 ponds in Wielkopolska, habitat type was the strongest predictor of rotifer distribution, regardless of any other parameter. Most rotifer species were associated with macrophyte-dominated ponds, thereby illustrating the high value of vegetated areas, even in small aquatic ecosystems [58]. Moreover, an extensive study, taking into account 65 ponds varying in, e.g. catchment type, pond morphology, the presence or lack of fish as well as overshading, proved that the type of habitat was the strongest driver of zooplankton species distribution. This indicates a prerequisite to examine ponds in relation to their microhabitats created by various macrophytes [36]. The results of other investigations, conducted on ponds differing in origin, anthropogenic transformation or morphology of a pond basin, indicated that biometric parameters of the habitat – dry biomass and stem length of plants in a unit of water volume [69] – belonged to the strongest predictors of zooplankton occurrence [25, 66, 70, 71]. Plant biometrics is not only an indicator of the degree of spatial complexity but also of the availability of ecological niches for animals. In most cases elodeids, the most complex habitat compared to architecturally simple helophytes and nymphaeids, hosted the most diverse communities of both rotifers and crustaceans [25, 27].

Zooplankton distribution is determined primarily by habitat type, defined by diverse ecological groups of aquatic and rush vegetation. The ecological significance of spatial heterogeneity for zooplankton is closely connected to a functional perspective relating distribution patterns to environmental processes [68]. The degree of heterogeneity of the plant substratum reflected by an increase in the diversity of littoral microhabitats, determined mainly by length of plant stems, significantly affects zooplankton communities, and their diversity is higher. While in the open water 10–20 species of zooplankton are usually encountered, in a complex plant habitat, the number of species will be much higher, accounting for 30–50 [10].

There is also a close and significant relationship between the body size of organisms and particular habitat types in ponds, which confirms the assumption that the size structure of rotifers, e.g. *Filinia longiseta* [67], and crustaceans, e.g. *Chydorus sphaericus* [66], may differ depending on the morphological type of plant. The obtained results showed that the littoral is an extremely complicated system of dependence between abiotic elements and the inhabiting organisms or those periodically living within macrophytes.

The analysis of the share of eutrophic species in particular zones (open water vs. macrophyte-dominated areas) showed that the open water zone had the highest share of such species, while the elodeids had the smallest [10]. This signifies the necessity to exclude plant zones of ponds from water quality assessments based on zooplankton indicator species. This is due to the fact that mesotrophic and eutrophic indicator species are typically pelagic forms, which find their optimum in the open water area.

4 Concluding Remarks

The results of studies conducted on small water bodies not only expand our knowledge on zooplankton diversity but also create a very rich source of understanding of the ecological state of these ecosystems. These should also be considered as effective arguments in support of measures that would lead to the protection and maintenance of those valuable ecosystems, particularly because there is a tendency for the number of ponds to decrease from year to year [10, 72–74]. Golus and Bajkiewicz-Grabowska [75] gave evidence that the hydrological functions of ponds are very variable throughout the year and highly depend on the level of water storage in the catchment of a water body. It is not only highly imperative that more attention should be paid to the creation of new ponds but also to recognise the need for revitalisation of existing ponds, which finally will contribute to the increase in overall biodiversity. A large part of Europe, especially southern and western, has already noted the need to renature small water bodies. Thus, many projects serve this purpose, aiming, i.e. to reverse the trend of decay and deterioration of their quality and creating new ponds that will be priority natural habitats of European importance in accordance with the EU Habitats Directive. Some initiatives of this type also appear in Poland; however, the approach of individuals who can contribute to the protection and restoration of a decent condition to the natural environment is also very important. Therefore, educational actions on the biodiversity associated with ponds should be continually developed. Even though water bodies subjected to less human impact create a refuge for various freshwater species [2], the diversity of organisms does not relate only to a low scale of anthropogenic transformation of the landscape [76]. It is therefore also important to preserve ponds within strongly transformed areas – intensively farmed arable lands or urban areas, where many species which are rare, endangered or have a very narrow ecological scale may exist [77, 78, 79]. This is why the basic duty of the ecologist is to recognise, monitor and then undertake activities in order to protect and maintain them for future generations. Thus, some researchers (e.g. [3]) have defined priorities that should be fulfilled, such as (1) application of reliable monitoring programmes for small aquatic ecosystems and (2) development of effective measures that will lead to protection of aquatic biodiversity. To protect biodiversity of small water bodies also requires their intense monitoring, and even though interest in ponds systematically increases, they still remain neglected by scientists and excluded from water management planning in many parts of Europe.

The examination of small water bodies fully confirms a thesis that great variation of zooplankton exists both in different ponds and in different habitats. This clearly shows the direction of future limnological research and heightens awareness of the need for the protection of the diversity of small freshwater ecosystems, since it is within them that the symptoms of global climatic changes will be soonest visible.

The assumption of [2] has been well demonstrated in the case of Polish ponds which support a very high richness of inhabiting species given their generally small size. This might be connected with the fact that individual ponds, even those situated in close vicinity, usually support distinct fauna, contributing to very high diversity on both regional and national scales [2]. Therefore, even if local alpha diversity referring to a single site and beta diversity calculated for spatial replacement of species between the sites of an area [80] is not very high, the regional diversity (gamma) of small water bodies can be great.

References

1. Downing JA, Prairie YT, Cole JJ, Duarte CM, Tranvik LJ, Striegl RG, McDowell WH, Kortelainen P, Caraco NF, Melack JM, Middelburg JJ (2006) The global abundance and size distribution of lakes, ponds and impoundments. Limnol Oceanogr 51:2388–2397
2. Williams P, Whitfield M, Biggs J, Bray S, Fox G, Nicolet P, Sear D (2004) Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. Biol Conserv 115:329–341
3. Biggs J, von Fumetti S, Kelly-Quinn M (2017) The importance of small waterbodies for biodiversity and ecosystem services: implications for policy makers. Hydrobiologia 793:3–39
4. Stendera S, Adrian R, Bonada N, Canedo-Arguelles M, Hugueny B, Januschke K, Pletterbauer F, Hering D (2012) Drivers and stressors of freshwater biodiversity patterns across different ecosystems and scales: a review. Hydrobiologia 696:1–28
5. Céréghino R, Biggs J, Oertli B, Declerck S (2008) The ecology of European ponds: defining the characteristics of a neglected freshwater habitat. Hydrobiologia 597:1–6
6. Oertli B, Biggs J, Céréghino R, Grillas P, Joly P, Lachavanne J-B (2005) Conservation and monitoring of pond biodiversity: introduction. Aquat Conserv 15:535–540
7. Kuczyńska-Kippen N, Pronin M (2018) Diversity and zooplankton species associated with certain hydroperiods and fish state in field ponds. Ecol Indic 90:171–178
8. Hill MJ, Biggs J, Thornhill I, Briers RA, Ledger M, Gledhill DG, Wood PJ, Hassall C (2018) Community heterogeneity of aquatic macroinvertebrates in urban ponds at a multi-city scale. Landsc Ecol 33:389–405
9. Dumont HJ, Segers H (1992) Estimating lacustrine zooplankton species richness and complementarity. Hydrobiologia 341:125–132
10. Kuczyńska-Kippen N (ed) (2009) Funkcjonowanie zbiorowisk planktonu w zróżnicowanych siedliskowo drobnych zbiornikach wodnych Wielkopolski. Bonami, Poznań, p 505
11. Bielańska-Grajner I, Ejsmont-Karabin J, Radwan S (2015) Rotifers Rotifera Monogononta. Freshwater Fauna of Poland, vol 32. Wydawnictwo Uniwersytetu Łódzkiego, Łódź, p 580

12. Karabin A (1985) Pelagic zooplankton (Rotatoria + Crustacea) variation in the process of lake eutrophication I. Structural and quantitative features. Ekol Pol 33:567–616
13. Rybak JI, Błędzki LA (2010) Słodkowodne skorupiaki planktonowe. Klucz do oznaczania gatunków. Wydawnictwa Uniwersytetu Warszawskiego, Wydawca, p 366
14. Jurasz W (2005) Wioślarki (Cladocera) zbiorników śródpolnych i jezior południowych Kujaw. Studium faunistyczno-ekologiczne [Cladocerans (Cladocera) of mid-field ponds and lakes of southern Kujawy. Fauna and ecological study]. Wyd. UŁ, Łódź, p 123
15. Pociecha A, Wilk-Woźniak E (2006) The life strategy and dynamics of selected species of phyto- and zooplankton in a dam reservoir during "wet" and "dry" years. Pol J Ecol 54:29–38
16. Goździejewska A, Tucholski S (2011) Zooplankton of fish culture ponds periodically fed with treated wastewater. Pol J Environ Stud 20:67–79
17. Skowronek E, Cudak A, Bielańska-Grajner I (2012) Effect of recreation on the species richness and diversity of rotifers in ponds. Int J Water Res Prot 4:795–799
18. Pejler B (1962) On the variation of the rotifer Keratella cochlearis (Gosse). Zool Bidr Upps 35:1–17
19. Czerniawski R, Popko R, Krepski T, Domagała J (2013) Invertebrates of three small ponds located in stream-pond system. Teka Kom Ochr Kszt Prod Przyr 10:14–22
20. Kuczyńska-Kippen N (2008) Spatial distribution of zooplankton communities between the *Sphagnum* mat and open water in a dystrophic lake. Pol J Ecol 56:57–64
21. Lokko K, Virro T (2014) The structure of psammic rotifer communities in two boreal lakes with different trophic conditions: Lake Võrtsjärv and Lake Saadjärv (Estonia). Oceanol Hydrobiol Stud 43:49–55
22. Iakovenko NS, Smykla J, Convey P, Kasparova EI, Kozeretska A, Trokhymets V, Dykyy I, Plewka M, Devetter M, Duris Z, Janko K (2015) Antarctic bdelloid rotifers: diversity, endemism and evolution. Hydrobiologia 761:5–43
23. Czerniawski R, Sługocki Ł, Kowalska-Góralska M (2016) Diurnal changes of zooplankton community reduction rate at lake outlets and related environmental factors. PLoS One 11(7): e0158837
24. Kuczyńska-Kippen N (2018) The use of bdelloids in reference to rotifer biocoenotic indices as an indicator of the ecological state of small field water bodies: the effect of macrophytes, shading and trophic state of water. Ecol Indic 89:576–583
25. Kuczyńska-Kippen N, Joniak T (2016) Zooplankton diversity and macrophyte biometry in shallow water bodies of various trophic state. Hydrobiologia 774:39–51
26. Dziuba MK, Cerbin S, Wejnerowski Ł (2013) Cladocera and Copepoda of the shallow eutrophic lake in Natura2000 area in western Poland. Pakistan J Zool 45:653–659
27. Celewicz-Gołdyn S, Kuczyńska-Kippen N (2017) Ecological value of macrophyte cover in creating habitat for microalgae (diatoms) and zooplankton (rotifers and crustaceans) in small field and forest water bodies. PLoS One 12(5):e0177317
28. Kosiba J, Wilk-Woźniak E, Krztoń W, Strzesak M, Pociecha A, Walusiak E, Pudaś K, Szarek-Gwiazda E (2017) What underpins the trophic networks of the plankton in shallow oxbow lakes? Microb Ecol 73:17–28
29. Havens KE (1991) Summer zooplankton dynamics in the limnetic and littoral zones of a humic acid lake. Hydrobiologia 215:21–29
30. Adamczuk M (2016) Past, present, and future roles of small cladoceran *Bosmina longirostris* (O.F. Müller, 1785) in aquatic ecosystems. Hydrobiologia 767:1–11
31. Dembowska EA, Napiórkowski P (2015) A case study of the planktonic communities in two hydrologically different oxbow lakes, Vistula River, Central Poland. J Limnol 74:346–357
32. Davies BR, Biggs J, Williams PJ, Lee JT, Thompson S (2008) A comparison of the catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic biodiversity in an agricultural landscape. Hydrobiologia 597:7–17
33. Brzęk G (1948) Studium limnologiczne nad zbiornikami wodnymi Wielkopolskiego Parku Narodowego pod Poznaniem [Limnological study on water bodies of the Wielkopolski National

Park near Poznań]. Praca monograficzna nad Przyrodą Wielkopolskiego Parku Narodowego pod Poznaniem. Pr Kom Biol PTPN 2, 2

34. Kamiński KZ (1975) Drugie stanowisko w Polsce *Simocephalus lusaticus* Herr (Cladocera, Daphniidae) [The second locality in Poland of *Simocephalus lusaticus* Herr (Cladocera, Daphniidae)]. Acta Hydrobiol 17:89–92
35. Jurasz W (2003) Fauna Cladocera Polski - aktualny stan wiedzy [Cladocera fauna of Poland - the present state of knowledge]. Przeg Zool 47:7–17
36. Kuczyńska-Kippen N, Wiśniewska M (2011) Environmental predictors of rotifer community structure in two types of small water bodies. Int Rev Hydrobiol 96:397–404
37. Karpowicz M (2017) Biodiversity of microcrustaceans (Cladocera, Copepoda) in a lowland river ecosystem. J Limnol 76:15–22
38. Jurasz W (1992) Wioślarki (Cladocera) Roztocza Środkowego [The water-fleas (Cladocera) of the Central Roztocze Upland]. Fragm Faun 35:301–310
39. Bielańska-Grajner I, Cudak A (2014) Effects of salinity on species diversity of rotifers in anthropogenic water bodies. Pol J Environ Stud 23:27–34
40. Kuczyńska-Kippen N, Basińska AM, Świdnicki K (2013) Specificity of zooplankton distribution in meteorite crater ponds (Morasko, Poland). Knowl Manag Aquat Ecosyst 409:08
41. Świdnicki K, Basińska AM, Pronin M, Kuczyńska-Kippen N (2016) Meteorite crater ponds as source of high zooplankton biodiversity. Biologia 71:1361–1368
42. Kolicka M, Dziuba MK, Zawierucha K, Kuczyńska-Kippen N, Kotwicki L (2015) Palm house – biodiversity hotspot or risk of invasion? Aquatic invertebrates: the special case of Monogononta (Rotifera) under greenhouse conditions. Biologia 70:94–103
43. Bielańska-Grajner I, Sowa A, Pasamonik A, Pielok K, Kolek L (2017) A new record of the rare rotifer species *Filinia opoliensis* in southern Poland. Chrońmy Przyr Ojcz 73:151–154
44. Ejsmont-Karabin J (2014) Rotifer invasion? On appearance and abundance of tropical species in lakes of North-Eastern Poland. Pol J Ecol 62:821–827
45. Ejsmont-Karabin J (2011) Does invasion of *Vallisneria spiralis* L. promote appearance of rare and new rotifer (Rotifera) species in littoral of the lakes heated by power station (Konin lakes, W. Poland)? Pol J Ecol 59:201–207
46. Ejsmont-Karabin J, Hutorowicz A (2011) Spatial distribution of rotifers (Rotifera) in monospecies beds of invasive *Vallisneria spiralis* L. in heated lakes. Oceanol Hydrobiol Stud 40:71–76
47. Basińska AM, Świdnicki K, Kuczyńska-Kippen N (2014a) Effect of surrounding trees and dry rush presence on spring zooplankton community in an urban pond complex. Ann Limnol 50:315–323
48. Brysiewicz A, Sługocki Ł, Wesołowski P, Czerniawski R (2017) Zooplankton community structure in small ponds in relation to fish community and environmental factors. Appl Ecol Environ Res 15:929–941
49. Kuczyńska-Kippen N, Joniak T (2010) The impact of water chemistry on zooplankton occurrence in two types (field versus forest) of small water bodies. Int Rev Hydrobiol 95:130–141
50. Basińska A, Kuczyńska-Kippen N (2009) Differentiated macrophyte types as a habitat for rotifers in small mid-forest water bodies. Biologia 64:1100–1107
51. Demetraki-Paleolog A, Tarkowska-Kukuryk M (2010) Structure of rotifer communities of restored small peat-bog reservoirs of Poleski National Park. Arch Environ Prot 36:21–30
52. Demetraki-Paleolog A (2014) Studies on sustainability of planktonic rotifer assemblages in select national park ponds and wetland reservoirs. Pol J Environ Stud 23:51–56
53. Klimaszyk P, Kuczyńska-Kippen N (2005) Peat-bog pool (Wielkopolski National Park) as a habitat of specific communities of zooplankton. Acta Agrophysica 7:375–382
54. Kuczyńska-Kippen N, Nagengast B, Bartkowska E, Horbajczuk L (2004) Differentiation of the structure of zooplankton communities in stands of water and rush vegetation of post-exploitation small water bodies in Poznań. Teka Kom Ochr Kszt Prod Przyr 1:122–129

55. Żmudziński L, Kornijów R, Bolałek J, Górniak A, Olańczuk-Meyman K, Pęczalska A, Korzeniewski K (2002) Słownik hydrobiologiczny, terminy, pojęcia, interpretacje. PWN, Warszawa, p 287
56. Dembowska E, Napiórkowski P (2012) Dlaczego warto chronić starorzecza? [Why do we have to protect old river beds?]. Kosmos 61:341–349
57. Joniak T, Kuczyńska-Kippen N (2016) Habitat features and zooplankton community structure of oxbows in the limnophase: reference to transitional phase between flooding and stabilization. Limnetica 35:37–48
58. Kuczyńska-Kippen N, Basińska AM (2014) Habitat as the most important influencing factor for the rotifer community structure at landscape level. Int Rev Hydrobiol 99:58–64
59. Klimowicz H (1967) Rotifers of astatic waters. Part II. Rotifers of small water bodies from the Mikołajki region. Pol Arch Hydrobiol 14:91–110
60. Yevdokimov NA, Yermokhin MV (2009) Zooplankton crustaceans of ephemeral waterbodies on the territory of various natural zones in Saratov Oblast. Inland Water Biol 2:59–66
61. Klimowicz H (1970) Wrotki (Rotatoria) wód astatycznych [Rotifers (Rotatoria) of astatic waters]. Zesz Nauk Inst Gosp Komunalnej Warszawa 30:253
62. Dodson SJ, Everhast WR, Jandl AK, Krauskopf SJ (2007) Effect of watershed land use and lake age on zooplankton species richness. Hydrobiologia 579:393–399
63. Declerck S, De Bie T, Ercken T, Hampel H, Schrijvers S, Van Wichelen J, Gillard V, Mandiki R, Losson B, Bauwens D, Keijers S, Vyverman W, Goddeeris B, De Meester L, Brendonck L, Martens K (2006) Ecological characteristics of small farm land ponds: 1 associations with land-use practices at multiple spatial scales. Biol Conserv 131:523–532
64. Pawlikiewicz P, Jurasz W (2017) Ecological drivers of cladoceran diversity in the central European city (Łódź, Poland): effects of urbanisation and size of the waterbody. Ann Zool Fennici 54:315–333
65. Ejsmont-Karabin J, Kuczyńska-Kippen N (2001) Urban rotifers - structure and densities of rotifer communities in water bodies of the Poznań agglomeration (Western Poland). Hydrobiologia 446(447):165–171
66. Basińska AM, Antczak M, Świdnicki K, Jassey VEJ, Kuczyńska-Kippen N (2014b) Habitat type as strongest predictor of the body size distribution of *Chydorus sphaericus* (O. F. Muller) in small water bodies. Int Rev Hydrobiol 99:382–392
67. Basińska A, Kuczyńska-Kippen N, Świdnicki K (2010) The body size distribution of *Filinia longiseta* (Ehrenberg) in different types of small water bodies in the Wielkopolska region. Limnetica 29/1:171–182
68. Pinel-Alloul B, Ghadouani A (2007) Spatial heterogeneity of planktonic microorganisms in aquatic systems. In: Franklin R, Mills A (eds) The spatial distribution of microbes in the environment. Springer, Dordrecht, pp 203–310
69. Nagengast B, Kuczyńska-Kippen N (2015) Macrophyte biometric features as an indicator of the trophic status of small water bodies. Oceanol Hydrobiol Stud 44:38–50
70. Pasztaleniec A, Karpowicz M, Strzałek M (2013) The influence of habitat conditions on the plankton in the Białe oxbow lake (Nadbużański Landscape Park). Limnol Rev 13:43–50
71. Joniak T, Kuczyńska-Kippen N, Gąbka M (2017) Effect of agricultural landscape characteristics on the hydrobiota structure in small water bodies. Hydrobiologia 793:121–133
72. Pieńkowski P (2004) Disappearance of ponds in landscape of Northern Europe as an effect of anthropogenic influence and global climate change. Pol J Environ Stud 13:192–196
73. Jeffries M (2012) Ponds and the importance of their history: an audit of pond numbers, turnover and the relationship between the origins of ponds and their contemporary plant communities in south-east Northumberland, UK. Hydrobiologia 689:11–21
74. Waldon B (2012) Natural value and endangerment of small water reservoirs in the Krajeńskie Lakeland (North-West Poland). Limnologica 42:320–327
75. Golus W, Bajkiewicz-Grabowska E (2017) Water circulation in the moraine ponds of northern Poland. Hydrobiologia 793:55–65

76. Pociecha A, Wilk-Woźniak E, Mróz W, Bielańska-Grajner I, Gadzinowska J, Walusiak E (2015) Biodiversity of rotifers in urban water reservoirs of southern Poland. Oceanol Hydrobiol Stud 44:335–342
77. Koperski P (2010) Urban environments as habitats for rare aquatic species: the case of leeches (Euhirudinea, Clitellata) in Warsaw freshwaters. Limnologica 40:233–240
78. Hassall C (2014) The ecology and biodiversity of urban ponds. WIREs Water 1:187–206
79. Hill MJ, Wood PJ, Mathers KL (2015) The aquatic macroinvertebrate biodiversity of urban ponds in a medium-sized European town (Loughborough, UK). Hydrobiologia 760:225–238
80. Whittaker RH (1972) Evolution and measurement of species diversity. Taxon 21:213–251

Structure of Ciliate, Rotifer, and Crustacean Communities in Lake Systems of Northeastern Poland

Jolanta Ejsmont-Karabin, Krystyna Kalinowska, and Maciej Karpowicz

Contents

Abstract Pelagic zooplankton communities are structured by ecological processes, like dispersal, and by biotic interactions and abiotic environmental conditions. In this paper we study a role of the processes in structuring zooplankton communities in relation to a character of connections between lakes in their systems. Studies were carried out in five lake systems in northeast Poland: the Great Masurian Lakes, lakes of the Krutynia River watershed, harmonic and dystrophic lakes in the Wigry National Park, and lakes in the Suwałki Landscape Park. Pelagic waters of all the studied lakes involved 89 ciliate, 129 rotifer, 40 cladoceran, and 22 copepod species. Forty-seven rotifer species were littoral, occasionally occurring in the pelagial.

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In four harmonic lake systems, there were the same most frequent species: among are rotifers *Keratella cochlearis* (100% frequency) and present in more than 75% *Ascomorpha saltans*, *Asplanchna priodonta*, *Collotheca mutabilis*, *Gastropus stylifer*, *Kellicottia longispina*, *Keratella quadrata*, *Polyarthra major*, *P. remata*, and *P. vulgaris* and crustaceans *Daphnia cucullata* and *Diaphanosoma brachyurum*. Among ciliates, the most common and frequent (100%) species for two harmonic lake systems was *Rimostrombidium humile*. Dystrophic lakes in the Wigry National Park had a completely different list of the most frequent ciliate and rotifer species, among which only one ciliate (*Halteria grandinella*) and one rotifer species (*Polyarthra remata*) occurred in more than 75% of the lakes. Our results suggest that the role of dispersal processes in structuring zooplankton communities is particularly important in the system of lakes that are connected directly or by short channels.

Keywords Biodiversity · Ciliata · Cladocera · Copepoda · Lakes · Rotifera

1 Introduction

Poland is relatively rich in lakes, which cover in total 2,810 km^2 [1]. The country consists of 7,081 lakes of area 10^4 m^2 or more. However, only 3,112 lakes range from 1 to 5×10^4 m^2. Most of the lakes occur in the north part of Poland (Masuria and Pomerania). They are mostly of postglacial origin, i.e., 8,000–13,000 years old [2]. Deep lakes are thermally stratified. Lakes in northeast Poland are usually eutrophic and meso-eutrophic. Mesotrophic lakes are not common here, and oligotrophic ones are nearly absent. A characteristic feature of this part of Poland is the presence of numerous lake systems, e.g., Great Masurian Lakes, lakes of the Krutynia River watershed, harmonic and dystrophic lakes in the Wigry National Park, and lakes of the Suwałki Landscape Park (Fig. 1). Extensive studies of zooplankton were conducted in all of the abovementioned lake systems.

Pelagic zooplankton communities are structured by ecological processes, like dispersal, and by biotic interactions and abiotic environmental conditions. Dispersal processes are especially important in systems of lakes, which may be more or less isolated within terrestrial landscapes. Role of the processes in structuring zooplankton communities in relation to a character of connections between lakes in their systems is still unknown. Zooplankton communities are effective indicators of environmental variations such as abiotic factors and changes in lake food webs [3]. Zooplankton may disperse passively among lakes and in dormant forms – among lake systems. There is however a discrepancy as regards the importance of dispersal as a factor determining zooplankton composition. It seems that the process is scale-dependent [4].

Dispersal limitation is probably less important than environmental control [3, 5], and spatial variation may result from historical effects [6]. Taking into account that zooplankton may disperse passively through overland and hydrological connections

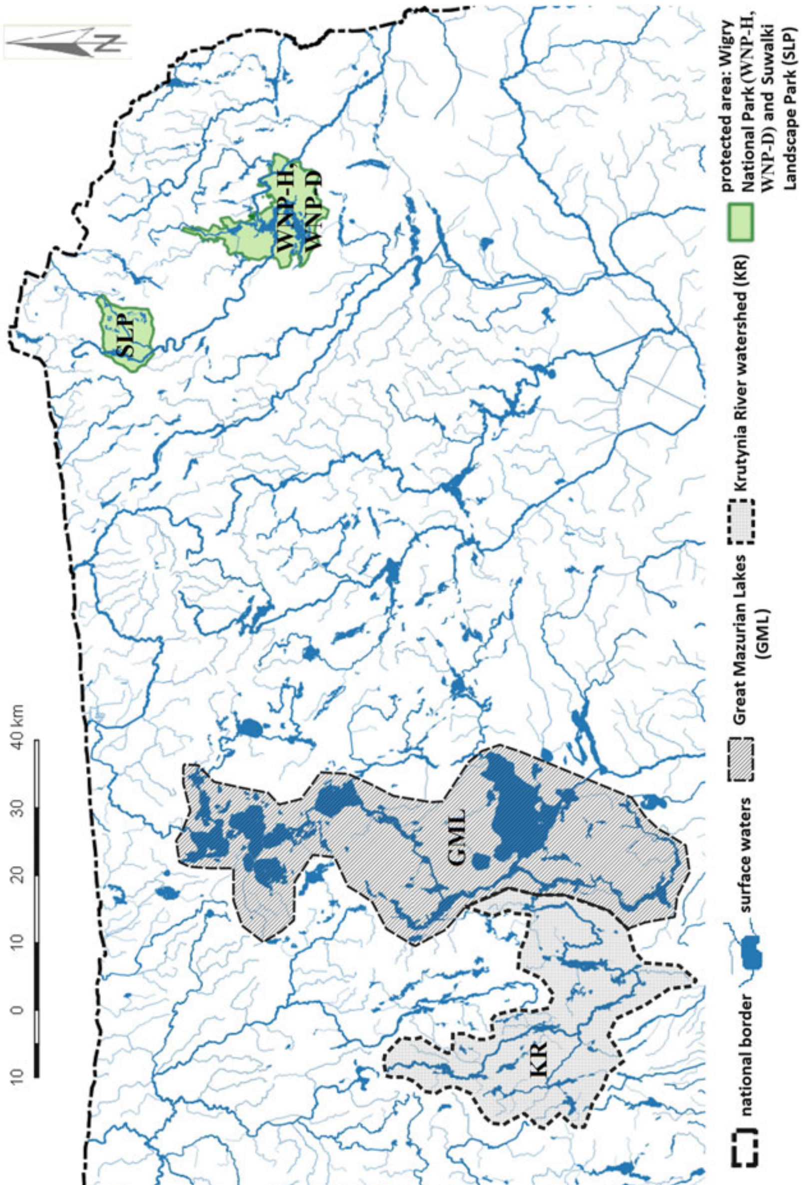

Fig. 1 Location of the studied lake systems

between lakes, we may expect different structures of zooplankton communities in different lake systems.

An aim of this study was to evaluate the effects of a character of different landscapes for structuring ciliate, rotifer, and crustacean communities. The studies quantified the relative role of dispersal processes in determining the species structure of zooplankton communities at two different spatial scales: local (i.e., within lake systems) and regional (between lake systems).

2 Methods

2.1 Study Area

Studies were carried out in five lake systems in northeast Poland (Fig. 1, Table 1). The study area is characterized by climatic conditions that represent patterns of weather typical of the moderate transition climate with continental influences. The multiannual mean air temperature does not exceed 6°C [7], which is among the lowest values in Poland.

The Great Masurian Lakes (GML) comprise three basic lake systems: northern basin drained by the River Węgorapa, central basin without outflow, and southern basin drained by the River Pisa [8]. The watershed line is situated between lakes Kisajno and Niegocin and separates the system into two hydrologically different parts. However, the course of the watershed line is variable and depends on the current hydrometeorological conditions and water management. The Great Masurian Lakes system consists of 31 lakes connected by natural and artificial channels or natural rivers and streams. They are mostly surrounded by agriculture areas and forests. Water quality assessment based on physical and chemical data for 1976–1977 [9] showed rapid eutrophication of the lakes in the central part of the GML system (lakes Niegocin, Jagodne, Tałty, Ryńskie, Mikołajskie). As a result, in the 1980s the lakes represented the gradient of eutrophication from mesotrophy to hypereutrophy. Lakes located above the watershed line (northern lakes) were less eutrophicated (e.g., Przystań, Mamry, Dargin, Łabap, Kisajno), while lakes located below the watershed (southern lakes) were more eutrophicated (e.g., Ryńskie, Tałty, Mikołajskie, Śniardwy, Bełdany), among which three were highly eutrophicated lakes (Szymoneckie, Szymon, Tałtowisko). Nowadays, after establishment of numerous sewage purification plants in the region, most of the lakes are meso-eutrophic [10]. There is no heavy industry in the watershed of the lakes. The main anthropogenic impact occurs through sewage and agricultural runoff. The lakes are similar with respect to the input of allochthonous organic matter because they have comparable agricultural, wetland, and urban areas [11]. During the summer season (June–August), the lakes are strongly influenced by recreational activities.

The Krutynia river-lake (KR) system is a set of 17 lakes linked by a river ca 100 km long (Table 1). The system is very popular among tourists and well known for routes for sailing and kayaking. A large part of the system is situated within the

Table 1 Characteristics of the studied lake systems of northeastern Poland (data are given only for lakes considered in this study)

Characteristics	Lake system				
	GML	KR	WNP-H	WNP-D	SLP
Number of lakes					
Total	21	17	12	20	25
Dimictic	15	9	12	0	11
Range of lake areas (10^4 m^2)	40–11,383	24–841	0.8–2,118.3	0.2–8.9	3.4–356.1
Depth of lakes (m)					
Mean	8.8	5.4	11.0	2.6	7.4
Range of mean	0.6–14.0	1.7–12.7	3.5–15.8	0.8–8.6	1.2–38.7
Range of maximum	3.0–47.0	3.2–51.0	11.0–73.0	1.5–15.0	2.4–108.5
Type of connections between lakes	Connected directly or via short channels	A river-lake-river system	Connected with small rivers, in Czarna Hańcza watershed	A lack of connections	Connected with small rivers in Szelmentka and Szeszupa watersheds
Drainage basins (mean for lakes of the system)	Forests – 37.0% Arable lands – 19.0% Meadows – 13.3% Urban areas – 7.9% Bogs – 22.0%	Forests – 51.6% Arable lands – 39.7% Meadows – 6.9% Urban areas – 1.4%	Forests – 55.2% Arable lands – 37.0% Meadows – 2.7% Bogs –2.6%	Forests – 91.2% Arable lands – 6.2% Meadows – 1.8% Bogs –0.3%	Forests – 24.0% Arable lands – 62.7% Meadows – 13.2%
Source of information	[1, 52, 53]	[54]	[13]	[13]	[55, 56]

GML the Great Masurian Lakes, *KR* lakes of the Krutynia River watershed, *WNP-H* harmonic lakes in the Wigry National Park, *WNP-D* dystrophic lakes in the Wigry National Park, *SLP* lakes in the Suwałki Landscape Park

Mazury Landscape Park with many nature reserves. The area of the lakes in KR system ranges from 24×10^4 m^2 (Lake Kujno) to 841×10^4 m^2 (Lake Mokre); their maximum depth is between 3.2 m (Lake Krutyńskie) and 51 m (Lake Mokre). Based on zooplankton indices, the lakes are mostly meso-eutrophic and eutrophic, except Lake Mokre which is mesotrophic [12].

Harmonic lakes in the Wigry National Park (WNP-H) are very diverse and include ca 30 lakes belonging to three drainage basins. These are typical deep channel lakes, moraine lakes, and shallow polymictic lakes. As a result, the set of lakes is very differentiated in relation to their area (from 0.8 to $2{,}118.3 \times 10^4$ m^2) and maximum depth (from 11.0 to 73.0 m) [13] (Table 1).

Dystrophic lakes in the Wigry National Park (WNP-D) are 16 small (area of $0.2–8.9 \times 10^4$ m^2) and shallow (maximum depth of 1.2–15 m) lakes without an outlet. The lakes are a very specific type of lakes due to a small quantity of mineral substances dissolved in their waters. The waters are strongly acidic and contain a remarkable content of dissolved organic matter. The share of humus substances fed from the catchment basin also plays an important role in the dystrophication of the lakes. The lakes have a characteristic yellow-brown color of waters [14]. This results in several typical characteristics: unique light climate (rapid light attenuation and changes in wavelength proportions), rapid warming of the epilimnion, and hence strong thermal stratification [15].

Lakes in the Suwałki Landscape Park (SLP) are as diverse as landscapes of the Park, with forests, meadows, and pastures. These are several tens of small and large lakes with the deepest lake in Poland – Lake Hańcza (105.6 m). Lake Hańcza Reserve has been created in 1963. The concentrations of nutrients in waters of the SLP lakes are much lower than in the other lakes of the region. Maintaining a low trophic level in the studied lakes is possible because of large buffering properties and low phosphorus loads from the catchment [16]. The catchments of the lakes are mostly unforested with extensive agriculture and population density less than 30 persons per square km [15].

2.2 Sampling and Analysis

Zooplankton was sampled in summer 2009–2015 using 2.6 L Limnos sampler at the deepest place in a lake at 1 m intervals, filtered through a conical plankton net with mesh size of 30 μm, and pooled together for epi-, meta-, and hypolimnion (in stratified lakes) or for the total water column in polymictic lakes. Ciliate samples were taken without filtration and fixed with Lugol's solution and then decanted to 10 mL and examined with a light microscope.

Ciliate, rotifer, and crustacean communities were identified and enumerated to species level (if possible) in samples equal usually to 0.5 L for ciliates, 1 L for rotifers, and 5 L for crustaceans. Zooplankton mean densities for water column were used.

3 Zooplankton in Lake Systems of Northeastern Poland

3.1 Zooplankton of the Great Masurian Lakes

Relatively little has been published about the composition, abundance, and role of ciliates in the lake system. First studies focusing on the ciliate communities were carried out in Lake Mikołajskie in 1970–1971 by Bownik-Dylińska [17]. The author determined 14 ciliate species, among which small Holotricha were the most numerous during the whole period of the study. Subsequent research was carried out after almost 30 years in eutrophic Lake Ryńskie in which 31 taxa were recorded [18]. According to the author, the lake was mainly dominated by bacterivorous ciliates, represented by small-sized Scuticociliatida and larger-sized Peritrichida; there was also a high percentage of predatory ciliates. Studies carried out during summer stratification in 2004–2006 by Chróst et al. [19] demonstrated that ciliate community was characterized by high variability over a short time scale and that food resources were probably an important factor determining the abundance of ciliates.

Kalinowska et al. [20] in studies conducted in the spring 2009 identified a total of 50 ciliate taxa and found that three ciliate orders, namely, Oligotrichida, represented by *Rimostrombidium* species, Prostomatida (mainly small *Urotricha* spp.), and Scuticociliatida, mainly composed of large-bodied *Histiobalantium bodamicum*, were dominant. The authors demonstrated that the studied lakes were characterized by relatively high ciliate numbers that ranged from 32.9 to 100.5 ind. mL^{-1}.

Studies carried out in the summer 2009 by Kalinowska (unpublished data) showed that the northern mesotrophic lakes were characterized by low numbers of ciliates (below 20 ind. mL^{-1}). Southern eutrophic lakes had relatively high ciliate numbers (70–90 ind. mL^{-1}), while lakes located at the end of the system were characterized by moderate numbers of ciliates (20–50 ind. mL^{-1}). Two ciliate orders were dominant. In lakes situated in the northern and central part of the system, Oligotrichida dominated, accounting for 40–50% of the total ciliate numbers. They were represented mainly by *Strombidium* sp., *Tintinnidium*, and small species from the genus *Rimostrombidium*. In the southern lakes, Haptorida, represented by small omnivorous species from the genus *Mesodinium* (*M. acarus* and *M. pulex*) were the most numerous (40–60% of the total numbers). The importance of Oligotrichida and Prostomatida decreased, while the importance of Haptorida and Scuticociliatida increased along with a eutrophication gradient (from northern to southern lakes). Close correlations were found between ciliates and both autotrophic and heterotrophic nanoflagellates.

Seasonal dynamics of planktonic ciliates, including winter under ice period, were studied in the eutrophic Lake Mikołajskie [17, 21], Lake Ryńskie [18], and the shallow hypereutrophic Lake Warnołty [Kalinowska et al., unpublished]. Bownik-Dylińska [17] showed that the mean numbers of protozoans (ciliates and testate amoebae) ranged from 0.50 to 8.0 ind. mL^{-1} with a maximum in spring, whereas Kalinowska [18] noted the highest numbers in September. Kalinowska and Grabowska [21] showed that during winter, ciliates were present in relatively high numbers (1.3–25.2 ind. mL^{-1}; values about four times lower than in the ice-free periods) with the peak in the middle of January. The authors did not find any relationship between ciliates and either physical or biological factors, suggesting that the ciliate community was mainly controlled bottom up by bacterial food resources.

Monitoring program of the Great Masurian Lakes started in 1986 and was undertaken to follow long-term changes in the lakes. In frames of the program, rotifers and crustaceans were sampled once a year during summer stagnation period since 1986 till 1996. Additional studies were carried out in the years 2011–2013. The preliminary results were published in Progress Reports of the Hydrobiological Station Mikołajki. Data on rotifer and crustacean abundance and species composition were used to establish zooplankton indices of lake trophy [22, 23]. Based on the structure of rotifer communities, observations revealed three periods in functioning of the GML: rapid eutrophication in 1986–1995, then 10 years of oligotrophication, and eutrophication of lakes in the southern part of the GML system [10]. In the first period, bacterivorous rotifers (*Keratella cochlearis*, *Brachionus angularis*, *Anuraeopsis fissa*, and others) played more and more important role. In the second period, indicators of low trophy (*Polyarthra major*, *Gastropus stylifer*, *Ascomorpha ecaudis*) appeared. At the end of the first period, the lowest density of Rotifera was noted in epilimnion of Lake Dargin (222 ind. L^{-1}), where colonial *Conochilus unicornis* dominated, whereas the highest one was found in Lake Ryńskie (28,600 ind. L^{-1}) in which rotifer community was dominated by *Anuraeopsis fissa*. In the year 2005, rotifer densities were markedly lower. The lowest rotifer abundance was noted for Lake Niegocin (50 ind. L^{-1}), the highest one (3,095 ind. L^{-1}) for Lake Nidzkie.

One of the lakes more intensively investigated was a Biosphere Reserve, Lake Łuknajno. Seasonal studies of zooplankton in the lake were a part of complex works carried out in the year 1993 [24]. Results of the research have shown [25] that a high rate of phytoplankton consumption by herbivorous zooplankton prevents the formation of phytoplankton blooms in this lake. The phosphorus regenerated by zooplankton is incorporated by bacterioplankton, and as a result of low bacterial consumption by bacterivorous zooplankton, it is withdrawn from circulation.

Studies of crustacean zooplankton of the Great Masurian Lakes revealed that the communities were built of typical planktonic species (*Daphnia cucullata*, *Bosmina coregoni*, *Bosmina longirostris*, *Diaphanosoma brachyurum*, *Thermocyclops oithonoides*, *Mesocyclops leuckarti*, *Eudiaptomus graciloides*). The biomass of the lake zooplankton was dominated by filter feeder *Daphnia cucullata*. In shallow lakes, the proportion of smaller species (*Ceriodaphnia quadrangula*, *Chydorus sphaericus*) was higher. The distinguishing feature of the Great Masurian Lakes is a low diversity of pelagic Cyclopoida, where only four species were found (Table 2). However, *Acanthocyclops americanus* was found only in the lakes of this system. Another distinguishing feature is the presence of Calanoida glacial relicts: *Eurytemora lacustris*, *Heterocope appendiculata*, and *Limnocalanus macrurus*. Among them, *Limnocalanus macrurus* was found only in this lake system (Dargin, Mamry, Kisajno).

3.2 Zooplankton of Lakes of the Krutynia River Watershed

Research of 17 lakes of the lake system was carried out in summer 1987 [12]. The studies revealed an impact of river connection on zooplankton of the linked lakes. Three assemblages of zooplankton characteristic of upper, middle, and

Table 2 List of pelagic zooplankton species and their frequency in the studied lake systems (abbreviations are given in Table 1)

Species	GML	KR	HW	DW	SLP
Ciliata					
Actinobolina radians (Stein)	S				
Amphileptus sp.	*				
Askenasia chlorelligera Krainer & Foissn.	S				S
Askenasia sp.	***			S	S
Askenasia volvox (Eichwald)	*****			**	***
Aspidisca cicada (Müller)	*				
Aspidisca lynceus (Müller)	S				
Astylozoon faurei Kahl	*				
Balanion planctonicum (Foissner et al.)	****				*
Belonophrya pelagica André	***				**
Bursaridium pseudobursaria (Fauré-Fre.)	*			S	
Chilodonella uncinata (Ehrenberg)	S				
Cinetochilum margaritaceum (Ehrenberg)	****			*	***
Codonella cratera (Leidy)	*****				**
Codonella cratera v. lariana Zacharias	S				
Coleps amphacanthus Ehrenberg	S				
Coleps hirtus hirtus (Müller)	***				***
Coleps hirtus lacustris Fauré-Fremiet	S				
Coleps hirtus viridis Ehrenberg	****				
Coleps nolandi Kahl	*				S
Coleps spetai Foissner	*				
Ctedoctema acanthocryptum Stokes	*				
Cyclidium sp.	****			***	*
Cyclotrichium viride Gajewskaja	*				
Cyrtolophosis mucicola Stokes	**			S	
Enchelys gasterosteus Kahl	S			**	
Epicarchesium pectinatum (Zacharias)	S				
Epistylis procumbens Zacharias	S				
Epistylis pygmaeum (Ehrenberg)	S				
Epistylis sp.	*				*
Frontonia leucas (Ehrenberg)	S				
Halteria grandinella (Müller)	*****			****	****
Hastatella radians Erlanger	S				
Histiobalantium bodamicum Krainer & Müll.	****				S
Holosticha pullaster (Müller)	S				
Hypotrichs	S				
Lagynophrya acuminata Kahl	****				**
Lembadion lucens (Maskell)	S			**	
Limnostrombidium viride (Stein)	*****			***	****
Litonotus sp.	S				S
Loxodes sp.	S				

(continued)

Table 2 (continued)

Species	GML	KR	HW	DW	SLP
Loxophyllum sp.	S				
Mesodinium acarus Stein	****				S
Mesodinium pulex (Claparède & Lachm.)	***				
Monodinium balb. balbianii Fabre-Dom.	****				*
Monodinium chlorelligerum Krainer	***				S
Monodinium sp.	*				
Nassula ornata Ehrenberg	*				
Paradileptus elephantinus Svec	S			S	
Paramecium sp.	*				
Pelagodileptus trachelioides (Zacharias)	**				
Pelagohalteria viridis (Fromentel)	****				****
Pelagolacrymaria sp.	S				
Pelagostrombidium sp.	**				**
Pelagovorticella natans (Fauré-Fremiet)	*				
Phascolodon vorticella Stein					S
Phialina spp.	*			*	
Rimostrombidium humile (Penard)	*****			***	*****
R. lacustris (Foissner, Skog. & Pratt)	****				**
Scuticociliates unidentified	****			**	***
Spathidium spp.	***			S	S
Spirostomum minus Roux	S				
Stentor coeruleus (Pallas)	S				
Stentor roeseli Ehrenberg	S				
Stentor sp.	S				
Stichotricha aculeata Wrześniowski	S				
Stichotricha secunda Perty	***				
Stokesia vernalis Wenrich	**			**	
Strombidium sp.	S				
Strombidium viride Stein	S				
Teuthophrys trisulca Chatton & De Beu.				S	
Tintinnidium fluviatile (Stein)	*****				***
Tintinnidium sp.	S				
Tintinnopsis cylindrata Kofoid & Campb.	S				
Trichodina pediculus Ehrenberg	*				S
Trithigmostoma sp.	*				
Trochilia minuta (Roux)	S				
Urotricha agilis (Stokes)	S			*	
Urotricha armata Kahl				*	
Urotricha farcta Claparède & Lachmann	****			*	***
Urotricha furcata Schewiakoff	*****			**	***
Urotricha globosa Schewiakoff				*	
Urotricha pelagica Kahl	****				*

(continued)

Table 2 (continued)

Species	GML	KR	HW	DW	SLP
Urotricha sp.	***				*
Vaginicola ingenita (Müller)	**				
Vorticella aquadulcis complex	***			*	****
Vorticella chlorellata Stiller	*				
Vorticella sp. 1	S			S	**
Vorticella sp. 2	S				
Rotifera					
Anuraeopsis fissa (Gosse)	***	***	**	**	****
Ascomorpha agilis Zacharias				S	
Ascomorpha ecaudis Perty	***	**	***	S	**
Ascomorpha ovalis (Bergendal)	***	**	****	*	***
Ascomorpha saltans Bartsch	****	****	****		****
Asplanchna brightwellii Gosse	**	*	*		*
Asplanchna priodonta Gosse	****	****	*****	**	****
Brachionus angularis Gosse	***	*	****	S	**
Brachionus calyciflorus Pallas	***	*	S		S
Brachionus diversicornis (Daday)	*	S			
Brachionus quadridentatus Hermann	S	S	S		S
Brachionus sessilis Varga			**		
Cephalodella auriculata (Müller)	S		*		S
Cephalodella catellina (Müller)	***	*			S
Cephalodella exigua (Gosse)	S			S	
Cephalodella gibba (Ehrenberg)	S	*			
Cephalodella gibboides Wulfert				S	
Cephalodella sterea (Gosse)	S				
Cephalodella ventripes (Dixon-Nuttall)	S				
Collotheca libera (Zacharias)	**		*		S
Collotheca mutabilis (Hudson)	****	****	****	*	****
Collotheca pelagica (Rousselet)	****	***	****	*	**
Colurella adriatica Ehrenberg	*	*			S
Colurella colurus (Ehrenberg)	*	S			S
Colurella obtusa (Gosse)	*	*		S	*
Colurella uncinata (Müller)	*	*	S		S
Colurella tesselata (Glasscott)				*	
Conochiloides dossuarius (Hudson)			S	***	S
Conochiloides natans (Seligo)	***				
Conochilus hippocrepis (Schrank)	***	*	***		*
Conochilus unicornis Rousselet	****	***	****	*	***
Encentrum lupus Wulfert					S
Epiphanes pelagica (Jennings)					S
Euchlanis deflexa Gosse	S				
Euchlanis dilatata Ehrenberg	****	**	S		*

(continued)

Table 2 (continued)

Species	GML	KR	HW	DW	SLP
Filinia brachiata (Rousselet)	S				
Filinia longiseta (Ehrenberg)	****	****	***		***
Filinia terminalis (Plate)	****	*	*****		**
Floscularia ringens (Linnaeus)				S	
Gastropus hyptopus (Ehrenberg)	S	*	**		***
Gastropus stylifer Imhof	****	****	****	*	****
Hexarthra mira (Hudson)				S	S
Kellicottia longispina (Kellicott)	*****	****	*****		****
Keratella cochlearis (Gosse)	*****	*****	*****	**	*****
Keratella hiemalis Carlin	***	*	**		*
Keratella irregularis (Lauterborn)			S		**
Keratella paludosa (Lucks)					**
Keratella quadrata (Müller)	****	****	****	S	****
Keratella serrulata (Ehrenberg)				S	
Keratella testudo (Ehrenberg)	S				
Keratella ticinensis (Callerio)				*	
Keratella valga (Ehrenberg)	S				
Lecane acus (Harring)				***	
Lecane agilis (Bryce)				*	
Lecane bulla (Gosse)	S				*
Lecane closterocerca (Schmarda)	**	*	*	*	S
Lecane elasma Harring & Myers				**	
Lecane flexilis (Gosse)	*		*	*	
Lecane galeata (Bryce)				***	
Lecane gwileti (Tarnogradski)	S				
Lecane hamata (Stokes)	S			S	
Lecane luna (Müller)	**	S	S		S
Lecane lunaris (Ehrenberg)	*	*		**	*
Lecane monostyla (Daday)				S	
Lecane perpusilla (Hauer)				*	
Lecane quadridentata (Ehrenberg)					S
Lecane stichaea (Harring)				*	
Lecane subtilis (Harring & Myers)				*	
Lepadella acuminata (Ehrenberg)				S	
Lepadella cristata (Rousselet)				S	
Lepadella ovalis (Müller)		S			
Lepadella patella (Müller)	**	*	S	*	*
Lepadella triba Myers				*	
Lindia torulosa Dujardin				S	
Lophocharis salpina (Ehrenberg)	*		S		S
Macrochaetus subquadratus Perty			S		
Monommata longiseta (Müller)	S	S			

(continued)

Table 2 (continued)

Species	GML	KR	HW	DW	SLP
Monommata maculata Harring & Myers				S	
Mytilina bisulcata (Lucks)			S	S	
Mytilina mucronata (Müller)	S				
Mytilina ventralis (Ehrenberg)	S				S
Notholca acuminata (Ehrenberg)	*				
Notholca foliacea (Ehrenberg)	**				
Notholca labis Gosse	**				
Notholca squamula (Müller)	**				
Plationus patulus (Müller)	S				S
Ploesoma hudsoni (Imhof)	*		*	S	S
Ploesoma truncatum (Levander)	*				
Polyarthra dolichoptera Idelson	***	**	****		***
Polyarthra euryptera Wierzejski	**	S	*	*	**
Polyarthra longiremis Carlin					S
Polyarthra major Burckhardt	****	****	****	S	****
Polyarthra minor Voigt				*	
Polyarthra remata Skorikov	*****	****	****	****	****
Polyarthra vulgaris Carlin	*****	****	****	**	****
Pompholyx complanata Gosse			S		S
Pompholyx sulcata Hudson	****	****	****	S	***
Proales fallaciosa Wulfert				S	
Proalides tentaculatus De Beauchamp	***	**			S
Scaridium longicaudum (Müller)	S				S
Synchaeta grandis Zacharias	S		S		S
Synchaeta kitina Rousselet	****	****	***		****
Synchaeta lakowitziana Lucks	***				
Synchaeta oblonga Ehrenberg	***	*	S		*
Synchaeta pectinata Ehrenberg	***	**	*	S	**
Synchaeta stylata Wierzejski	**	*			
Synchaeta tremula (Müller)	**	S			
Taphrocampa selenura Gosse					S
Testudinella parva (Ternetz)	S				S
Testudinella patina (Hermann)	S	S			S
Testudinella truncata (Gosse)		S			
Trichocerca brachyura (Gosse)	S				
Trichocerca capucina (Wierz. and Zach.)	****	****	***	*	****
Trichocerca collaris (Rousselet)				S	
Trichocerca cylindrica (Imhof)	***	*	S	**	**
Trichocerca dixon-nuttalli	**		*		
Trichocerca longiseta (Schrank)	S				
Trichocerca porcellus (Gosse)	**	**	S		S
Trichocerca pusilla (Lauterborn)	****	****	*		***

(continued)

Table 2 (continued)

Species	GML	KR	HW	DW	SLP
Trichocerca rattus (Müller)	S	S			S
Trichocerca rousseleti (Voigt)	****	****	***	S	****
Trichocerca similis (Wierzejski)	****	****	****	**	**
Trichocerca simoneae DeSmet				**	
Trichocerca stylata (Gosse)	**	*			
Trichocerca taurocephala (Hauer)	S				
Trichocerca tenuior (Gosse)	S	S			
Trichocerca tigris (Müller)	S				
Trichocerca weberi (Jennings)	S	S			
Trichotria pocillum (Müller)	*				S
Cladocera					
Acroperus harpae (Baird)					*
Alona guttata Sars				*	
Alona quadrangularis (Müller)			S		
Alonella excisa (Fischer)				**	
Alonella nana (Baird)			S	S	*
Bosmina (Bosmina) longirostris (Müller)	***	*	**	**	****
Bosmina (Eubosmina) coregoni Baird	****	*****	***		**
Bosmina (Eubosmina) crassicornis Lillj.	***	***	*		*
Bythotrephes longimanus Leydig	*		S		S
Camptocercus lilljeborgi Schödler					S
Camptocercus rectirostris Schödler				S	
Ceriodaphnia pulchella Sars			S		S
Ceriodaphnia quadrangula (Müller)	***		***	***	***
Chydorus ovalis Kurz				*	S
Chydorus sphaericus (Müller)	****	*****	***	*	*
Daphnia cristata Sars	**	**	*		**
Daphnia cucullata Sars	****	*****	****		****
Daphnia galeata Sars	*		S		
Daphnia hyalina Leydig	***	***	**		*
Daphnia longiremis Sars			S		S
Daphnia longispina (Müller)			**	*	S
Daphnia obtusa Kurz				S	
Diaphanosoma brachyurum (Liévin)	****	****	****	*	***
Disparalona rostrata (Koch)			S	S	
Graptoleberis testudinaria (Fischer)			S		
Leptodora kindtii (Focke)	***	***	**		**
Peracantha truncata (Müller)					S
Phreatalona protzi (Hartwig)	S				S
Pleuroxus aduncus (Jurine)				S	
Pleuroxus striatoides Sramek-Husek				S	
Pleuroxus trigonellus (Müller)	S				

(continued)

Table 2 (continued)

Species	GML	KR	HW	DW	SLP
Polyphemus pediculus (Linnaeus)					S
Pseudochydorus globosus (Baird)					S
Scapholeberis microcephala Sars				S	
Scapholeberis mucronata (Müller)				*	*
Sida crystallina (Müller)			S		*
Simocephalus vetulus (Müller)	S			S	*
Cyclopoida					
Acanthocyclops americanus (Marsh)	***				
Acanthocyclops vernalis (Fischer)		*			
Cyclops abyssorum Sars			S		S
Cyclops scutifer Sars		*	*		*
Cyclops strenuus Fischer			*		
Cyclops vicinus Uljanin			*		S
Diacyclops bicuspidatus (Claus)			S		
Ectocyclops phaleratus (Koch)			**		*
Macrocyclops albidus (Jurine)			S		
Megacyclops viridis (Jurine)			S		S
Mesocyclops leuckarti (Claus)	****	*****	****	**	****
Metacyclops minutus (Claus)			S		
Microcyclops rubellus (Lilljeborg)				**	
Paracyclops fimbriatus (Fischer)			**		
Thermocyclops crassus (Fischer)	S	****		S	*
Thermocyclops oithonoides (Sars)	****	*****	****		****
Calanoida					
Eudiaptomus gracilis (Sars)	**		***	*	**
Eudiaptomus graciloides (Lilljeborg)	****	*****	****	**	***
Eurytemora lacustris (Poppe)	*		*		S
Heterocope appendiculata Sars	S		S		*
Limnocalanus macrurus Sars	*				

Explanations: ***** 100% frequency, **** 75–99%, *** 50–74%, ** 25–49%, * 10–24%, S <10%

lower courses of the River Krutynia were identified. In the upper course, rotifer communities were dominated by detritobacteriophagous species (*Keratella cochlearis*, *Pompholyx sulcata*, and *Conochilus unicornis*). Crustacea were dominated by *Eudiaptomus graciloides*. Biomass of the lake zooplankton of the middle part of the system was dominated by nanophytoplanktivorous species of the rotifer genera *Polyarthra* and *Synchaeta* and filter feeder *Daphnia cucullata*. Nanophytoplanktivorous species of Rotifera play even more significant role in lakes in the lower part of the River Krutynia, but with reference to Crustacea, the dominant were species of the genus *Mesocyclops*. However, river waters did not seem to influence zooplankton structure [12].

3.3 Zooplankton of Harmonic Lakes in the Wigry National Park

Zooplankton of the largest lake in the Wigry National Park, i.e., Lake Wigry, was studied relatively early. The creation of a field station at Lake Wigry in 1920 resulted in the intensive studies of zooplankton in this lake. Already from 1922, the works describing the zooplankton assemblages of Lake Wigry and nearby lakes began to appear [26–33]. The studies revealed that crustacean communities were typical of oligotrophic and mesotrophic lakes with a large share of cold-stenotherm glacial relicts (*Eurytemora lacustris*, *Heterocope appendiculata*, *Daphnia longiremis*). However, the studies covered mostly crustaceans; investigations of Rotifera were focused on psammon communities [34]. Studies of zooplankton carried out by Karabin and Ejsmont-Karabin [35] in 1986 were focused on the assessment of trophic state of Lake Wigry. They revealed relatively high species richness of rotifer and crustacean communities and low (although various at different parts of the lake) trophic status of Lake Wigry. However, the cold-stenotherm glacial relicts were found occasionally in the 1980s. In studies carried out in 1997 on the Hańczańska Bay of the lake, Jabłońska and Paturej [36] noticed strong domination of *Keratella cochlearis* and constancy of occurrence of this species.

First studies of rotifer fauna of harmonic lakes (WNP-H) in the region were carried out in summer 1986, and their results were published 6 years later [37]. Rotifer community of the studied 16 WNP-H lakes was built of 46 taxa, and species indicatory of low trophy occurred in 10 lakes, whereas those typical of the high trophy occurred in 8 lakes. In 1986, generally, the lakes were characterized by relatively low trophy; however, only Lake Białe Wigierskie was mesotrophic [37]. A survey of zooplankton of 14 lakes carried out in 1995 [38] confirmed trophic status of the lakes established 10 years earlier [37].

Last studies of crustacean zooplankton carried out in 2011–2016 indicated an improvement of environmental conditions in Lake Wigry with a high share of glacial relicts in hypolimnion [39]. The abundance of *Eurytemora lacustris* in the deep water of the North Basin of the Lake Wigry was around 8 ind. L^{-1} (Karpowicz and Kalinowska, unpublished), a value which is close to the winter maximum of this species [40]. Feature of Lake Wigry is a large variety of crustacean zooplankton, the occurrence of many genera from this group, and co-occurrence of many species of the same genus. In particular, it deals with genus *Daphnia*, where at one station 3–5 taxa (species or subspecies) were found [41]. In general, in Lake Wigry, six species from the genus *Daphnia* and four species from the genus *Cyclops* were found (Table 2). The crustacean community of the studied 16 harmonic lakes in the Wigry National Park was built of 20 Cladocera species, 12 Cyclopoida species, and 4 Calanoida species (Table 2).

There are no published data on planktonic ciliates of WNP-H lakes. Unpublished results of studies on the ciliate assemblages in Lake Wigry at eight stations conducted by Kalinowska in summer 2002 indicated relatively low ciliate numbers that ranged from 1.9 to 5.7 ind. mL^{-1}. The stations differed in the dominance structure, *Codonella cratera* dominated at three stations, *Strombidium* sp. at other three stations, and *Halteria grandinella* and *Tintinnidium fluviatile* at the remaining two stations, respectively.

3.4 Zooplankton of Dystrophic Lakes in the Wigry National Park

Preliminary studies of zooplankton of dystrophic lakes in the Wigry National Park (WNP-D) were limited to crustaceans and were carried out before World War II at the Hydrobiological Station in Wigry [28, 30, 31, 42].

The first quantitative and qualitative studies of rotifer and crustacean fauna of WNP-D lakes were done in 1986 and published in 1992 [43]. The studies conducted in pelagic waters revealed relatively low number of rotifer species and rotifer densities ranging from 26 to 1,202 ind. L^{-1}. A survey of rotifer fauna inhabiting a nearshore of the lakes gave the markedly higher number of rotifer species [44]. The list of 105 species found in the lakes was dominated by the genera of *Lecane* (22 species) and *Trichocerca* (15 species). Similar results provided the survey of crustacean fauna, which had a markedly higher number of crustacean species and much higher densities in nearshore zones [45]. The studies of the zooplankton in the peat-bog zones surrounding humic lakes revealed high densities of zooplankton and high number of rotifer and crustacean zooplankton (Ejsmont-Karabin and Karpowicz, unpublished).

The recent studies of pelagic crustaceans in dystrophic lakes in the WNP revealed that the communities were composed of the cosmopolitan species (*Ceriodaphnia quadrangula*, *Bosmina longirostris*, *Mesocyclops leuckarti*, *Diaphanosoma brachyurum*), species preferring acidic waters (*Alonella excisa*, *Scapholeberis microcephala*, *Microcyclops rubellus*), and species typical of aquatic vegetation and *Sphagnum* (e.g., *Pleuroxus aduncus*, *Simocephalus vetulus*, *Alona guttata*) (Table 2). The densities of Crustacea in the pelagic zone of humic lakes ranged from 1 to 656 ind. L^{-1}, with the highest abundance in epilimnion (Ejsmont-Karabin and Karpowicz, unpublished). The last study suggests that relatively productive humic lakes do not offer many niches for zooplankton because of the sharp thermal gradient which results in a shallow layer of oxygenated waters [46].

Pioneering studies of ciliates in four dystrophic lakes of the Wigry National Park were performed in 1993–1994 [47]. A total of 65 species of ciliates were found. However, the vast majority of species lived in the mud and *Sphagnum* mat, and only six species were found in the pelagic zone. Pelagic forms were represented by *Askenasia* sp., *Bursaridium pseudobursaria*, and *Paradileptus elephantinus* (found in one lake only, namely, Suchar I), *Teutophrys trisulca* (found in Suchar III), *Stokesia vernalis* (recorded in two lakes – Suchar I and II), and *Urotricha armata* (present in all lakes).

From these years until now, there are no published data on ciliate communities. Studies carried out by Kalinowska (unpublished results) in the pelagial zone of 20 lakes in summer 2002 revealed that ciliate numbers ranged from about 0.1 ind. mL^{-1} in Suchar V and Suchar VII to 175.6 ind. mL^{-1} in Sucharek k/Bryzgla. The high ciliate numbers were also found in lakes Ślepe Zielone (121.6 ind. mL^{-1}), Suchar I (58.6 ind. mL^{-1}), and Suchar Zachodni (49.9 ind. mL^{-1}). In the remaining lakes, the numbers were low and did not exceed 15 ind. mL^{-1}. The dominant taxa were *Halteria*, *Strombidium*, *Strobilidium*, *Cinetochilum*, *Cyclidium*, *Urotricha*, and *Vorticella*.

3.5 Zooplankton of Lakes in the Suwałki Landscape Park

For the most of the lakes in the Suwałki Landscape Park (SLP), the first research of Rotifera was done in summer 1983–1985 by Karabin and Ejsmont-Karabin [48]. Their research covered 25 lakes. Rotifers in the lakes constituted 75–90% of zooplankton density. In total, 51 rotifer species were recognized, with *Keratella cochlearis* present in all of the lakes. Among the most characteristic features of rotifer zooplankton of the lakes was the co-occurrence of many (4–5) species from the genera *Trichocerca* and *Polyarthra.* Two lakes were different than the rest of the lakes of the region, i.e., Lake Hańcza, which was characterized by the lowest trophic state, and Lake Wistuć, the most eutrophic one. Their rotifer fauna was poor in species, and in Lake Hańcza *Conochilus hippocrepis* dominated, whereas in Lake Wistuć the highest densities achieved small detritophage, *Anuraeopsis fissa*.

Studies of rotifer fauna of 17 lakes of SLP carried out in July 2015 showed that there were differences between shallow and deep lakes. Shallow lakes were more differentiated as regards species structure of rotifer fauna, and originality of their rotifer communities was markedly higher. Species richness of rotifer fauna was in the lakes dependent on their trophic status as well as biomass of diatoms and green algae [49]. Long-term studies conducted in the years 1983–1985, 2009, 2012, and 2015 [50] revealed that during 25–32 years, in most lakes of the Suwałki Landscape Park, there were no changes in rotifer communities indicative of low trophy.

As many as 25 species of Cladocera, 8 Cyclopoida, and 4 Calanoida were found in the SLP lakes. The lakes with low trophic status were characterized by low numbers of crustaceans and domination of *Daphnia cucullata* [16]. High species diversity of the pelagic Crustacea gives a very specific value to these lakes, particularly by the occurrence of relict and rare species, e.g., *Eurytemora lacustris* (in Lake Hańcza and Lake Szurpiły), *Heterocope appendiculata* (in Lake Szurpiły and Lake Kameduł), *Bythotrephes longimanus* (in Lake Hańcza), and *Daphnia hyalina* (in Lake Perty). The last study revealed that thermocline zones of unpolluted lakes of SLP were a favorable place for plankton communities. The vertical distribution of large crustacean zooplankton was similar to the distribution of phytoplankton. Especially, *D. cucullata* was strongly related to the phytoplankton distribution and reached maximum densities in deep chlorophyll layers [15].

So far, the only study on the planktonic ciliates conducted in lakes of the Suwałki Landscape Park in summer 2002 is that of Kalinowska (unpublished data) who showed that the numbers of ciliates ranged from 3.7 ind. mL^{-1} in lakes Czarne and Wysokie to 48.5 ind. mL^{-1} in Lake Białe. In most lakes, Oligotrichida, represented by *Strombidium*, *Halteria*, *Rimostrombidium*, and *Tintinnidium*, dominated. Peritrichida were the most numerous only in two lakes (Wiżajny and Małda). Prostomatida (mainly *Coleps* spp.) predominated in Lake Ingiel, while Haptorida (represented mainly by *Askenasia* sp.) predominated in Lake Wysokie.

4 Patterns of Zooplankton Distribution Within and Between Lake Systems

A total of 89 ciliate species were found in three lake systems of northeastern Poland (Table 2). Among them, 32 species had frequencies <10%. Most of these sporadically occurring species were mainly found in the shallow hypertrophic lakes. The most common species (with a frequency of above 75%) for all three systems were *Halteria grandinella*, *Urotricha furcata*, and *Rimostrombidium humile*. In lakes of the GML system, seven ciliate species reached 100% of frequency (*Askenasia volvox*, *Codonella cratera*, *Halteria grandinella*, *Limnostrombidium viride*, *Rimostrombidium humile*, *Tintinnidium fluviatile*, and *Urotricha furcata*), and as many as 34 species occurred sporadically. In lakes of the SLP system, only one species (*Rimostrombidium humile*) was recorded at a frequency of 100%, while ten species had frequencies <10%. In the dystrophic lakes (WNP-D), only two species, namely, *Halteria grandinella* and *Limnostrombidium viride*, reached high frequency (95% and 70%, respectively), seven species occurred sporadically, while others were found in less than 55% of lakes.

In all of the studied lake systems, as many as 129 rotifer species were recorded from pelagic waters (Table 2). However, 47 species were found sporadically. These were mostly littoral species that casually appeared in pelagial. However, even between the littoral species, there were some more often met in pelagial both in harmonic and dystrophic lakes. These were, for example, *Colurella uncinata*, *Lecane luna*, *L. lunaris*, and *Lepadella patella.* Four harmonic lake systems were relatively similar with regard to the most frequent rotifer species. The list of species found in more than 75% in lakes of the systems includes *Ascomorpha saltans*, *Asplanchna priodonta*, *Collotheca mutabilis*, *Gastropus stylifer*, *Kellicottia longispina*, *Keratella quadrata*, *Polyarthra major*, *P. remata*, and *P. vulgaris*. One species, *Keratella cochlearis*, was recorded in all harmonic lakes.

Table 3 Established for lakes of the studied lake systems summer values of mean number of pelagic species of zooplankton

	Ciliata	Rotifera	Crustacea
GML	24.6 ± 2.9	15.8 ± 4.1	11.9 ± 2.5
KR		19.4 ± 5.6	10.6 ± 1.3
WPN-H		18.3 ± 3.4	11.4 ± 3.4
WPN-D	6.4 ± 3.2	9.5 ± 3.8	3.9 ± 1.4
SLP	12.8 ± 2.8	14.6 ± 5.9	9.3 ± 2.8

Abbreviations as in Table 1

Dystrophic lakes in the Wigry National Park had a completely different list of the most frequent rotifer species (Table 2). Only one species, *Polyarthra remata*, occurred in more than 75% of the studied lakes, and three species, *Conochiloides dossuarius*, *Lecane acus*, and *L. galeata*, were met in 50–74% of the lakes.

Pelagic waters of all the studied lakes involved 40 cladoceran and 22 copepod species. However, 19 species were found sporadically (Table 2), and among them there were rare species (e.g., *Camptocercus lilljeborgi*, *Pleuroxus striatoides*, *Scapholeberis microcephala*) and glacial relicts (*Eurytemora lacustris*, *Heterocope appendiculata*, *Limnocalanus macrurus*, *Bythotrephes longimanus*, and *Daphnia longiremis*). Among the glacial relicts, *Limnocalanus macrurus* was found only in the Great Masurian Lakes. The harmonic lake systems (GML, KR, WNP-H, SLP) were relatively similar as regards the most frequent crustacean species, with domination of *Daphnia cucullata*, *Bosmina coregoni*, *Diaphanosoma brachyurum*, *Mesocyclops leuckarti*, and *Thermocyclops oithonoides*. The pelagic crustacean fauna in the dystrophic lakes (WNP-D) revealed some particularities. Beside the cosmopolitan species, there were species preferring acidic waters and species typical to aquatic vegetation and *Sphagnum*. Generally, in the pelagic zone of dystrophic lakes, there were low numbers of crustacean zooplankton. Only two species reached higher numbers in humic lakes: *Bosmina longirostris* (up to 535 ind. L^{-1}) and *Ceriodaphnia quadrangula* (up to 233 ind. L^{-1}).

The mean number of pelagic zooplankton species differed between the studied lake systems (Table 3). However, four harmonic systems showed some similarities in the species richness of rotifers and crustaceans. Thus, the most evident differences were found between harmonic lake systems and a dystrophic one. The highest mean number of ciliates and crustaceans (24.6 ± 2.9 and 11.9 ± 2.5 species lake^{-1}, respectively) was recorded in lakes of the Great Masurian Lakes system. However, the number of crustacean species in lakes of the Krutynia River watershed and harmonic lakes in the Wigry National Park was only slightly lower. In case of rotifers, the highest number (19.4 ± 5.6 species lake^{-1}) was noted in lakes of the Krutynia River watershed. The dystrophic lakes in the Wigry National Park were characterized by the lowest mean number of zooplankton species (6.4 ± 3.2, 9.5 ± 3.8, and 3.9 ± 2.8 species lake^{-1} for ciliates, rotifers, and crustaceans, respectively).

5 Conclusions

A lake may be treated as a patch of the landscape structure associated with other lakes of the system through the processes of transport and exchange of the ecological information in the form of individuals or species [51]. The analyzed groups of lakes in northeastern Poland also reflected this statement.

1. The richest taxonomic composition of ciliates (85 species in total) and rotifers (91 species in total) was found in lakes of the GML system. Crustaceans were richest in species in lakes of both the SLP (37 species) and WNP-H (35 species) systems. Poor taxonomic composition (24 ciliate, 55 rotifer, and 22 crustacean species) was observed in dystrophic lakes (WNP-D). Thirteen ciliate species (i.e., 15%) were common for three lake systems, while twenty-three rotifer (18%) and five crustacean (8%) species were common for all five lake systems.
2. In all of the studied lake systems, the abundance of ciliates, rotifers, and crustaceans differed clearly between lakes of the same system and usually increased with the increasing lake trophy.
3. In all four harmonic systems, the mean number of pelagic zooplankton species was more or less similar. The highest number of very common and frequent (100%) species of ciliates (seven species) was found in the GML system, rotifers (four species) both in the GML and WNP-H systems, and crustaceans (six species) in the KR system.
4. The dystrophic lakes were characterized by a lack of ciliate, rotifer, and crustacean species with frequency of 100%, and only one ciliate (*Halteria grandinella*) and one rotifer (*Polyarthra remata*) species had frequency of 75–99%, indicating that each lake may have a specific species composition.

All of the above facts suggest that both trophic status and the type of connections between lakes may have an effect on the structure of pelagic zooplankton communities and that the role of dispersal processes in determining the species structure of these communities is particularly important in the system of lakes that are connected directly or by short channels as well as connected with rivers. It seems that in the dystrophic lake system, a lack of connections between lakes may restrict the species number of zooplankton communities in addition to the content of poorly available dissolved organic carbon and low pH.

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References

1. Choiński A (1991) Katalog jezior Polski. Część II. Pojezierze Mazurskie (Catalog of Polish lakes. Part II. Mazury lake district). Wyd Nauk UAM, Poznań. (in Polish)

2. Dobrowolski KA, Halba R, Lewandowski K (1998) Survey of wetlands in Poland. In: Dobrowolski KA, Lewandowski K (eds) The strategy of wetland protection in Poland. Institute of Ecology PAS Publ Office, Dziekanów Leśny, pp 7–37
3. Anas MUM, Meegahage BJ, Evans MS, Jeffries DS (2017) Scale-dependent effects of natural environmental gradients, industrial emissions and dispersal processes on zooplankton metacommunity structure: implications for the bioassessment of boreal lakes. Ecol Indic 82:484–494
4. Declerk SAJ, Coronel JS, Legendre P, Brandonck L (2011) Scale dependency of processes structuring metacommunities of cladocerans in temporary pools of High-Andes wetlands. Ecography 34:296–305
5. Cottenie K (2005) Integrating environmental and spatial processes in ecological community dynamics. Ecol Lett 8:1175–1182
6. Heino J (2013) The importance of metacommunity ecology for environmental assessment research in the freshwater realm. Biol Rev Camb Philos Soc 88:166–178
7. Górniak A (2000) Klimat województwa podlaskiego (Climate of the Podlaskie Voivodship). IMGW, Bialystok, pp 1–119. (in Polish)
8. Bajkiewicz-Grabowska E (2008) Obieg wody w systemie Wielkich Jezior Mazurskich (Water circulation in the Great Masurian Lake System). In: Jasser I, Robak S, Zdanowski B (eds) Ochrona i rekultywacja wód Wielkich Jezior Mazurskich narzędziem rozwoju naukowego, gospodarczego, społecznego i kulturowego regionu (Protection and reclamation of water in the Great Masurian Lakes as a tool of scientific, economic social and cultural development of the region). Wyd. IRS, Olsztyn, pp 19–29. (in Polish)
9. Soszka H, Cydzik D, Kudelska D (1979) Ocena stanu czystości Wielkich Jezior Mazurskich (The assessment of water quality in the Great Masurian Lakes). Z W IKŚ Wyd Akcydensowe, Warszawa, pp 1–67. (in Polish)
10. Ejsmont-Karabin J (2013) An analysis based on rotifer indices of the effects of water and sewage management on water quality in the system of interconnected glacial lakes. Limnol Rev 13:191–194
11. Chróst RJ, Siuda W (2006) Microbial production, utilization, and enzymatic degradation of organic matter in the upper trophogenic layer in the pelagial zone of lakes along a eutrophication gradient. Limnol Oceanogr 51:749–762
12. Karabin A, Ejsmont-Karabin J (1996) Struktura, obfitość oraz zróżnicowanie zooplanktonu jezior rzeki Krutyni (Pojezierze Mazurskie) (Structure, abundance and differentiation of the zooplankton of the Krutynia river lakes). In: Hillbricht-Ilkowska A, Wiśniewski RJ (eds) Funkcjonowanie systemów rzeczno-jeziornych w krajobrazie pojeziernym: rzeka Krutynia (Pojezierze Mazurskie). Zesz Nauk Kom Człowiek i Środowisko, vol 13. pp 155–172. (in Polish)
13. Bajkiewicz-Grabowska E, Hillbricht-Ilkowska A, Zdanowski B (1992) Ocena podatności na degradację, stan czystości wód i tempa eutrofizacji jezior (Evaluation of susceptibility to degradation, water quality status and the rate of eutrophication of lakes). In: Zdanowski B (ed) Lakes of Wigry National Park. Trophic status and protection directions. Ossolineum Wrocław-Warszawa-Kraków, Zesz Nauk Człowiek i Środowisko, vol 3:165–189. (in Polish)
14. Drzymulska D, Zieliński P (2013) Developmental changes in the historical and present-day trophic status of brown water lakes. Are humic water bodies a uniform aquatic ecosystem? Wetlands 33:610–919
15. Karpowicz M, Ejsmont-Karabin J (2017) Effect of metalimnetic gradient on phytoplankton and zooplankton (Rotifera, Crustacea) communities in different trophic conditions. Environ Monit Assess 189:367. https://doi.org/10.1007/s10661-017-6055-7
16. Jekatierynczuk-Rudczyk E, Zieliński P, Grabowska M, Ejsmont-Karabin J, Karpowicz M, Więcko A (2014) The trophic status of Suwałki Landscape Park lakes based on selected parameters (NE Poland). Environ Monit Assess 186:5101–5121
17. Bownik-Dylińska L (1975) Ecosystem of the Mikołajskie Lake. Dynamics and biomass of free living planktonic protozoans. Pol Arch Hydrobiol 22:65–72

18. Kalinowska K (2004) Bacteria, nanoflagellates and ciliates as components of the microbial loop in three lakes of different trophic status. Pol J Ecol 52:19–34
19. Chróst RJ, Adamczewski T, Kalinowska K, Skowrońska A (2009) Abundance and structure of microbial loop components (bacteria and protists) in lakes of different trophic status. J Microbiol Biotechnol 19:858–868
20. Kalinowska K, Guśpiel A, Kiersztyn B, Chróst RJ (2013) Factors controlling bacteria and protists in selected Mazurian eutrophic lakes (North-Eastern Poland) during spring. Aquat Biosyst 9:9. https://doi.org/10.1186/2046-9063-9-9
21. Kalinowska K, Grabowska M (2016) Autotrophic and heterotrophic plankton under ice in a eutrophic temperate lake. Hydrobiologia 777:111–118. https://doi.org/10.1007/s10750-016-2769-8
22. Ejsmont-Karabin J (2012) The usefulness of zooplankton as lake ecosystem indicators: rotifer trophic state index. Pol J Ecol 60:339–350
23. Ejsmont-Karabin J, Karabin A (2013) The suitability of zooplankton as lake ecosystem indicators: crustacean trophic state index. Pol J Ecol 61:561–573
24. Karabin A, Ejsmont-Karabin J, Kornatowska R (1997) Eutrophication processes in a shallow, macrophyte-dominated lake - factors influencing zooplankton structure and density in Lake Łuknajno (Poland). Hydrobiologia 342/343:401–409
25. Ejsmont-Karabin J, Karabin A, Kornatowska R, Królikowska J, Ozimek T (1996) The effect of zooplankton on phosphorus cycling in a shallow, hard-water and macrophyte-dominated lake. Ekol Pol 44:259–270
26. Lityński A (1922) Dane ogólne o jeziorach Wigierskich (General data on Wigry lakes). Sprawozdania Stacji Hydrobiologicznej w Wigrach 1:7–14. (in Polish)
27. Lityński A (1925) Próba klasyfikacji biologicznej jezior Suwalszczyzny na zasadzie składu zooplanktonu (An attempt to classify the lakes of the Suwałki region according to the zooplankton composition). Sprawozdanie Stacji Hydrobiologicznej w Wigrach 1:37–56. (in Polish)
28. Lityński A (1925) Uzupełnienie do wykazu wioślarek (Cladocera) znalezionych na terenie Wigierskim (Supplement to the list of cladocerans found in Wigry). Sprawozdanie Stacji Hydrobiologicznej w Wigrach 1:57–58. (in Polish)
29. Bowkiewicz J (1926) O występowaniu wioślarek eupelagicznych (On the occurrence of eupelagic cladocerans). Arch Hydrobiol Ryb 1:255–266. (in Polish)
30. Bowkiewicz J (1934a) Materiały do fauny Cladocera i Copepoda jezior Suwalszczyzny (Materials for the Cladocera and Copepoda fauna of the Suwalszczyzna lakes). Fragm Faun Musei Zool Pol 2:143–146. (in Polish)
31. Bowkiewicz J (1934b) Materiały do planktonu jezior Suwalszczyzny (Materials for plankton of the Suwalszczyzna lakes). Fragm Faun Musei Zool Pol 2:151–156. (in Polish)
32. Bowkiewicz J (1935) *Daphnia (Cophaloxus) longiremis* G.O. Sars nowy dla fauny Polski gatunek wioślarek (*Daphnia (Cophaloxus) longiremis* G.O. Sars a new species of cladocerans for Polish fauna). Fragm Faun Musei Zool Pol 2:243–247. (in Polish)
33. Koźmiński Z (1933) Badania morfometryczne i ekologiczne nad oczlikami (Cyclopidae) z grupy *Strenuus* (Morphometric and ecological studies on cyclopides (Cyclopidae) from the *Strenuus* group). Arch Hydrobiol Ryb 7:59–140
34. Wiszniewski J (1934) Les rotiferes psammiques. Ann Mus Zool Pol 10:339–399
35. Karabin A, Ejsmont-Karabin J (1992) Zooplankton Jeziora Wigry (Zooplankton of Lake Wigry). In: Zdanowski B (ed) Lakes of Wigry National Park. Trophic status and protection directions. Ossolineum Wrocław-Warszawa-Kraków. Zesz Nauk Człowiek i Środowisko, vol 3. pp 101–116. (in Polish)
36. Jabłońska I, Paturej E (1999) The domination and constancy of occurrence of invertebrate communities in the Hańczańska Bay, Lake Wigry, NW Poland. Arch Hydrobiol 41:249–253
37. Leszczyńska-Kowalska J (1992) Zooplankton mezo- i eutroficznych jezior Wigierskiego Parku Narodowego (Zooplankton of meso- and eutrophic lakes of Wigry National Park). In: Zdanowski B (ed) Lakes of Wigry National Park. Trophic status and protection directions.

Ossolineum Wrocław-Warszawa-Kraków. Zesz Nauk Człowiek i Środowisko, vol 3. pp 117–127. (in Polish)
38. Ejsmont-Karabin J, Karabin A (1999) Zespół stratyfikowanych termicznie jezior Wigierskiego Parku Narodowego – próba waloryzacji ekologicznej przy użyciu wskaźników zooplanktonowych (Thermally stratified lakes of Wigry National Park – attempted ecological valorisation using zooplankton indices). In: Zdanowski B, Kamiński M, Martyniak A (eds) Funkcjonowanie i ochrona ekosystemów wodnych na obszarach chronionych. Wyd. IRŚ, Olsztyn-Kortowo, pp 289–302. (in Polish)
39. Karpowicz M, Górniak A (2013) Zooplankton skorupiakowy jezior harmonijnych Wigierskiego Parku Narodowego a trofia wód (Crustacean zooplankton of harmonious lakes in Wigry National Park and relationship with the trophic state). Monitoring Środowiska Przyrodniczego 14:97–101. (in Polish)
40. Błędzki LA, Rybak JI (2016) Freshwater Crustacean Zooplankton of Europe. Cladocera & Copepoda (Calanoida, Cyclopoida) key to species identification. Springer, Basel, p 918
41. Kamiński M (1999) Lake Wigry - the lake "adopted" by International Association of Theoretical and Applied Limnology (SIL "Lake Adoption" Project). Pol J Ecol 47:215–224
42. Minkiewicz S (1924) Dalsze badania na fauną Harpacticidae Jezior Wigierskich (further research of Harpacticidae fauna of Wigry lakes). Spraw Stacji Hydrob Na Wigrach 2 (3):67–90. (in Polish)
43. Tunowski J (1992) Zooplankton jezior dystroficznych Wigierskiego Parku Narodowego (Zooplankton in dystrophic lakes in Wigry National Park). In: Zdanowski B (ed). Lakes of Wigry National Park. Trophic status and protection directions. Ossolineum Wrocław-Warszawa-Kraków. Zesz Nauk Człowiek i Środowisko, vol 3. pp 129–138. (in Polish)
44. Ejsmont-Karabin J (1999) Zespoły Rotifera strefy przybrzeżnej małych, humusowych jezior Wigierskiego Parku Narodowego z zaznaczeniem gatunków nowych i rzadkich w faunie Polski (Rotifer communities of shore zones of small humic lakes in Wigry National Park). In: Zdanowski B, Kamiński M, Martyniak A (eds) Funkcjonowanie i ochrona ekosystemów wodnych na obszarach chronionych. Wyd. IRŚ, Olsztyn-Kortowo, pp 389–403. (in Polish)
45. Karabin A (1999) Zespoły Crustacea strefy przybrzeżnej humusowych jezior Wigierskiego Parku Narodowego (Crustacea communities of shore zones of humic lakes in Wigry National Park). In: Zdanowski B, Kamiński M, Martyniak A (eds) Funkcjonowanie i ochrona ekosystemów wodnych na obszarach chronionych. Wyd. IRŚ, Olsztyn-Kortowo, pp 405–415. (in Polish)
46. Karpowicz M, Ejsmont-Karabin J (2018) Influence of environmental factors on vertical distribution of zooplankton communities in humic lakes. Ann Limnol 54:17. https://doi.org/10.1051/limn/2018004
47. Czapik A, Fyda J (1995) Wstępne badania nad mikro- i meiofauną sucharów Wigierskiego Parku Narodowego (Preliminary studies of micro- and meiofauna of "suchars" in the Wigry National Park). Notatki Faunistyczne-Faunistic Notes. Przegląd Zoologiczny 39:65–72
48. Karabin A, Ejsmont-Karabin J (1994) Zespoły zooplanktonu jezior Suwalskiego Parku Krajobrazowego (Zooplankton communities in the lakes of Suwałki Lanscape Park). In: Hillbricht-Ilkowska A, Wisniewski RJ (eds) Jeziora Suwalskiego Parku Krajobrazowego. Zesz Nauk Komitetu Człowiek i Środowisko. pp 215–242. (in Polish)
49. Ejsmont-Karabin J, Górniak A, Karpowicz M (2016) Diversity of rotifer communities in lakes of the Suwalki Landscape Park. Limnol Rev 16:207–211
50. Ejsmont-Karabin J, Górniak A, Jekatierynczuk-Rudczyk E, Karpowicz M, Więcko A (2016) Long-term changes in the trophic state of Suwalki lakes – an analysis by means of indices based on abundance and composition of their rotifer fauna. Limnol Rev 16:199–205
51. Hillbricht-Ilkowska A (1999) Current challenges and the recommended directions of research in the ecology of freshwaters. Arch Hydrobiol 41:17–27
52. Kufel L (1999) Dimictic versus polymictic masurian lakes: similarities and differences in chlorophyll – nutrients - SD relationships. Hydrobiologia 408/409:1–6

53. Bajkiewicz-Grabowska E, Hillbricht-Ilkowska A, Kajak Z, Kufel L (1989). Charakterystyka fizjograficzna zlewni i limnologiczna większych jezior, ich stan troficzny i czystość wód, podatność na eutrofizację i aktualne zagrożenia (Physiographic characteristics of the catchment and limnology of larger lakes, their trophic status and water quality, susceptibility to eutrophication and current threats). In: Hillbricht-Ilkowska A (ed) Jeziora Mazurskiego Parku Krajobrazowego. Zesz Nauk Komitetu Człowiek i Środowisko, vol 1. pp 45–108. (in Polish)
54. Bajkiewicz-Grabowska E (1996) Odporność jezior systemu rzeczno-jeziornego Krutyni (Pojezierze Mazurskie) i podatność ich zlewni na degradację (Resistance of the Krutynia river-lake system [Mazurian Lakeland] and susceptibility of their drainage basins to degradation). In: Hillbricht-Ilkowska A, Wiśniewski RJ (eds) Funkcjonowanie systemów rzeczno-jeziornych w krajobrazie pojeziernym: rzeka Krutynia (Pojezierze Mazurskie). Zesz Nauk Komitetu Człowiek i Środowisko, vol 13. pp 36–43
55. Bajkiewicz-Grabowska E (1994) Charakterystyka fizyczno-geograficzna i hydrograficzna Suwalskiego Parku Krajobrazowego (Physico-geographical and hydrographical characteristics of Suwałki Landscape Park). In: Hillbricht-Ilkowska A, Wisniewski RJ (eds) Jeziora Suwalskiego Parku Krajobrazowego. Zesz Nauk Komitetu Człowiek i Środowisko, vol 7. pp 15–32. (in Polish)
56. Borowiak D, Nowiński K, Grabowska K (2016) A new bathymetric survey of the Suwałki Landscape Park lakes. Limnol Rev 16:185–197

Aquatic Macroinvertebrate Biodiversity in Freshwaters in Northeastern Poland

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Contents

Abstract Biodiversity is a significant element that describes the ecological state of waterbodies. Eutrophication is a widespread problem that has an impact on water habitats and leads to the succession of sensitive species. Habitat degradation results

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in significant, predictable decreases in taxonomic diversity. We assessed benthic macroinvertebrate community structure (mainly families) in 9 rivers, 9 ponds, and 23 lakes in northeastern Poland. Mollusca, Annelida, Arthropoda, and Nemathelminthes were the 4 phyla represented, and 76 Insecta families, 5 taxa of Crustacea, and 12 Mollusca families were identified. A total of 91 taxa (mainly families) were recorded in all of the waterbodies studied. Diptera, Trichoptera, Ephemeroptera, and Gastropoda were the major components of the benthic macroinvertebrate communities in the aquatic habitats studied. The biodiversity values of the macroinvertebrate assemblages in the rivers and lakes studied were similar. This likely resulted from the similar number of habitats in both types of waters. Differences in biodiversity among the various waterbodies could be evidence of the moderate, diversified anthropogenic pressure to which they are subjected. The analysis of similarities indicated that in terms of the benthic macroinvertebrate communities, the waterbodies studied formed three groups, which, with just one exception, consisted separately of rivers, ponds, and lakes.

Keywords Aquatic macroinvertebrates · Lakes · Ponds · Rivers · Taxonomic diversity

1 Introduction

Lakes dominate the landscape of northeastern Poland. In general, lentic ecosystems encompass a large variety of waterbodies differing in size and ecological processes. The limited economic value of small waterbodies has meant that their biodiversity potential has tended to be overlooked [1], and ponds have been studied far less than lakes, but they are important components of freshwater biodiversity on regional scales [2]. Rivers provide suitable habitats for diverse macroinvertebrate taxa, but the high biodiversity of lotic ecosystems is reduced by human actions in their catchment basins that cause excess nutrient levels through intense agriculture or discharges of point sources of municipal or industrial wastewaters.

The main pressures affecting the integrity of lakes are eutrophication, acidification, and alterations of hydrology and geomorphology [3], and river ecosystems are threatened by industrialization and urbanization processes that alter land use and produce wastes that contribute to growing, long-lasting problems [4]. Habitat degradation results in significant, predictable decreases in taxonomic diversity [5], and the loss of sensitive taxa can have highly adverse effects on ecosystem function if lost species are dominant contributors to ecosystem processes [6]. Structural changes in invertebrate community composition could be a response to stress (increased or decreased environmental factors) [7]. Decreased biological diversity stemming from environmental degradation can be assessed as reduced taxa richness (response of less tolerant species to stressors decreases the persistence or re-establishment of previous densities of taxa in degraded habitats [8]) or decreased diversity index values (loss of sensitive taxa and the strong domination of those able to persist [9]).

One of the main challenges in ecology is to identify biodiversity and determine how it is arranged on different spatial scales [10]. Declines in biodiversity are happening faster in freshwater ecosystems than in terrestrial ones [11].

Aquatic macroinvertebrates are the most commonly used bioindicators of environmental condition in lakes and rivers [12, 13], and macrozoobenthos is one of the ecological components of water ecosystems (in addition to phytoplankton, macrophytes, and phytobenthos) in assessments of the ecological status of waterbodies in the Water Framework Directive [14]. Benthic macroinvertebrates identified to the family level may be sufficient to detect differences for the rapid assessment of water quality or to demonstrate biotic relationships on broader scales [15], while the taxonomic structure is also an important ecological tool used to describe spatial and temporal changes in aquatic environments.

This study aimed to compile an inventory of macroinvertebrate families inhabiting the selected aquatic ecosystems of northeastern Poland. The comparative biodiversity of these waterbodies was assessed in terms of their aquatic macroinvertebrate assemblages.

2 Selected Waterbodies of Northeastern Poland

Data for the present study were collected from 41 waterbodies located across northeastern Poland (Fig. 1). Different types of aquatic ecosystems were studied, including rivers, lakes, and ponds.

2.1 Rivers

Data on macroinvertebrates were collected from sites on nine rivers, of which some were in the Vistula River catchment basin (six rivers) and three rivers were in the Pregoła River catchment basin.

The longest stream among those studied was the Narew river, the length of which in Polish territory is 448 km. This stream is a right-bank tributary of the Vistula River that originates in the Belovezhskaya Forest in western Belarus and flows through eastern Poland to the Vistula River near Warsaw. A part of the river valley between Suraż and Rzędziany forms the Narew National Park, protecting the precious anastomosing stretch [16]. One of its tributaries is the Wkra River, the source of which is located in an area of drained peat bogs near Lake Kownatki. The confluence of this river with the Narew River is near Pomiechówek. The section of the river of the upper course situated between the towns of Nidzica and Szymany has been subjected to revitalization treatments, including stocking with fish to rebuild the ichthyofauna of the river [17]. The valley of the lower course of the Wkra River is

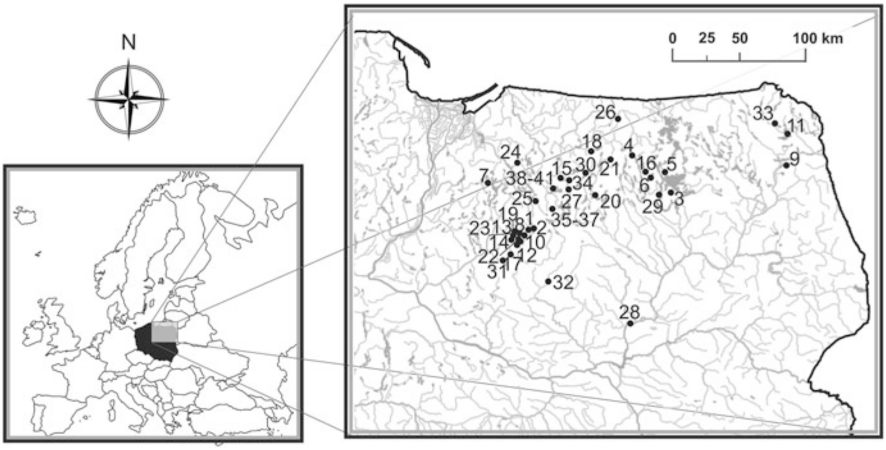

Fig. 1 The study area showing location of river, lake, and pond sampling sites Lakes: 1, Dąbrowa Mała; 2, Dąbrowa Wielka; 3, Jegocin; 4, Kiersztanowskie; 5, Łuknajno; 6, Majcz Wielki; 7, Płaskie; 8, Rumian; 9, Sajno; 10, Tarczyńskie; 11, Wigry; 12, Grądy; 13, Hartowieckie; 14, Kiełpińskie; 15, Kortowskie; 16, Kuc; 17, Lidzbarskie; 18, Ławki; 19, Neliwa; 20, Purdy; 21, Stryjewskie; 22, Zarybinek; 23, Zwiniarz; rivers: 24, Drela; 25, Drwęca; 26, Guber; 27, Łyna; 28, Narew; 29. Nidka; 30, Pisa; 31, Wel; 32, Wkra; ponds: 33, Arkadia; 34, unnamed pond in Olsztyn; 35–37, unnamed ponds in Olsztynek; 38–41, unnamed ponds in Tomaszkowo

included in the European Ecological Natura 2000 Network Programme of protected sites and shares some coverage with the nature reserve "Dolina Wkry".

The Drwęca River is second-order stream and a right-bank tributary of the Vistula River. The main tributary of the Drwęca River that contains lake catchments is the Wel River. Its source is near the village of Drwęck in the moraine hills. The river valleys of Drwęca and its larger tributaries belong to the Natura 2000 Network, and there is an ichthyological nature reserve in the catchment area. Releasing juvenile Atlantic sturgeon (*Acipenser oxyrinchus*) into the Drwęca was initiated in 2006 in an attempt to restore populations of this species to the Baltic Sea, from which it was extirpated many years ago [18].

The Łyna River originates from water-rich sources located near the village of Łyna. It is a second-order stream that flows into the Pregola River that then flows into the Vistula Lagoon. In order to protect backward erosion (retreating slope), which is a very rare phenomenon in the lowlands, and to protect spring area of the Łyna river, the landscape and morphological reserve was established [19]. Near Olsztyn, the river flows through the Las Warmiński reserve. The Łyna River has many tributaries, the largest of which are the right-bank Pisa and Guber rivers, which were also studied.

The remaining streams, such as the Wel, are tributaries of larger rivers, or they flow into lakes. The Nidka and Drela are small rivers; the Nidka flows into lakes Bełdany and Nidzkie, while the Drela flows into Lake Ruda Woda.

2.2 Lakes

The investigation focused on 23 lakes located in northeastern Poland (Fig. 1). Most of the lakes studied were in the Vistula River catchment basin (18 lakes), but 4 lakes were in the Pregoła River catchment basin, and 1 lake was in the Niemen River catchment basin.

According to the typology proposed by Kondracki [20], 21 lakes are located in the macroregion of the Masurian Lake District, while lakes located in the Wel River basin are in the macroregion of the Chełmińsko-Dobrzyńskie Lake District, lakes Wigry and Sajno are in the macroregion of the Lithuanian Lake District, and Lake Płaskie is in the macroregion of the Iława Lake District.

Among all the lakes studied, the largest and deepest is the mesotrophic Lake Wigry with a surface area of 2,115 ha and a maximum depth of 74.2 m. This lake is located in the Wigry National Park and is considered to be the cradle of Polish hydrobiology. In addition to Lake Wigry, ten of the lakes also belong to the group of lakes with surface areas exceeding 125 ha.

Lake Łuknajno has the second largest surface area of 623 ha. It is a shallow meso-eutrophic lake dominated by macrophytes with a 3 m maximum depth [21]. Due to its natural values, it has been protected as a nature reserve since 1937 and a UNESCO Biosphere Reserve since 1977.

Lake Płaskie located in the Iławskie Lake District has the third largest surface area of 620 ha. Part of area of the lake is protected under the European environmental protection network Natura 2000 and the Iława Lake District Landscape Park.

The largest of the lakes located in the Wel River catchment is Lake Dąbrowa Wielka with a surface area of 615 ha and a maximum depth of 34.7 m. The surface areas of lakes Rumian, Dąbrowa Mała, Tarczyńskie, Grądy, and Lidzbarskie range from 100 to 200 ha, while the remaining lakes are smaller than 100 ha (lakes Hartowieckie, Kiełpińskie, Zarybinek, Zwiniarz, and Neliwa). Six of them are located in the protected area of the Wel Landscape Park, while Lake Lidzbarskie is in its buffer zone. In 2006 Neliwa Lake Nature Reserve was designated to protect natural habitats of eutrophic lake and rare, endangered, and vulnerable plants and animal species.

The nine remaining lakes are spread across northeastern Poland (Fig. 1). The largest of these is the eutrophic Lake Sajno located on the Augustów Plain. Its surface area is 522 ha. Lake Jegocin is the deepest lake with a maximum depth of 36.1 m. The surface area of lakes Majcz Wielki, Jegocin, Ławki, and Kiersztanowskie ranges from 100 to 200 ha. Lakes Kuc, Kortowskie, Stryjewskie, and Purdy are smaller than 100 ha.

Agricultural land use in the catchment area exerts a major influence on the water quality of most of the lakes investigated. Tourism and recreation are also developing dynamically in the catchment area, and the investigated lakes are also exploited by inland fisheries [22].

2.3 Ponds

A total of eight ponds in the Masurian Lake District and one in the Lithuanian Lake District were sampled (Fig. 1). The ponds were located in three land use types, typical of European lowland landscapes: floodplain meadows (four ponds), arable agricultural lands (three ponds), and urban environments (two ponds). Four of them are golf course ponds in Tomaszkowo. Three are located near the sewage treatment plant in Olsztynek. We selected two urban ponds for a more detailed study. The ponds in Olsztyn and Suwałki (Arkadia) are used for recreational purposes.

All nine ponds are permanent and contain water year round.

3 Biodiversity of Benthic Macroinvertebrates

A total of 91 taxa (identified mainly to the family level) were recorded from all the waterbodies studied (Tables 1, 2, 3 and 4). Diptera, Oligochaeta, Trichoptera, Ephemeroptera, Bivalvia, and Gastropoda were the major components of the benthic macroinvertebrate communities in the waterbodies studied.

The families Chironomidae and Naididae were noted in almost all (36 of 37) of the waterbodies studied.

3.1 Rivers

The benthic macroinvertebrate assemblages were represented by 78 taxa (mainly families) from the classes Insecta (53), Bivalvia (3), and Gastropoda (8) and the subclass Oligochaeta (2). The number of taxa observed per river ranged from 19 to 55 (Fig. 2). The minimum number of taxa was noted in the Pisa River, and the maximum number of taxa was recorded in the Wel River. The macroinvertebrate taxa richness in the rivers studied averaged 32.

The majority of the insect taxa belonged to the orders Trichoptera (12 families) and Diptera (12) followed by Ephemeroptera (7 families), while Gastropoda were represented by 7 families.

The families Asellidae, Chironomidae, Baetidae, Planorbidae, Erpobdellidae, Naididae, and Hydropsychidae were recorded in all of the rivers investigated.

Shannon diversity index values for each of the rivers ranged from 0.71 to 1.23 (Fig. 3). The minimum value of this index was noted in the Pisa River, while the maximum was noted in the Narew River. The biodiversity index of the macroinvertebrate assemblages in the rivers studied averaged 0.94.

Table 1 Checklist (+ for presence) of families and the higher taxa of aquatic macroinvertebrates identified in rivers within the study period

Order	Family	Drela	Drwęca	Guber	Łyna	Narew	Nidka	Pisa	Wel	Wkra
Bivalvia	Dreissenidae								+	
	Sphaeriidae		+	+	+	+	+	+	+	+
	Unionidae				+	+	+		+	
Coleoptera	Dytiscidae		+	+	+			+	+	
	Elmidae	+								+
	Gyrinidae	+		+	+	+			+	
	Haliplidae		+						+	+
	Limoniidae								+	
Crustacea	Asellidae	+	+	+	+	+	+	+	+	+
	Cambaridae							+		
	Corophiidae					+				
	Gammaridae	+		+	+	+			+	+
Diptera	Athericidae		+	+	+				+	
	Ceratopogonidae	+	+	+	+	+	+		+	
	Chironomidae	+	+	+	+	+	+	+	+	+
	Culicidae	+				+				
	Limoniidae	+	+	+		+	+		+	+
	Psychodidae		+							
	Sciomyzidae		+			+				
	Ptychopteridae		+							
	Rhagionidae		+							
	Simuliidae	+	+	+	+			+	+	+
	Tabanidae	+	+	+		+			+	
	Tipulidae	+	+	+		+				

(continued)

Table 1 (continued)

Order	Family	Drela	Drwęca	Guber	Łyna	Narew	Nidka	Pisa	Wel	Wkra
Ephemeroptera	Baetidae	+	+	+	+	+	+	+	+	+
	Caenidae	+	+		+			+	+	
	Ephemerellidae	+		+		+				
	Ephemeridae		+	+	+		+	+	+	+
	Heptageniidae			+		+			+	
	Leptophlebiidae		+	+	+	+			+	
	Siphlonuridae								+	
Gastropoda	Acroloxidae		+							
	Bithyniidae			+	+	+	+		+	
	Lymnaeidae	+	+	+	+	+	+	+	+	
	Neritidae	+			+				+	+
	Physidae				+	+	+	+	+	
	Planorbidae	+	+	+	+	+	+	+	+	+
	Valvatidae		+	+	+	+	+			
	Viviparidae	+			+	+	+		+	
Heteroptera	Aphelocheiridae			+	+				+	
	Corixidae								+	+
	Nepidae				+	+			+	
	Notonectidae					+			+	
Hirudinea	Erpobdellidae	+	+	+	+	+	+	+	+	+
	Glossiphoniidae	+	+	+	+		+		+	+
	Haemopidae	+	+						+	
	Hirudinidae								+	
	Piscicolidae			+		+	+			
Hydrachnidia								+	+	
Megaloptera	Sialidae		+	+	+		+		+	+
Nematoda					+					

Lepidoptera	Crambidae	+		+	+	+			+	
Odonata	Calopterygidae				+	+			+	+
	Coenagrionidae				+	+		+	+	
	Corduliidae				+			+		
	Gomphidae					+			+	
	Lestidae				+					
	Libellulidae				+		+		+	
	Platycnemididae				+				+	
Oligochaeta	Lumbricidae		+							
	Naididae	+	+	+	+	+	+	+	+	+
Plecoptera	Chloroperlidae								+	
	Leuctridae		+							
	Nemouridae		+	+					+	
	Taeniopterygidae		+							
Trichoptera	Brachycentridae					+			+	
	Goeridae								+	
	Glossosomatidae				+		+	+		
	Hydropsychidae	+	+	+	+	+	+	+	+	+
	Leptoceridae			+		+				
	Limnephilidae	+	+	+	+				+	+
	Molannidae								+	
	Odontoceridae				+					
	Polycentropodidae	+	+	+	+	+			+	
	Psychomyiidae								+	
	Rhyacophilidae				+				+	
	Sericostomatidae		+						+	
Turbellaria					+					+
H′		0.75	0.71	1.04	0.76	1.23	1.12	0.87	1.19	0.80
Evenness		0.53	0.46	0.69	0.46	0.79	0.83	0.68	0.68	0.61

Table 2 Checklist (+ for presence) of families and the higher taxa of aquatic macroinvertebrates identified in lakes >125 ha within the study period

Order	Family	Dąbrowa Mała	Dąbrowa Wielka	Jegocin	Kiersztanowskie	Łuknajno	Majcz Wielki	Płaskie	Rumian	Sajno	Tarczyńskie	Wigry
Bivalvia	Dreissenidae	+	+	+	+		+		+	+	+	+
	Sphaeriidae	+	+	+	+	+	+		+	+	+	+
	Unionidae	+	+		+	+		+	+	+	+	+
Coleoptera	Chrysomelidae					+						+
	Dytiscidae	+								+		
	Elmidae										+	+
	Haliplidae	+	+	+			+					
Crustacea	Asellidae	+	+	+	+	+	+	+	+	+	+	+
	Cambaridae		+									
	Gammaridae						+		+	+		+
Diptera	Ceratopogonidae	+	+	+	+	+	+	+	+	+	+	+
	Chaoboridae	+	+	+	+	+	+	+	+	+	+	+
	Chironomidae	+	+	+	+	+	+	+	+	+	+	+
	Culicidae						+					
	Limoniidae		+	+			+					
	Sciomyzidae		+									
	Tabanidae											+
Ephemeroptera	Baetidae	+	+	+	+		+	+	+	+		+
	Caenidae	+	+	+	+	+	+	+	+	+	+	+
	Ephemeridae						+			+		
	Heptageniidae									+		
	Leptophlebiidae		+	+	+		+			+		
	Siphlonuridae	+				+						
Gastropoda	Acroloxidae	+			+		+		+		+	
	Bithyniidae	+	+	+	+	+	+	+	+	+	+	+
	Hydrobiidae	+			+					+		+
	Lymnaeidae	+	+	+	+		+	+	+	+	+	+
	Neritidae	+	+		+					+		+
	Physidae	+										
	Planorbidae	+	+	+	+		+	+		+		
	Valvatidae	+			+	+		+				
	Viviparidae	+	+	+	+		+	+	+	+		

Heteroptera	Corixidae	+			+				+		+	+
	Nepidae									+		
Hirudinea	Erpobdellidae	+	+	+	+	+	+	+	+	+	+	+
	Glossiphoniidae	+	+	+	+	+	+	+	+	+	+	+
	Piscicolidae								+	+	+	+
Hydrachnidia		+	+	+	+		+	+	+	+	+	
Megaloptera	Sialidae	+	+	+	+	+	+	+	+	+	+	
Nematoda								+				
Lepidoptera	Crambidae				+			+		+		
Odonata	Aeshnidae				+		+					
	Coenagrionidae	+	+	+	+	+	+	+		+	+	
	Corduliidae		+	+			+	+				
	Gomphidae				+				+			
	Lestidae					+						
	Libellulidae		+	+			+					
	Platycnemididae			+	+		+	+	+	+	+	
Oligochaeta	Enchytraeidae								+		+	
	Lumbriculidae	+	+	+	+							+
	Naididae	+	+	+	+	+		+	+	+	+	+
Ostracoda					+		+					
Trichoptera	Ecnomidae			+	+		+	+				+
	Hydroptilidae	+			+	+		+	+		+	
	Leptoceridae	+	+	+	+	+	+	+	+	+	+	+
	Limnephilidae	+	+	+	+		+	+	+	+	+	+
	Molannidae	+	+	+	+		+	+	+	+	+	
	Phryganeidae	+		+	+		+	+		+	+	
	Polycentropodidae	+	+	+	+	+	+	+	+	+		+
	Psychomyiidae				+		+		+	+	+	
Turbellaria		+					+		+	+		
H′		1.07	1.05	1.17	0.97	0.53	0.99	0.86	0.76	1.09	0.49	0.54
Evenness		0.67	0.70	0.78	0.61	0.41	0.63	0.59	0.50	0.70	0.34	0.38

Table 3 Checklist (+ for presence) of families and the higher taxa of aquatic macroinvertebrates identified in lakes <125 ha within the study period

Order	Family	Grądy	Hartowieckie	Kiełpińskie	Kortowskie	Kuc	Lidzbarskie	Ławki	Neliwa	Purdy	Stryjewskie	Zarybinek	Zwiniarz
Bivalvia	Dreissenidae	+	+	+	+	+	+			+	+	+	+
	Sphaeriidae		+	+	+	+	+	+	+	+	+		
	Unionidae	+	+	+	+		+		+	+	+	+	+
Coleoptera	Chrysomelidae				+	+		+					
	Dytiscidae			+		+			+		+		
	Haliplidae			+	+	+		+					
Crustacea	Asellidae	+	+	+	+	+	+	+	+	+	+	+	+
	Gammaridae					+	+		+				+
Diptera	Ceratopogonidae	+	+	+	+	+	+	+	+	+	+	+	+
	Chaoboridae	+	+	+	+	+	+	+	+	+	+	+	+
	Chironomidae	+	+	+	+	+	+	+	+	+	+	+	+
	Culicidae					+							
	Ephydridae	+						+	+				
	Limoniidae			+	+	+						+	
	Sciomyzidae							+					
	Tabanidae			+		+	+					+	
	Tipulidae											+	
Ephemeroptera	Baetidae		+	+	+	+		+	+	+	+		
	Caenidae	+	+	+	+	+	+	+	+	+	+	+	+
	Ephemeridae			+	+					+			
	Leptophlebiidae			+		+		+		+	+		
Gastropoda	Acroloxidae		+		+	+	+			+	+		
	Bithyniidae	+	+	+	+	+	+	+	+	+	+	+	+
	Hydrobiidae			+	+		+	+		+			
	Lymnaeidae		+	+	+	+	+	+	+	+	+	+	+
	Neritidae			+	+		+			+		+	
	Planorbidae			+	+	+		+		+	+		
	Valvatidae	+	+		+	+	+	+	+		+	+	+
	Viviparidae			+	+	+	+		+	+			
Heteroptera	Corixidae	+	+		+		+			+		+	+

Hirudinea	Erpobdellidae	+	+	+	+	+	+	+		+	+	+	+
	Glossiphoniidae	+	+	+	+	+	+	+		+	+	+	+
	Haemopidae												+
	Piscicolidae	+		+		+	+			+	+	+	
Hydrachnidia		+	+	+	+	+	+	+	+	+	+	+	+
Megaloptera	Sialidae			+	+	+	+	+	+	+	+	+	+
Nematomorpha								+			+		
Lepidoptera	Crambidae					+		+					
Neuroptera	Sisyridae										+		
Odonata	Coenagrionidae	+	+	+	+	+	+	+		+	+	+	+
	Corduliidae			+		+			+	+			
	Libellulidae	+	+	+	+	+		+	+	+		+	
	Platycnemididae			+	+		+			+	+		
Oligochaeta	Enchytraeidae		+					+					+
	Lumbricidae			+									
	Lumbriculidae			+		+	+			+	+		
	Naididae	+	+	+	+	+	+	+	+	+	+	+	+
Ostracoda					+			+			+		
Trichoptera	Ecnomidae		+	+	+	+		+		+	+		
	Hydroptilidae	+		+	+			+			+		+
	Leptoceridae	+	+	+	+	+	+	+	+	+	+		+
	Limnephilidae	+	+	+	+	+	+	+	+	+	+	+	+
	Molannidae	+	+	+	+	+	+		+	+		+	+
	Phryganeidae		+	+	+	+		+		+	+	+	
	Polycentropodidae	+	+	+		+			+		+		+
	Psychomyiidae			+		+				+		+	+
Turbellaria					+	+				+		+	
H′		0.49	0.58	0.81	0.88	1.35	0.67	0.62	0.78	1.00	0.70	0.82	0.61
Evenness		0.36	0.40	0.50	0.56	0.83	0.45	0.41	0.56	0.64	0.46	0.56	0.43

Table 4 Checklist (+ for presence) of families and the higher taxa of aquatic macroinvertebrates identified in ponds within the study period

Order	Family	Arkadia pond	Olsztyn pond	Olsztynek pond 1	Olsztynek pond 2	Olsztynek pond 3	Tomaszkowo pond 1	Tomaszkowo pond 2	Tomaszkowo pond 3	Tomaszkowo pond 4
Bivalvia	Dreissenidae	+								
	Sphaeriidae	+						+		
	Unionidae		+				+			
Coleoptera	Chrysomelidae		+							
	Dytiscidae		+				+	+		
	Gyrinidae		+							
Crustacea	Asellidae						+		+	
Diptera	Ceratopogonidae	+	+	+	+	+	+	+	+	+
	Chaoboridae				+		+	+	+	+
	Chironomidae	+	+	+	+	+	+	+	+	
	Culicidae		+							
	Limoniidae	+								
	Tipulidae									+
Ephemeroptera	Baetidae	+	+				+	+	+	+
	Caenidae	+			+	+	+		+	
	Ephemerellidae			+		+				
	Ephemeridae	+								
	Leptophlebiidae	+								
	Siphlonuridae		+							
Gastropoda	Lymnaeidae	+	+					+		+
	Planorbidae		+					+		
Heteroptera	Aphelocheiridae		+							
	Corixidae	+	+				+	+	+	+
	Nepidae		+							
	Notonectidae	+	+				+	+		
Hirudinea	Erpobdellidae	+	+			+	+	+	+	+
	Glossiphoniidae	+	+					+	+	+
Hydrachnidia		+	+		+					
Nematoda		+								
Nematomorpha		+								

Lepidoptera	Crambidae	+				+				
Odonata	Aeshnidae	+								
	Coenagrionidae	+	+							
	Corduliidae								+	
	Gomphidae							+		
	Libellulidae	+	+							+
Oligochaeta	Naididae	+	+	+	+	+	+	+	+	+
Ostracoda		+								
Trichoptera	Leptoceridae	+					+		+	+
	Limnephilidae	+					+	+	+	
	Molannidae	+								
	Phryganeidae						+	+		
	Polycentropodidae	+								
H′		0.58	0.77	0.17	0.35	0.34	0.73	0.79	0.39	0.58
Evenness		0.40	0.58	0.29	0.44	0.40	0.62	0.66	0.35	0.56

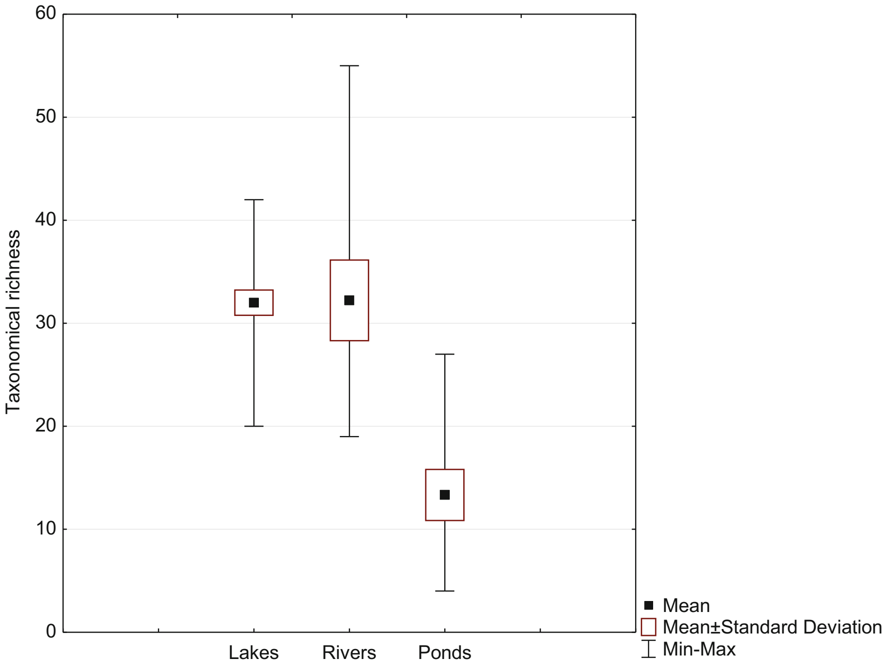

Fig. 2 The values of mean, minimum, and maximum taxon richness (S) recorded in three kinds of studied waterbodies

3.2 Lakes

The benthic macroinvertebrate assemblages were represented by 67 taxa (mainly families) from the classes Insecta (38), Gastropoda (9), and Bivalvia (3) and from the subclasses Hirudinea (4) and Oligochaeta (4). The taxa numbers observed in each lake ranged from 20 to 42 (Fig. 2). The minimum number of taxa was noted in Lake Łuknajno, while the maximum number of taxa was noted in Lake Kuc. The macroinvertebrate taxa richness in the lakes studied averaged 31.

The majority of insect taxa belonged to the orders Diptera (nine families), Trichoptera (eight), and Odonata (seven) followed by Ephemeroptera (six families), while Gastropoda were represented by nine families.

The families Asellidae, Ceratopogonidae, Chaoboridae, Chironomidae, Caenidae, Bithyniidae, Erpobdellidae, Glossiphoniidae, and Limnephilidae were found in all of the lakes investigated. Moreover, in 21 of the lakes studied, water mites (Hydrachnidia of the class Arachnida) were recorded.

Shannon diversity index values in each of the lakes ranged from 0.49 to 1.34 (Fig. 3). The minimum index values were noted in Lake Tarczyńskie and Lake Grądy, and the maximum number of taxa was noted in Lake Kuc. The biodiversity index of macroinvertebrate assemblages in the lakes investigated averaged 0.82.

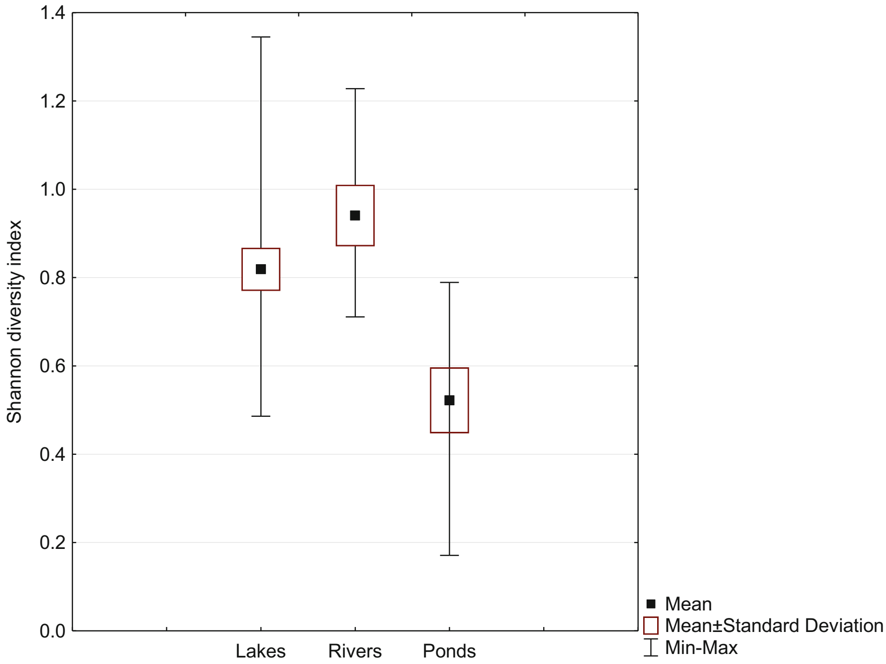

Fig. 3 The values of mean, minimum, and maximum of Shannon diversity index (H′) recorded in three kinds of studied waterbodies

3.3 Ponds

The benthic macroinvertebrate assemblages were represented by 43 taxa (mainly families) from the classes Insecta (32), Bivalvia (3), and Gastropoda (2), and the subclass Hirudinea (4). Taxa numbers observed in each of the ponds ranged from 4 to 27 (Fig. 2). The minimum number of taxa was noted in pond no. 1 in Olsztynek, and the maximum number of taxa was noted in Arkadia pond in Suwałki. The macroinvertebrate taxa richness in the ponds studied averaged 13.

The majority of insect taxa belonged to the orders Diptera (six families), Ephemeroptera (six), and Trichoptera (five) and the suborder Heteroptera (four families), while Bivalvia was represented by three families.

The families Asellidae, Ceratopogonidae, Chaoboridae, Chironomidae, Caenidae, Bithyniidae, Erpobdellidae, Glossiphoniidae, and Limnephilidae were found in all of the ponds investigated. Moreover, in each of ponds studied, water mites (Hydrachnidia of the class Arachnida) were recorded.

Shannon diversity index values in each of the ponds ranged from 0.17 to 0.79 (Fig. 3). The minimum index value was noted in pond no. 1 in Olsztynek, and the maximum number of taxa was found in pond no. 2 in Tomaszkowo. The biodiversity index of the macroinvertebrates assemblages in the ponds investigated averaged 0.52.

4 Macroinvertebrate Communities

The similarity dendrogram indicated that there are three groups of waterbodies and one pond (Arkadia) that were separate from the others (Fig. 4). Ponds, lakes, and rivers were clustered together at a moderate similarity level. Among the groups, the lakes showed greater similarity than the other types of aquatic ecosystems.

4.1 *Nematoda and Nematomorpha*

These organisms belonging to three taxonomic sister groups were confirmed only in some of the waterbodies studied; moreover, they occurred only in small numbers.

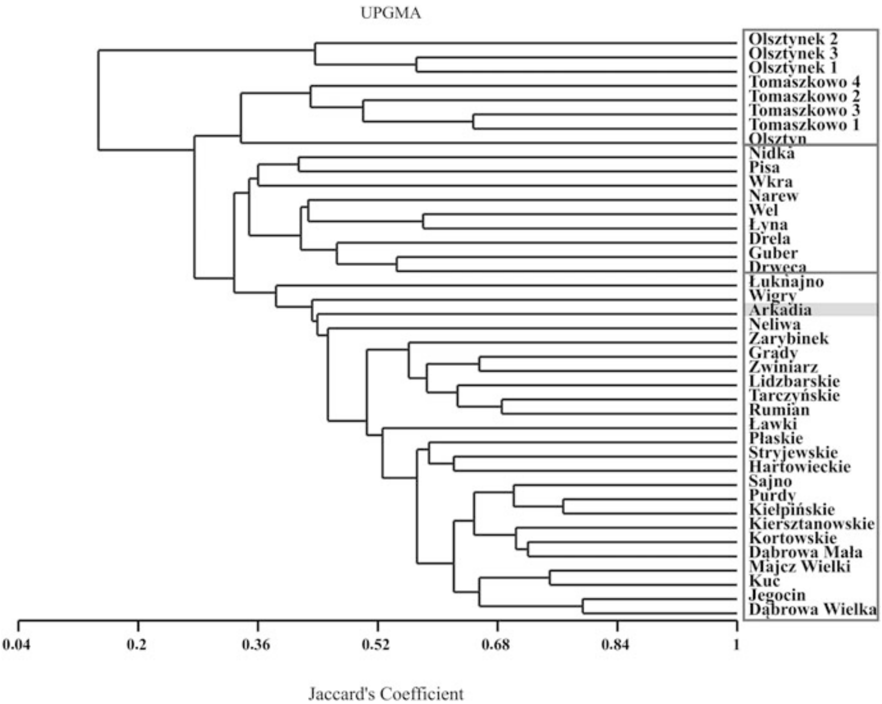

Fig. 4 Dendrogram based on Jaccard similarity index of macroinvertebrate assemblages of studied waterbodies

4.2 Hirudinea

Representatives of the five families occurring in Poland were noted in the waterbodies studied. The representatives of two families, Erpobdellidae and Glossiphoniidae, were particularly widespread. The first of these families includes carnivorous leeches that feed on worms, insects, or molluscs [23], while the second includes leeches that parasitize molluscs, birds, and reptiles.

4.3 Oligochaeta

Oligochaetes from the family Naididae occurred in all the types of waterbodies investigated, which probably stemmed from the fact that the species belonging to this family are well adapted to conditions prevailing on the bottoms of waterbodies of poor ecological status. Some of them show several mechanisms of metabolic adaptations to occupy also saprobial habitats [24]. Additionally, representatives of all the other oligochaete families occurring in Poland that have aquatic forms were noted in the lakes studied.

4.4 Crustacea

Among the crustaceans that occurred in the waterbodies studied, the most common was *Asellus aquaticus*, a representative of the Asellidae family, which inhabited all of the rivers and lakes studied, and it was not noted in only seven of the ponds studied. *Asellus aquaticus* occurs worldwide in temperate parts of the northern hemisphere and is relatively tolerant to a range of pollutants [25]. Representatives of the Gammaridae family with more specific habitat requirements were only noted in some of the rivers and lakes studied, and they were not found in any of the ponds.

4.5 Bivalvia

The occurrence of bivalves was noted primarily in lakes, while representatives of this phylum occurred less frequently in rivers; however, they were noted in some ponds. The invasive species *Dreissena polymorpha* was particularly abundant; the usefulness of this species in controlling eutrophication was investigated over 30 years ago, and it succeeded in removing dissolved nutrients from waters [26]. Small bivalve species belonging to the Sphaeriidae family occurred equally frequently in rivers and lakes. However, despite being described as a typical element of small waterbodies [27], they were rarely recorded in the ponds studied. Large species of bivalves belonging to the Unionidae family occurred mainly in lakes. Freshwater bivalve species provide important ecosystem functions and are known to play a role of ecosystem engineers [28].

4.6 Gastropoda

The malacofauna of the waterbodies studied were represented by nine families, and distinctly more diverse assemblages of snails were noted in the lakes; however, in this respect, similarly abundant assemblages of snails occurred in the Łyna, Wel, Narew, and Nidka rivers (Tables 1, 2 and 3).

4.7 Diptera

Belonging to a very diverse order of the class Insecta, flies have adapted to various freshwater aquatic environments, from the smallest occurring in tree hollows to areas of lake bottoms with depths exceeding 70 m. This order includes taxa unrivalled in environments where chemical water parameters are unsuitable for other aquatic invertebrates, such as the larvae of the families Chaoboridae [29] or Syrphidae. Larvae from the family Chaoboridae were noted in all of the lakes studied and were also recorded in more than half of the ponds. They spend the day in the water column or burrowing into the sediment, migrating at night into the epilimnion to forage on zooplankton [30]. Representatives of the family Chironomidae were recorded in almost all of the studied waterbodies, and they are common inhabitants of most aquatic habitats and often dominate aquatic communities in both abundance and species richness. They occur on all continents, including Antarctica [31].

4.8 Ephemeroptera

Mayflies usually occur in lakes, ponds, streams, and rivers that are fairly clean and well oxygenated [32], and mayfly taxa diversity in still water habitats is generally poor [33]. The number of families noted in the lakes studied did not exceed five; however, the occurrence of representatives of six families was confirmed in the rivers, which was certainly linked to their good ecological status [34].

4.9 Odonata

Dragonflies and damselflies play central role in the conservation of freshwater habitats [35]. Like many European countries, Poland has compiled its own odonate Red List [36]. Larval dragonflies and damselflies appear to be associated with littoral zone and shoreline habitat integrity [37], and in rivers they might be indicators of habitat degradation caused by recreational fishing [38]. The results indicate that there is greater dragonfly and damselfly richness inhabiting rivers than lakes or ponds.

4.10 Trichoptera

Caddisflies are an order rich in families that have developed adaptations to the conditions in which they occur. Representatives of the case-building caddisfly families such as Goeridae and the net-spinning caddisflies such as Hydropsychidae are present in lotic environments, while other families, such as Molannidae and Phryganeidae, are characteristic of lentic environments [39, 40]. Limnephilidae, which is considered the most common caddisfly family, includes cosmopolitan taxa from both environments [41]. The results indicate that the greatest caddisflies richness was noted in the Wel River. Larvae belonging to the Leptoceridae and Limnephilidae family were recorded in almost all of the studied lakes and were also noted in half of the ponds and some of the rivers. Hydropsychidae were noted in all of the rivers studied and were not found in any of the lentic waterbodies.

5 Conclusions

Our study permits us to conclude that the results of taxonomical richness highlight the large contribution many rivers make to macroinvertebrate biodiversity and that these macroinvertebrate communities are distinct compared to those of lentic waterbodies in lowland landscapes. The constant complement of macroinvertebrate assemblages despite waterbody type comprised representatives of cosmopolitan families such as Chironomidae, Naididae, and Asellidae, which are very significant elements of organic matter cycling in freshwater ecosystems. The biodiversity index values obtained for lakes and rivers were similar. The biodiversity and community composition noted in ponds across a range of environments (especially urban) provided vital information.

References

1. Davies BR, Biggs J, Williams PJ, Lee JT, Thompson S (2008) A comparison of the catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic biodiversity in an agricultural landscape. Hydrobiologia 597:7–17
2. Hamerlík L, Svitok M, Novikmec M, Ocadlik M, Bitušík P (2013) Local, among-site, and regional diversity patterns of benthic macroinvertebrates in high altitude waterbodies: do ponds differ from lakes? Hydrobiologia 723:41–52
3. Poikane S, Johnson RK, Sandin L, Schartau AK, Solimini AG, Urbanič G, Arbačiauskas KS, Aroviita J, Gabriels W, Miler O, Pusch MT, Tim H, Böhmer J (2016) Benthic macroinvertebrates in lake ecological assessment: a review of methods, intercalibration and practical recommendations. Sci Total Environ 543:123–134
4. Ding Y, Shan B, Zhao Y (2015) Assessment of river habitat quality in the Hai River Basin, Northern China. Int J Environ Res Public Health 12:11699–11717
5. Carlisle DM, Hawkins CP, Meador MR, Potapova M, Falcone J (2008) Biological assessments of Appalachian streams based on predictive models for fish, macroinvertebrate, and diatom assemblages. J N Am Benthol Soc 27:16–37
6. Vaughn CC (2010) Biodiversity losses and ecosystem function in freshwaters: emerging conclusions and research directions. Bioscience 60:25–35
7. Woodcock TS, Huryn AD (2005) Leaf litter processing and invertebrate assemblages along a pollution gradient in a Maine (USA) headwater stream. Environ Pollut 134:363–375
8. Townsend CR, Hildrew AG (1994) Species traits in relation to a habitat templet for river systems. Freshw Biol 31:265–275
9. Jones FC (2008) Taxonomic sufficiency: the influence of taxonomic resolution on freshwater bioassessments using benthic macroinvertebrates. Environ Rev 16:45–69
10. Richardson DM, Whittaker RJ (2010) Conservation biogeography – foundations, concepts and challenges. Divers Distrib 16:313–320
11. Dudgeon D, Arthington AH, Gessner MO, Kawabata ZI, Knowler DJ, Leveque C, Naiman RJ, Prieur-Richard AH, Soto D, Stiassny MLJ, Sullivan CA (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. Biol Rev 81:163–182
12. Hellawell JM (1986) Biological indicators of freshwater pollution and environmental management. Elsevier Applied Science, London
13. Rosenberg DM, Resh VH (1993) Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York
14. European Commission (2000) Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy
15. Schmidt-Kloiber A, Nijboer RC (2004) The effect of taxonomic resolution on the assessment of ecological water quality classes. Hydrobiologia 516:269–283
16. Marcinkowski P, Giełczewski M, Okruszko T (2018) Where might the hands-off protection strategy of anastomosing rivers lead? A case study of Narew National Park. Pol J Environ Stud 27:2647–2658
17. Goździejewska A, Skrzypczak A, Furgała-Selezniow G, Koszałka J, Mamcarz A (2010) Zooplankton in the Nida River (the upper Wkra river) subjected to revitalization treatments. Pol J Nat Sci 25:387–400
18. Kapusta A, Skóra M, Duda A, Morzuch J, Kolman R (2011) Distribution and growth of juvenile Atlantic sturgeon released into Drwęca and Wisłoka rivers (Poland). Arch Pol Fish 19:69–76
19. Parszuto K, Tandyrak R, Grochowska J, Deleżuch U (2017) The validity of reserve protection of the seepage spring areas of Łyna river for tourism development and preservation of the quality of its water. Folia Turistica 44:63–85
20. Kondracki J (2001) Geografia regionalna Polski. PWN Warszawa, Wydawn
21. Kufel I, Kufel L (1997) Eutrophication processes in a shallow, macrophyte-dominated Lake – nutrient loading to and flow through lake Łuknajno (Poland). Hydrobiologia 342/343:387–394

22. Błachuta J, Pasztaleniec A (2011) Study area. In: Soszka H (ed) Ecological status assessment of the waters in the Wel River catchment. Wydawnictwo IRŚ, Olsztyn, p 320
23. Jeuniaux C (1968) Nutrition and digestion. In: Florkin M, Scheer BT (eds) Chemical zoology. Academic Press, London, p 660
24. Hoffmann H, Hipp E, Sedlmeier UA (1987) Aerobic and anaerobic metabolism of the freshwater oligochaete Tubifex sp. Hydrobiologia 155:157
25. Maltby L (1991) Pollution as a probe of life-history adaptation in *Asellus aquaticus* (Isopoda). Oikos 61:11–18
26. Piesik Z (1983) Biology of *Dreissena polymorpha* (Pall) settling on stylon nets and the role of this mollusc in eliminating the seston and the nutrients from the watercourse. Pol Arch Hydrobiol 30:353–361
27. Piechocki A, Dyduch-Falinowska A (1993) Mięczaki, Małże. Pol Tow Hydrobiol, Fauna słodkowodna Polski, zeszyt 7A, PWN, Warszawa
28. Vaughn CC, Hakenkamp CC (2001) The functional role of burrowing bivalves in freshwater ecosystems. Freshw Biol 46:1431–1446
29. Liljendahl-Nurminen A, Horppila J, Eloranta P, Malinen T, Uusitalo L (2002) The seasonal dynamics and distribution of *Chaoborus flavicans* in adjacent lake basins of different morphometry and degree of eutrophication. Freshw Biol 47:1283–1295
30. Borkent A (1981) The distribution and habitat preferences of the Chaoboridae (Culicomorpha: Diptera) of the holarctic region. Can J Zool 59:122–133
31. Ferrington LC (2008) Global diversity of non-biting midges (Chironomidae; Insecta-Diptera) in freshwater. Hydrobiologia 595:447–455
32. Sowa R (1962) Materiały do poznania Ephemeroptera i Plecoptera w Polsce. Acta Hydrobiol 4:205–224
33. Bauernfeind E, Soldán T (2012) The mayflies of Europe (Ephemeroptera). Apollo Books, Ollerup
34. Soszka H (2011) Ocena stanu ekologicznego wód zlewni rzeki Wel. Wytyczne do zintegrowanej oceny stanu ekologicznego rzek i jezior na potrzeby planów gospodarowania wodami w dorzeczu. Wydawnictwo IRS, Olsztyn
35. Clausnitzer V, Kalkman VJ, Ram M, Collen B, Baillie JEM, Bedjaniç M, Darwall WRT, Dijkstra K-DB, Dow R, Hawking J, Karube H, Malikova E, Paulson D, Schütte K, Suhling F, Villanueva R, von Ellenrieder N, Wilson K (2009) Odonata enter the biodiversity crisis debate: the first global assessment of an insect group. Biol Conserv 142:1864–1869
36. Bernard R, Buczyński P, Tończyk G, Wendzonka J (2009) Atlas rozmieszczenia ważek (Odonata) w Polsce. Bogucki Wydawnictwo Naukowe, Poznań
37. Butler R, de Maynadier P (2008) The significance of aquatic and shoreline habitat integrity to the conservation of lacustrine damselflies (Odonata). J Insect Conserv 12:23–36
38. Müller Z, Jakab T, Toth A, Devai G, Szallassy N, Kiss B, Horvath R (2003) Effect of sports fisherman activities on dragonfly assemblages on a Hungarian river floodplain. Biodivers Conserv 12:167–179
39. Mackay RJ, Wiggins GB (1979) Ecological diversity in Trichoptera. Annu Rev Entomol 24:185–208
40. Wiberg-Larsen P, Brodersen KP, Birkholm S, Grøn PN, Skriver J (2000) Species richness and assemblage structure of Trichoptera in Danish streams. Freshw Biol 43:633–647
41. de Moor FC, Ivanov VD (2008) Global diversity of caddisflies (Trichoptera: Insecta) in freshwater. Hydrobiologia 595:393–407

Biocontamination of the Aquatic Ecosystems of Northeastern Poland

Izabela Jabłońska-Barna and Jacek Koszałka

Contents

Abstract This study provides an overview of the presence of alien species in selected northeastern Polish aquatic ecosystems. The analysis included the western part (Polish) of the Vistula Lagoon, 16 rivers, 12 lakes, and 10 small water bodies. In the study area, a total of 22 non-native taxa were recorded, with the highest frequency of *Dreissena polymorpha*. This species has been present in Polish waters for over 100 years. The largest number of alien species and the highest level of biocontamination were recorded in the benthofauna of the Vistula Lagoon – 18 taxa – which corresponds to "bad" ecological status. The "bad" ecological status was observed both in Vistula river and Lake Lidzbarskie. No taxa of foreign origin were found in the studied small water bodies.

Keywords Lakes · Macrozoobenthos · Non-native species · Ponds · Rivers

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1 Introduction

One of the most important threats to biodiversity are alien species (synonyms: introduced, non-native, nonindigenous, foreign, allochthonous, and exotic species). Introduction of subsequent species is happening in front of our eyes. As a result of these processes, the taxonomic composition of domestic fauna and flora is constantly reorganized. Due to the potential and observed impact of these species on biota indigenous species, alien species are considered as one of the types of water pollution – biocontamination [1, 2]. Frequently the emergence of new, non-native species in ecosystems results in qualitative and quantitative changes in the assemblages of fauna and flora. When species of foreign origin create stable populations in areas beyond their natural range, they can transform the natural environment [3]. They can have a big impact on the structure and functioning of the ecosystem they have entered.

The Convention on Biological Diversity [4] distinguishes among alien species a group that after introducing (resettling) into a new area has a negative impact on native species, habitats, or ecosystems – the so-called invasive species. Potential impacts of these species on the ecosystem include competition for food and space, changes in the habitat, changes in the predator-prey interaction, parasitism, toxicity, and domination in the community (large quantitative changes in the structure of the community) [5].

The most exposed to invasions are large water ecosystems, connected by shipping lanes or channels. In the area of the northeastern Poland (the area located within the administrative borders of the country, covering the right bank of Vistula and Bug basins), such reservoirs are the Vistula Lagoon and the Vistula river. Arbačiauskas et al. [1] showed seven alien species in the benthic macroinvertebrate assemblages of the Vistula river. Jabłońska-Barna et al. [6] in the Polish part of the Vistula Lagoon showed the presence of 12 alien benthic species. In subsequent years of research, the list of non-native species rose by next taxa [5, 7]. It shows that introduction of alien species is a dynamic process that requires constant monitoring but also detailed taxonomic designations. We know very little about the alien species Oligochaeta. This taxonomic group in most studies is determined only as higher systematic units (subclass, family). The lack of historical data on the species composition of zoocenoses and the human impact on their spread causes that a part of taxa is described as cryptogenic species – a species of unknown origin that cannot be described as native or non-native. Their classification is often debatable; therefore some authors treat them as cryptogenic species while others as non-native species. This applies to, among others, species belonging to oligochaete worms *Potamothrix hammoniensis* or *Tubifex blanchardi* (www.iop.krakow.pl/ias; [8]).

When determining the taxonomic composition of alien species within individual ecological formations, it is important to assign individual taxa to ecological groups. In the aquatic ecosystems of our country, *Orconectes limosus* is quite widely spread. It is, like other representatives of Decapoda (e.g., *Rhithropanopeus harrisii*, *Eriocheir sinensis*), often included into necton or necto-benthos, and its presence is overlooked when considering the taxonomic composition of bottom zoocenosis. The following study takes into account the presence of representatives of this systematic group, including them to the macrofauna associated with the bottom of water bodies. To benthic fauna did not include the occurring in the Vistula Lagoon *Cordylophora caspia* (Pallas, 1771) and *Balanus improvisus* (Darwin, 1854), which should be considered as fouling, not benthic species.

The aim of this paper is to review literature and authors' own data on non-indigenous and cryptogenic species introduced to the water bodies of northeastern Poland.

2 Alien Species in the Zoocenoses of Aquatic Ecosystems

In the years 2009–2018, the taxonomic composition of bottom fauna of selected aquatic ecosystems located in the northeast of Poland was studied, including the presence of non-native and cryptogenic species. The research covered the western part (Polish) of the Vistula Lagoon, 16 rivers, 12 lakes, and 10 small water bodies located in Olsztyn and the vicinity (Fig. 1, Tables 1, 2, and 3).

In total, 22 taxa of foreign origin were found in the studied ecosystems (Tables 1, 2, and 3).

The most frequent species was *Dreissena polymorpha*. This species is one of the first Ponto-Caspian taxa that inhabited the Baltic Sea basin [9]. The dispersal of this

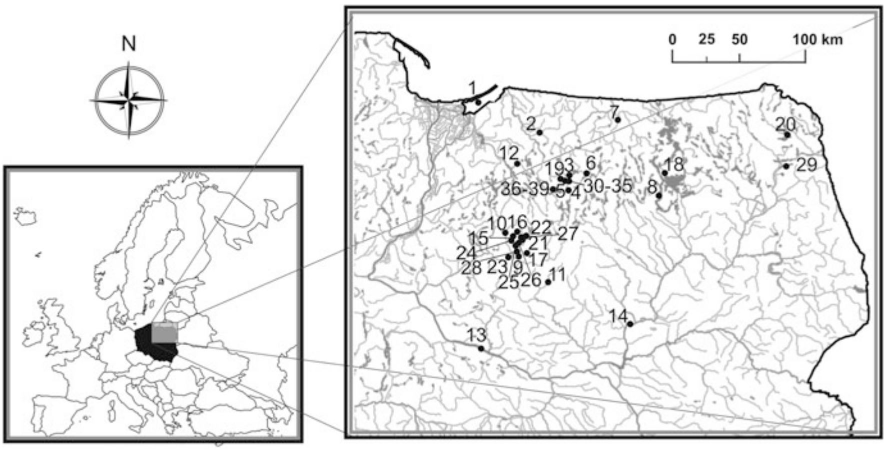

Fig. 1 Research area. Distribution of research stations; surveyed water bodies have been marked with numbers: 1, Vistula Lagoon; rivers: 2, Miłakówka; 3, Wadąg; 4, Łyna; 5, Kortówka; 6, Pisa Warmińska; 7, Guber; 8, Nidka; 9, Wel; 10, Bałwanka; 11, Drela; 12, Wkra; 13, Wisła; 14, Narew; 15, Struga Kiełpińska; 16, Katlewska Struga; 17, Płośniczanka; lakes: 18, Łuknajno; 19, Kortowskie; 20, Wigry; 21, Zarybinek; 22, Rumian; 23, Lidzbarskie; 24, Kiełpińskie; 25, Grądy; 26, Tarczyńskie; 27, Neliwa; 28, Jeleńskie; 29, Sajno; 30–35, small water bodies in the city area (Olsztyn); 36–39, small water bodies near village Naterki

Table 1 Alien species in benthic fauna assemblages of the eastern part of the Vistula Lagoon and the integrated biocontamination index (IBCI)

Non-native taxa	
Dreissena polymorpha Pallas, 1771	+
Potamopyrgus antipodarum (Gray, 1843)	+
Rhithropanopeus harrisii Gould, 1841	+
Eriocheir sinensis Milne-Edwards, 1853	+
Orconectes limosus (Rafinesque, 1817)	+
Marenzelleria sp.	+
Gammarus tigrinus Sexton, 1935	+
Pontogammarus rubustoides (G.O. SARS, 1894)	+
Dikerogammarus haemobaphes (Eichwald, 1841)	+
Obesogammarus crassus (G. O. Sars, 1894)	+
Dikerogammarus villosus (Sowinsky, 1894)	+
Potamothrix moldaviensis (Vejdovský and Mrázek, 1903)	+
Potamothrix vejdovsky Vejdovský and Mrázek, 1903	+
Limnodrilus cervix Brinkhurst, 1963	+
Potamothrix hammoniensis (Michaelsen, 1901)	+
Tubifex blanchardi Vejdovský, 1891	+
Palaemon elegans Rathke, 1837	+
Rangia cuneata (Sowerby I, 1832)	+
IBCI	4

species began at the end of the eighteenth century, at a time when shipping trade became increasingly important and many canals were built to link different navigable river systems in Europe [10]. It is considered as a highly invasive species. It colonizes even very distant areas. An example is the expansion directed to the Great American Lakes [11]. This species is identified by the International Union for Conservation of Nature (IUCN) as 1 of the 100 of the World's Worst Invasive Alien Species [12] and as 1 of the 100 of the Worst Invasive Alien Species for Europe (http://www.europe-aliens.org/speciesTheWorst.do). In the research area, the presence of representatives of this species was found in the Vistula Lagoon, six rivers and nine lakes (Tables 1, 2, and 3).

Non-native species recorded in the aquatic ecosystems of northeastern Poland originate mainly from two areas: from the Ponto-Caspian region and from the waters of North America. The presence of the Ponto-Caspian species in the fauna of Poland and Europe is mainly due to the migration of inland corridors resulting from the construction of sewers connecting the rivers of Europe and Asia [10]. They connected the isolated sub-basins of the Ponto-Caspian region seas with the Baltic Sea catchment. North American species have reached the waters of our country mainly through ballast waters, although some of them have been deliberately introduced into European waters by humans. The species deliberately introduced into European waters is native to North America spiny-cheek crayfish *Orconectes limosus*. It was introduced to Europe in 1890 and up to now is recorded from

Table 2 Alien species in benthic fauna assemblages of rivers and the site-specific (SBCI) or integrated biocontamination index (IBCI)

Non-native taxa	Rivers															
	Miłakówka	Wadąg	Łyna	Kortówka	Pisa Warmińska	Guber	Nidka	Wel	Bałwanka	Drela	Wkra	Wisła	Narew	Struga Kiełpińska	Katlewska Struga	Płośniczanka
Dreissena polymorpha Pallas, 1771	+	+	+	+				+				+				
Potamopyrgus antipodarum (Gray, 1843)			+									+				
Chelicorophium curvispinum (G.O. Sars, 1895)													+			
Orconectes limosus (Rafinesque, 1817)			+			+						+				
Dikerogammarus haemobaphes (Eichwald, 1841)												+				
Pontogammarus rubustoides (G.O. Sars, 1894)												+				
Potamothrix heuscheri Bretscher, 1900											+					
Potamothrix hammoniensis (Michaelsen, 1901)			+								+	+				
Branchiura sowerbyi Beddard, 1892												+				
Chaetogammarus ischnus (Stebbing, 1898)												+				
SBCI/IBCI	1	1	3	1	1	0	0	1	0	0	1	4	1	0	0	0

Table 3 Alien and cryptogenic species in benthic fauna assemblages of lakes and the site-specific (SBCI) or integrated biocontamination index (IBCI)

Non-native taxa	Lakes											
	Łuknajno	Kortowskie	Wigry	Zarybinek	Rumian	Lidzbarskie	Kiełpińskie	Grądy	Tarczyńskie	Neliwa	Jeleńskie	Sajno
Dreissena polymorpha Pallas, 1771		+	+	+	+	+	+	+	+	+		
Potamopyrgus antipodarum (Gray, 1843)			+			+	+					
Potamothrix hammoniensis (Michaelsen, 1901)				+	+	+	+	+	+			
Potamothrix heuscheri Bretscher, 1900					+	+	+	+	+			
Orconectes limosus (Rafinesque, 1817)		+			+	+						
SBCI/IBCI	0	1	1	1	3	4	3	2	2	1	0	0

22 territories [13]. *O. limosus* has many features that enhance its survival, giving it an advantage over native species [14, 15]. The reason of colonization success and at the same time the huge threat for native biota is the resistance to crayfish plague, high fertility and tolerance for significant levels of water pollution. Unlike European crayfishes, some populations of spiny-cheek crayfish mate twice a year, in autumn and spring [15]. In the analyzed area, the presence of *O. limosus* was found in the Vistula Lagoon, three lakes and three rivers (Tables 1, 2, and 3), including the Vistula river. It is also worth noting that we did not find it on the studied section of the Narew river, despite the fact that in the literature there was a confirmed presence of *O. limosus* in the Narew basin. However, it should be assumed that occurrence of this species in the studied area is wider. In the area of Warmia, Mazury, and Mazovia, *O. limosus* is a common species [16, 17].

The other species introduced intentionally into the waters of continental Europe is the North American *Gammarus tigrinus*. It was introduced into British waters from the offloaded ballast waters of vessels travelling between North America and Europe [18]. In 1957, *G. tigrinus* was deliberately introduced to German [19] and then Dutch rivers, from where it began migration along the southern shores of the Baltic Sea [20]. The research carried out for this work showed the presence of representatives of this expansive newcomer, just like the *Palaemon elegans* and *Obesogammarus crassus*, only in the area of the Vistula Lagoon. In the period of research, *G. tigrinus* was dominant in the middle part of lagoon and the remaining taxa in the coastal zone (A. Rychter and I. Jabłońska-Barna, unpublished data). In addition to the North American *G. tigrinus* in the Vistula Lagoon, four other species of gammarids with Ponto-Caspian origin were found (Table 1). One of them is *Pontogammarus robustoides*. This amphipod is relatively large and aggressive, effectively hunting for other small invertebrate animals. As an eurytopic and omnivorous species, similarly to *Dikerogammarus haemobaphes* and *D. villosus*, it can be a competitor and a threat to the local benthofauna, also for native Amphipoda species. Their influence on local shellfish fauna communities is best illustrated by their English-language names: "demon shrimp" – *Dikerogammarus haemobaphes* – and "killer shrimp" *Dikerogammarus villosus*. *D. villosus* is 1 of the 100 of the Worst Invasive Alien Species for Europe (http://www.europe-aliens.org/speciesTheWorst.do).

It should be underlined that despite its expansiveness and the presence of *D. haemobaphes* in the channel connecting the lakes Śniardwy and Łuknajno [21], during the research to this study, it was not found in Lake Łuknajno. Its presence on the Narew section was also not observed. In Narew and Vistula, the presence of other invaders was confirmed – corophioid amphipod species of Ponto-Caspian origin – *Chelicorophium curvispinum*. The checklist of non-native Crustacea closing Ponto-Caspian *Obesogammarus crassus* originated from the Atlantic shore of North America – *Rhithropanopeus harrisii* – and originated from Southeastern Asia, *Eriocheir sinen*sis. All these species were recorded only in the Vistula Lagoon. Despite that we know that the occurrence of Chinese mitten crab *E. sinensis* is wider. Its number periodically increases in the Baltic Sea. In the years when the population density increases, the number of its quotations is also growing outside the area of brackish water. Chinese mitten crab, due to its high environmental tolerance, is able to acclimate and occurs in salt, brackish, and fresh waters. In inland waters it was found in the Vistula river near Włocławek, in Warta and Drwęca rivers and Lake Wydminy [22]. A representative of this species was also found in the Lake Wulpińskie located near Olsztyn (Jabłońska-Barna and Palińska-Żarska, unpublished) which indicates the constantly increasing range of the species in the waters of Poland. This species is identified by the International Union for Conservation of Nature as 1 of the 100 of the World's Worst Invasive Alien Species [12] and as 1 of the 100 of the Worst Invasive Alien Species for Europe (http://www.europe-aliens.org/speciesTheWorst.do).

In many cases the emergence of new species from distant geographical areas in benthofauna is related to the possibility of overcoming geographical barriers. The worldwide intensification of ship traffic has become the main factor in the introduction of species from remote geographical regions. The organisms are transferred beyond their natural ranges, to distant places mainly through ballast waters. The species that appeared in the waters of Europe using this vector is among others *Potamopyrgus antipodarum*. In Europe, the presence of representatives of this species was found for the first time at the Thames estuary around 1859 [23]. On the European continent, it appeared in 1887 in the Wismar Bay and in inland waters in 1916 [24]. Currently, it appears in almost all of Europe. In surface waters, this snail has a very large spreading potential. The presence of the operculum and the hard shell allows it to pass through the digestive tract of fish, and more importantly it is well adapted to passive spreading on birds (found on the ducks' beaks, it is resistant to drying out) [25–27]. The most important, however, is the fact that these animals reproduce parthenogenetically and under favorable conditions. The beginning of a new population (colonizing another water body) can be initiated by a single individual. Representatives of this species in investigated water bodies were found as well at the bottom of brackish water of the Vistula Lagoon, as in flowing and standing fresh waters.

Maritime transport is the pathway of introduction of two "youngest" arrivals from North America – *Limnodrilus cervix* and *Rangia cuneata*. Both species were found in the Vistula Lagoon at the same time [6, 28]; however, *R. cuneata* in the western part of the lagoon (territorially belonging to Poland) reached a year later [29]. Since then *R. cuneata* has been systematically expanding its range and its share in abundance and biomass of benthofauna [7, 29, 30]. There is only one report on the presence of *Limnodrilus cervix* in the waters of our country and on southern coast of the Baltic Sea [6]. There is no information on the possible spread of representatives of this species in the waters of the Vistula Lagoon and adjacent water ecosystems. An important reason is the lack of accurate taxonomic designations. In many studies, representatives of this systematic group are identified to the level of a subclass or families. Among worms of the subclass Oligochaeta recorded in the studied lake ecosystems, attention should be paid to the occurrence of *Potamothrix hammoniensis*. This species has been found in the profundal zone of several eutrophic lakes: Rumian, Zarybinek, Grądy, Tarczyńskie, Lidzbarskie, and Kiełpińskie

(Table 3). Literature data indicate the interaction of this species with representatives of *Tubifex tubifex* [31] – *Potamothrix hammoniensis* displaces *T. tubifex*. In the case of the lakes studied, there is no historical data confirming this trend; however, only representatives of *P. hammoniensis* were found within the studied zoocenoses [32]. It should also be emphasized that both species coexist in the zoocenosis of the Vistula Lagoon bottom [6], Włocławek Dam Reservoir [33] and river Łyna (Table 2). It should also be noted that *P. hammoniensis*, similarly to *Tubifex blanchardi*, is identified as cryptogenic species (www.iop.krakow.pl/ias). Detailed taxonomic identification of Oligochaeta carried out in the discussed aquatic ecosystems revealed the presence of other three species of Ponto-Caspian origin and one of Asian origin. These are *Potamothrix heuscheri*, *P. moldaviensis*, *P. vejdovsky*, and *Branchiura sowerbyi*. The last of these species was noted only in the Vistula river – in Włocławek Dam Reservoir (Jabłońska-Barna and Hliwa, unpublished) – and according to the authors' knowledge, it is the first report about the presence of this species in the Vistula. The checklist of Annelida alien species closes with spionid polychaetes of the genus *Marenzelleria*. The first *Marenzelleria* observation in the Vistula Lagoon was at the end of the 1980s of the last century [34]. At the beginning of this century, Sikorski and Bick [35] described five species, four of which occur in brackish areas and the species found in the Baltic represents *M. neglecta* [36]. Blank et al. [37] show that there are present three species in the Baltic and in the south part *M. neglecta* occurs together with *M. viridis*, but in Delivering Alien Invasive Species Inventories for Europe, *M. viridis* is described as a synonym for *M. neglecta*. Despite the fact that in the base of alien species in Poland (www.iop.krakow.pl/ias) there are listed three species from genus *Marenzelleria* occurring in the Polish zone of the Baltic Sea, in this study it has been adopted that there is one species in the Vistula Lagoon – *M. neglecta*. This species is identified as 1 of the 100 of the Worst Invasive Alien Species for Europe (http://www.europe-aliens.org/speciesTheWorst.do). However, there is no evidence of negative impact of this taxon on native biocenosis in Polish waters [38].

3 Level of Biocontamination

The largest number of non-native species was observed in the zoocenosis of the Vistula Lagoon bottom – 18 (Table 1). This water reservoir as a part the Baltic Sea is a geologically young postglacial basin. The native fauna and flora concentrations display a relatively low degree of biodiversity with a marked dominance of a few species [7]. Therefore currently, a wide-scale biological reorganization of this reservoir and the whole Baltic Sea is being witnessed. It results primarily from human activity and manifests itself in invasion of new species. Until recently, new species came mainly from the Ponto-Caspian region by channels connecting the sea basins. Nowadays the principal vectors for the invasion to this reservoir seem to be marine transport. The most numerous group of alien species among the macrozoobenthos of the Vistula Lagoon are Malacostraca, with the dominance of gammarids. Four species recorded in Vistula Lagoon are identified as 1 of the 100 of the Worst Invasive Alien Species for Europe (http://www.europe-aliens.org/speciesTheWorst.do).

In total, the presence of ten alien species was found within the studied watercourses (Table 2). The most numerous group, as in the case of the Vistula Lagoon, were Malacostraca. The largest number of alien species was recorded in the Vistula river – eight. Our list does not fully match the markings of other authors. Probably in the middle section of the watercourse, there are present more alien species that were shown by Arbačiauskas et al. [1] and Dumnicka and Poznańska [33]. The presence of such a large number of species from other geographical areas is mainly the result of two components: the location of the river within the central migration corridor from the Ponto-Caspian region [10] and direct connection with the Baltic Sea, where a significant number of taxa originate from remote geographical regions via shipping [7]. The number of non-native species in the other tested watercourses is much lower, and in 7 out of 16 tested watercourses (44%), no foreign species were found (Table 2).

In the studied lakes, the largest number of alien species was found in the eutrophic lake Lidzbarskie – five. There were present representatives of all taxa of foreign origin found in the studied lake ecosystems. Only in three of the studied group of lakes (23%), no species of macrozoobenthos of foreign origin were found (Table 3). Among the studied water ecosystems, small water bodies stand out, in which representatives of benthic alien species were not found at all. It should be added that in these water reservoirs there were representatives of rare and protected species ([39]; authors' own data, unpublished) which emphasizes the role of this type of water bodies in preserving biodiversity.

On the basis of the number of alien species and quantitative relations of macrozoobenthos foreign origin, five classes of biocontamination for simple sampling site (SBCI – site-specific biocontamination index) or for a few sampling sites in one water body (IBCI – integrated biocontamination index) that corresponded directly to five ecological quality classes in the sense of Water Framework Directive [40] can be estimated [1]. The site-specific biocontamination index (SBCI) of each locality/sampling site was determined from the proportion of nonindigenous species in the taxonomic composition (ratio of the number of alien taxonomic orders to the total number of orders in the community) and in the total abundance of the macrozoobenthos assemblage (ratio of the abundance of alien taxa specimens to the total abundance of the assemblage) [1, 2]. The highest level of biocontamination (IBCI = 4, "bad" ecological status) is observed in the Vistula Lagoon and in the Vistula river (Tables 1 and 2). It is consistent with the observations of the other authors cited above [1, 5]. Within the investigated rivers, high biocontamination was observed only in the Łyna (IBCI = 3, "poor" ecological status). The remaining watercourses were characterized by "high" and "good" ecological status. No foreign species were found in seven rivers (Table 2). Among the studied lakes, Lidzbarskie indicated severe biological contamination (IBCI = 4, "bad" ecological status) and Kiełpińskie and Rumian high biological contamination (IBCI = 3, "poor" ecological status) (Table 3). Only in three of the studied lakes, no presence of alien species was observed. As mentioned above, all the examined small water bodies were characterized by the "high" ecological status.

4 Conclusions

The level of biocontamination of the studied aquatic ecosystems is varied. As expected, the largest number of alien species is observed in the Vistula Lagoon and the Vistula river. Referring to the data found in literature in both water bodies, a continuous increase in the number of species from distant geographical areas is observed. The considerable number and abundance of non-native species testify to the high level of biopollution of these water reservoirs. In the studied aquatic ecosystems, four taxa rated among the 100 most invasive alien species in Europe were found. Alien species were more often found in lakes than rivers. It should be emphasized that in the area of the examined small water bodies no presence of non-native species was observed. Nevertheless, the emergence of new species is a dynamic process that requires constant observation.

References

1. Arbačiauskas K, Semenchenko V, Grabowski M et al (2008) Assessment of biocontamination of benthic macroinvertebrate communities in European inland waterways. Aquat Invasions 3(2):211–230
2. Panov VE, Alexandrov B, Arbačiauskas K et al (2009) Assessing the risks of aquatic species invasions via European inland waterways: from concepts to environmental indicators. Integr Environ Assess Manag 5(1):110–126
3. Olenin S, Alemany F, Cardoso AC et al (2010) Marine Strategy Framework Directive, Task Group 2 Report – non-indigenous species (April 2010). Joint Report ICES and JRC European Commission EUR 24342 – EN 2010
4. Convention on Biological Diversity (2002) https://www.cbd.int/convention/articles
5. Rychter A, Dobrzycka-Krahel A, Jabłońska-Barna I (2018) Gatunki obce (Alien species). In: Bolałek J (ed) Zalew Wiślany na początku XXI wieku (The Vistula Lagoon at the beginning of the 21st century). PWN, Warsaw, pp 312–326
6. Jabłońska-Barna I, Rychter A, Kruk M (2013) Biocontamination of the western Vistula Lagoon (south-eastern Baltic Sea, Poland). Oceanologia 55(3):751–763
7. Dumnicka E, Jabłońska-Barna I, Rychter A (2014) First record of a new alien species Limnodrilus cervix Brinkhurst, 1963 (Annelida, Clitellata) in the Vistula Lagoon (Southern Baltic Sea). Oceanologia 56(1):151–158
8. Dumnicka E (2016) Alien Naididae species (Annelida: Clitellata) and their role in aquatic habitats in Poland. Biologia 71(1):16–23
9. Orlova MI, Panov VE (2004) Establishment of zebra mussel, Dreissena polymorpha Pallas, in the Neva Estuary (Gulf of Finland, Baltic Sea): distribution, population structure and possible impact on local unionid bivalves. Hydrobiologia 514:207–217

10. Bij de Vaate A, Jażdżewski K, Ketelears HAM et al (2002) Geographical patterns in range extension of Ponto-Caspian macroinvertebrate species in Europe. Can J Fish Aquat Sci 59(7):1159–1174
11. Olenin S, Orlova M, Minchin D (1999) *Dreissena polymorpha* (Pallas, 1771). In: Gollasch S, Minchin D, Rosenthal H, Voigt M (eds) Case histories on introduced species: their general biology, distribution, range expansion and impact. Logos-Verlag, Berlin, pp 37–42
12. Lowe SJ, Browne M, Boudjelas S (2000) 100 of the world's worst invasive alien species. IUCN/SSC Invasive Species Specialist Group (ISSG), Auckland. http://www.issg.org/pdf/publications/worst_100/english_100_worst.pdf
13. Kouba A, Petrusek A, Kozák P (2014) Continental-wide distribution of crayfish species in Europe: update and maps. Knowl Manag Aquat Ecosyst 413:5
14. Schulz R, Śmietana P (2001) Occurrence of native and introduced crayfish in Northeastern Germany and Northwestern Poland. Bull Fr Peche Piscic 361:629–641
15. Aklehnovich A, Razlutskij V (2013) Distribution and spread of spiny-cheek crayfish Orconectes limosus (Rafinesque, 1817) in Belarus. BioInvasions Rec 2(3):221–225
16. Krzywosz T, Białokoz W, Chybowski Ł (1995) Rak pręgowaty w wodach województwa suwalskiego (Spinycheek crayfish in the waters of the Suwalki Voivodeship). Komunikaty Rybackie 2:8–11
17. Strużyński W, Śmietana P (1999) On the distribution of crayfish in Poland. Freshwater Crayfish 12:825–829
18. Hynes HBN (1955) Distribution of some freshwater Amphipoda in Britain. Int Assoc Theor Appl Limnol Proc 12:620–628
19. Ruoff K (1968) Experimentelle Untersuchungen über den in die Weser eingebürgerten amerikanischen Bachflohkrebs *Gammarus tigrinus* Sexton. Arch Fisch 19(2–3):134–158
20. Nijsen H, Stock JH (1966) The amphipod *Gammarus tigrinus* Sexton, 1939, introduced in the Netherlands (Crustacea). Beaufortia 13(160):197–206
21. Jażdżewski K (2003) An invasive Ponto-Caspian amphipod *Dikerogammarus haemobaphes* (Eichwald, 1841) conquers Great Masurian Lakes, north-eastern Poland. Fragmenta faunistica 46:19–25
22. Grabda E (1973) Krab wełnistoszczypcy, *Eriocheir sinensis* Milne-Edwards, 1853 w Polsce (Chinese mitten crab, *Eriocheir sinensis* Milne-Edwards, 1853 in Poland). Przegląd Zoologiczny 17:46–49
23. Boycott AC (1936) The habitats of freshwater Mollusca in Britain. J Anim Ecol 5:111–186
24. Jaeckel SGA (1962) Ergänzungen und Berichtigungen zum rezenten und quartären Vorkommen der mitteleuropäischen Mollusken. In: Brohmer, Ehrman, Ulmer, Die Tierwelt Mitteleuropas 2(1):25–294
25. Bondensen P, Kaiser E (1949) Hydrobia (Potamopyrgus) jenkinsi (Smith) in Denmark illustrated by its ecology. Oikos 1:252–281
26. Haynes AB, Taylor B, Varley M (1985) The influence of the mobility of *Potamopyrgus jenkinsi* (E.A. Smith) (Prosobranchia: Hydrobiidae) on its spread. Arch Hydrobiol 103:497–508
27. Økland J (1990) Lakes and snails. Universal Book Services, Oegstgeest, 516 pp
28. Rudinskaya LV, Gusev AA (2012) Invasion of the North American wedge clam *Rangia cuneata* (G.B. Sowerby I, 1831) (Bivalvia: Mactridae) in the Vistula Lagoon of the Baltic Sea. Russ J Biol Invasions 3(3):220–229
29. Warzocha J, Drgas A (2013) The alien gulf clam (Rangia cuneata G. B. Sowerby I, 1831) (Mollusca: Bivalvia: Mactridae) in the Polish part of the Vistula Lagoon (SE Baltic). Folia Malacol 21(4):291–292
30. Warzocha J, Szymanek L, Witalis B, Wodzinowski T (2016) The first report on the establishment and spread of the alien clam *Rangia cuneate* (Mactridae) in the Polish part of the Vistula Lagoon (southern Baltic). Oceanologia 58(1):54–58
31. Tarnao T (1996) *Tubifex tubifex* (Müller, 1774) (Oligochaeta, tubificidae) in the profundal of Estonian lakes. Internationale Revue der gesamten Hydrobiologie und Hydrographie 81(4):589–596

32. Koprowska L, Koszałka J, Bielecki A (2016) Makrobezkręgowce wybranych ekosystemów wodnych Welskiego Parku Krajobrazowego (Macroinvertebrates of selected water ecosystems of Welski Landscape Park). In: Hołdyński CZ (ed) Welski Park Krajobrazowy. Monografia przyrodnicza. Mantis, Olsztyn, pp 96–105
33. Dumnicka E, Poznańska M (2006) Novel Polish recordings of rare aquatic Oligochaeta species. Oceanol Hydrobiol Stud 35(2):111–120
34. Żmudziński L, Chubarova-Solovjeva S, Dobrowolski Z et al (1996) Expansion of the spionid polychaete *Marenzelleria viridis* in the southern part of the Baltic Sea. In: Proceedings of the 13th Baltic Marine Biologists symposium, Jurmala, Aug 31–Sept 4 1993, pp 127–129
35. Sikorski AV, Bick A (2004) Revision of Marenzelleria Mesnil, 1896 (Spionidae, Polychaeta). Sarsia: North Atlantic Mar Sci 89(4):253–275
36. Warzocha J, Gromisz S, Woźniczka A, Koper M (2005) Distribution of *Marenzelleria* cf *viridis* (Polychaeta: Spionidae) along the Polish coast of the Baltic Sea. 2005. Oceanol Hydrobiol Stud 34(supl. 1):227–237
37. Blank M, Laine AO, Jürss K, Bastrop R (2008) Molecular identification key based on PCR/RFLP for three polychaete sibling species of the genus *Marenzelleria*, and the species' current distribution in the Baltic Sea. Helgol Mar Res 62:129–141
38. Warzocha J (2011) *Marenzelleria neglecta* Sikorski & Bick 2004. In: Głowaciński Z, Okarma H, Pawłowski J, Solorz W (eds) Alien species in the fauna of Poland. 1. An overview of the status. IOP PAN, Kraków, pp 160–164
39. Jabłońska-Barna I, Rychter A, Macias N (2011) Bentofauna drobnego zbiornika wodnego na terenie Olsztyna (The benthic fauna of a small water body in Olsztyn). Rozprawy Naukowe i Zawodowe PWSZ w Elblągu, Instytut Politechniczny 13:59–65
40. Water Framework Directive (2000/60/EC) European Community 2000

Cyanobacteria and Toxic Blooms in the Great Mazurian Lakes System: Biodiversity and Toxicity

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Contents

Abstract The Great Mazurian Lakes System (GMLS), located in the northeastern part of Poland, is an extremely valuable area in terms of natural environment value, tourism, and local economy. The system is divided into two parts – the northern meso-eutrophic and the southern eutrophic. GMLS are lakes with very high taxonomic diversity of phytoplankton, and cyanobacteria are very often predominant in the species composition and biomass. The presence of cyanobacteria belonging to 14 different families from the orders of *Nostocales*, *Oscillatoriales*, *Synechococcales*, and *Chroococcales* was recorded throughout the system. The GMLS has undergone significant changes over the recent decades which affected the taxonomic composition and dominant species of cyanobacteria. Particularly the southern part was subject to significant changes, from rapid eutrophication in the 1970s and 1980s, resulting in massive blooms of cyanobacteria, to a significant improvement in water quality in the 1990s and a reduction of cyanobacteria biomass. However, cyanobacteria are the dominant component of phytoplankton up to the present, although there are no dense blooms in recent years. Many of the cyanobacteria taxa in the GMLS can potentially produce toxins. Hepatotoxic microcystins are the most common cyanotoxins in freshwater, and in GMLS

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they sometimes reached significant concentrations in water. Studies have shown that the main producers of microcystins in GMLS are genera *Microcystis* and *Planktothrix*.

Keywords Bloom · Cyanobacteria · Cyanotoxins · GMLS · Microcystins

1 Introduction

Cyanobacteria are one of the first forms of life that appeared on Earth [1, 2]. They are prokaryotic microorganisms, single cells, or colony-forming (like large aggregates, filamentous forms). Most cyanobacteria are aerobic photoautotrophs, some of them may also be mixotrophic, under certain environmental conditions (e.g., with insufficient light intensity) [3]. All cyanobacterial cells contain photosynthetic dyes like chlorophyll$_a$ and phycocyanin from phycobiliproteins group, which make them have a blue-green color. In addition, the cell may also contain auxiliary dyes such as carotenoids and other phycobiliproteins, e.g., phycoerythrin or allophycocyanin [4].

Cyanobacteria can live in very diverse environments, even with extreme conditions such as hot springs [5], snow, or ice [6, 7], as well as places with very high salinity [8]. However, their main habitat is water, both freshwater and marine ecosystems.

Cyanobacteria have some adaptations and features, which under certain conditions allow them to win a competitive advantage with other phytoplanktonic microorganisms. Auxiliary photosynthetic dyes cause that cyanobacteria can use in photosynthesis green, yellow, and orange part of the sunlight spectrum (500–650 nm), which are rarely used by other phytoplankton organisms. This enables living in deeper parts of water bodies, in turbid water with a large biomass of phytoplankton or other factors limiting the availability of light [8, 9].

Aerotopes (gas "vacuums") are cylindrical structures in the cytoplasm, filled with gas, which allows the regulation of the vertical position of cells in the water column [10]. Thanks to this ability, cyanobacteria can accumulate at depths with the most favorable conditions, e.g., with the appropriate intensity of light and carbon dioxide or biogenic compound availability. Aerotopes also contribute to the formation of dense blooms, because they can keep cyanobacteria cells on the surface [11].

Atmospheric nitrogen fixation is another process favoring the dominance of some cyanobacterial species. Vegetative cells of genera such as *Aphanizomenon*, *Cylindrospermopsis*, *Dolichospermum* (formerly *Anabaena*), *Nodularia*, or *Nostoc* can develop heterocysts in which molecular nitrogen (N_2) is reduced directly to ammonium nitrogen (NH_4^+) using the nitrogenase enzyme. In addition, the low ratio of nitrogen to phosphorus in water favors the development of cyanobacteria. The optimal ratio of TN/TP (total nitrogen to total phosphorus) for eukaryotic phytoplankton microorganisms is 16–23:1, while for cyanobacterial strains, it is 10–16:1 [12].

Cyanobacteria are also more resistant to zooplankton feeding than other phytoplankton microorganisms. Filamentous cyanobacteria (e.g., *Planktothrix* spp.) can clog the filtration apparatus of many species of zooplankton animals and thus reduce their efficiency of food intake [13, 14]. In addition, the nutritional value of cyanobacteria cells is low, due to the low content of unsaturated fatty acids [15].

On the other hand, there are several aspects in which cyanobacteria lose the competition with eukaryotic phytoplankton. Primarily the growth rate of cyanobacteria is usually much lower than in eukaryotic algae [16]. Slow growth rates require long water retention times to reach significant cell numbers. Therefore, cyanobacteria do not form blooms in flowing water, such as rivers.

The biomass and taxonomic structure of phytoplankton in water depend on the concentration of key biogenic elements, primarily nitrogen and phosphorus. At high concentrations of biogenic elements, the phytoplankton biomass increases, but also its biodiversity decreases. Species composition is dominated by cyanobacteria, especially large filamentous taxa such as *Aphanizomenon*, *Dolichospermum*, *Limnothrix*, *Microcystis*, or *Planktothrix* [17]. Advanced eutrophication is therefore closely related to the dominance of cyanobacteria in water [17, 18] and may eventually lead to the occurrence of mass cyanobacterial blooms, which are highly dangerous due to the possibility of release of cyanotoxins into the water.

The second most important factor, promoting the dominance of cyanobacteria is the elevated temperature. Cyanobacteria reach the maximum growth rate at a temperature higher than optimal temperatures for other phytoplankton species [19]. Global warming causes earlier stratification and later destratification of lakes in a temperate climate, which extends the period with optimal conditions for the development of cyanobacteria [11]. Climate change also significantly contributes to expanding the geographical coverage of some taxa. *Cylindrospermopsis raciborskii* was initially described as a species occurring in tropical or subtropical regions, but now is already widespread in lakes in northern Germany [20], and also appears in some Polish lakes [21]. Along with the predicted further climate warming [22, 23], cyanobacteria and the toxic blooms will become an increasingly important global environmental problem [24].

Cyanobacteria produce many metabolites that can be dangerous to animals and humans. Cyanotoxins are produced by a wide spectrum of cyanobacterial species, marine or freshwater, both colony and filamentous forms. Toxins produced by cyanobacteria are usually classified due to their effect on mammalian organisms, and there are three main groups distinguished: hepatotoxins (cyclic peptides, such as microcystins or nodularins, and alkaloids such as cylindrospermopsins), neurotoxins (e.g., anatoxin-a, saxitoxin, BMAA), and dermatotoxins (e.g., aplysiatoxin, lyngbyatoxin) [25]. The most widespread and most common type of cyanotoxins in freshwater ecosystems are hepatotoxic microcystins. These are cyclic heptapeptides which can be produced by many genera, e.g., *Aphanizomenon*, *Dolichospermum*, *Microcystis*, *Nostoc*, *Phormidium*, *Planktothrix*, *Pseudanabaena*, *Synechococcus*, and *Synechocystis* [8, 25–27]. Toxicity of microcystins for animal cells is associated with inhibition of the key cellular enzymes – protein phosphatases [25]. Microcystins are synthesized non-ribosomally by large enzymatic complexes, encoded by the *mcy* gene cluster. In cyanobacteria population toxic and nontoxic strains of one species can coexist, so the percentage of toxigenic cells indicates whether the bloom, if occur, will be toxic.

The Great Mazurian Lakes System (GMLS), which is part of the Mazurian Lake District, is located in the northeastern part of Poland (Fig. 1).

It is an extremely valuable area in terms of natural environment value as well as for the local fisheries economy. However, the basis of the region's economy is tourism. In recent years, over one million people visit Mazurian Lake District annually (Statistics Poland data for 2016). The GMLS includes glacial origin lakes of different size and depth, connected naturally or with channels built in the nineteenth century. This region was formed around 10–15 thousand years ago during the Baltic glaciation [28].

The system is divided into two parts – the northern meso-eutrophic part, which includes the lakes Mamry (with a bay called Przystań Lake), Dargin, Łabap, and Kisajno, and the southern eutrophic part covering the lakes Niegocin, Boczne, Jagodne, Szymoneckie, Szymon, Tałtowisko, Ryńskie, Tałty, Mikołajskie, Bełdany, and Śniardwy (Fig. 1). The watershed is located around the Giżycki Canal and Lake Kisajno [29], although its location may be changeable [30]. The Pisa River is the

Fig. 1 Map of the Great Mazurian Lakes System area. The numbers indicate the lakes described in this chapter: 1, Przystań; 2, Mamry; 3, Kirsajty; 4, Dargin; 5, Łabap; 6, Kisajno; 7, Niegocin; 8, Boczne; 9, Jagodne; 10, Szymoneckie; 11, Szymon; 12, Tałtowisko; 13, Ryńskie; 14, Tałty; 15, Mikołajskie; 16, Bełdany; 17, Śniardwy

main outflow from the southern part of the GMLS; it flows into the Narew River, which connects with the Vistula River through the Zegrze Reservoir. Northern part belongs to the Pregoła catchment. Both parts of the GMLS substantially differ in anthropopressure level. Northern part of catchment basin is much less populated, while in the southern part, more towns, villages, and tourist centers are located.

In the species composition and biomass of phytoplankton in the GMLS, cyanobacteria are very often predominant [31, 32]. There is also a very high taxonomic diversity of cyanobacteria, morphologically different species appear, and many of them can also potentially produce toxins.

2 Biodiversity of Cyanobacteria in the Great Mazurian Lakes System

GMLS are lakes with very high taxonomic diversity of cyanobacteria (Table 1). The collected data from publications available from the 1970s allowed to state that the presence of cyanobacteria belonging to the orders of *Nostocales*, *Oscillatoriales*, *Synechococcales*, and *Chroococcales* was recorded throughout the system [31, 33–43]. Literature data indicate that cyanobacteria in GMLS belong to 14 different families and 26 genera (Table 1). It should be noted that the taxonomic classification of cyanobacteria and some species names have been changing over the years, and in this report, they are given according to the present arrangements (Algae.Base.org) [44].

Over the years, changes in the taxonomic composition and dominant species of cyanobacteria in the phytoplankton of Mazurian Lakes have been observed. They were different depending on the lake, its trophic state, surrounding area, or external conditions. Although all the lakes in the system are connected, the abundance of cyanobacteria can be very different, especially between the lakes of the northern and southern parts. The GMLS has undergone significant changes over the past few decades. In 1970s and 1980s, very fast eutrophication processes were observed, mainly in the southern lakes. The main reasons were bad agricultural practices (such as overfertilization and lack of vegetative riparian buffers around lakes), draining sewage to the lakes from urban areas and industry, as well as increasing tourism [45]. This had a devastating effect on the GMLS ecosystem, causing frequent and massive cyanobacterial blooms. The situation improved in the 1990s, when agriculture started to be less ecologically damaging, for example, the use of fertilizers decreased significantly [45]. The modernization of wastewater treatment plants in the region (e.g., in Giżycko) also contributed to the improvement. After modernization favorable changes were observed quickly in Lake Niegocin – a decrease in biomass of cyanobacteria and the dominance of other phytoplankton species [38]. This tendency, when cyanobacteria did not dominate in phytoplankton, remained only for a few years [38]. However, from that time to present, no dense, surface blooms are observed in GMLS, and the biomass of phytoplankton does not usually exceed 8 mg L^{-1}, which is considered a threshold limit for algal blooms.

Table 1 All species of cyanobacteria observed in the Great Mazurian Lakes System from the 1970s to present

Order	Family	Species
Nostocales	*Aphanizomenonaceae*	*Anabaenopsis* sp.
		Aphanizomenon flos-aquae
		Aphanizomenon gracile
		Aphanizomenon skujae
		Aphanizomenon yezoense
		Cuspidothrix issatschenkoi
		Dolichospermum sp.
		Dolichospermum affine
		Dolichospermum flos-aquae
		Dolichospermum lemmermanii
		Dolichospermum planctonica
		Dolichospermum spiroides
	Gloeotrichiaceae	*Gloeotrichia echinulata*
Oscillatoriales	*Microcoleaceae*	*Planktothrix agardhii*
		Planktothrix agardhii var. suspensa
	Oscillatoriaceae	*Lyngbya* sp.
Synechococcales	*Pseudanabaenaceae*	*Limnothrix redekei*
		Pseudanabaena catenata
		Pseudanabaena limnetica
		Pseudanabaena mucicola
	Leptolyngbyaceae	*Leptolyngbya thermalis*
		Planktolyngbya limnetica
	Romeriaceae	*Romeria gracilis*
	Coelosphaeriaceae	*Coelosphaerium* spp.
		Snowella lacustris
		Snowella litoralis
		Woronichinia naegeliana
	Merismopediaceae	*Aphanocapsa* spp.
		Merismopedia spp.
	Synechococcaceae	*Cyanobium* sp.
		Cyanodictyon spp.
		Lemmermanniella pallida
		Rhabdoderma lineare
		Synechococcus sp.
Chroococcales	*Aphanothecaceae*	*Aphanothece* spp.
	Chroococcaceae	*Chroococcus* sp.
		Chroococcus minimus
		Chroococcus turgidus
	Gomphosphaeriaceae	*Gomphosphaeria* sp.

(continued)

Table 1 (continued)

Order	Family	Species
	Microcystaceae	*Microcystis aeruginosa*
		Microcystis flos-aquae
		Microcystis ichtyoblabe
		Microcystis smithii
		Microcystis viridis
		Microcystis wesenbergii

Based on literature data [31, 33–43]

Rapid changes in phytoplankton structure or changes in dominant species are observed in many lakes of the system. They may indicate a low stability of the ecosystem and its high susceptibility to changes caused by various factors, such as the availability of nutrients. Temperature is also an important factor – an example can be the exceptionally hot summer of 2010, when there was practically the total domination of cyanobacteria in the phytoplankton structure, in contrast to the next year, when the water temperature was on average 2.6°C lower, and the cyanobacteria in most lakes did not dominate [43, 45].

The most visible taxonomic change in recent years concerns practically the entire system. Despite incomplete historical data, it can be stated that by the end of the first decade of the twenty-first century, the dominant taxa in different lakes were changing, but most often they were cyanobacteria from the genera *Aphanizomenon*, *Dolichospermum*, *Microcystis*, and *Planktothrix* [33–39, 41]. In recent years, however, the domination in summer of *Pseudanabaena limnetica* is observed [31, 42, 43]. Previously this species never had such a large share in the taxonomic composition.

2.1 Northern Part

The northern part of GMLS has been much less eutrophicated in the last decades than the southern part of the system. Available historical data, mainly from Lake Mamry, indicate that in the 1960s, 1970s, and 1980s of the twentieth century, the biomass of phytoplankton in these lakes was usually low (e.g., 0.7–2 mg L^{-1} in July 1976) and cyanobacteria were not dominant component of phytoplankton [35, 46, 47]. In the summer months in 1986–1989, the biomass of phytoplankton in Lake Mamry was still low (0.1–1.5 mg L^{-1}), cyanobacteria accounted for about 20% of biomass, and the dominant groups were diatoms, cryptophytes, and dinoflagellates [36]. The exception in the northern part was Lake Kirsajty, the only small, shallow, and polymictic lake in this part, in which both in 1976 and in 1988, cyanobacteria predominated in the summer in the phytoplankton structure (Table 2). In 1988, they were mainly *Aphanizomenon gracile* and *Leptolyngbya thermalis* [35, 39]. Data from the 1990s and the first decade of the twenty-first century are unfortunately very limited. However, research conducted in Lake Mamry

Table 2 The dominance of cyanobacteria (defined as >50% of the phytoplankton biomass) in the northern part of the Great Mazurian Lakes System in the last few decades and dominant taxa (if available)

Period	Lakes						References
	Przystań	Mamry	Kirsajty	Dargin	Łabap	Kisajno	
1960s, 1970s	nd	–	nd	nd	nd	nd	[36]
Summer 1976	nd	nd	+	nd	nd	nd	[35]
Summer 1986	nd	–	–	nd	nd	nd	[36, 39]
Summer 1987	nd	–	–	nd	nd	nd	[36, 39]
Summer 1988	nd	–	+ *Ag* *Lt*	nd	nd	nd	[36, 39]
Summer 1989	nd	–	nd	nd	nd	nd	[36]
Summer 2000	nd	+	+ *Lr* *Ag* *Lt*	nd	nd	nd	[36, 39]
Summer 2001	nd	+	+ *Ma* *Lt*	nd	nd	nd	[36, 39]
Summer 2010	+ *Psl*	+ *Psl*	nd	+ *Psl*	+ *Ge* *Ag*	–	[43]
Summer 2011	+ *Psl*	+ *Psl*	nd	–	–	–	[43]
Summer 2012	nd	+	nd	nd	nd	+	[42]

Based on literature data [35, 36, 39, 42, 43]
+ cyanobacteria predominate in phytoplankton biomass, – cyanobacteria do not dominate in phytoplankton biomass, *nd* no data, *Ag Aphanizomenon gracile*, *Ge Gloeotrichia echinulata*, *Lr Limnothrix redekei*, *Lt Leptolyngbya thermalis*, *Ma Microcystis aeruginosa*, *Psl Pseudanabaena limnetica*

in 2000 and 2001 indicates that there has been a significant change in the structure of phytoplankton in this lake. The total biomass of phytoplankton in these years was about two times higher than at the end of the 1980s, and the share of cyanobacteria (mainly *Microcystis aeruginosa*, *Aphanizomenon flos-aquae*, *Limnothrix redekei*, and *Leptolyngbya thermalis*) in the summer reached even 80% biomass [36]. In these years, surface blooms of *Gloeotrichia echinulata* were also observed [36]. Lake Kirsajty was studied in the same years and cyanobacteria predominated there and also the summer phytoplankton. In August 2000 the dominant species were *Limnothrix redekei*, *Aphanizomenon gracile*, and *Leptolyngbya thermalis*, in August 2001 *Microcystis aeruginosa*, and in September 2001 *Microcystis aeruginosa* and *Leptolyngbya thermalis* (Table 2) [39].

Studies concerning a larger number of lakes from the northern part were conducted in 2010 and 2011 on Przystań, Mamry, Dargin, Łabap, and Kisajno lakes (Table 2) [43]. In the summer months, the biomass of cyanobacteria reached high values only in Lake Łabap (average 12.8 mg L^{-1}), while in the other lakes, it did not exceed 2 mg L^{-1}. Cyanobacteria usually dominated in the phytoplankton during the summer months, especially in August, often reaching over 90%. One exception was the Lake Kisajno, in which in these 2 years, there was no dominance of cyanobacteria. At that time, there was a change in the taxonomic structure of cyanobacteria in comparison to previously analyzed periods. The most frequently dominant species was *Pseudanabaena limnetica*, which reached even about 90% share in the biomass of cyanobacteria in Przystań and Mamry lakes. The dominance of this species is a significant change compared to previous studies that have never shown a large biomass of this taxon in the northern lakes before. In addition to *Pseudanabaena limnetica*, in Lake Łabap in the summer of 2010, higher numbers also reached *Gloeotrichia echinulata* and *Aphanizomenon gracile* [43].

The most recent studies on the composition of phytoplankton in the northern lakes concern years 2011–2013, when the lakes Mamry and Kisajno were studied (Table 2) [42]. The average share of cyanobacteria in the total phytoplankton biomass was 43–44% in the summer months, and the maximum value was 83% in Lake Kisajno in July 2012. Lake Kisajno, in which in 2010 and 2011 there was no dominance of cyanobacteria, in 2012 completely changed the structure of phytoplankton. In 2012 and 2013, the most dominant species in the northern lakes was still *Pseudanabaena limnetica* [42].

2.2 Southern Part

The southern part of the GMLS differs significantly in trophic status from the northern part. While the northern lakes are mainly classified into meso-eutrophy, southern lakes in the last decades were defined as eutrophic or sometimes even hypereutrophic [43, 45, 48]. For this reason, the abundance and taxonomic structure of cyanobacteria in these lakes differs from those of the northern part.

Already in the 1970s and 1980s, it was reported that the biomass of phytoplankton often exceeded 8 mg L^{-1}, which is a characteristic value for eutrophic lakes [34, 35, 38, 49]. However, historical data on the taxonomic composition of cyanobacteria in these lakes are quite limited. The report by Chróst [33] describing Lake Mikołajskie indicates that in the early autumn of 1972 and 1973, cyanobacteria dominated the phytoplankton. *Microcystis aeruginosa*, *Microcystis wesenbergii*, and *Dolichospermum flos-aquae* reached the highest numbers, and *Aphanizomenon flos-aquae* also appeared in smaller quantities. Data from Lake Niegocin show that in the 1970s and 1980s, summer phytoplankton was dominated by filamentous species, mainly *Planktothrix agardhii* and *Aphanizomenon flos-aquae* [38].

Available data from the last decade of the twentieth century mainly concern Lake Niegocin (Table 3). Changes in the biomass of phytoplankton and its taxonomic composition during this period were largely related to the modernization of the wastewater treatment plant, from which sewage was discharged into the lake. Before the modernization, until 1994, the biomass of phytoplankton reached even 8.2 mg L^{-1}, while after this process, in the years 1995–1999, it never exceeded 2.9 mg L^{-1} [38]. Until 1995, cyanobacteria most often dominated in the summer and autumn phytoplankton, reaching even 96% of biomass. The dominant species were

Table 3 The dominance of cyanobacteria (defined as >50% of the phytoplankton biomass) in the southern part of the Great Mazurian Lakes System in the last few decades and dominant taxa (if available)

Period	Lakes 1	2	3	4	5	6	7	8	9	10	11	References
1970s–1980s	+ *Pa* *Afa*	nd	nd	nd	nd	nd	nd	nd	+ *Dfa* *Ma* *Mw*	nd	nd	[33, 38]
S 1991	+ *Pa*	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S-A 1992	+ *Pa* *Afa* *Lt*	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S-A 1993	+ *Pa* *Afa*	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
A 1994	+	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S-A 1995	+ *Dfa* *Pa* *Lt*	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S-A 1996	–	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S 1997	–	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
A 1998	–	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S-A 1999	+ *Asp* *Dsp* *Lsp*	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S-A 2000	–	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S-A 2001	+ *Asp* *Dsp* *Lp*	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[38]
S 2002	nd	nd	+ *Pa*	+ *Pll* *Lr* *Asp*	+ *As* *Lr*	+ *Afa* *Ag* *Pa* *Psl*	nd	nd	nd	nd	nd	[37]
S 2007	+ *Ma*	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	[41]

(continued)

Table 3 (continued)

Period	Lakes											References
	1	2	3	4	5	6	7	8	9	10	11	
S 2010	+ *Ag*	+ *Ma*	+ *Psl*	+ *Psl*	+ *Psl*	+ *Psl*	+ *Psl*	+ *Psl* *Ge*	+ *Psl*	+ *Ag* *Psl*	+ *Psl*	[43]
S 2011	–	–	–	–	+ *Ag*	–	+ *Psl*	+ *Psl*	–	–	+ *Psl*	[43]
S 2012	–	nd	nd	nd	+	+	nd	+	+	+	+	[42]

Based on literature data [33, 37, 38, 41–43]
Lakes (numbers according to Fig. 1): 7, Niegocin; 8, Boczne; 9, Jagodne; 10, Szymoneckie; 11, Szymon; 12, Tałtowisko; 13, Ryńskie; 14, Tałty; 15, Mikołajskie; 16, Bełdany; 17, Śniardwy. S, summer; A, autumn; S-A, summer-autumn
+ cyanobacteria predominate in phytoplankton biomass, – cyanobacteria do not dominate in phytoplankton biomass, *nd* no data. *Afa*, *Aphanizomenon flos-aquae*; *Ag*, *Aphanizomenon gracile*; *As*, *Aphanizomenon skujae*; A_{sp}, *Aphanizomenon* sp.; *Dfa*, *Dolichospermum flos-aquae*; D_{sp}, *Dolichospermum* sp.; *Ge*, *Gloeotrichia echinulata*; *Lt*, *Leptolyngbya thermalis*; L_{sp}., *Leptolyngbya* sp.; *Lr*, *Limnothrix redekei*; *Ma*, *Microcystis aeruginosa*; *Mw*, *Microcystis wesenbergii*; *Pll*, *Planktolyngbya limnetica*; *Pa*, *Planktothrix agardhii*; *Psl*, *Pseudanabaena limnetica*

filamentous *Planktothrix agardhii*, *Aphanizomenon flos-aquae*, and *Leptolyngbya thermalis*. Since 1996 in summer more often occurs the dominance of dinoflagellates or the codominance of dinoflagellates-cyanobacteria. In autumn the domination of diatoms has been more frequent [38]. At the turn of the century, the situation in the Lake Niegocin again began to change (Table 3). Already in 1999 there was a re-dominance of cyanobacteria in the summer, mainly of the genera *Aphanizomenon*, *Dolichospermum*, and *Leptolyngbya*. In the next 2 years, the increase of phytoplankton biomass to a similar level as before the modernization of the sewage treatment plant was observed, and the predominant taxa of cyanobacteria were *Aphanizomenon* sp., *Dolichospermum* sp., *Leptolyngbya* sp., as well as *Microcystis aeruginosa* [38]. Further studies of cyanobacteria in Lake Niegocin were made in the summer of 2007 and confirmed their domination in phytoplankton biomass (over 90%) [41]. Analysis at that time showed a large taxonomic diversity of cyanobacteria, but the dominant species was *Microcystis aeruginosa*. Besides this species, cyanobacteria from the genera *Chroococcus*, *Aphanizomenon*, *Dolichospermum*, and *Planktolyngbya* also occurred in significant numbers [41].

Four lakes of the southern part of the GMLS (Jagodne, Szymoneckie, Szymon, and Tałtowisko) were examined for biomass and taxonomic composition of phytoplankton in the summer of 2002 [37]. In all lakes, cyanobacteria predominated, reaching 53.5% (Tałtowisko) to over 92% (Jagodne, Szymoneckie) of phytoplankton biomass. In lakes Jagodne and Szymoneckie, cyanobacteria reached at that time a very high biomass, 9.8 and 15.5 mg L^{-1}, respectively. The taxonomic composition was dominated by filamentous genera, primarily *Planktothrix*, *Planktolyngbya*, *Limnothrix*, *Aphanizomenon*, and *Pseudanabaena* spp. (Table 3).

Extensive analysis of biomass and diversity of cyanobacteria in the southern part of GMLS in 2009–2011 is described by Siuda et al. [43]. There were 11 lakes studied – Niegocin, Boczne, Jagodne, Szymoneckie, Szymon, Tałtowisko, Ryńskie, Tałty, Mikołajskie, Bełdany, and Śniardwy. The average summer biomass of cyanobacteria in these years was very different in individual lakes and ranged from 0.7 mg L^{-1} in Lake Niegocin to almost 11 mg L^{-1} in Lake Tałty. The dominance of cyanobacteria in the phytoplankton composition was visible primarily in the summer of 2010, when they accounted for 75–100% of the phytoplankton biomass in almost all lakes, except for lakes Niegocin and Boczne (Table 3). The predominant species of cyanobacteria was the filamentous *Pseudanabaena limnetica*, which dominated almost all lakes, again with the exception of lakes Niegocin and Boczne, where *Aphanizomenon gracile* and *Microcystis aeruginosa* were predominantly observed. In the summer of 2011, cyanobacteria were less abundant, and their share in the phytoplankton biomass did not exceed 70%, with the exception of Lake Śniardwy. The dominance of cyanobacteria was not observed in lakes Niegocin, Boczne, Jagodne, Szymoneckie, Tałtowisko, Mikołajskie, and Bełdany (Table 3). Again, the most frequently occurring species was *Pseudanabaena limnetica*, although the dominance of *Aphanizomenon gracile* was observed in Lake Szymon [43].

The latest study on six southern lakes (Niegocin, Tałtowisko, Tałty, Mikołajskie, Bełdany, Śniardwy) shows that the average summer biomass of phytoplankton in 2011–2013 did not exceed 6 mg L^{-1} in any of the lakes, and the average share of cyanobacteria in phytoplankton was 16–57% [42]. In the summer of 2012, the highest share of cyanobacteria was recorded in Lake Tałtowisko (82%). Again, the most common taxa were *Pseudanabaena limnetica*, but *Planktolyngbya limnetica*, *Limnothrix redekei*, *Planktothrix agardhii*, and *Planktothrix agardhii* var. suspensa also appeared sometimes in large numbers (Table 3).

3 Toxic Cyanobacteria in the Great Mazurian Lakes System

Most of the data about cyanobacterial toxins in GMLS is related to hepatotoxic microcystins, which are the most common cyanotoxins in freshwaters. The first study on microcystin concentration was carried out in 2002 on the lakes Jagodne, Szymoneckie, Szymon, and Tałtowisko [37]. The HPLC analysis showed quite high toxin concentrations, 4.8–12.1 μg L^{-1}, while the safe limit value for recreational waters is 5 μg L^{-1} according to the World Health Organization [50]. The highest concentration was recorded in Lake Szymoneckie, and the detected variants of microcystins were dmMC-RR (desmethyl-RR), MC-YR, and MC-LR [37].

In the summer of 2007, concentration of microcystins was analyzed in Lake Niegocin [41]. The highest concentration was noted at the beginning of September;

however, it was much lower than in the lakes analyzed in 2002 (<0.2 μg L^{-1}). The microcystin variants detected during these studies were MC-LR, MC-RR, and MC-YR.

Subsequent analyses of the microcystin concentration in the GMLS were conducted from spring to autumn in 2012 and 2013 and concerned with both lakes of the northern (Mamry, Kisajno) and southern part (Niegocin, Tałtowisko, Ryńskie, Tałty, Mikołajskie, Bełdany, Śniardwy) [42]. Also, in these years, the concentration in water was low (0–0.3 μg L^{-1} in 2012 and 0–0.6 μg L^{-1} in 2013). Microcystins were practically not detected in Lake Mamry (except one sample in which the concentration was 0.1 μg L^{-1}), while the highest concentration was recorded in Lake Ryńskie. In addition to the dissolved microcystin, the total concentration of microcystins (together dissolved in water and present in cyanobacteria cells) was also tested. This analysis showed that the total concentration of microcystins may be 2–20 times higher than dissolved microcystins. The highest total concentration was recorded in September 2013 in Lake Mikołajskie (2.1 μg L^{-1}). In 2012 and 2013, seven variants of microcystin were detected in GMLS, the most common was demethylated variant [Asp3]MC-RR [42].

Because in the population of one cyanobacteria species, both toxic and nontoxic strains can occur, and toxic and nontoxic cells cannot be distinguished by microscopic methods, until recently it was not known which taxa in GMLS are responsible for the production of toxins. Among many taxa occurring in the Mazurian Lakes, microcystins can produce the genera *Anabaenopsis*, *Aphanizomenon*, *Aphanocapsa*, *Dolichospermum*, *Limnothrix*, *Microcystis*, *Planktothrix*, *Pseudanabaena*, *Snowella*, *Synechococcus*, and *Woronichinia* [8, 24, 51–56]. However, only the molecular methods allow to determine which of them actually have genes responsible for toxin production. In the summer and early autumn of 2011, *mcy*A gene sequences from four lakes were obtained (Mamry, Tałtowisko, Mikołajskie, Bełdany). Sequences showed that in Lake Mamry toxicity genes belong to cyanobacteria from genus *Microcystis* while in the other three lakes to the *Planktothrix* genus [31].

In the publication from Bukowska et al. [42], it was found that the *mcy*A, *mcy*D, and *mcy*E genes, responsible for microcystin production, occur in all lakes studied in 2012 and 2013 (Mamry, Kisajno, Niegocin, Tałtowisko, Ryńskie, Tałty, Mikołajskie, Bełdany, and Śniardwy). In the whole GMLS, the main producers of microcystins were genera *Planktothrix* and *Microcystis*, which at that time were not the dominant taxa in the biomass of cyanobacteria. The participation of toxic cells in the populations of these genera was also investigated by real-time PCR method. In the case of *Planktothrix*, in all samples almost the entire population consisted of toxic cells (75–100%), and the highest average share was recorded in Lake Mikołajskie (95%). It was different in the case of *Microcystis* – there was a very large variation in the proportion of toxic cells between samples (0–100%). The highest average share was recorded in Lake Mamry (21%) and the lowest in Lake Mikołajskie (0.2%). The proportion of *Microcystis* cells in the population was also variable over time; no seasonal pattern was observed [42].

There are no studies on the presence of cyanotoxins other than microcystins in the GMLS. One of the harmful toxins is cylindrospermopsin, also classified as hepatotoxin. One of its producers – *Cylindrospermopsis raciborskii* – is quite rapidly spreading in temperate climate lakes, although it was recently considered as a tropical or subtropical species [20]. Studies conducted in the summer of 2014 did not show the presence of this species in Mazurian Lakes [57]. However, cylindrospermopsins may be also produced by other genera (e.g., *Aphanizomenon*, *Dolichospermum*, *Planktothrix*), which are sometimes present in the GMLS in large numbers; thus they can potentially be dangerous [8, 24, 56, 58]. It is similar with neurotoxins such as anatoxin-a or saxitoxin, which potential producers are also cyanobacteria found in the GMLS (*Aphanizomenon*, *Cuspidothrix*, *Dolichospermum*, *Planktothrix*) [26, 58–60].

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References

1. Brock TD (1973) Evolutionary and ecological aspects of the cyanophytes. The biology of the blue green algae. Blackwell Scientific Publications, Oxford, pp 487–500
2. Schopf JW (1996) Cyanobacteria. Pioneers of the early earth. Contributions in phycology, Berlin, pp 13–32
3. Fay P (1965) Heterotrophy and nitrogen fixation in *Chlorogloea fritschii*. J Gen Microbiol 39:11–20
4. Douglas SE (1994) Chloroplast origins and evolution. The molecular biology of cyanobacteria. Kluwer Academic Publishers, Dordrecht, pp 91–118
5. Castenholz RW (1973) Ecology of blue-green algae in hot springs. The biology of blue-green algae. Blackwell Scientific Publications, Oxford, pp 379–414
6. Kol E (1968) Kryobiologie. I. Kryovegetation. Die Binnengewässer, Band XXIV, E. Schweizerbart'sche Verlagsbuchhandlung, Stuttgart, p 216
7. Laamanen M (1996) Cyanoprokaryotes in the Baltic Sea ice and winter plankton. Arch Hydrobiol Suppl 117. Algol Stud 83:423–433
8. Chorus I (1999) Bartram toxic cyanobacteria in water: a guide to their public health consequences, monitoring and management. E&FN Spon, London
9. Kardinaal WEA, Visser PM (2005) Dynamics of cyanobacterial toxins: sources of variability in microcystin concentrations. In: Huisman JH, Matthijs CP, Visser P (eds) Harmful cyanobacteria. Springer, Dordrecht, pp 41–63
10. Walsby AE (1978) The gas vesicles of aquatic prokaryotes. In: Relations between structure and functioning in the prokaryotic cell, 28th symposium of the society for general microbiology. Cambridge University Press, Cambridge, p 338
11. Paerl HW, Huisman J (2008) Climate. Blooms like it hot. Science 320:57–58
12. Schreurs H (1992) Cyanobacterial dominance, relation to eutrophication and lake morphology. PhD thesis, University of Amsterdam, Amsterdam
13. Gliwicz ZM, Lampert W (1990) Food thresholds in *Daphnia* species in the absence and presence of blue-green filaments. Ecology 71:691–702

14. DeMott WR, Gulati RD, van Donk E (2001) Daphnia food limitation in three hypereutrophic Dutch lakes: evidence for exclusion of large-bodied species by interfering filaments of cyanobacteria. Limnol Oceanogr 46:2054–2060
15. Huisman J, Hulot FD Population dynamics of harmful Cyanobacteria. Factors affecting species composition. In: Huisman JH, CP Matthijs, P Visser (eds), Harmful cyanobacteria, 143–176. Springer, Dordrecht 2005
16. Reynolds CS, Wiseman SW, Clarke MJO (1984) Growth- and loss-rate responses of phytoplankton to intermittent artificial mixing and their potential application to the control of planktonic algal biomass. J Appl Ecol 21:11–39
17. Dokulil MT, Teubner K (2000) Cyanobacterial dominance in lakes. Hydrobiologia 438:1–12
18. Pizzolon L, Tracanna B, Prósperi C, Guerrero JM (1999) Cyanobacterial blooms in Argentinean inland waters. Lakes Reserv Res Manage 4:101–105
19. Robarts RD, Zohary T (1987) Temperature effects on photosynthetic capacity, respiration, and growth rates of bloom-forming cyanobacteria. N Z J Mar Freshw Res 21:391–399
20. Wiedner C, Rücker J, Brüggemann R, Nixdorf B (2007) Climate change affects timing and size of populations of an invasive cyanobacterium in temperate regions. Oecologia 152:473–484
21. Kokociński M, Meriluoto J, Spoof L, Gągała I, Jurczak T, Rejmonczyk E, Mankiewicz-Boczek J (2011) Distribution and potential producers of cylindrospermopsin in western Poland. Eur J Phycol 46:130–131
22. Bates BC, Kundzewicz ZW, Wu S, Palutikof JP (2008) Climate change and water. Technical paper of the intergovernmental panel on climate change, IPCC Secretariat, Geneva, p 210
23. IPCC (2013) Climate change 2013: the physical science basis. In: Stocker TF, Qin D, Plattner G-K, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM (eds) Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, 1535 pp. https://doi.org/10.1017/CBO9781107415324
24. Paerl HW, Huisman J (2009) Climate change: a catalyst for global expansion of harmful cyanobacterial blooms. Environ Microbiol Rep 1:27–37
25. Metcalf JS, Codd GA (2004) Cyanobacterial toxins in the water environment. Foundation for Water Research
26. Sivonen K, Börner T (2008) Bioactive compounds produced by cyanobacteria. The Cyanobacteria: molecular biology, genomics and evolution. Caister Academic Press, Norfolk
27. Jasser I, Callieri C (2017) Picocyanobacteria: the smallest cell-size cyanobacteria. Handbook of cyanobacterial monitoring and cyanotoxin analysis. Wiley, Chichester
28. Siuda W (2008) Fosfor, azot i węgiel a ewolucja statusu troficznego Wielkich Jezior Mazurskich. Ochrona i Rekultywacja Wód Wielkich Jezior Mazurskich Narzędziem Rozwoju Naukowego, Gospodarczego, Społecznego i Kulturowego Regionu. Wydawnictwo Instytutu Rybactwa Śródlądowego, Olsztyn, pp 110–115. [In Polish]
29. Mikulski Z (1966) Bilans wodny Wielkich Jezior Mazurskich. Państwowy Instytutu Hydrologiczno-Metorologiczny, Warszawa. [In Polish]
30. Bajkiewicz-Grabowska E (2008) Obieg wody w systemie Wielkich Jezior Mazurskich. Ochrona i Rekultywacja Wód Wielkich Jezior Mazurskich Narzędziem Rozwoju Naukowego, Gospodarczego, Społecznego i Kulturowego Regionu. Wydawnictwo Instytutu Rybactwa Śródlądowego, Olsztyn. [In Polish]
31. Bukowska A, Bielczynska A, Karnkowska-Ishikawa A, Chrost RJ, Jasser I (2014) Molecular (PCR-DGGE) versus morphological approach: analysis of taxonomic composition of potentially toxic cyanobacteria in freshwater lakes. Aquat Biosyst 10:2–11
32. Napiórkowska-Krzebietke A (2015) Cyanobacterial bloom intensity in the ecologically relevant state of lakes – an approach to Water Framework Directive implementation. Oceanol Hydrobiol Stud 44:97–108
33. Chróst RJ (1975) Inhibitors produced by algae as an ecological factor affecting bacteria in water ecosystem. I. Dependence between phytoplankton and bacteria development. A Microbiol Pol 7:125–133

34. Spodniewska I (1978) Phytoplankton as the indicator of lake eutrophication. I. Summer situation in 34 Masurian lakes in 1973. Ekol Pol 26:53–70
35. Spodniewska I (1979) Phytoplankton as the indicator of lake eutrophication. II. Summer situation in 25 Masurian lakes in 1976. Ekol Pol 27:481–496
36. Napiórkowska-Krzebietke A, Hutorowicz A (2005) Long-term changes of phytoplankton in Lake Mamry Północne. Oceanol Hydrobiol Stud 34:217–228
37. Mankiewicz J, Komárková J, Izydorczyk K, Jurczak T, Tarczynska M, Zalewski M (2005) Hepatotoxic cyanobacterial blooms in the lakes of northern Poland. Environ Toxicol 20:499–506
38. Napiórkowska-Krzebietke A, Hutorowicz A (2006) Long-term changes of phytoplankton in Lake Niegocin, in the Masurian Lake region, Poland. Oceanol Hydrobiol Stud 35:209–226
39. Napiórkowska-Krzebietke A, Hutorowicz A (2007) Long-term changes in the biomass and composition of phytoplankton in a shallow, flow-through lake Kirsajty (Masurian Lakeland, Poland). Pol J Nat Sci 22:512–524
40. Jasser I, Karnkowska-Ishikawa A, Kozłowska E, Królicka A, Łukomska-Kowalczyk M (2010) Composition of picocyanobacteria community in the Great Mazurian Lakes: isolation of phycoerythrin-rich and phycocyanin-rich ecotypes from the system – comparison of two methods. Pol J Microbiol 59:21–31
41. Jakubowska N, Zagajewski P, Gołdyn R (2013) Water blooms and cyanobacterial toxins in lakes. Pol J Environ Stud 22:1077–1082
42. Bukowska A, Kaliński T, Koper M, Kostrzewska-Szlakowska I, Kwiatowski J, Mazur-Marzec H, Jasser I (2017) Predicting blooms of toxic cyanobacteria in eutrophic lakes with diverse cyanobacterial communities. Sci Rep 7:8342
43. Siuda W, Kauppinen ES, Kaliński T, Chróst RJ, Kiersztyn B (2017) The relationship between primary production and respiration in the photic zone of the Great Mazurian Lakes (GMLS), in relation to trophic conditions, plankton composition and other ecological factors. Pol J Ecol 65:303–323
44. Komárek J, Kaštovský J, Mareš J, Johansen JR (2014) Taxonomic classification of cyanoprokaryotes (cyanobacterial genera) 2014, using a polyphasic approach. Preslia 86:295–335
45. Kauppinen ES (2013) Trophic state of the Great Masurian Lakes system in the past, present and future – causes mechanisms and effects of changes. PhD thesis, Faculty of Biology, University of Warsaw, Warszawa
46. Hillbricht-Ilkowska A, Spodniewska I (1969) Comparison of the primary production of phyto-plankton in three lakes of different trophic type. Ekol Pol 17:241–261
47. Spodniewska I (1986) Planktonic blue-green algae of lakes in north-eastern Poland. Ekol Pol 34:151–183
48. Chróst RJ, Siuda W (2006) Microbial production, utilization, and enzymatic degradation of organic matter in the upper trophogenic water layer in the pelagial zone of lakes along the eutrophication gradient. Limnol Oceanogr 51:749–762
49. Krzywosz W (1993) Long-term changes in the composition and biomass of the phytoplankton in lake Niegocin (Great Masurian Lakes, Poland). Arch Ryb Pol 1:187–203
50. Tarczyńska M, Mankiewicz-Boczek J (2005) Risk of microcystins occurrence in water, guidelines of WHO. Ecohydrol Hydrobiol 5:7. [In Polish]
51. Maršálek B, Bláha L, Babica P (2003) Analyses of microcystins in the biomass of *Pseudanabaena limnetica* collected in Znojmo reservoir. Czech Phycol 3:195–197
52. Carmichael WW, Li R (2006) Cyanobacteria toxins in the Salton Sea. Saline Syst 2:5
53. Humpage A (2008) Toxin types, toxicokinetics and toxicodynamics. Cyanobacterial harmful algal blooms: state of the science and research needs. Springer Science, New York
54. Furtado ALFF, Calijuri MD, Lorenzi AS, Honda RY, Genuario DB, Fiore MF (2009) Morphological and molecular characterization of cyanobacteria from a Brazilian facultative wastewater stabilization pond and evaluation of microcystin production. Hydrobiologia 627:195–209

55. Santos MCR, Muelle H, Pacheco DMD (2012) Cyanobacteria and microcystins in lake Furnas (S. Miguel Island-Azores). Limnetica 3:107–118
56. Boopathi T, Ki J-S (2014) Impact of environmental factors on the regulation of cyanotoxin production. Toxins 6:1951–1978
57. Kokocinski M, Gągała I, Jasser I, Karosiene J, Kasperoviciene J, Kobos J, Koreiviene J, Soininen J, Szczurowska A, Woszczyk M, Mankiewicz-Boczek J (2017) Distribution of invasive *Cylindrospermopsis raciborskii* in the East-Central Europe is driven by climatic and local environmental variables. FEMS Microbiol Ecol 93:fix035
58. Kurmayer R, Christiansen G (2009) The genetic basis of toxin production in cyanobacteria. Fr Rev 2:31–50
59. Rücker J, Stüken A, Nixdorf B, Fastner J, Chorus I, Wiedner C (2007) Concentrations of particulate and dissolved cylindrospermopsin in 21 *Aphanizomenon*-dominated temperate lakes. Toxicon 50:800–809
60. Ballot A, Fastner J, Wiedner C (2010) Paralytic shellfish poisoning toxin-producing cyanobacterium *Aphanizomenon gracile* in Northeast Germany. Appl Environ Microbol 76:1173

Microorganisms as Sanitary State Bioindicators of Flowing Waters in Poland

Iwona Gołaś, Anna Gotkowska-Płachta, and Jacek Arkadiusz Potorski

Contents

Abstract The article presents changes in the quantitative and qualitative composition of indicator bacteria in three Polish rivers – Vistula, Oder, and Łyna River – in the last 30 years. The analyzed rivers are situated in the catchment area of the Baltic Sea, and they play important economic, tourist and environmental roles. The sanitary and bacteriological status of these ecosystems was determined based on the counts of total coliform bacteria (TCB), fecal coliform bacteria (FCB), fecal enterococci bacteria (FEB), and *Escherichia coli*. The study involved a review of the literature and the reports published by Polish inspectorates of environmental protection. Water samples collected from all rivers in urban areas were characterized by the highest TCB, FCB, FEB, and *E. coli* counts throughout the analyzed period. High contamination levels contributed to epidemiological risks in rivers, in particular in the proximity of discharge points of treated effluents. As a result, most of the analyzed water samples were graded as purity class III and/or as non-class (polluted beyond permissible standards) waters. Since 2003, Polish laws have been harmonized with

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the provisions of the Water Framework Directive (2000/60/EC), and the range of indicator bacteria was restricted to FEB and *E. coli* only in water bodies used as sources of water for human consumption and/or for bathing. In Poland, treated wastewater does not have to be disinfected before it is discharged to surface waters. For this reason, contamination with indicator bacteria (TCB, FCB, FEB, and *E. coli*) continues to be high in water bodies that act as receptacles of wastewater despite considerable reduction of microbiological contamination (97–99%) in treated effluents. The above applies particularly to rivers where contamination spreads. For this reason, the concentration of indicator bacteria should be the main criterion in evaluations of river water purity.

Keywords Coliform bacteria · Fecal bacteria · Indicator bacteria · River · Water

1 Introduction

River ecosystems are dynamic habitats that undergo constant and often dramatic changes from the source to the river mouth. These changes are associated with the natural flow of rivers as well as natural and anthropogenic runoffs. Rivers are contaminated with surface runoffs from agricultural land as well as with water and effluents from urban areas [1–5]. These sources of contamination carry fecal pathogens such as viruses, bacteria, and protozoa to rivers and decrease the microbiological quality of these ecosystems. The above increases epidemiological risks and makes river water less suitable for human consumption, industrial use, and recreation [6–9]. The bacteriological quality and safety of rivers should be regularly controlled to minimize the relevant risks. The identification of all pathogens is very difficult, laborious, and costly, which is why microbiological evaluations rely on indicator microorganisms. Indicator bacteria have been long used to determine the contamination of water with fecal bacteria, intestinal parasites, and pathogens. Indicator bacteria include total coliform bacteria (TCB), fecal coliform bacteria (FCB), and fecal enterococci bacteria (FEB) which are saprophytes that colonize the digestive tract of humans and animals [10–12]. These bacteria are also used as microbiological indicators because they are absent from waters that are not contaminated with fecal matter, they have longer survival times than pathogens, and bacterial counts are always higher than pathogen counts and are proportional to the degree of contamination. Indicator bacteria are also more resistant to disinfectants, and they do not proliferate, or their proliferation is inhibited in aquatic environments. Indicator bacteria are also used in microbiological evaluations of water ecosystems because they can be detected with the involvement of simple, inexpensive, and relatively rapid laboratory methods that produce repeatable results.

Water is essential for human life and the economy, which is why evaluations of the microbiological safety of water and the protection of water resources are the priority concerns in all countries around the world. In Poland and other European Union member states, the Water Framework Directive (WFD) is the leading document which sets the standards for managing the quality of surface waters [13]. In the WFD, water assessments are based on living organisms as biological indicators of the ecological status of aquatic environments. The main emphasis is placed on the quantitative and qualitative composition of phytoplankton, phytobenthos, macrophytes, zoobenthos, and ichthyofauna. However, these organisms are not direct indicators of fecal contamination, whereas microbiological indicators (FCB, *E. coli*, FEB) are the most sensitive markers of fecal pollution. Unfortunately, according to the WFD [13], indicator bacteria have to be identified only in assessments of drinking water and in bathing areas. The Polish laws relating to water quality monitoring had to be harmonized with the provisions of the WFD [13], and as a result, indicator bacteria no longer have to be identified in assessments of surface water. For this reason, the presence of TCB and FCB in water is determined by very few scientific institutions in Poland. These microorganisms act as vectors in the transmission of various genes, and they are good bioindicators of fecal contamination and epidemiological risks in aquatic ecosystems [1–4, 6, 7, 14, 15]. Indicator bacteria play a particularly important role in river assessments. Rivers flow through areas characterized by various levels of anthropogenic pressure; they carry microbiological pollutants and increase epidemiological risks in the seas into which they are emptied. Rivers Vistula, Oder, and Łyna River are such lotic ecosystems which play a very important role in Poland. The microbiological status of rivers of key economic significance should be monitored regularly to prevent adverse changes in water quality. The identification of fecal contaminants supports the initiation of remedy measures to restore the bacteriological safety of rivers.

In view of the significance of indicator bacteria in aquatic habitats, the aim of this study was to analyze changes in the sanitary and bacteriological status of three major Polish rivers in the past 30 years. The study was conducted based on a review of the literature and the reports published by Polish inspectorates of environmental protection.

2 Materials and Methods

2.1 Study Area

Spatial and temporal changes in the counts of indicator bacteria were determined in water samples collected from three Polish rivers: Vistula, Oder, and Łyna River. The rivers are situated in the Baltic Sea catchment, and they play very important roles in the economy, the tourism industry, and the environment. The Vistula and the Oder are the largest Polish rivers that span more than 80% of the national territory. Łyna River is the largest river in northeastern Poland.

2.1.1 Vistula River

The Vistula is the longest Polish river and the largest river in the Baltic Sea catchment. It has a length of 1,048 km, and it passes through eight Polish regions. The Vistula has its source in Beskid Śląski, and it empties into the Gdańsk Bay of the Baltic Sea. Its upper course and tributaries cover 54.0% of Poland's territory. The drainage basin of the Vistula has an area of 168,700 km^2 [16]. The Vistula is composed of three principal sections: upper, middle, and lower. Rivers San and Bug are its largest tributaries. In the upper section, the Vistula has the characteristic features of a mountain river with a high gradient (up to 5%) and an uneven profile (numerous rapids and cascades), and it carries significant quantities of rock debris. The middle section has an estimated length of 210 km. The river channel widens to 25 m; it has a meandering course and is deprived of the characteristic features of a mountain river. The middle Vistula consists of a network of braided channels separated by numerous islands, oxbow lakes, and side canals. The Vistula forms a large delta in the region of Żuławy, around 50 km from its estuary. The delta forks out into two rivers, Leniwka and Nogat. The Nogat River empties into the Vistula Lagoon, and the section of Leniwka known as the Dead Vistula flows into the Gdańsk Bay. The Vistula has 20 tributaries, and the major tributaries are the following rivers: Soła, Dunajec, San, Wieprz, Narew, Drwęca, Pilica, Bzura, and Brda. The Vistula flows through large Polish cities, including Cracow, Sandomierz, Warsaw, Płock, Toruń, Grudziądz, Świecie, and Tczew. It is characterized by significant changes in water level and the movement of rock debris from soil erosion. Agricultural land stretching along Vistula's course is used mainly for cattle grazing. The Vistula is an important north-south ecological corridor in Europe. This natural and untamed river contributes to the movement of living organisms between Scandinavia and Africa. The Vistula is a popular tourist and recreational destination [17].

2.1.2 Oder River

The Oder is Poland's second largest river in terms of length (742 km) and drainage basin area (106,000 km^2). Oder's drainage basin covers approximately 34% of the national territory. The river begins in the Oder Mountains in the Czech Republic, and it empties into the Szczecin Lagoon. The river's lower course delineates the Polish-German border. Nearly 55% of Oder's catchment is situated at an altitude of 100–300 m asl and 21% above 300 m asl. The catchment area covers nearly the entire Sudetes mountain range, and most rain water is carried downstream. The Oder has upper, middle, and lower sections. The upper Oder has a length of 272.3 km between the source and the tributary of Nysa Kłodzka. The middle Oder has a length of 374.8 km between Nysa Kłodzka and Gozdowice. The lower Oder spans a distance of 95 km between Gozdowice and the Szczecin Lagoon [18]. Oder's main tributaries on Polish territory are the following rivers: Nysa Kłodzka, Oława, Ślęża, Bystrzyca, Kaczawa, Bóbr, Nysa Łużycka, Kłodnica, Mała Panew, Stobrawa, Widawa, Barycz, Warta, Myśla, and Ina [19]. The Oder flows through several large Polish cities: Racibórz, Kędzierzyn-Koźle, Opole, Wrocław, Krosno Odrzańskie, Słubice, and Szczecin. The river is characterized by a high average flow rate of 100–1,000 m^3/s, which contributes to high water levels, particularly in late spring and summer, and frequently causes floods [20]. The Oder is supplied with both treated and untreated wastewater, including from illegal discharge points, along its entire course [21, 22]. Due to considerable pollution, the Oder is not used for recreational purposes and poses a significant environmental threat. The river's main tributaries and inadequate management of effluents throughout the entire catchment area indirectly contribute to the poor quality of river water. In the middle and lower sections, the water quality is influenced mostly by the inflow of pollutants from Upper and Lower Silesia, Germany, and the Czech Republic. The Oder is one of the major Polish rivers, and its valley is an important ecological corridor of great natural value. There are seven national parks in the river's drainage basin. Numerous oxbow lakes, canals, and marshes are the habitats of valuable plant and bird species. The Oder is regulated along its entire course, and it is the major inland waterway in Poland which is used by Central European countries without sea access.

2.1.3 Łyna River

The Łyna River is the main river of the region of Warmia and Mazury in northeastern Poland. It rises near the village of Łyna River at an altitude of 153 m asl in the Łyna River Springs nature reserve. The reserve is characterized by retrogressive erosion, a rare phenomenon in lowland areas. The Polish section of the Łyna River has a length of 190 km, and the catchment basin spans an area of 5,700 km^2. Outside Poland, the Łyna River feeds into the Pregola River in the Kaliningrad Region of Russia. The Łyna River has a total length of 263.7 km. It flows through the following ribbon lakes: Brzeźno Duże, Kiernoz Mały, Kiernoz Wielki, Łańskie Lake, and Ustrych. The Łyna River is a lakeland river which frequently changes its direction. It flows slowly, forms numerous meanders, and crosses ravines where it resembles a mountain stream. The width of the river valley ranges from less than 20 m to around 5 km, and the average valley depth is 1.5–2.5 m. The Łyna River has an average annual flow rate of 7–35 m^3/s, and the highest values are noted in spring [1, 23, 24]. The river flows through forests, meadows, agricultural land, and urban areas. Arable land, meadows, and pastures occupy approximately 53% of the catchment basin, and the share of forests exceeds 26% [24]. In urban areas, the Łyna River crosses numerous villages and five cities: Olsztyn, Dobre Miasto, Lidzbark Warmiński, Bartoszyce, and Sępopol. The river is a receptacle of treated wastewater from urban areas (approx. 41,000 m^3/day) which is the main point source of pollution [25]. The upper section of the river intersects the Warmia Forest nature reserve (with an area of 1,600 ha) which is inhabited by many protected animal and plant species. The forest is part of the Łyna River Middle Valley protected landscape area. In its lower course, the Łyna River crosses the Sępopol Plain which is part of the Ostoja

Warmińska refuge of the Natura 2000 network of nature protection areas [26]. Due to its unique geomorphological and geographic features, the Łyna River is a tourist attraction and a popular kayaking destination.

2.2 Sampling Sites

Water samples from the Vistula, Łyna River, and Oder were collected for microbiological analyses in various periods from 37 sites at the most characteristic points, depending on the scope of the analysis and research objectives. The location of sampling sites is presented in Fig. 1.

The studied section of the Vistula had a length of 153 km. Water samples were collected between the cities of Wyszogród and Toruń [27] from April 1999 to February 2000 at five sampling sites:

– Site 1 – Wyszogród (582 and 588 km along Vistula's course)
– Site 2 – Płock (632 km along Vistula's course)

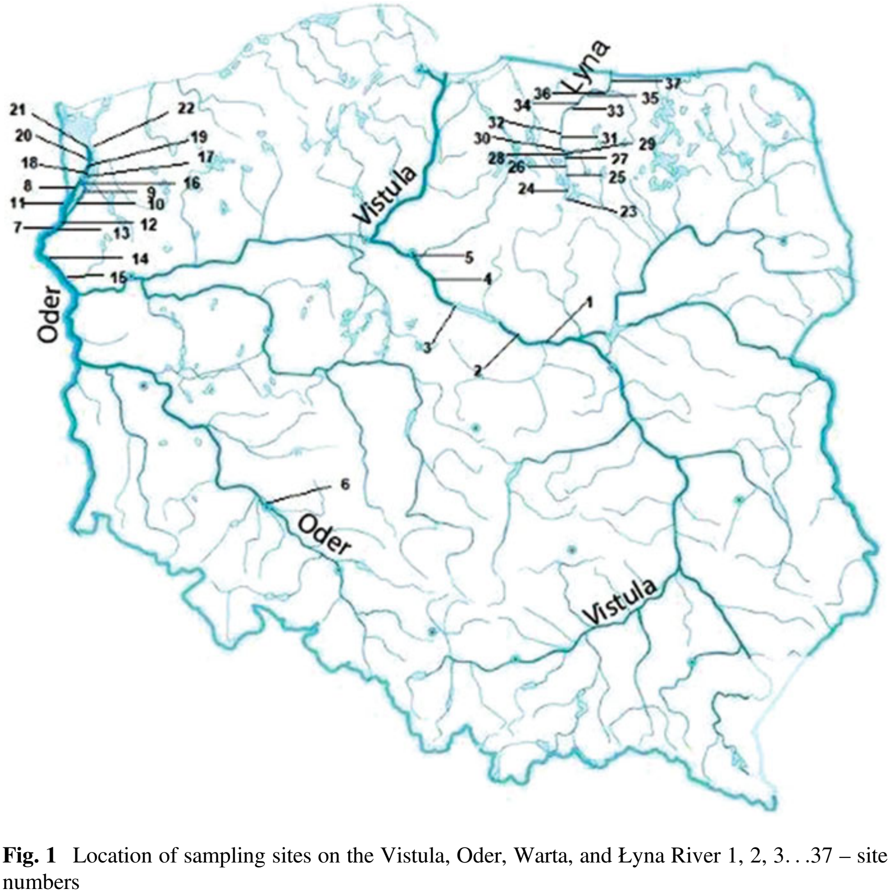

Fig. 1 Location of sampling sites on the Vistula, Oder, Warta, and Łyna River 1, 2, 3. . .37 – site numbers

– Site 3 – Włocławek (674 km along Vistula's course)
– Site 4 – Nieszawa (695 km along Vistula's course)
– Site 5 – Toruń (735 km along Vistula's course)

Water purity in the Oder was monitored by collecting water samples in the following sites:

– Site 6 – stream gauge at 249 km along Oder's course, situated directly upstream from Wrocław, in 1992 and 2001 [22]
– Site 7 – stream gauge in Krajnik Dolny at 690 km along Oder's course, in 2007–2011 [28]
– Sites 8–20 (13 sampling sites) – situated near Szczecin, between stream gauges in the towns of Gozdowice (645.3 km along Oder's course) and Police, from April 1998 to March 1999 [29]
– Sites 21 and 22 – situated in municipal bathing areas at the point where the Oder empties into the Szczecin Lagoon in the towns of Stępnica and Trzebież, from May to October 2006 [9]

Water samples from the Łyna River were collected for microbiological analyses between the winter of 2011 and the fall of 2012 in 15 sampling sites between the river source and the Polish-Russian border:

– Sites 23–37 – situated in headwaters, Kurki, Ustrych, Ruś, Posorty, Olsztyn, Redykajny, Knopin, Kosyń, Lidzbark, Bartoszyce, Sępopol, and Stopki [3]

2.3 Water Sampling Procedure

Water samples from the Vistula (sites 1–5) and the Łyna River (sites 23–37) were collected in spring, summer, fall, and winter at 3-month intervals. Water samples from the Oder (sites 6–23) were collected at monthly intervals.

Water samples from rivers Vistula, Oder, and Łyna River were collected from the main stream at a depth of 0.3–0.5 m. The samples were collected directly into sterile glass bottles according to the Polish Standards [30] and Standard Methods [31]. In most cases, water samples were analyzed within 24 h after collection, and they were stored at 4°C until analysis.

2.4 Microbiological Analyses

Water samples from the Vistula, Oder, and Łyna River were subjected to microbiological analyses to determine the counts of indicator bacteria: TCB, FCB, FEB, and *E. coli*. The analytical methods are presented in Table 1.

Table 1 Microbiological analyses of Polish rivers water samples

Indicator bacteria	Units	Method recommendation	River	Number of sampling sites	References
TCB	Titer	Polish standard [32]	Vistula	1–5	[27]
	cfu[a]/100 mL	Polish standard [33]	Oder	21–22	[9]
		Polish standard [34]	Łyna River	23–37	[3]
FCB	MPN[b]/100 mL	Polish standard [35]	Vistula	1–5	[27]
			Oder	8–20	[29]
	Titer	Polish standard [35]	Vistula	1–5	[27]
			Oder	6	[22]
			Oder	7	[28]
	cfu/100 mL	Polish standard [34]	Łyna River	23–37	[3]
E. coli	cfu/100 mL	Polish standard [33]	Oder	21–22	[9]
FEB	cfu/100 mL	Polish standard [36]	Vistula	1–5	[27]
			Łyna River	23–37	

[a]Colony forming units
[b]Most Probable Number

3 Results and Discussion

3.1 *Bacterial Indicators of Water Quality (TCB, FCB, and FEB) in the Vistula River*

Water samples collected from the Vistula were analyzed by Donderski and Wilk [27] who observed considerable variations in the levels of indicator bacteria (TCB, FCB, FEB) along the river's course. Fecal coliform (FCB) counts differed across sampling sites and were highest (2.4×10^4 MPN/100 mL) at sites 2 and 5 in the vicinity of the large cities of Toruń and Płock (Fig. 2) According to Geldreich [37], FCB titers higher than 2,000 point to the presence of *Salmonella* bacteria in the aquatic environment with nearly 100% probability. In the Vistula, this critical value was significantly exceeded in the sampling sites in Toruń and Płock, thus contributing to epidemiological risks in those regions. Coliform titers (TCB and FCB) were determined at 0.0014 and 0.004, which points to fresh contamination with fecal matter and the non-class grade of the evaluated samples [38]. In the remaining water samples from the Vistula, FCB counts were around tenfold lower and did not exceed 2.5×10^3 MPN/100 mL, with titer values of 0.008 (TCB) to 0.04 (FCB). The above samples were moderately contaminated and were graded as purity class III waters [38]. A rising trend in the counts of fecal enterococci (FEB) was also observed in the water samples from the Vistula. The counts of FEB ranged from several hundred to several thousand colony forming units (cfu) per 100 mL, with a rising trend in the proximity of large cities. The FCB and FEB identified in the samples from the lower Vistula were saprophytic enteric bacteria that colonize the digestive tract of humans

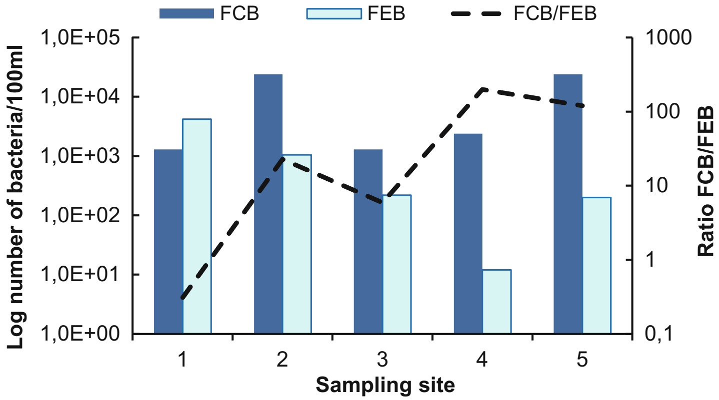

Fig. 2 Number of fecal coliforms (MPN FCB/100 mL) and fecal enterococci (cfu FEB/100 mL) and the FCB/FEB ratio in water sampled from the Vistula

and higher animals. Their presence in the aquatic environment is indicative of fresh fecal contamination, and these bacteria could be accompanied by other pathogens from diseased individuals. Fecal coliforms and fecal enterococci are reliable indicators of water quality. Their counts are indicative not only of direct fecal contamination but also of the type of effluents discharged to a body of water. The source and origin of fecal pollutants can be identified based on the FCB/FEB ratio. Fecal coliforms account for 96.4% of human fecal microbiota [31], whereas FEB are the predominant microorganisms in the feces of both domesticated and wild animals [37]. For this reason, FCB/FEB ratios below 0.7 point to animal sources of contamination, and ratios of 0.7–4.0 are characteristic of mixed (human and animal) contamination. Waters are contaminated with human feces when the FCB/FEB ratio exceeds 4.0. In the Vistula, the above ratio exceeded 4.0 in most samples, which points to the human origin of fecal contamination. The analyzed ratio was determined at 0.31 only in the typically agricultural municipality of Wyszogród (site 1) where farmland occupies more than 62% of municipal territory, and it was indicative of fecal pollution of animal origin (Fig. 2).

3.2 Counts of Fecal Coliform Bacteria (FCB) and E. coli in the Oder River

The analyses of water samples collected from the Oder in the area of Wrocław (site 6) revealed that safe levels of indicator bacteria were exceeded in both 1992 and 2001 [22]. In 1992, FCB titers were below 0.01 in most samples and were determined at 0.01 to 0.1 in the remaining samples. Based on the provisions of the Regulation of the Minister of Environmental Protection, Natural Resources, and Forestry, the above samples were graded as purity class III (40%) and as non-class (60%) waters. In 1992, elevated FCB titers in the Oder were accompanied by higher BOD5 values and increased concentrations of dissolved substances. In most of the studied samples, the above parameters were characteristic of purity class III and non-class waters [22]. Fecal contamination was also strongly correlated with the analyzed physicochemical parameters of water in Łyna River [1, 3] and Drwęca [2]. Similar trends have been described in the international literature [39–41]. These results indicate that land use types in catchment areas and the physicochemical parameters of water bodies significantly influence the concentrations of fecal bacteria in aquatic ecosystems. In 2001, a minor improvement was noted in the microbiological quality of the Oder. A total of 26 water samples were analyzed. In 5% of the samples, FCB titers were determined at 0.1–1.0, and they were graded as waters of purity class II. In 55% of the samples, FCB titers ranged from 0.1 to 0.01, and they were graded as waters of purity class III, whereas 40% of the samples were graded as non-class (Fig. 3).

The minor improvement in the microbiological quality of the Oder in 2001 could probably be attributed to the collapse of local industrial plants which had polluted the river and its tributaries. Plant closure decreased the quantity of wastewater discharged to the river. The construction and modernization of wastewater treatment plants also contributed to the quality of the aquatic ecosystem in the Oder [22].

The results of a monitoring study conducted in the Oder in 2007–2011 (an evaluation of the surface waters in the region of Western Pomerania) [28] revealed changes in the microbiological quality of water sampled in Krajnik Dolny (site 7). The results of FCB titers differed by several orders in magnitude between sampling periods. The lowest levels of microbial contamination were noted in 2011 when FCB titers ranged from 0.1 to 0.43. Microbial contamination was highest (FCB titers of 0.0001–0.2) in 2007–2009 (Fig. 4).

In 2010–2011, an increase in the above parameter testified to a decrease in FCB concentration in the Oder. The observed improvement in the river's microbiological purity resulted from the construction of new wastewater treatment plants and the growing popularity of organic farming, which reduced the quantity of wastewater,

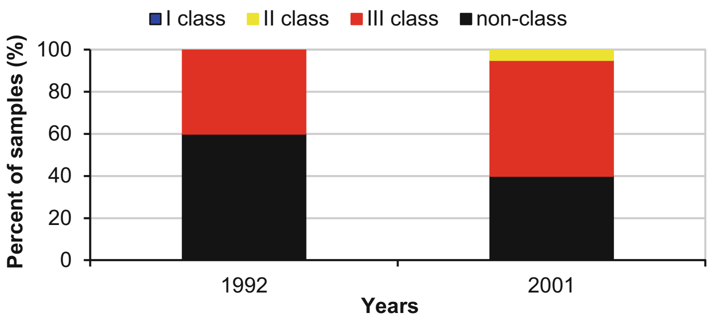

Fig. 3 Percentage distribution of water samples collected from the Oder River upstream of the city of Wrocław (site 6) into water purity classes meeting the requirements of the Regulation of the Minister of Environmental Protection, Natural Resources, and Forestry [38]

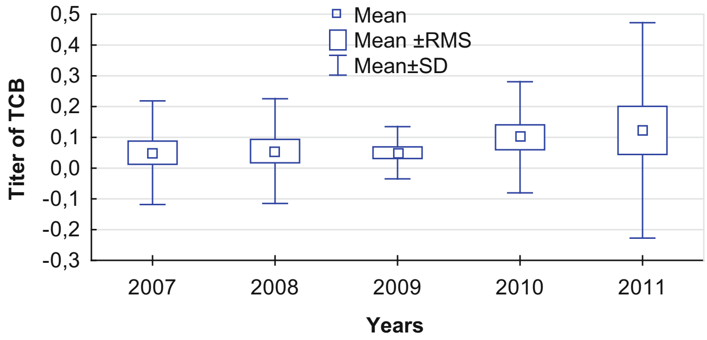

Fig. 4 Mean and standard deviation (±SD) of FCB titers in water samples collected from the Oder in the village of Krajnik Dolny near Szczecin (site 7) in 2007–2011. RMS – random mean square

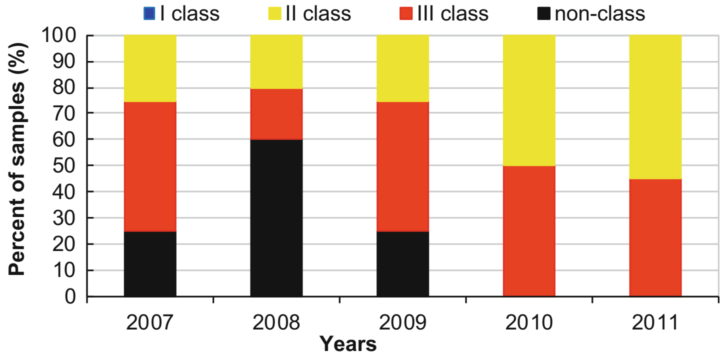

Fig. 5 Percentage distribution of water samples collected from the Oder in Krajnik Dolny into water purity classes meeting the requirements of the Regulation of the Minister of Environmental Protection, Natural Resources, and Forestry [38]

surface runoffs, and rainwater evacuated to the Oder [28]. These processes increased the percentage of water samples graded as waters of purity class II [38] from 20–25% in 2007–2009 to 50–55% in 2010–2011 (Fig. 5). Land-use type and the degree of urbanization and industrialization in the catchment area significantly influence the contamination of surface waters with heterotrophic microorganisms, including FCB. In new and upgraded wastewater treatment plants, the abundance of fecal bacteria in treated wastewater can be effectively reduced by 97–99% relative to raw wastewater [7, 42]. Despite the above, an increase in their abundance is not inhibited in water bodies that are receptacles of treated wastewater. The above was confirmed by Godela et al. [43] who analyzed water samples from the Warta River in Częstochowa. The above authors observed that treated wastewater discharged by the WARTA sewage treatment plant increased *E. coli* counts in the analyzed river. An increase in *E. coli* and FCB counts was also noted in the Łyna River which receives treated wastewater from five urban areas situated on its banks [1, 3].

In 1998–1999, the microbiological analyses of water sampled from the Oder in the area of Szczecin revealed differences in the MPN values of FCB which reached several orders of magnitude [29]. Samples collected at sites 8, 9, 10, and 11 above Szczecin were characterized by the lowest fecal contamination. Fecal coliform counts ranged from 1.1×10^2 to 1.0×10^3/100 mL in the vast majority of the samples. Higher values ($\geq 1.0 \times 10^3$ MPN/100 mL) were determined only sporadically in water sampled from sites 8, 9, 10, and 11. Microbiological contamination was somewhat higher in samples of Oder water collected in the Cedynia Landscape Park (sites 12, 13, 14, 15) and in an industrial area (sites 19, 20) where FCB counts ranged from 1.1×10^3 to 1.0×10^4/100 mL in more than 50% of the samples. Water samples from a municipal area (sites 16, 17, 18) were most contaminated, and most of them were characterized by FCB counts higher than 1.1×10^4/100 mL. These findings indicate that anthropogenic pressure is one of the key determinants of water purity, and it is associated with the manner in which aquatic ecosystems and their catchment areas are used as well as the observance of rigorous standards in wastewater treatment plants. Other authors also reported very high levels of fecal bacteria in water bodies receiving industrial and municipal effluents [1–3, 15, 44–47]. In view of the provisions of the Regulation of the Minister of Environmental Protection, Natural Resources, and Forestry [38], most water samples collected from the Oder upstream of Szczecin and in the Cedynia Landscape Park corresponded to purity class II. In contrast, 60–92% of the samples collected in the municipal area were non-class waters (Fig. 6). Water samples characterized by high FCB counts ($>1.1 \times 10^3$ MPN/100 mL) were also contaminated with *Campylobacter* spp. bacteria, which increased the epidemiological risk. The presence of *Campylobacter* spp. was significantly correlated with water purity class [29]. Microbiological contamination with other potentially pathogenic bacteria, including *Pseudomonas aeruginosa* and *Aeromonas hydrophila*, also significantly contributed to epidemiological risks in other rivers in Poland [2, 6, 7, 48–51] and the world [52–54]. In view of the above, surface waters can be regarded as potential reservoirs and important vehicles for bacterial transmission [29].

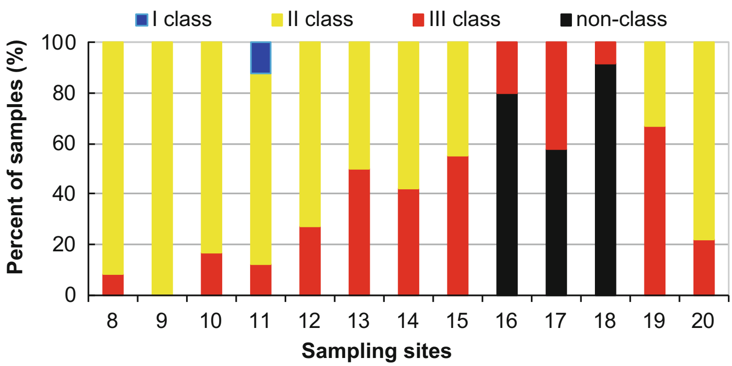

Fig. 6 Percentage distribution of water samples collected from the Oder near Szczecin in 1998–1999 into water purity classes meeting the requirements of the Regulation of the Minister of Environmental Protection, Natural Resources, and Forestry [38]

Point and nonpoint sources of pollution in the Oder River increased fecal contamination levels in municipal bathing areas where the river discharges into the Szczecin Lagoon [9]. In water samples collected from two municipal bathing areas in the summer, *E. coli* counts increased more than 30-fold. In samples collected from bathing areas in Trzebież (site 21) and Stępnica (site 22), average *E. coli* counts were determined at 110 and 150 cfu/100 mL, respectively. In the samples collected from both sites, the increase in *E. coli* was accompanied by a rise in TCB counts (Fig. 7).

The results were analyzed in the 3D ocean model and the Lagrangian particle tracking model to reveal that the observed increase in TCB and *E. coli* concentrations resulted from insufficient wastewater treatment in the city of Szczecin. Weather and hydrological conditions were responsible for differences in bacterial transfer within a radius of up to 20 km [9]. Numerous authors have reported that the microbiological contamination of rivers increases downstream. The above poses a significant problem in rivers that flow through large urban areas. In Poland, treated effluents can be reservoirs of bacteria because even highly advanced treatment processes that effectively remove phosphorus and nitrogen do not completely eliminate bacterial pathogens from wastewater [42]. The provisions of the Regulation of the Minister of the Environment on the requirements for the evacuation of wastewater to water and ground [55] and successive amendments [56, 57] do not account for microbiological parameters and do not place wastewater treatment plants under the obligation to disinfect effluents in the last stages of treatment. Treated wastewater is not disinfected before it is discharged, which systematically increases microbiological contamination in Polish rivers [58]. In consequence, rivers are reservoirs of microbiological pollutants, and they increase the epidemiological risk in the seas into which they are emptied. In Poland, long-term studies revealed growing levels of microbiological pollution in the Łyna River [1, 3], Drwęca [2], and Vistula [27]. Similar observations were made by Kacar [4] who found that the five largest rivers in Turkey (Meriç, Bakırçay, Gediz, Küçükmenderes, and

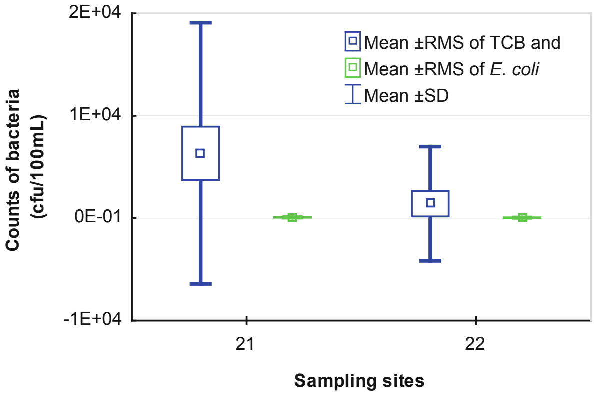

Fig. 7 Mean and standard deviation (±SD) counts of TCB and *E. coli* (cfu/100 mL) in municipal bathing areas in Trzebież (site 21) and Stępnica (site 22) at the Oder estuary in the Szczecin Lagoon in 2006. RMS – random mean square

Büyük Menderes) were the main routes of FCB and FEB transmission to the Aegean Sea. Kirschner et al. [59] demonstrated that fecal bacteria carried by the Danube, a river that flows through ten countries, pose epidemiological risks on an international scale. Goel et al. [60] found that fecal contamination in the River Yamuna in India increased the prevalence of various diseases. They observed that an increase in TCB and FCB counts in the Yamuna was accompanied by a rise in the abundance of pathogenic parasites, helminth eggs, and coliphages.

3.3 Counts of Indicator Bacteria (TCB, FCB, and FEB) in the Łyna River

In the Łyna River, TCB, FCB, and FEB counts differed across sampling sites. The highest average FCB concentration (2.2–5.3 $\times$ 10^3 cfu/100 mL) was noted at sites 28, 29, and 30 in the area of Olsztyn (Fig. 8). According to the World Health Organization [12], an increase in the counts of heat-resistant fecal coliform bacteria (FCB) in excess of 1 $\times$ 10^3 cfu/100 mL can pose an epidemiological risk. The concentration of FEB was also elevated in the sections of the river flowing in urban areas, and it peaked (5.6 $\times$ 10^2 and 2.1 $\times$ 10^3 cfu/100 mL) in samples collected 200 m downstream from the discharge points of treated wastewater in Kosyń and Lidzbark Warmiński at sites 32, 34, and 35 (Gotkowska-Płachta, data not published).

The total counts of indicator bacteria (TCB, FCB, and FEB) were lowest in forests (sites 23, 24, 25, 26) where anthropogenic pressure was limited. At those

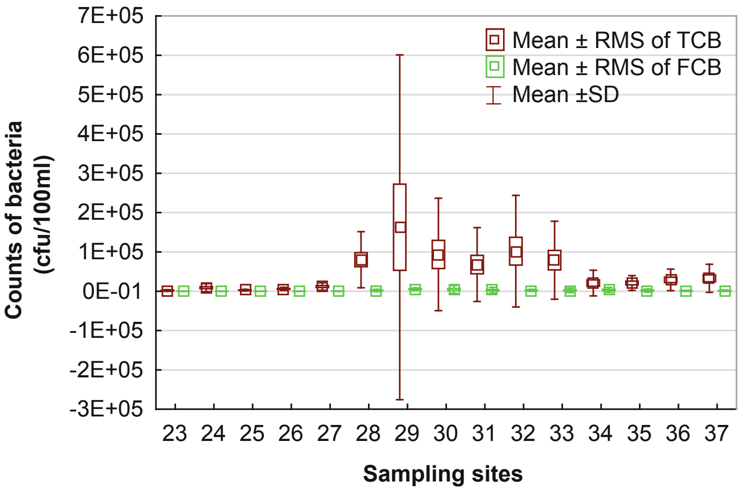

Fig. 8 Mean and standard deviation (±SD) counts of TCB, FCB (cfu/100 mL) in water samples collected from the Łyna River 2011–2012. RMS – random mean square

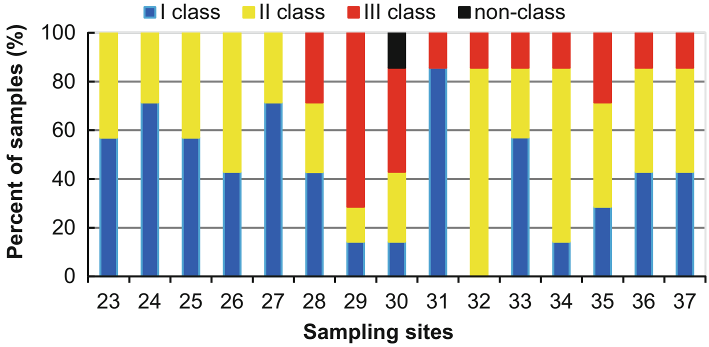

Fig. 9 Percentage distribution of water samples collected from the Łyna River in 2011–2012 into water purity classes meeting the requirements of the Regulation of the Minister of Environmental Protection, Natural Resources, and Forestry [38]

sites, bacterial counts were generally tenfold lower than in agricultural and urban areas. In most samples, the FCB/FEB ratio exceeded 7.0, which indicates that the river was contaminated mainly with fecal bacteria of human origin.

According to Feachem [61] and Geldreich [62], the FCB/FEB ratio is higher in human feces than in animal excreta, and this parameter is used to determine the origin of fecal contamination (human, animal, or mixed) in aquatic ecosystems.

Based on TCB, FCB, and FEB counts in the samples collected in forests (sites 23, 24, 25, 26), 43–71% of the samples were graded as waters of purity class I, and 29–57% of samples were graded as waters of purity class II [38]. In agricultural areas (sampling sites 27, 31, 36, and 37), 43–86%, 0–43%, and 0–14% of the samples were graded as waters of purity class I, class II, and class III, respectively. In urban areas (sampling sites 28, 29, 30, 32, 33, 34, 35), 0–57% of the samples were graded as class I waters, 14–86% of the samples – as class II waters, 14–71% of the samples – as class III waters, and 14% of the samples – as non-class waters (Fig. 9).

High abundance of indicator bacteria is often observed in urban areas where wastewater is discharged to rivers. Such observations were made in the Danube downstream from Budapest and Bucharest [59], in the Seine downstream from Paris [63], and in the Thames downstream from London [64]. According to the above studies, the decrease in water quality and the increase in the abundance of indicator bacteria are closely associated with the quantity and type of wastewater discharged to the river as well as river flow rates which influence the rate of purification processes. Regardless of their source, TCB, FCB, and FEB in surface waters and bottom deposits are indicative of fecal contamination of human or animal origin [46, 50, 63]. Indicator bacteria are very often accompanied by potentially pathogenic bacteria and pathogens such as *Aeromonas hydrophila*, *Pseudomonas aeruginosa*, *Listeria monocytogenes*, *Salmonella* spp., *Campylobacter* spp., *Vibrio* spp., and *Yersinia* spp. [8, 40, 65–68].

4 Conclusions

A review of the literature revealed quantitative and qualitative differences in the communities of indicator bacteria (TCB, FCB, FEB, and *E. coli*) in the Polish rivers Vistula, Oder, and Łyna River. Their counts differed by several orders of magnitude, depending on the analyzed river, sampling site, sampling period, and the applied analytical method. The abundance of indicator bacteria increased downstream in all rivers, regardless of the sampling period and the applied analytical method. In all evaluated water ecosystems, contamination with TCB, FCB, FEB, and *E. coli* was highest in samples collected from urban areas where municipal wastewater was the main source of pollution. The counts of indicator bacteria exceeded safe limits in the vicinity of municipal wastewater treatment plants, which could be attributed to the fact that Polish wastewater treatment plants are not legally required to disinfect effluents in the last stages of treatment [55–57]. Most water samples collected from the Vistula, Oder, and Łyna River downstream from wastewater discharge points were graded as purity class III waters and non-class waters. Despite the fact that treatment processes effectively eliminate microbiological contaminants (in 97–99%), the analyzed samples posed an epidemiological risk. Microbiological pollutants are transmitted downstream, which renders river water unsafe for health. Microbiological contamination also increases the concentrations of pathogenic bacteria and the pool of virulence genes, and it contributes to antibiotic resistance in water ecosystems. For this reason, the abundance of indicator bacteria should be the main criterion in evaluations of river water purity.

References

1. Glińska-Lewczuk K, Gołaś I, Koc J et al (2016) The impact of urban areas on the water quality gradient along a lowland river. Environ Monit Assess 188:624–638. https://doi.org/10.1007/s10661-016-5638-z
2. Gołaś I (2011) The effects of anthropogenic and environmental factors on the bacteriological quality of water in the upper reach of the Drwęca River. Dissertations and monographs 163. Publishing House of the University of Warmia and Mazury in Olsztyn, Olsztyn. [in Polish]
3. Gotkowska-Płachta A, Gołaś I, Korzeniewska E et al (2016) Evaluation of the distribution of fecal indicator bacteria in a river system depending on different types of land use in the southern watershed of the Baltic Sea. Environ Sci Pollut Res 23:4073–4085. https://doi.org/10.1007/s11356-015-4442-6
4. Kacar A (2011) Analysis of spatial and temporal variation in the levels of microbial fecal indicators in the major rivers flowing into the Aegean Sea, Turkey. Ecol Indic 11:1360–1365. https://doi.org/10.1016/j.ecolind.2011.02.010
5. Ma L, Mao G, Liu J et al (2016) Spatial temporal changes of bacterioplankton community along an Exhorheic River. Front Microbiol 7:250. https://doi.org/10.3389/fmicb.2016.00250
6. Harnisz M, Gołaś I, Pietruk M (2011) Tetracycline-resistant bacteria as indicators of antimicrobial resistance in protected waters – the example of the Drwęca River nature reserve (Poland). Ecol Indic 11:663–6682. https://doi.org/10.1016/j.ecolind.2010.09.005

7. Korzeniewska E, Harnisz M (2012) Culture-dependent and culture independent methods in evaluation of emission of *Enterobacteriaceae* from sewage to the air and surface water. Water Air Soil Pollut 223:4039–4046. https://doi.org/10.1007/s11270-012-1171-z
8. Rodrigues V, Ramaiah N, Kakti S, Samant D (2011) Long-term variations in abundance and distribution of sewage pollution indicator and human pathogenic bacteria along the central west coast of India. Ecol Indic 11:318–327. https://doi.org/10.1016/j.ecolind.2010.05.010
9. Schippmann B, Schernewski G, Gräwea U (2013) *Escherichia coli* pollution in a Baltic Sea lagoon: a model-based source and spatial risk assessment. Int J Hyg Environ Health 216:408–420. https://doi.org/10.1016/j.ijheh.2012.12.012
10. Małecka M, Donderski W (2006) Heterotrophic bacteria inhibiting water of the River Brda on the Bydgoszcz town. Balt Coast Zone J Ecol Protect Coast 10:31–46
11. USEPA (2002) Implementation guidance for ambient water quality criteria for bacteria. EPA-823-B-02-003. US Environmental Protection Agency, Office of Water, Washington
12. World Health Organization (WHO) (2001) Bathing water quality and human health: faecal pollution. Outcome of an expert consultation. World Health Organization, Farnham
13. Water Framework Directive (WFD) (2000) Directive 2000/60/EC of the European Parliament and of the council of 23 October 2000 establishing a framework for community action in the field of water policy. J Eur Union
14. Gotkowska-Płachta A, Niewolak S, Korzeniewska E (2003) Vertical distribution and seasonal changes in the number of bacterioplankton in the water of Lake Hańcza, particularly in the period of reservoir summer stratification. EJPAU Ser Environ Dev 6:2
15. Gotkowska-Płachta A, Niewolak S, Korzeniewska E (2005) Pollution degree and sanitary state indicator bacteria as the indicators of the purity of Lake Hańcza waters. Arch Environ Prot 31(2):53–68
16. Vistula River. http://www.m188.webd.pl/index.php?option=com_content&view=article&id=64&Itemi=64. Accessed 7 Mar 2015
17. Vistula River (2001) Vistula – the last wild river of modern Europe. http://www.ekologia.pl/srodowisko/ochrona-srodowiska/wisla-ostatnia-dzika-rzeka-wspolczesnej-europy-ekologiczne-znaczenie-wisly,435.html. Accessed Feb 2001
18. Dubicki A, Radczuk L, Adynkiewicz-Piragas M et al (2005) Surface waters. Ecophysiographic report for the region of Lower Silesia. Regional Urban Planning Office, Wrocław. [in Polish]
19. IMS-Odra – monitoring data from the Odra river basin. http://www.mkoo.pl/index.php?mid=2&lang=PL. Accessed 12 Jan 2008
20. Kowalewski Z (2006) Floods in Poland – types, frequency and protection systems. Water-environment-rural areas. Institute of Land Improvement and Grasslands, Falenty 6, 1(16), pp 207–22. [in Polish]
21. Andrzejczak E, Kędziora R, Kozar-Makocka H et al (2002) Environmental report. City report. Szczecin City Office, Szczecin. [in Polish]. http://www.szczecin.pl/swiat/raport2002/srodowisko/stan.htm
22. Bożek A, Dubicki A, Dziewanowski M, Kwiatkowska-Szygulska B (2002) Surface waters. In: Smolnicki K, Szykasiuk M (eds) Environmental report for the city of Wrocław 2002. A.P.W. Bierońscy, Wrocław, pp 50–65. [in Polish]. http://docplayer.pl/8676184-Informator-o-stanie-srodowiska-wroclawia-2002.html
23. Glińska-Lewczuk K, Burandt P (2011) Effect of river straightening on the hydrochemical properties of floodplain lakes: observations from the Lyna and Drwęca Rivers, N Poland. Ecol Eng 37:786–795. https://doi.org/10.1016/j.ecoleng.2010.07.028
24. Kalinowski A (2011) The issue of the snowmelt flooding on post-glacial rivers based on the example of the Lyna River. In: Glińska-Lewczuk K (ed) Issues of landscape conservation and water management in rural areas. Contemporary problems of management and environmental protection, vol 7. University of Warmia and Mazury, Olsztyn, pp 203–220. http://www.uwm.edu.pl/environ/vol07/vol07_chapter15.pdf
25. Budzyńska D (2012) Environmental report for the Region of Warmia and Mazury, 2011. Environmental Protection Inspectorate. Regional inspectorate of environmental protection in

Olsztyn. Environmental Monitoring Library, Olsztyn. [in Polish]. http://arc.wfosigw.olsztyn.pl/pliki/raport_2011.pdf
26. Natura 2000 network of nature protection areas. http://natura2000.gdos.gov.pl/natura2000/pl/%20proste.php. Accessed 29 Sept 2015
27. Donderski W, Wilk I (2002) The sanitary state of water in the Vistula river between Wyszogród and Toruń. Pol J Environ Stud 11(5):509–515
28. An evaluation of the surface waters in the Region of Western Pomerania (2012) In: Lands-berg M (ed) Environmental report for the region of Western Pomerania, 2011. Regional Inspectorate of Environmental Protection, Szczecin, pp 30–93. [in Polish]. https://www.wios.szczecin.pl/bip/files/4780126ACCBB4B60B164E13E30BFA5A7/Ocena%20stanu%20w%C3%B3d.pdf
29. Daczkowska-Kozon E, Brzostek-Nowakowska J (2001) *Campylobacter* spp. in waters of three main Western Pomerania water bodies. Int J Hyg Environ Health 203:435–443. https://doi.org/10.1078/1438-4639-00048
30. Polish Standard PN-74/C-04620/02. Water and wastewater. Sampling of surface water for the physical, chemical and bacteriological analysis. [In Polish]
31. APHA (1992) In: Greenberg AE, Clesceri LS, Eaton AD (eds) Standard methods for the examination of water and wastewater, 18th edn. Washington
32. Polish Standard PN-75/C-04615/05. Water and wastewater. Microbiological assays. Determination of coliform bacteria with the test-tube fermentation method. [in Polish]
33. Polish Standard PN-EN ISO 9308-1:2004. Water quality. Detection and enumeration of *Escherichia coli* and coliform bacteria. Part 1: membrane filtration method. [in Polish]
34. Polish Standard PN-EN ISO 9308-1:2004/AC:2009. Water quality. Detection and enumeration of *Escherichia coli* and coliform bacteria. Part 1: Membrane filtration method. [in Polish]
35. Polish Standard PN-75/C-04615/07. Water and wastewater. Microbiological assays. Determination of fecal coliform bacteria with the test-tube fermentation method. [in Polish]
36. Polish Standard PN-82/C-04615/25. Water and wastewater. Water and sewage. Microbiological assays. Determination of fecal enterococci by the membrane filtration method (FM) and the tube method. [in Polish]
37. Geldreich EE (1970) Applying bacteriological parameters to recreational water quality. J Am Water Works Assoc 62:113–120
38. Regulation of the Minister of Environmental Protection. Natural Resources and Forestry of 16 December 1991, Journal of Laws, 115, item 503. [in Polish]
39. Dzyuban AN (2005) The environmental conditions of the Sheksna reservoir: assessment based on microbiological investigations. Water Resour 32(1):62–72
40. Fries JS, Characklis GW, Noble RT (2008) Sediment-water exchange of *Vibrio* sp. and fecal indicator bacteria: implications for persistence and transport in the Neuse River Estuary, North Carolina, USA. Water Res 42:941–950. https://doi.org/10.1016/j.watres.2007.09.006
41. Huachang H, Jianwen Q, Liang Y (2010) Environmental factors influencing the distribution of total and fecal coliform bacteria in six water storage reservoirs in the Pearl River Delta region. Chin J Environ Sci 22(5):663–668. https://doi.org/10.1016/S1001-0742(09)60160-1
42. Filipkowska Z (2003) Sanitary and bacteriological aspects of sewage treatment. Acta Microbiol Pol 52(Suppl):57–66
43. Godela A, Lewańska M, Myga-Nowak M (2017) Changes in surface water microflora caused by discharge of purified sewage into rivers. World Scientific News 75:26–32
44. Almeida CA, Quintar S, González P, Mallea MA (2007) Influence of urbanization and tourist activities on the water quality of Potrero de los Funes River (San Luis – Argentina). Environ Monit Assess 133:459–465. https://doi.org/10.1007/s10661-006-9600-3
45. Evanson M, Ambrose RF (2006) Sources and growth dynamics of fecal indicator bacteria in a coastal wetland system and potential impacts to adjacent waters. Water Res 40:475–486. https://doi.org/10.1016/j.watres.2005.11.027

46. Garcia-Arminsen T, Servais P (2004) Enumeration of viable *E. coli* in rivers and wastewaters by fluorescent in situ hybridization. J Microbiol Method 58:269–279. https://doi.org/10.1016/j.mimet.2004.04.014
47. Olapade OA, Weage EA (2010) Comparison of fecal indicator bacterial populations in surface waters of the Kalamazoo River, USA. Microbes Environ 25(1):41–44. https://doi.org/10.1264/jsme2.ME09136
48. Gołaś I, Korzekwa K, Harnisz M et al (2009) Influence of fishery management and environmental factors on occurrence of heterotrophic, hemolytic and mesophilic bacteria and *Aeromonas hydrophila* in waters of Drwęca River, Poland. Arch Environ Prot 35(2):27–40
49. Korzekwa K, Gołaś I, Harnisz M (2012) Evaluation of anthropogenic pollution in the river water based on genetic diversity of *Aeromonas hydrophila*. Arch Environ Prot 38(3):41–50
50. Niewolak S, Gołaś I (2000) Estimation of degree pollution and sanitary and bacteriological state of underground and surface waters in Omulewski Aquifer area after liquidation of large cattle and pig farms. Pol J Environ Stud 9(6):487–503
51. Niewolak S, Opieka A (2000) Potentially pathogenic microorganisms in water and bottom sediments in the Czarna Hańcza River. Pol J Environ Stud 9(3):183–194
52. Chandra R, Singh S, Raj A (2006) Seasonal bacteriological analysis of Gola River water contaminated with pulp paper mill waste in Uttaranchal, India. Environ Monit Assess 118:393–406. https://doi.org/10.1007/s10661-006-1508-4
53. Pianetti A, Sabatini L, Bruscolini F et al (2004) Fecal contamination indicators, *Salmonella*, *Vibrio* and *Aeromonas* in water used for the irrigation of agricultural products. Epidemiol Infect 132:231–238. https://doi.org/10.1017/S095026880300181
54. Pirnay J-P, Matthijs S, Colak H et al (2005) Global *Pseudomonas aeruginosa* biodiversity as reflected in a Belgian river. Environ Microbiol 7(7):969–980. https://doi.org/10.1111/j.1462-2920.2005.00776.x
55. Regulation of the Minister of the Environment of 8 July 2004 on the requirements for the evacuation of wastewater to water and ground and substances that are particularly harmful to the aquatic environment, Journal of Laws, No. 168, item 1763. [in Polish]
56. Regulation of the Minister of the Environment of 24 July 2006 on the requirements for the evacuation of wastewater to water and ground and substances that are particularly harmful to the aquatic environment, Journal of Laws, No. 137, item 984. [in Polish]
57. Regulation of the Minister of the Environment off 18 November 2014 on the requirements for the evacuation of wastewater to water and ground and substances that are particularly harmful to the aquatic environment, Journal of Laws, item 1800. [in Polish]
58. Jóźwiakowski K, Podbrożna D, Kopczacka K et al (2018) The state of water and wastewater management in the municipalities of the Roztocze National Park. J Ecol Eng 19(1):255–262. https://doi.org/10.12911/22998993/84764
59. Kirschner AKT, Kavka GG, Velimirov B et al (2009) Microbiological water quality along the Danube River: integrating data from two whole-river surveys and a transnational monitoring network. Water Res 43:3673–3684. https://doi.org/10.1016/j.watres.2009.05.034
60. Goel V, Grad IE, Kumar A, Verma NK (2008) Quantitative study on microbial pollution of River Yamuna at Delhi. Inst Eng J 88:56–60
61. Feachem R (1975) An improved role for faecal coliform to faecal streptococci ratios in the differentiation between human and non-human pollution sources. Water Res 9(7):689–690
62. Geldreich EE (1976) Fecal coliform and fecal streptococcus density relationships in waste discharges and receiving waters. Crit Rev Environ Control 6:349–369
63. Servais P, Garcia-Armisen T, George I, Billen G (2007) Fecal bacteria in the rivers of the Seine drainage network (France): sources, fate and modelling. Sci Total Environ 375(1–3):152–167. https://doi.org/10.1016/j.scitotenv.2006.12.010
64. Tryland I, Surman S, Berg JD (2002) Monitoring faecal contamination of the Thames estuary using semi-automated early warning system. Water Sci Technol 46(3):25–31. https://doi.org/10.2166/wst.2002.0047

65. Horman A, Rimhanen-Finne R, Maunula L et al (2004) *Campylobacter* spp., *Giardia* spp., *Cryptosporidium* spp., *Noroviruses*, and indicator organisms in surface water in southwestern Finland, 2000–2001. Appl Environ Microbol 70:87–95. https://doi.org/10.1128/AEM.70.1.87-95.2004
66. Korzeniewska E (2005) *Escherichia coli* and other groups of opportunistically pathogenic and pathogenic bacteria in the offshore water of the Lake Wigry. Arch Environ Prot 31(2):69–84
67. Lipp EK, Farrah SA, Rose JB (2001) Assessment and impact of microbial fecal pollution and human enteric pathogens in a coastal community. Mar Pollut Bull 42:286–293. https://doi.org/10.1016/S0025-326X(00)00152-1
68. Wilkes G, Edge T, Gannon V et al (2009) Seasonal relationships among indicator bacteria, pathogenic bacteria, *Cryptosporidium* oocysts, *Giardia* cysts, and hydrological indices for surface waters within an agricultural landscape. Water Res 43:2209–2223. https://doi.org/10.1016/j.watres.2009.01.033

Sources, Occurrence, and Environmental Risk Assessment of Antibiotics and Antimicrobial-Resistant Bacteria in Aquatic Environments of Poland

Ewa Korzeniewska and Monika Harnisz

Contents

Abstract Antimicrobial compounds are widely used in human and veterinary medicine to protect human and animal health, to prevent economic losses, and to help to ensure a safe food supply. After administration, the antibiotics and their transformant products (TPs) may pass through the sewage system and end up in the environment, mainly in the water bodies. Apart from antibiotics and their TPs, antibiotic-resistant bacteria (ARB) and antibiotic resistance genes (ARGs), which have been identified as emerging pollutants of concern, enter ecosystems with treated wastewater and livestock manure. In the environment altered by human activity, the occurrence of bacteria resistant to almost all known antibiotics has been confirmed. Despite being universally considered relatively important, land runoff, drainage and seepage are not the sites with the most striking occurrence of the transfer of resistance genes among different species of bacteria. The most significant hot-spots are wastewater treatment plants (WWTPs). Together with treated wastewater which is released from WWTPs, antibiotics, their TPs, ARB, and ARGs can penetrate the surface water, rural groundwater supplies, drinking water, soil, and plants growing in soil irrigated with contaminated water. It creates a direct risk to human and animal health because drugs, ARGs, and ARB transported

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to the environment can be transferred back to people and animals. The authors of this article aim to familiarize readers with the phenomenon of antibiotic resistance in Poland: its occurrence, ranges of antibiotic concentration, and the number of ARB and ARGs occurring in Polish surface waters as well as main sources of these contaminants.

Keywords Antibiotics · Antibiotic resistance · Lakes · Rivers · Wastewater treatment plant

1 Introduction

Nowadays, the occurrence of antibiotics in the environment is of a great concern due to the large amounts of these compounds being used in both human medicine and veterinary and due to their status as the pollutants responsible for bacterial resistance. They can be present in different environmental areas as parent drugs or as transformed compounds (TPs), resulting in mixtures with higher ecotoxicities than those consisting of individual compounds. Unfortunately, antibiotics are often metabolized in the organism only partially and are excreted as the parent compound or as metabolites in urine and feces, in case of human medicines into wastewaters [1, 2] and in, e.g., manure from animal farms in case of veterinary [3–6]. These soluble compounds are poorly removed from wastewater in WWTPs [1, 7, 8] and may reach surface waters, where some drugs can be detected at concentrations up to a few μg/L [9–11].

When pharmaceuticals reach the environment, they may undergo biotic or abiotic degradation, such as hydrolysis or photolysis. As pharmaceuticals are designed to be taken mainly orally, they are usually resistant to hydrolysis, which is why either direct or indirect photolysis may be considered the main degradation route of these compounds in the environment, especially in water. Pharmaceutical compounds widely detected in aquatic environments have been shown to accumulate in plants, where their environmental presence can be a cause for concern as some pharmaceuticals have been found to have adverse effects on aquatic or plant life [12–15]. Antibiotics and their TPs entering the environment can affect the evolution of the community structure [16, 17], which may have a close relationship with the ecosystem functions [18, 19]. In the environment, the presence of antibiotics even in subinhibitory concentrations can be associated with chronic toxicity. Furthermore, products of antibiotic phototransformation may be even more toxic to inhabitants of natural environment than parent drugs [20].

One of the most negative consequences of excessive and widespread use of antibiotics is antibiotic resistance. In the environment, antibiotic-producing bacteria occur naturally, colonizing plants and aquatic animals. However, pharmaceuticals residues in environment can negatively affect aquatic biota and microbial population. They can influence the secondary resistance of bacterial development or – and what is believed to be more important – drug-resistant bacteria entering the environment. Antibiotic resistance, according to OECD estimates, can cause over 700,000 deaths worldwide [21]. The large-scale mixing of environmental bacteria with antibiotic-resistant microorganisms from anthropogenic sources poses a risk that new resistant strains may arise [8, 22]. Antibiotic resistance also brings significant financial losses. In OECD countries, they can reach around 0.03% of gross domestic product (GDP) in 2020, 0.07% in 2030, and 0.16% in 2050, and this year it is to lead to total losses amounting to USD 2.9 trillion [23]. Due to this, the identification of the main sources of these contaminants is the first step to prevent their dissemination into environment.

The main problem associated with pharmaceuticals compounds entering the environment, besides the development of ARB, is the antibiotic resistance genes (ARGs), which can cause a decrease in the therapeutic potential of both human and animal drugs. Therefore, understanding the characteristics and behavior of microorganisms from the most diverse environmental niches is important to take actions in order to attenuate the emergence and dissemination of resistance. Improved surveillance and monitoring of antimicrobial residues and antimicrobial resistance in environment are the greatest and most urgent global challenges requiring increased attention and coherence at the international, national, and regional levels [24].

The opinions of researchers as to whether environmental concentration of antibiotics can enhance the development and the spread of resistance in environment are divided. Xu et al. [25] reported strong negative correlation between concentration of oxytetracycline and *tet* genes in WWTPs' effluents. Gao et al. [26] stated that the concentrations of antibiotics presented in environment are high enough to exert a selective pressure on clinically relevant bacteria by complete or partial inhibition of growth of wild-type (sensitive to antimicrobials) bacterial populations. Tello et al. [27] demonstrated a positive correlation between sulfonamide content and the concentrations of bacteria resistant to that drug, but they found no such correlation for tetracycline. Similar observations were made by other authors. Some of them [28, 29] reported correlations between bacterial resistance and antibiotic concentrations in environment, whereas others found the relationship between those parameters statistically insignificant [30]. Harnisz [11] found correlation only between antibiotic-resistant bacteria (ARB) and concentration of new generation antibiotics but did not find correlation between ARB and concentration of old generation antibiotics. It suggests that bacteria resistant to old generation drugs are abundant in the environment due to the transfer of resistance genes rather than to selective pressure exerted on those microorganisms by antibiotics.

The main aim of this article is to familiarize readers with the prevalence of the antibiotic resistance phenomenon in Poland: its occurrence, ranges of antibiotic concentration, and the number of ARB and ARGs occurring in Polish surface waters as well as main sources of these contaminants.

2 The Sources of Antibiotics, ARB, and ARGs Contamination

Both human and veterinary pharmaceuticals and, in consequence, the huge numbers of ARB and ARGs are introduced to the environment through a variety of pathways. In the case of human medicines, they are mainly excreted as parent compounds or metabolites in urine and feces into wastewater [8, 31–33] or, in the case of veterinary, as a fertilizer in animal farms [34, 35]. The environment pollutants are divided into point and nonpoint sources pollution (Fig. 1). The main point dischargers are factories and sewage treatment plants, which release treated wastewater. Nonpoint source pollution (NPS) generally results from land runoff, precipitation, atmospheric deposition, drainage, seepage, or hydrologic modification. NPS pollution, unlike pollution from industrial and sewage treatment plants, comes from many diffuse sources. It is caused by rainfall or snowmelt moving over and through the ground. As the runoff moves, it picks up and carries away natural and human-made pollutants, finally depositing them into lakes, rivers, wetlands, coastal waters, and groundwaters [36].

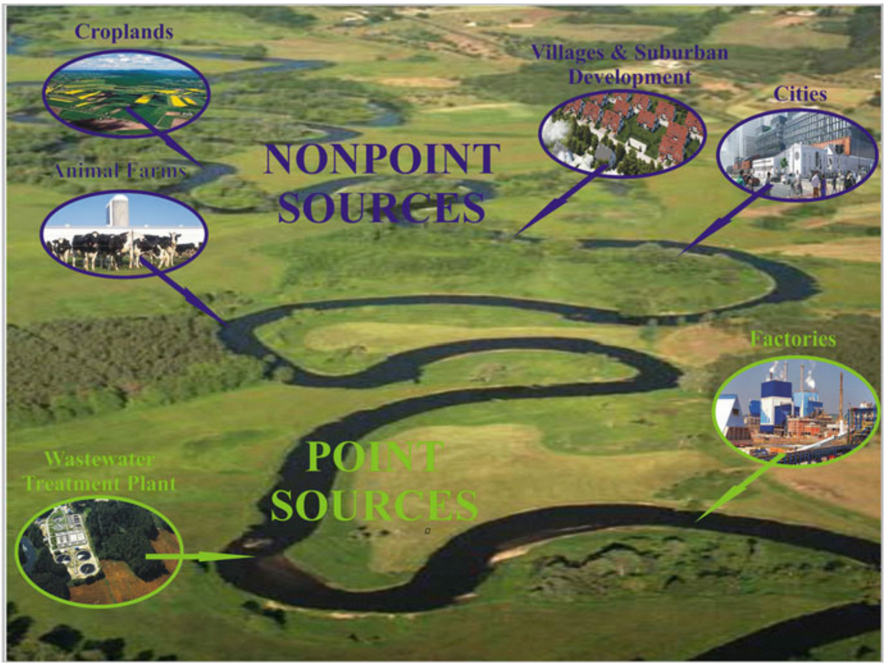

Fig. 1 Point and nonpoint sources of water pollution

2.1 WWTPs as Sources of Contamination

Generally, WWTPs have a crucial role in the protection of the environment, in particular – the natural water bodies. The removal of organic matter, chemical pollutants, and undesirable microorganisms from sewage, using combinations of physicochemical and biological treatments, was a major technological achievement of the last century, allowing the return of water of a good quality to the environment [37, 38]. However, WWTPs' effluents can constitute a significant source of ARB and ARGs as these treatment systems were not originally designed to remove these kind of contaminants [8, 39]. The final WWTPs' effluents release to the environment contain high amounts of antibiotics and their TPs as well as a huge number of ARB, many of which are of animal or human origin [37, 40, 41]. Many of these bacteria harbor acquired antibiotic resistance genes (ARGs) and are potential carriers contributing to the dissemination of these genes in the environmental microbiome [42, 43]. Most of ARB and ARGs enter ecosystems with WWTPs' effluents, specifically if the influents of WWTPs include hospital wastewater [42, 44–47]. The low efficacy of hospital sewage treatment or lack of any sewage treatment may contribute to the dissemination of multidrug-resistant bacteria (MDR) from hospital effluents to the municipal sewage and then to the environment either with treated sewage or directly into the water bodies (lakes/rivers) [43, 48, 49]. As such, these bacteria are considered a potential threat to humans and/or animals health since they may lead to more cases of difficult-to-treat infections. Moreover, although only part of the ARB released from WWTPs will be able to cause disease in humans or animals, the risk of enriching the environmental resistome either through selection or horizontal gene transfer (HGT) and therefore contributing to the emergence of resistance in pathogenic bacteria cannot be neglected [37]. WWTPs bring together ARB, antibiotic residues, and other potential selectors that favor the selection toward these bacteria and, simultaneously, offer a rich supply of nutrients and close cell-to-cell interaction, capable of facilitating the horizontal transfer of ARGs [42, 50].

Industrial and municipal wastewater discharged into waters or into the ground in 2016 in Poland reached up to 8,895.2 hm^3 (7,605.4 and 1,289.8 hm^3, respectively, for industrial and municipal). Wastewater requiring treatment ranged up to 2,166 hm^3 of which 95.2% was treated [51]. Biological treatment processes at wastewater treatment plants (WWTPs) are the most common methods of sewage treatment. At the end of 2016 in Poland, there were 3,319 working WWTPs, including 22 mechanical, 2,461 biological, and 836 with increased nutrients removal system [51]. Biological WWTPs are mainly located in villages and small cities (2,089), and most of them use activated sludge technology with sequencing batch reactors (SBR). The results of Korzeniewska and Harnisz [42] study indicated inefficient removal of ARB and ARGs by conventional Polish WWTPs with four various modifications of the wastewater treatment technologies (anaerobic/anoxic/oxic – A2/O system, mechanical-biological system, SBR, and mechanical-biological system with elevated removal of nutrients) based on activated sludge technological solutions. Furthermore, their findings clearly show that the abundance of ARB and

ARGs in municipal WWTPs based on activated sludge technological solution used in treatment sewage is directly or indirectly related to the applied modification of sewage treatment system. They reported a significant increase in the percentage of *Escherichia coli* resistant to the new generation antibiotics cefotaxime and doxycycline reaching up to 47–66% in total counts of this species, especially in effluents from WWTPs with A2/O and with SBR system where the value of hydraulic retention time (HRT) was the highest. The results of their studies showed statistically significant correlations between the number of some groups of analyzed ARB and ARGs and HRT. The HRT value can therefore be the cause of the low efficacy of these kinds of WWTPs. Wang et al. [52] and Kumar et al. [53] also reported similar significant effects of HRT on bacterial community composition and behavior in WWTPs' bioreactors. Therefore WWTPs' effluents are still an important reservoir of ARGs which can be transferred to other microorganisms, especially in surface water where WWTPs' effluents are discharged [43, 49, 54].

2.2 Other Sources of Contamination

Unjustified use of antibiotics in animal husbandry is banned in the European Union since 2006; however, such cases do occur. The scale of antibiotic use in animal production in Poland is not fully known. Breeders' assurances about nonuse of antibiotics are denied by official data on the veterinary medicines sales. They show that only in 5 years (2011–2015) in Poland, the sales of veterinary antibiotics increased by 23%, and Poland is at the forefront of the countries in Europe in terms of the consumption of antibiotics in animal farms [55].

Due to the increasing production of livestock for slaughter, the amount of produced manure and liquid manure is also growing [51]. A significant part of them is used for fertilizing land used to cultivation. According to the Institute of Soil Science and Plant Cultivation (IUNG) estimates, the total annual production of manure in Poland is about 80 million tons, liquid manure about 13 million m^3, and slurry about 7.5 million m^3 (http://www.agronews.com.pl/artykul/wykorzystanie-nawozow-naturalnych-pod-uprawy/). Therefore, due to pharmaceuticals used in animal husbandry, ARB and ARGs can be discharged with these natural fertilizers during their land application into the soil and agricultural products [56].

Other important routes of pharmaceutical, ARB, and ARGs release include wastewater irrigation and biosolid fertilization [57, 58]. The quantity of sewage sludge from industrial and municipal WWTPs in Poland in 2016 ranged up to 947,200 tones, and over 20% of it was applied in agriculture (Fig. 2).

There is a shortage of this kind of studies in Poland; only studies conducted by Popowska et al. [59] are devoted to this subject. They detected bacteria with higher resistance to erythromycin and exposure to streptomycin and oxytetracycline in manure-amended soils and soils from agricultural systems with a history of antibiotic use than in microorganisms from non-manure-amended soils.

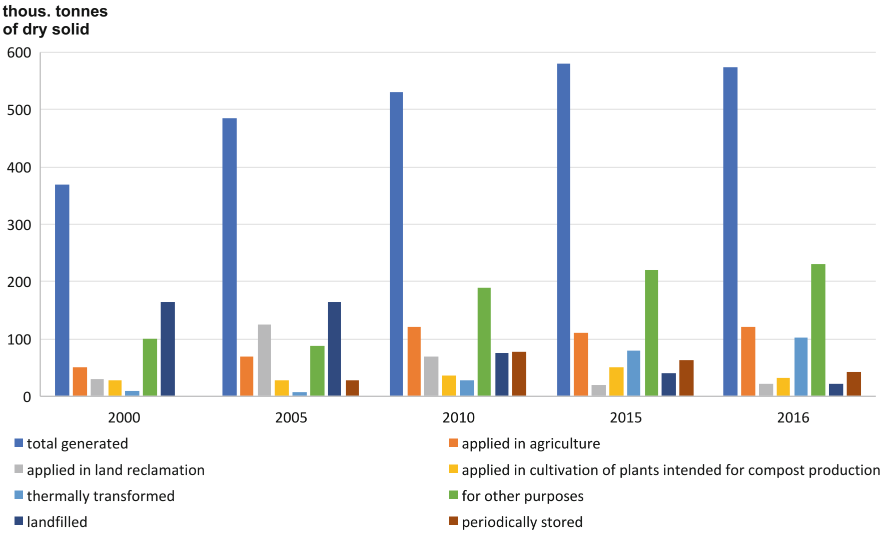

Fig. 2 Dealing with sewage sludge from municipal WWTPs in 2000–2016 years in Poland

Some of ARB and ARGs can be also released to the environment by wild animals. Mokracka et al. [60] detected that the numerous multiresistant microorganisms of different genera may inhabit wild animals, with a possibility of transferring ARGs to another animals or to the environment.

3 The Concentration of Antibiotics in Polish Surface Water

In accordance with the recommendations of the European Commission, member countries of the EU are required to monitor the drug consumption both in human medicine [61] and veterinary [55]. These recommendations are related to emergence and spread of antibiotic resistance. Due to its worldwide range, the problem of microbial antibiotic resistance has been recognized as a priority in the area of public health by many organizations all over the world, e.g., World Health Organization, European Parliament, European Centre for Disease Prevention and Control, Centers for Disease Control and Prevention, and Food and Drug Administration. These organizations recommend the rationalization of drug use by quantitative and qualitative monitoring of their use and also by monitoring the resistant bacterial strains spread.

According to ECDC [61] data, the consumption of antibiotics for systemic use in 2016 in the Polish community (primary care sector) and in the hospital sector was 24.0 and 1.36 DDD per 1,000 inhabitants per day, respectively. It was higher than the EU/EEA population-weighted mean (21.9 and 2.0 DDD per 1,000 inhabitants per day in primary care sector and in the hospital sector, respectively). In the last

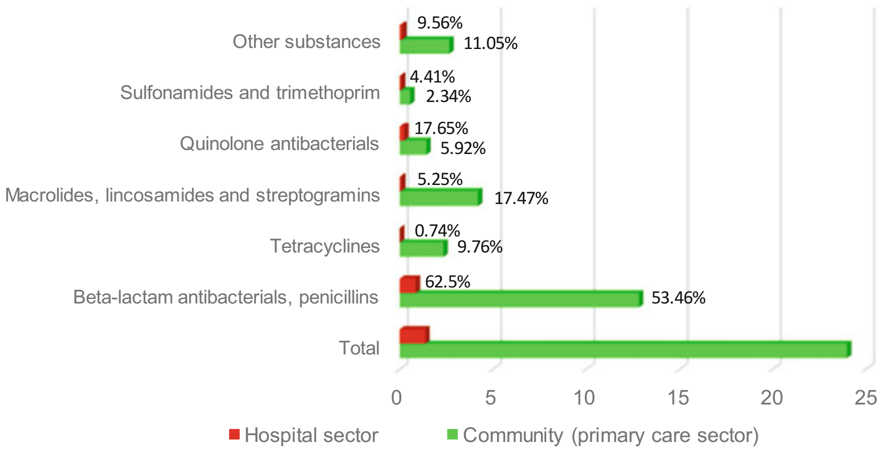

Fig. 3 Consumption of antibacterials for systemic use in the community (primary care sector) and the hospital sector expressed in DDD per 1,000 inhabitants and per day in 2016 (according to ECDC [61])

few years (2012–2016), there was no trend suggesting an increase in antibiotic consumption in the Polish community, while in the EU, population-weighted mean consumption slightly increased. The most frequently used antibiotics in Poland in 2016 were penicillins and other beta-lactams which ranged up to 53.46 and 62.5% of the total consumption in primary care and hospital sector. Macrolides, lincosamides and streptogramins (17.47 and 5.25%), and tetracyclines (9.76 and 0.74%) also can be counted into frequently used antibiotics. Besides these groups of drugs, quinolones, sulfonamides, and trimethoprim as well as other antibacterials constituting 19, 31, and 31.62% of the total consumption were used (Fig. 3).

Due to the high consumption of antibiotics, some of these compounds and their TPs can be released to the wastewater and with WWTPs' effluents – to surface water reservoirs. Ziembińska-Buczyńska et al. [62], who analyzed concentrations of erythromycin, sulfamethoxazole, and trimethoprim in WWTP's effluent in Zabrze (south of Poland), detected erythromycin at the level lower than 20 ng/L. The concentration of sulfamethoxazole ranged from 1,003 ± 146 ng/L in winter to 1,226 ± 205 ng/L in summer. Trimethoprim was present in outflowing wastewater at concentrations 369 ± 29 and 283 ± 49 ng/L in winter and summer, respectively. Harnisz et al. [39] found concentration of tetracyclines antibiotics in treated wastewater in Olsztyn (north of Poland) from limit of detection to 0.64 μg/L. The above findings are consistent with the results reported by other authors [63–65] and confirm that conventional WWTPs cannot completely remove antibiotics from wastewater leading to contamination of the environment and a high risk to the receiving water.

There are only few data on the concentration of antibiotics in Polish surface waters. Gbylik-Sikorska et al. [66], who studied the occurrence of veterinary antibiotics and chemotherapeutics in fresh water of the rivers and lakes in various parts of Poland, noted the highest concentration of aminoglycosides and beta-lactams (up to 10 μg/L) in water samples. Macrolides and tetracyclines were detected in concentration up to 5 and 50 ng/L. Kasprzyk-Hordern et al. [67] analyzed concentrations of antibiotics in downstream of Warta River in Poznań (central-western Poland). They found the level of antibiotics in a few ng/L to single μg/L depending on compound and sampling point. The highest concentration of antibiotics was observed for carbamazepine, metoprolol, and tramadol (678–794, 101–155, and 895–2,108 ng/L, respectively) after releasing of WWTPs' effluent to Warta River. On the contrary Adamek et al. [68] did not find concentration of sulfamethoxazole above the detection limit in any sample collected from heavily polluted Brynica and Czarna Przemsza rivers of Upper Silesian Industrial Region (south of Poland). Giebułtowicz et al. [8], who analyzed the occurrence of various antimicrobial agents in water of Vistula River (the longest river in Poland), noted the highest concentration of macrolides in the water sampling points near to the discharge of the WWTP's effluent reached 631, 442, and 456 ng/L for azithromycin, clarithromycin, and erythromycin, respectively. In Europe, azithromycin has been found up to 15 ng/L in Germany and 90 ng/L in Italy. In Spanish rivers, its maximum concentration ranges from 18 to up to 569 ng/L. Clarithromycin has been detected at concentrations up to 2,330 ng/L in France, up to 100 ng/L in Germany and Italy, and 78 ng/L in Sweden. Erythromycin concentrations in European rivers are up to 362 ng/L in Spain, 302 ng/L in Germany, 131 ng/L in France, 100 ng/L in the Netherlands, and 121 ng/L in the UK [69]. Comparing the concentration of the antimicrobials in Poland and in other European countries, we can observe higher level of clarithromycin and azithromycin and lower concentration of erythromycin in Polish surface waters. The results can be partly explained by the consumption data published by the ECDC [61].

4 The Concentration of ARB and ARGs in Polish Surface Water

The intensive use of antibiotics in human medicine results in the continuous release of ARB and ARGs into the hospital wastewater and subsequently into wastewater treatment plants and the environment [45, 46]. Nutrient-rich wastewater, the temperature of wastewater, and very high abundance of bacteria in the wastewater favor optimal growth of microorganisms. Due to very high concentrations of bacterial cells in those facilities, genetic information is easily transferred between microorganisms [42, 54]. Antibiotic resistance is determined by genes located on the bacterial chromosome or mobile elements, such as plasmids, transposons, and integrons, which are efficient vectors for the spread of these genes between bacteria [70]. In both soil and water, gene transfer frequencies are thought to be low, and the main factor limiting gene transfer in these environments seems to be nutrient availability as a factor controlling bacterial density and activity [71]. Wastewater treatment plants, in particular activated sludge chambers and biological filters, are hot spots for gene transfer. The results of many studies [38, 39, 42, 48, 49, 54, 72] indicate inefficient removal of ARB and ARGs by conventional Polish WWTPs and that bacteria released in WWTP's effluents may have an ability to actively spread resistance genes among indigenous microorganisms.

Despite the fact that wastewater treatment plants are characterized by a high efficiency in pollutants removal, a large pool of drug-resistant bacteria and drug resistance genes are introduced into collector water along with treated wastewater. Osińska et al. [50] pointed out that the proportion of fluoroquinolones-resistant bacteria in total counts of bacteria in downstream river water (DRW) samples (below WWTP's effluent discharged) increased radically compared to upstream river water (URW) from 0.7 to 5.76%. Moreover, due to the inflow of WWTP's effluent to the Łyna River (north-eastern Poland), Osińska et al. [49] noted also significant increases of the number of *E. coli* resistant to beta-lactams (from 26.1 to 57.5%), tetracycline (from 6.5 to 8.5%) and fluoroquinolones (from 2.2 to 5.0%) in the population of the total *E. coli* in DRW samples compared to URW samples. They also noted significant increase of multidrug-resistant strains in DRW samples (from 0 to 32%). They also found the least diverse virulence genes in *E. coli* from URW samples, whereas the strains isolated from DRW samples carried virulence genes characteristic of several concurrent pathotypes (*eae*, *bfp*A, CVD432, LT gene, ST gene, *ipa*H, *stx*1, *stx*2). It was also proven that treated wastewater is a source of tetracycline resistance genes and that they affect the number of genes associated with drug insensitivity in treated wastewater collectors [39]. Harnisz et al. [39] reported that relative concentrations (normalized according to the number of copies of 16S rRNA gene) of *tet* genes (*tet*A, *tetB*, and *tet*L) were significantly higher in URW of the Łyna River (northeastern Poland) samples as compared to water collected after the discharge of treated wastewater. Moreover, the authors found a statistically significant positive correlation between doxycycline concentration and the *tet*B gene concentration in river water samples. Such correlations were not found for older generation drugs (tetracycline and oxytetracycline), which probably is due to the fact that resistance to these drugs has spread among bacteria resulting from their long-term use.

The results of study conducted by Harnisz and Korzeniewska [43] indicate that WWTPs could be a major source of ESBL-positive (bla_{TEM}, bla_{OXA}, bla_{SHV}) *Aeromonas* spp. harboring tet (*tet*A, *tet*L, *tet*O, *tet*S, *tet*A(P)), virulence (*ahh*1) and integrase (*int*I1) genes. They also observed that a huge number of these isolates can persist in river water. Treated wastewater discharged to the aquatic environment increases the prevalence of multidrug-resistant aeromonads, determinants of antibiotic resistance, and virulence in DRW samples relative to river water not polluted with effluents from WWTPs (URW). Korzeniewska and Harnisz [54] and Korzeniewska et al. [73] isolated numerous *Enterobacteriaceae* bacteria harboring various *bla* genes ($bla_{\mathrm{CTX\text{-}M\text{-}3}}$, $bla_{\mathrm{CTX\text{-}M\text{-}9}}$, $bla_{\mathrm{SHV\text{-}2}}$, $bla_{\mathrm{SHV\text{-}5}}$, and $bla_{\mathrm{TEM\text{-}1}}$) in Łyna River (northeastern Poland) water receiving WWTP's effluent. Similar results were obtained by Koczura et al. [74] who noted that *E. coli* isolated from downstream Warta River (central-western Poland), compared to those recovered from the upstream river, were more frequently resistant to kanamycin, cephalothin, co-trimoxazole (sulfamethoxazole+trimethoprim), trimethoprim, and fluoroquinolones. Moreover, the prevalence of *int*I-positive isolates which displayed broader antibiotic resistance ranges was higher in DRW than URW samples. Koczura et al. [75] also observed that the discharge of treated wastewater significantly increased the frequency of *int*I1 among culturable bacteria as well as the number of sulfonamide resistance genes *sul*1 in the sediment of Warta River.

The results of Giebułtowicz et al. [8] also show that WWTPs are the main source of the majority of antimicrobials, resistant bacteria, and genes in the aquatic environment. In their study the authors noted the presence of the *erm*B gene, coding the resistance to macrolides, lincosamides, and streptogramin simultaneous to the high macrolide levels in almost all sampling sites the water of the Vistula River. Another ubiquitous gene detected by these scientists was int1, an element of the 5′-conserved segment of class 1 integrons that encode site-specific integrase. In water environment they also detected numerous multidrug-resistant bacteria of Gram-negative and *Enterococcus faecium* and *E. faecalis*. *Enterococcus* genus are the most common bacteria frequently polluting surface water and groundwater and are commonly used in Poland as a measure of microbial water quality.

High prevalence of numerous ARB and ARGs is also reported in other Polish surface water reservoirs, even without direct inflow of WWTPs' effluents, but these kinds of studies are rather rare. Koczura et al. [76] reported the presence of antimicrobial resistance and virulence-associated genes in *int*I-positive coliform bacteria in the waters of recreational lakes in central-western Poland. Moreover, they noted also the presence of genes typical for enterotoxigenic and Shiga toxin-producing *E. coli*. The widespread dissemination of ARB and ARGs to the environment is very worrying and consequently should be the subject of constant scrutiny.

5 Conclusions

The wide geographic spread of antibiotics and antibiotic-resistant strains in the environment, especially in surface water reservoirs, may be connected with their transmission between hospitals, WWTPs, animal farms, agriculture, and environment. Although wastewater treatment processes reduce the concentration of pharmaceuticals and ARB and ARGs in the wastewater, some of them can remain in the sewage effluents. The bacterial removal rates in the WWTPs, even above to 99%, may not prevent the dissemination of ARB and ARGs from the WWTPs into the water bodies, which are receivers of WWTPs effluent. This poses a public health risk, which needs future evaluation and monitoring.

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References

1. Brown KD, Kulis J, Thomson B et al (2006) Occurrence of antibiotics in hospital, residential, and dairy, effluent, municipal wastewater, and the Rio Grande in New Mexico. Sci Total Environ 366:772–783
2. Gulkowska A, Leung HW, So MK et al (2008) Removal of antibiotics from wastewater by sewage treatment facilities in Hong Kong and Shenzhen, China. Water Res 42:395–403
3. Kemper N (2008) Veterinary antibiotics in the aquatic and terrestrial environment. Ecol Indic 8:1–13
4. Pan X, Qiang Z, Ben W, Chen M (2011) Residual veterinary antibiotics in swine manure from concentrated animal feeding operations in Shandong Province, China. Chemosphere 84:695–700
5. Wei R, Ge F, Huang S et al (2011) Occurrence of veterinary antibiotics in animal wastewater and surface water around farms in Jiangsu Province, China. Chemosphere 82:1408–1414
6. Ji X, Shen Q, Liu F et al (2012) Antibiotic resistance gene abundances associated with antibiotics and heavy metals in animal manures and agricultural soils adjacent to feedlots in Shanghai; China. J Hazard Mater 235:178–185
7. Batt AL, Kim S, Aga DS (2006) Enhanced biodegradation of iopromide and trimethoprim in nitrifying activated sludge. Environ Sci Technol 40:7367–7373
8. Giebułtowicz J, Tyski S, Wolinowska R et al (2018) Occurrence of antimicrobial agents, drug-resistant bacteria, and genes in the sewage-impacted Vistula River (Poland). Environ Sci Pollut Res 25:5788–5807
9. Jiang L, Hu X, Yin D et al (2011) Occurrence, distribution and seasonal variation of antibiotics in the Huangpu River, Shanghai, China. Chemosphere 82:822–828
10. Zou S, Xu W, Zhang R et al (2011) Occurrence and distribution of antibiotics in coastal water of the Bohai Bay, China: impacts of river discharge and aquaculture activities. Environ Pollut 159:2913–2920
11. Harnisz M (2013) Total resistance of native bacteria as an indicator of changes in the water environment. Environ Pollut 174:85–92
12. Wu X, Dodgen LK, Conkle JL et al (2015) Plant uptake of pharmaceutical and personal care products from recycled water and biosolids: a review. Sci Total Environ 536:655–666
13. Carter LJ, Williams M, Böttcher C et al (2015) Uptake of pharmaceuticals influences plant development and affects nutrient and hormone homeostases. Environ Sci Technol 49(20):12509–12518
14. Paltiel O, Fedorova G, Tadmor G et al (2016) Human exposure to wastewater-derived pharmaceuticals in fresh produce: a randomized controlled trial focusing on carbamazepine. Environ Sci Technol 50(8):4476–4482
15. Arnnok P, Singh RR, Burakham R et al (2017) Selective uptake and bioaccumulation of antidepressants in fish from effluent-impacted Niagara River. Environ Sci Technol 51(18):10652–10662
16. Aminov RI, Mackie RI (2007) Evolution and ecology of antibiotic resistance genes. FEMS Microbiol Lett 271:147–161
17. Baran W, Adamek E, Ziemianska J et al (2011) Effects of the presence of sulfonamides in the environment and their influence on human health. J Hazard Mater 196:1–15
18. Schmitt H, van Beelen P, Tolls J et al (2004) Pollution-induced community tolerance of soil microbial communities caused by the antibiotic sulfachloropyridazine. Environ Sci Technol 38:1148–1153
19. Costanzo SD, Murby J, Bates J (2005) Ecosystem response to antibiotics entering the aquatic environment. Mar Pollut Bull 51:218–223
20. Jiao S, Zheng S, Yin D et al (2008) Aqueous photolysis of tetracycline and toxicity of photolytic products to luminescent bacteria. Chemosphere 73:377–382

21. O'Neill J (2014) The review on antimicrobial resistance antimicrobial resistance: tackling a crisis for the health and wealth of nations. https://amr-review.org/sites/default/files/160518_Final%20paper_with%20cover.pdf
22. Wellington EMH, Boxall ABA, Cross P et al (2013) The role of the natural environment in the emergence of antibiotic resistance in gram-negative bacteria. Lancet Infect Dis 13:155–165
23. Cecchini M, Langer J, Slawomirski L (2015) Antimicrobial resistance in G7 countries and beyond: economic issues, policies and options for action. OECD, Paris. https://www.oecd.org/els/health-systems/Antimicrobial-Resistance-in-G7-Countries-and-Beyond.pdf
24. World Health Organization (WHO) (2015) Global action plan on antimicrobial resistance. http://www.who.int/antimicrobial-resistance/global-action-plan/en/
25. Xu J, Xu Y, Wang H et al (2015) Occurrence of antibiotics and antibiotic resistance genes in a sewage treatment plant and its effluent-receiving river. Chemosphere 119:1379–1385
26. Gao P, Munir M, Xagoraraki I (2012) Correlation of tetracycline and sulfonamide antibiotics with corresponding resistance genes and resistant bacteria in a conventional municipal wastewater treatment plant. Sci Total Environ 421–422:173–183
27. Tello A, Austin B, Telfer TC (2012) Selective pressure of antibiotic pollution on bacteria of importance to public health. Environ Health Perspect 120:1100–1106
28. Bouki C, Venieri D, Diamadopoulos E (2013) Detection and fate of antibiotic resistant bacteria in wastewater treatment plants: a review. Ecotoxicol Environ Safe 91:1–9
29. Zhu YG, Johnson TA, Su JQ et al (2013) Diverse and abundant antibiotic resistance genes in Chinese swine farms. PNAS 110:3435–3440
30. Takasu H, Suzuki S, Reungsang A (2011) Fluoroquinolone (FQ) contamination does not correlate with occurrence of FQ-resistant bacteria in aquatic environments of Vietnam and Thailand. Microbes Environ 26:135–143
31. Guo X, Feng C, Zhang J (2017) Role of dams in the phase transfer of antibiotics in an urban river receiving wastewater treatment plant effluent. Sci Total Environ 607–608:1173–1179
32. Guo X, Yan Z, Zhang Y (2018) Behavior of antibiotic resistance genes under extremely high-level antibiotic selection pressures in pharmaceutical wastewater treatment plants. Sci Total Environ 612:119–128
33. Le Corre KS, Ort C, Kateley D (2012) Consumption-based approach for assessing the contribution of hospitals towards the load of pharmaceutical residues in municipal wastewater. Environ Int 45:99–111
34. Marti R, Tien YC, Murray R (2014) Safely coupling livestock and crop production systems: how rapidly do antibiotic resistance genes dissipate in soil following a commercial application of swine or dairy manure? Appl Environ Microbiol 80:3258–3265
35. Tien YC, Li B, Zhang T (2017) Impact of dairy manure pre-application treatment on manure composition, soil dynamics of antibiotic resistance genes, and abundance of antibiotic-resistance genes on vegetables at harvest. Sci Total Environ 581–582:32–39
36. EPA (2017) United States Environmental Protection Agency. https://www.epa.gov/nps/what-nonpoint-source
37. Manaia CM, Rocha J, Scaccia N et al (2018) Antibiotic resistance in wastewater treatment plants: tackling the black box. Environ Int 115:312–324
38. Osińska A, Korzeniewska E, Harnisz M et al (2017) Impact of type of wastewater treatment process on the antibiotic resistance of bacterial populations. In: EKO-DOK E38 web of conferences 17:00070
39. Harnisz M, Korzeniewska E, Ciesielski S et al (2015) Tet genes as indicators of changes in the water environment: relationships between culture-dependent and culture-independent approaches. Sci Total Environ 505:704–711
40. Berendonk TU, Manaia CM, Merlin C et al (2015) Tackling antibiotic resistance: the environmental framework. Nat Rev Microbiol 13:310–317
41. Wei Z, Feng K, Li S et al (2018) Exploring abundance, diversity and variation of a widespread antibiotic resistance gene in wastewater treatment plants. Environ Int 117:186–195

42. Korzeniewska E, Harnisz M (2018) Relationship between modification of activated sludge wastewater treatment and changes in antibiotic resistance of bacteria. Sci Total Environ 639:304–315
43. Harnisz M, Korzeniewska E (2018) The prevalence of multidrug-resistant *Aeromonas* spp. in the municipal wastewater system and their dissemination in the environment. Sci Total Environ 626:377–383
44. Ahn Y, Choi J (2016) Bacterial communities and antibiotic resistance communities in a full-scale hospital wastewater treatment plant by high-throughput pyrosequencing. Water 8:580
45. Korzeniewska E, Harnisz M (2013) Beta-lactamase producing *Enterobacteriaceae* in hospital effluents. J Environ Manag 123:1–7
46. Lien LTQ, Lan PT, Chuc NTK et al (2017) Antibiotic resistance and antibiotic resistance genes in *Escherichia coli* isolates from hospital wastewater in Vietnam. Int J Environ Res Public Health 14:69
47. Wang Q, Wang P, Yang Q (2018) Occurrence and diversity of antibiotic resistance in untreated hospital wastewater. Sci Total Environ 621:990–999
48. Lekunberri I, Villagrasa M, Balcázar JL et al (2017) Contribution of bacteriophage and plasmid DNA to the mobilization of antibiotic resistance genes in a river receiving treated wastewater discharges. Sci Total Environ 601–602:206–209
49. Osińska A, Korzeniewska E, Harnisz M et al (2017) The prevalence and characterization of antibiotic-resistant and virulent *Escherichia coli* strains in the municipal wastewater system and their environmental fate. Sci Total Environ 577:367–375
50. Osińska A, Monika Harnisz M, Korzeniewska E (2016) Prevalence of plasmid-mediated multidrug resistance determinants in fluoroquinolone-resistant bacteria isolated from sewage and surface water. Environ Sci Pollut Res 23:10818–10831
51. GUS (2017) Główny Urząd Statystyczny, Ochrona środowiska (Central Statistical Office, Environment protection). Statistical Information and Elaborations, Warsaw
52. Wang X, Cheng B, Ji C et al (2017) Effects of hydraulic retention time on adsorption behaviours of EPS in an A/O-MBR: biofouling study with QCM-D. Sci Rep 7:2895
53. Kumar K, Singh GK, Dastidar MG et al (2014) Effect of mixed liquor volatile suspended solids (MLVSS) and hydraulic retention time (HRT) on the performance of activated sludge process during the biotreatment of real textile wastewater. Water Res Ind 5:1–8
54. Korzeniewska E, Harnisz M (2013) Extended-spectrum beta-lactamase (ESBL)-positive *Enterobacteriaceae* in municipal sewage and their emission to the environment. J Environ Manag 128:904–911
55. EMA (2018) European Medicines Agency. Science Medicine Health. http://www.ema.europa.eu/ema/index.jsp
56. Li J, Xin Z, Zhang Y et al (2017) Long-term manure application increased the levels of antibiotics and antibiotic resistance genes in a greenhouse soil. Appl Soil Ecol 121:193–200
57. Grossberger A, Hadar Y, Borch T et al (2014) Biodegradability of pharmaceutical compounds in agricultural soils irrigated with treated wastewater. Environ Pollut 185:68–177
58. Marti R, Scott A, Tien YC (2013) Impact of manure fertilization on the abundance of antibiotic-resistant bacteria and frequency of detection of antibiotic resistance genes in soil and on vegetables at harvest. Appl Environ Microbiol 79(18):5701–5709
59. Popowska M, Rzeczycka M, Miernik A et al (2012) Influence of soil use on prevalence of tetracycline, streptomycin, and erythromycin resistance and associated resistance genes. Antimicrob Agents Chemother 56:1434–1443
60. Mokracka J, Koczura R, Kaznowski A (2012) Transferable integrons of gram-negative bacteria isolated from the gut of a wild boar in the buffer zone of a national park. Ann Microbiol 62:877–880
61. ECDC (2018) European Centre for Disease Prevention and Control. An agency of the European Union. Country overview of antimicrobial consumption. http://www.ecdc.europa.eu/en/activities/surveillance/esac-net/pages/index.aspx

62. Ziembińska-Buczyńska A, Felis E, Folkert J (2015) Detection of antibiotic resistance genes in wastewater treatment plant – molecular and classical approach. Arch Environ Prot 41(4):23–32
63. Rodriguez-Mozaz S, Chamorro S, Marti E (2015) Occurrence of antibiotics and antibiotic resistance genes in hospital and urban wastewaters and their impact on the receiving river. Water Res 69:234–242
64. Hu J, Zhou J, Zhou S (2018) Occurrence and fate of antibiotics in a wastewater treatment plant and their biological effects on receiving waters in Guizhou. Process Safe Environ Prot 113:483–490
65. Lin H, Lia H, Chena L et al (2018) Mass loading and emission of thirty-seven pharmaceuticals in a typical municipal wastewater treatment plant in Hunan Province, Southern China. Ecotoxicol Environ Safe 147:530–536
66. Gbylik-Sikorska M, Posyniak A, Mitrowska K et al (2014) Occurrence of veterinary antibiotics and chemotherapeutics in fresh water, sediment, and fish of the rivers and lakes in Poland. Bull Vet Inst Pulawy 58:399–404
67. Kasprzyk-Hordern B, Dinsdale RM, Guwy AJ (2007) Multi-residue method for the determination of basic/neutral pharmaceuticals and illicit drugs in surface water by solid-phase extraction and ultra performance liquid chromatography–positive electrospray ionisation tandem mass spectrometry. J Chromat A 1161:132–145
68. Adamek E, Baran W, Sobczak A (2016) Assessment of the biodegradability of selected sulfa drugs in two polluted rivers in Poland: effects of seasonal variations, accidental contamination, turbidity and salinity. J Hazard Mater 313:147–158
69. Carvalho IT, Santos L (2016) Antibiotics in the aquatic environments: a review of the European scenario. Environ Int 94:736–757
70. Deng Y, Bao X, Ji L et al (2015) Resistance integrons: class 1, 2 and 3 integrons. Ann Clin Microbiol Antimicrob 14:45–55
71. Seveno NA, Kallifidas D, Smalla K et al (2002) Occurrence and reservoirs of antibiotic resistance genes in the environment. Rev Med Microbiol 13:15–27
72. Makowska N, Koczura R, Mokracka J (2016) Class 1 integrase, sulfonamide and tetracycline resistance genes in wastewater treatment plant and surface water. Chemosphere 144:1665–1673
73. Korzeniewska E, Korzeniewska A, Harnisz M (2013) Antibiotic resistant Escherichia coli in hospital and municipal sewage and their emission to the environment. Ecotoxicol Environ Safe 91:96–102
74. Koczura R, Mokracka J, Jabłońska L et al (2012) Antimicrobial resistance of integron-harboring Escherichia coli isolates from clinical samples, wastewater treatment plant and river water. Sci Total Environ 414:680–685
75. Koczura R, Mokracka J, Taraszewska A et al (2016) Abundance of class 1 integron-integrase and sulfonamide resistance genes in river water and sediment is affected by anthropogenic pressure and environmental factors. Microb Ecol 72:909–916
76. Koczura R, Krysiak N, Taraszewska A et al (2015) Coliform bacteria isolated from recreational lakes carry class 1 and class 2 integrons and virulence-associated genes. J Appl Microbiol 119:594–603

Riverine Fish Fauna in Poland

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Contents

Abstract At present Poland is drained from the Carpathians, located in the south-east, to the Baltic Sea, located in the north-west. In the Pleistocene, however, much of Poland experienced several glaciations, when ice sheets advanced south and retreated. Each one destroyed many river basins and their fish but created periodic connections with the North Sea and Ponto-Caspian catchments. This enabled repeated colonization from fish refugia, which has been discovered using genetic lineage analysis.

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The scientific knowledge of rivers and fish in Poland has much increased since World War II, owing to the Scientific Annual of the Polish Angling Association, and then European Community/Union initiatives, which classified rivers into 26 abiotic and 15 fish types. Most rivers belong to barbel and bream zones, while environmental factors affect assemblage composition more than historic ones. Fish in 90% of river systems have been investigated, but in only a few monitored more than twice. Out of 53 native fish and lamprey species, the LC category comprises half, other categories 10% each, while EW, EX and DD ones only solitary species.

Native fishes are threatened by invasions of non-native ones, 24 of which have appeared in Poland since WWII, by pollution, which was severe in the 1960s–1980s but has now much decreased, and by river regulation and impounding, which continuously stress fish populations. The Polish Angling Association administers most inland waters, 0.5 million ha, which are stocked with fish and exploited by 1.5 million anglers.

Keywords Exploitation · Fish assemblages · Fish fauna origin · Human impact · River network

1 Characterisation of River Basins and River Network Development in Poland

The direction of most rivers' flow in Poland is from the south-east to north-west of the country, which reflects the general inclination of the country's terrain and which appeared together with the development of landscape in the Neogene and Quaternary Periods. Yet, there are two major barriers to surface runoff that modify the direction of the flow. One barrier, in a considerable part of southern Poland, is the Meta-Carpathian Wall, which forces the Vistula, the only watercourse able to cross this barrier, to flow north-east in its upper course. The barrier also makes all rivers draining the northern side of the Carpathian Mountains and the southern slopes of the wall the Vistula's tributaries. The other barrier, in north-eastern Poland, is an old tectonic structure of the East European platform, the Masuria-Suwałki elevation, which forces the Vistula to change its direction from the meridional to north-western one close to the Warszawa (Warsaw) city. The Oder, the other major river in Poland, flows around the wall on the western side. As both the Vistula and the Oder empty into the Baltic Sea, 99.7% of the whole present territory of Poland belongs to the Baltic Sea catchment and only 0.3% to the watersheds of the Black and North Seas [1] (Fig. 1).

The basins of major rivers flowing in the country are asymmetrical: in the Vistula River basin the ratio of the left surface area to the right one is 27:73, in the Oder River basin 30:70. This asymmetry results from the general slope of the country. In mountainous and upland areas, the limits of watersheds are distinct but become less clear in the Polish Lowlands. The density of river networks in Poland is very diverse.

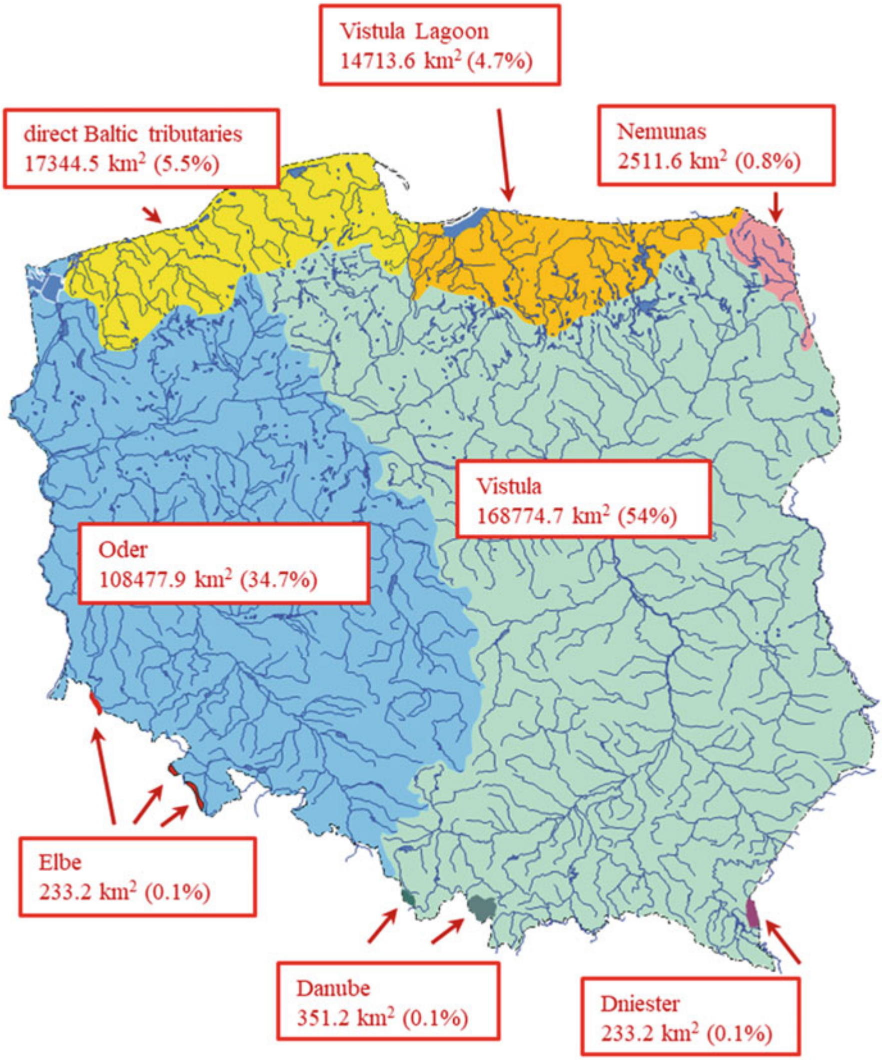

Fig. 1 River basins in Poland

It is highest in the Carpathian and Sudetes Mountains (high precipitation values, low permeability of riverbeds, and low evaporation) and about one fourth of the mountain density in high areas built of calcareous rocks (considerable infiltration of water into cracked and karst substrate). In lowlands, a dense network occurs only where the substrate is little permeable. Rivers flowing in Poland display a rainy-snowy (or snowy-rainy, dependent on which source of water is more abundant) water regime, which is characterized by two water maximums over the year: in spring (when snow and ice cover melts), which results in high water levels in the lowlands, and in July-August (maximum rainfall), which results in high water levels particularly in mountainous rivers. Lowest water levels (baseflow) occur in September and October [2].

Because both the Vistula and the Oder have their outflows in northern Poland and sources close to the country's southern border, an important feature of present river networks is the lack of transit elements. However, over the history of rivers' development, surface runoff assumed, at various spatial and temporal scales, alternative directions to that from the south-east to north-west. Particularly often this took place in the Pleistocene, when fluvioglacial waters and inflows from mountainous areas were directed to iceless areas, to the catchments of the North and Black Seas, directly adjacent to them on the east and west sides. The consequences of such events are visible in the present landscape of the country. In particular, the proof of interbasin connections is the presence of ice sheet marginal valleys originating from these periods, which have recently been used by Man to build navigable channels in the east-west orientation.

In general, the more southward a glaciation in the Vistula basin extended, the greater was the role of connections with eastern drainages, i.e. those of the Black Sea, in older glaciations. The ease of connecting with the catchment of the North Sea, due to the general inclination of the surface, gained importance in the course of younger glaciations. The first diversion of running waters to the west occurred during the recession of the Warta glaciation, when the Krzna and Pilica Rivers' ice sheet marginal valleys became connected with the upper parts of the Warta, Prosna and Widawa Rivers' valleys, with the Wrocław-Magdeburg ice-marginal valley and farther west with the Lower Weser [3]. The Pra-Oder River, with the exception of the episode of the maximum San-2 glaciation, constantly drained its waters to the North Sea.

The weight of ice sheet covering bedrock led to the creation of stresses, which were relieved after deglaciations. The post-glaciation activation of the halokinetic zones of the mid-Polish Swell was of particular importance for the system of river networks. The elevated structures slowed down water flow in some ice-marginal valleys, for example, in the Warsaw-Berlin one near the town of Łęczyca or Toruń-Eberswalde one near the town of Bydgoszcz. A more subtle influence of this process was manifest in the development of extensive depressions, through which today's Vistula flows in its lower reaches.

No less important fact for the evolution of the Polish river systems was the emergence of the general running waters' recipient, the Baltic Sea. It appeared between 14,000 and 5,000 years ago [4]. The recipient is younger than the elemental parts of its catchment, i.e. the Vistula and Oder Rivers. The current Holocene exoretic river system was created from the Neogene endorean system. The transformation is a result of tectonic processes of bedrock, initiated earlier than the Pleistocene glaciation cycle [5]. These processes in their last act contributed to the creation of a vast depression, which became the Baltic Sea when it had become filled with water.

The history of the formation of major rivers is a testimony to the diversity of their age. The further south we proceed, the older valleys become in Poland. The oldest preserved features of the modern river networks are present in the Carpathians and the Sudetes Mountains. Rivers in these areas have flowed along troughs at least since the Neogene. The Bardo antecedent river gorge of the Nysa Kłodzka River indicates that the river is older than the Bardzkie Mountains, an old variscite orogeny mountain range uplifted in the Alpine orogenesis, that run across it.

There have been two types of river valley evolution in Poland: downward expansion and upward expansion. The Vistula and Oder Rivers are examples of the downward one. Their lengths increased together with the progress of deglaciation (Figs. 1 and 2) [7]. Being linear depressions of terrain, valley forms were used

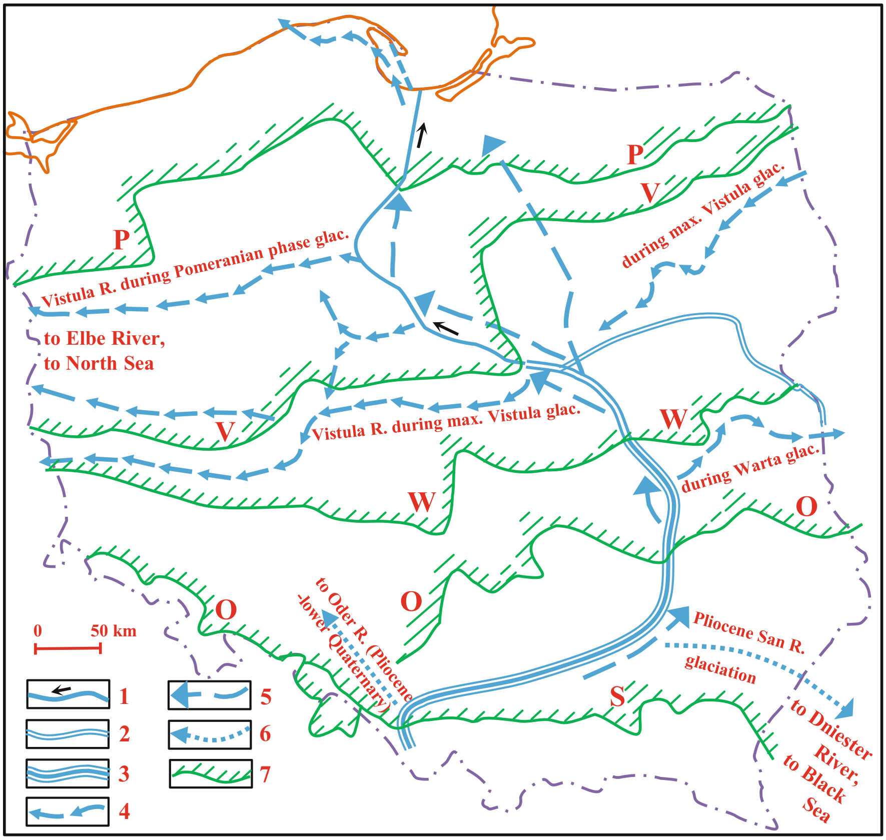

Fig. 2 The age of various sections of the Vistula valley and changes in the direction of flows (after [6], modified). 1 – Lower course of the Vistula after the recession of the latest ice sheet. 2 – Courses of the Vistula and Bug Rivers not changed after the latest interglacial period. 3 – Upper Vistula course older than glaciations (except its section upstream of the Kraków city, which was included after the San glaciation). 4 – Directions of outflow along grand valleys in the course of ice sheet stays. 5 – Ancient sections of the Vistula during interglacial periods. 6 – Old flow courses at the foot of the Carpathian Mountains before glaciations or during the maximal glaciation. 7 – Extents of major glaciations, of the following rivers: *S* San, *O* Oder, *W* Warta, *V* Vistula, *P* Pomeranian phase

by the advancing ice sheet when they corresponded to the basic direction of transgression. Therefore, the valleys were forms older than glaciations and, having their form developed in the late Pliocene and early Pleistocene Periods, were subject to further evolution. Large glaciations (San-1, San-2, Oder) also led to the obliteration of the old river system, which resulted in the creation of a new network of river valleys, such as that in the Silesian Upland [8].

On the other hand, glaciations caused significant changes in the course of mountainous rivers flowing in the foreground of the Carpathians and Sudetes mountain belt, to which the ice sheet had advanced. During the San-2 glaciation and ensuing total blocking of water outflow to the north, the Carpathian rivers led their waters to the Dniester River basin, along a developed form of urstromtal, the so-called Rynna Podkarpacka [9]. At the same time, the Bóbr and Kwisa Rivers diverged their courses in the Sudetes Mountains: through their gorges, these rivers started flowing westwards [10]. Besides, during the Oder glaciation in the eastern part of the Sudetes region, the swollen floodwaters of the Pra-Oder River started flowing into the Pra-Vistula River and farther to the Sandomierz ponding reservoir [8].

Ponding reservoirs were an integral part of the Pleistocene river drainage systems in Poland. The damming of waters took place due to fresh forms of glacial accumulation or just in front of the ice sheet. The largest reservoir was created at today's Masovian hydrographic node. At the time of the Warsaw ponding reservoir, the Vistula glaciated an important link in the outflow network from the Nemunas to the lower Elbe Rivers [3]. The outflow from this reservoir was first running west and then, together with the progressing Vistulian deglaciation, to the north-west [5]. The glaciation of the Oder contributed to the formation of a large lake in today's Sandomierz Basin, in which cone-shaped deltas were the mouths of Carpathian rivers [9].

Ice barriers and stagnation at their fronts caused an increase of the erosion base. In the upper sections of the valleys, a change into grassy channels took place. When, as a result of deglaciation, the erosion base diminished, the rivers became deeper, with erosive potential dominating over cumulative one. At that time there were breakthroughs and events of crossing tectonic (Polish Upland) or orographic accumulation barriers (including Trzebnickie and Dalkowskie Hills) by increased river discharges. The Vistula River gorge near the village of Tyniec was probably created during the South-Polish deglaciation [8]. Another important fact is the emergence of the Vistula River gorge at the town of Sandomierz, following the Oder deglaciation [11].

At the end of the Vistulian Epoch, there was a change of the Vistula's westward flow: from one along the Toruń-Eberswalde ice-marginal valley to the northern direction. The Vistula River gorge at the present village of Fordon became then created, which was probably a violent event and occurred as a result of removing a weaker morphological barrier by dammed Vistula waters, which could hardly fit in the halokinetically clamped western course [12]. The Pra-Oder River similarly reacted to the recession of given glaciers. The youngest Oder River gorge to the north, near the village of Cedynia, was created in the late Vistulian [3]. Older river gorges, near the Opole, Ścinawa and Nowa Sól towns, document the consistent breaking of landscape barriers, formerly blocking water flow to the north, at the expense of the western directed outflow of waters [13].

At the moment when the phenomena of bifurcation ended and a decisive outflow of the waters of the Vistula and Oder Rivers to the forming Baltic Sea began, the Pleistocene stage of the development of the river valley networks had been completed. In the Holocene, rivers flow along the shortest route to the north, through groundbreaking, meridion-oriented valleys, using small-scale sections of ice deposits [3]. The most pronounced breakthrough in rivers' activity was registered in the late Younger Dryas and early Eoholocene Periods. After a relatively cool continental climate, there occurred a rapid warming. The average July temperature rose from 10–13 to 16°C. This rapid change in climatic conditions became reflected in fluvial processes. The widths of riverbeds and the meander radii became reduced to one third-fourth of their former values. The number of river water channels also significantly decreased. In the Younger Dryas Period, river flows were 50–70% higher than those of the present day, but before Holocene full flows were approximately 8–20 times higher than those in the Eoholocene Period.

The Holocene is the latest part of the Quaternary Period, covering the latest approx. 11,500 BP years. During that period, the transformation of the river drainage systems in Poland has not been as spectacular as in the Pleistocene Period. However, it is a period of a further modification of river networks, in which the main roles have been played by:

1. The development of the Baltic Sea, forming the coastline and estuary conditions for rivers
2. Erosion and sedimentation processes co-shaping the natural characteristics of river networks
3. Anthropopressure modifying in a direct and indirect way river networks [3]

Significant changes have occurred in the area of postglacial waters in the Holocene, when river drainage systems have developed on the basis of proglacial valleys and postglacial gullies, gradually reducing the relatively large range of surfaceless areas. The evolution of the young glacial sculpture began with the melting of buried ice [14]. With the lapse of time, vegetation has been gradually gaining importance as it preserves land sculpture and leads to the deposition of organic substances. River valleys in the glaciation zone represent a river and lake system, unique in the whole country, especially in its central and eastern parts [14].

Human activities are manifest, among others, in the reconstruction of Pleistocene connections of river systems in the Polish Lowlands. The ice-marginal valleys, which became the axes of canals linking the Oder with the Elbe, the Oder with the Vistula and the Vistula with the Nemunas and Pripyat, were conducive to the construction of these canals.

2 Origin of European Fish Fauna

The distribution and genetic structure of temperate zone fish species have been shaped by both present processes and past history [15], in which the possibility of colonization was very important. At present fish colonization is possible only from the Baltic Sea (because almost all Poland is drained there), along few artificial canals, and as a result of other human-related fish translocations. In the past the most important historical events were climatic fluctuations during Quaternary glaciations [15, 16], i.e. in the recent ca. two and a half million years, when northern limits of occurrence ranges of fish species contracted south in cold periods and expanded north after the climate warmed [15, 17].

The climate changes were expressed mostly in glaciations, when ice sheet up to several kilometre thick advanced from the north of Poland to the south, and then recessed. There have been at least several major glaciations in the Quaternary, and each of them affected a considerable area of Poland. As regards fish fauna, the glaciations exerted both destructive and creative impacts. The former impact was destruction of many river systems and their fish or forcing the fish to crowd in undestroyed systems south to the ice sheet. The latter impact was change in the courses of rivers and establishment of periodic connections with river systems and sea catchments east and west of Poland and thus enabling fish colonization from other areas. Consequently, it seems that much different fish faunas may have been created in Poland several times, each time in each glacial-interglacial cycle.

Most organisms presently distributed in Europe have survived in populations outside the glaciation ice sheet, in climatically more favourable regions (glacial refugia). These events have left their genomic signatures across both animal and plant taxa. In population studies, using of modern DNA techniques provides the possibility of tracing lineages, routes of expansion and identification of relevant refugia [15]. However, while it is easier to trace whence a given fish species colonized Poland, it is more difficult to determine when, i.e. after which glaciation, the colonisation occurred.

Northern regions have generally been colonized from four major temperate refugia located in the southern peninsulas of Iberia, Italy and the Balkan-Greece as well as in the Ponto-Caspian region [15, 18, 19]. Besides, additional northern refugia in proximity to the European ice sheet margins, generally defined as the Atlantic refuge, might have also existed [20, 21], particularly for freshwater fish [22–28].

These historical events caused a clear difference in diversity between northern and southern European biotas [16, 29]. The south of Europe provides much more opportunity for species to find a suitable habitat due to great landscape heterogeneity and climate [15], thereby allowing the divergence and accumulation of several genomes, as well as retaining allelic diversity. This process led to the emergence of new species [16]. On the other hand, if a river basin has been isolated from its neighbours for a long time, allopatric speciation is likely to occur [30]. There are also recognizable areas of endemism throughout these regions, i.e. Central peri-Mediterranea (including West Balkan province) and

Ponto-Caspian Europe [29–32]. In contrast, the uniform fish fauna of the central and northern part of Europe, separated from the southern part by mountain ranges, was recolonized by Danubian fish species during interglacial and postglacial periods [29, 33]. This explains why central, western and northern Europe are characterized by low species richness while Ponto-Caspian Europe by high species richness [30]. Besides, the low number of fish species in central Europe appears to be related to its smooth topography, which favoured connections between river basins [34]. Additionally, postglacial expansion was remarkably rapid for many species and often led to loss of genetic variation [16].

Dispersal of primary freshwater fishes strictly depends on available water routes, and their phylogeographic distributions most reliably reflect the historical and geographical attributes of a region, including reversals of river flows, temporary connections between different drainages and the process of alternate isolation and interconnection among rivers and lakes [35, 36]. The evolutionary history of freshwater fishes is closely related not only to the geological evolution of a region but also depends on ecology, biology and physiology of species. Consequently, patterns of recolonization vary among species [37].

Some European areas constitute intraspecific contact zones for genomes expanding from different glacial refugia. Postglacial secondary contacts lead to the formation of hybrid zones, and many of these have been described across Europe [16]. If the contact zones for different species interfere, they form suture zones. There are four main suture zones described in Europe [18], and their occurrence is best explained by being a consequence of past or present geographical barriers for dispersal [38].

Recent molecular studies on the phylogeography of some freshwater European fishes have allowed to discover source areas and routes of their postglacial migration [22, 24, 39–41]. These studies revealed distinct clades in western (Atlantic), central (Danube) and eastern (Ponto-Caspian) Europe and pointed out the important role of the Danube in postglacial recolonization of European drainages (i.e. [42–44]). Probably, the Danube constituted the most ancestral glacial refuge for European freshwater fish (e.g. chub, barbel, bitterling, perch, grayling, bullhead), and they dispersed from there over much of Europe following the retreat of the glacial ice sheet [22–24, 27, 36, 39, 40, 45–47]. But in a growing number of phylogeographical studies, the Ponto-Caspian region is being identified as an equally important refuge not only for species mentioned above (e.g. [42, 48]). This area was implied as the only refuge and colonization source for a wide western-Palaearctic range in the case of European catfish [49, 50] and spined loach [37] (Fig. 3). Thus, the extent to which these refuges contributed to the origin of modern populations varied between species.

The general model of the colonization of non-Mediterranean Europe by freshwater fishes comprises two main routes, i.e. the ‘eastern colonization route’, including the northward spread of lineages from the Ponto-Caspian refugia (in a single step), and the ‘western colonization route’ often occurring in two waves, which is called two-step expansion scenario [22, 24, 36, 39, 46]. The eastern (Ponto-Caspian) lineage and western lineage (comprising Danubian and Atlantic sublineages) often came into contact in the Elbe and Odra Rivers [22, 36]. For European chub, isolation

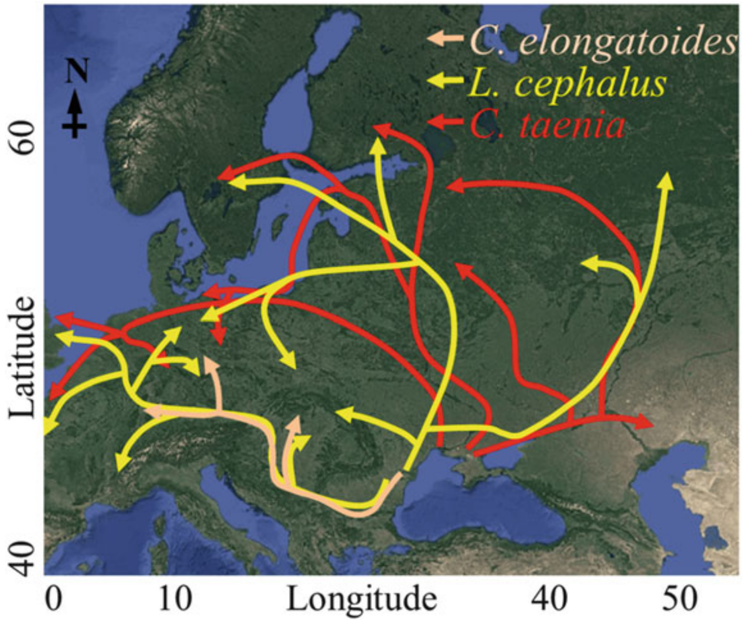

Fig. 3 Colonization routes of Europe by Danube loach, chub and spined loach

of the Rhone Rive populations from the ancestral western lineage supports the two-step expansion scenario from the Danubian refuge and the existence of a secondary glacial (Atlantic) refugium, probably located in the Rhone Basin during the Last Glacial Maximum. The eastern populations of this species clearly originate from the eastern lineage of survivors in a refugium located on the periphery of the Black and Caspian seas, and probably entered the Baltic areas as far as the Odra and northern Elbe during the Holocene [22, 36] (Fig. 3).

The western-Atlantic lineages of some species, e.g. chub and burbot, recently underwent rapid population expansion as indicated by a similar pattern of low genetic diversity identified using mtDNA [28, 36]. It seems that reduction in genetic diversity could be a general pattern in western European rivers. The next typical feature of postglacial dispersion common to European freshwater fishes is that the main mtDNA lineages came into secondary contact in the same central European areas [24, 27, 36, 42, 46], forming suture or hybrid zones [21]. Location of secondary contact zones between the western and eastern lineages in drainage basins of the Rhine, Elbe and Oder Rivers was detected for other European freshwater fishes with wide contemporary distributions, e.g. chub, barbel, vimba bream [51] and bitterling [52]. Besides, a species recolonization pattern can be modified depending on its life history mode and other biology characters (mobility, dispersal abilities, homing behaviour) [37]. For instance, the ability of cold water species to ‘cross’ high mountains was recently pointed out for the bullhead, which occurs on both sides of the Alps [53]. Generally, the number of recent major genetic lineages within each species reflects the number of refugia from which recolonization proceeded [51].

2.1 Focus on the Area of Poland

The area of Poland lies in the Atlantic-Baltic Province of the Euro-Mediterranean subregion of the Holarctic region [29, 54, 55]. Only a small part of the area is drained to the Danube and Dniester basins, both belonging to the Ponto-Caspian Province. A vast majority of the country area belongs to the Baltic Sea catchment and is drained primarily by the Vistula and Oder River systems (Fig. 1). It is evident that each basin has a specific fish species composition [56], as well as different genetic population structure of certain species, determined by postglacial history and long-term adaptation processes. Despite this fact both the Vistula and Oder River systems are noted to have very similar autochthonous fish fauna [56], lack of endemic species and much lower diversity as compared with southern and south-eastern Europe [30, 57, 58]. For this reason the fish fauna of Poland, with its uniform populations, should be considered as a smaller part of fauna inhabiting a more extensive area, defined as Central Europe [30], due to a similar pattern of postglacial recolonization from Pleistocene refugia, as well as previous Tertiary migration [23, 59]. The Dnieper-Nemunas-Vistula and the Dniester-San-Vistula routes could have been used by both individual species and groups of fish species originating from the Ponto-Caspian area, e.g. white-finned gudgeon and spirlin [60]. After the connection between the Vistula and Oder Rivers was lost in the Holocene, the last colonizers, white-eye bream, Kessler's gudgeon and Romanian barbel, managed to reach only the Vistula basin [56]. Another direction of migrants' inflow was the Atlantic refugium, accessible through the connection of the Rhine-Elbe-Oder-Vistula Rivers [61], as in the case of European perch. Some of the species of suboceanic origin (e.g. sea lamprey, river lamprey, European brook lamprey, Atlantic salmon, sea trout, eel, ninespine stickleback, three-spine stickleback) could have come from the north, probably under the pressure of a glacier expanding its range [57]. In addition, with the decline of the Weichselian (Vistulian) glaciation, boreal species (European whitefish, vendace) and Euro-Siberian species (lake minnow, European minnow, ide, Siberian bullhead) enriched the fish fauna of the southern Baltic catchment [60].

Populations of grayling inhabiting drainages of central Europe (e.g. Nemunas, Vistula, Oder, Elbe, Weser) and southern Scandinavia represent the same lineage of mtDNA [27, 44]. It is hypothesized that grayling has survived the last ice age in some periodically ice-free rivers of the Elbe or Vistula drainage, but the pattern of the colonization of Europe by this species, especially concerning its eastern distribution range, is still unresolved [27]. Until this point is clarified, it is assumed that grayling is a species of Euro-Siberian origin [60].

Fairly well recognized is the phylogeography of cobitids [37, 62]. In the case of golden loach, the previous morphological separation described by Witkowski [61] and Economidis and Nalbant [63] appears to be in agreement with its evolutionary distinctiveness [62]. This species was formerly considered to be closely related to or even conspecific with *S. balcanica* [60], while in fact it is related to lineages occurring across the Caucasus and the Caspian area and is included in the *S. baltica* monophyletic lineage [62]. Consequently, golden loach occurring in the Strwiąż River, a tributary of the Dniester River [64] is now considered *S. baltica* [62]. Similarly, spined loach has recolonized Europe from two separate refuges located in the Ponto-Caspian region (one in the middle or upper part of the Southern Bug River and the other – in the Dnieper or Volga River basins) [37] (Fig. 3).

A quite different recolonization route was detected for Danube loach, which has dispersed from a Danubian refuge and has been able to cross the Danube-Elbe and Danube-Oder water divides [65–67] (Fig. 3). The occurrence of this species is limited only to the upper part of the Oder River [68]. In this area, *C. elongatoides* has met with *C. taenia* and their hybridization led to the emergence of diplo-polyploid hybrid complexes [69, 70]. *C. elongatoides* and *C. taenia* are widespread throughout the Oder River basin but were never found to co-occur, conforming to previously reported parapatric distribution of both species on a continent-wide scale. *C. taenia* mostly occupy the main river but also enter some tributaries, while *C. elongatoides* exclusively occur in smaller tributaries [71]. The presence of a third species of the *Cobitis* genus, *C. tanaitica*, in the upper Oder River drainage requires further research.

Similar as *C. elongatoides*, bullhead has recently colonized the upper part of the Oder River [53], as a consequence of human translocations [52]. Another explanation of the invasion of the Oder drainage basin by this species are changes in the hydrological network during the Holocene, which probably allowed it to cross water divides owing to the mechanism of river capture [66]. Bullhead displays one of the most diverse genetic patterns of freshwater fishes [43], because its colonization happened along two major routes in several waves. Both the Oder (except for its upper part) and Vistula drainage basins are inhabited by the south-eastern lineage [23], originating from the Ponto-Caspian refugium, e.g. from the Dnieper system [41]. Molecular data have confirmed the morphological pattern recorded by Witkowski [61, 72], who indicated that the Vistula is dominated by the bullhead of a prickled morphology, and not by the western smooth form. This confirms the presence of a secondary contact zone for this species located in Poland with the western lineage, which invaded the upper drainage of the Oder River.

The morphological variation of three-spined stickleback [73] also points to the existence of a hybrid zone in the territory of Poland for lineages originated from Ponto-Caspian and Atlantic refuges. North-western freshwater populations of this species probably originate from the Atlantic refuge located in Brandenburg or from the Baltic Sea [74], while populations inhabiting the south-eastern part of the country are of Ponto-Caspian origin [73]. Variation in plate number in different populations probably occurs independently, due to multiple colonization events [75].

The view of chub origin in the Oder River has changed recently [36]. Chub recolonized central Europe from both Danubian and Ponto-Caspian refuges [22]. The eastern lineage (Ponto-Caspian) has entered the Baltic Sea catchment as far as the Oder River and both lineages came into contact in the Elbe River. However, a western (Danubian) origin of the Oder population of chub has been revealed [36].

Phylogeographical patterns for vimba bream [51] and stone loach [76] are very similar to those reported for bitterling [45] as only eastern lineages colonized both the Oder and Vistula drainage basins. In the case of bitterling, there is a notable exception in the upper Oder River, which is inhabited by fish from the Danube basin (Black Sea catchment) – a western clade, which colonized the rest of Europe from the Danubian refuge. This exception may be explained by human translocations across river basins [52]. The Danube refuge served as a single source for the colonization of central and northern Europe (including the Oder and Vistula Rivers) in the case of weatherfish [77], burbot [28] and barbel [24].

To sum up, the recolonization of the Baltic Sea catchment has been undertaken by fish species many times and may have taken place from different dispersion centres within the same refugium (e.g. a few centres were discovered along the Danube). Fishes have used various migration routes, which was caused by the changing hydrographic network in the Pleistocene and Holocene [56]. The present distribution of fish species is continuously modified due to human factors (e.g. accidental transport related to aquaculture, artificial canals connecting different river basins) [36].

3 General View of Contemporary Riverine Fish Fauna

3.1 History of Fish Assemblage Surveys in Poland

Despite the existence of source data on fish occurring in Poland since the seventeenth century and legal acts regarding their protection [78], more comprehensive information on the distribution of species in rivers that flow within the contemporary Polish state originates from the second half of the nineteenth century [79–81]. Most of the information concerns species of economic importance, i.e. sturgeon, Atlantic salmon and trout, and is scattered in numerous pieces in various sources. These pieces have recently been summarized in several review papers, which, however, include mostly data on fish occurrence in the rivers of northern [82, 83] and south-eastern Poland [84]. A summary of data on the distribution of fish and lamprey species in the whole area of Poland, up to 1973 and based on a very extensive literature, has also been presented [85].

An intensification of physiographic fish inventory studies started as late as after World War II. The breakthrough year was 1987, when the Polish Angling Association decided to cofinance research on the structure of fish and lamprey assemblages in river basins in Poland and to publish the resulting reports in the newly launched journal of the *Scientific Annual of the Polish Angling Association* (Roczniki Naukowe PZW). Since that year at least several respective reports, whose text is in Polish but with English abstract, summary and captions for figures and legends of tables, have been published each year. The reports may be openly accessed on the journal's webpage, www.pzw.org.pl/roczniki/cms/1635/, since 2007. A summary of riverine fish inventory research in the period of 1987–2007 was presented [86] several years ago. In the Appendix to that paper, the complete bibliography of

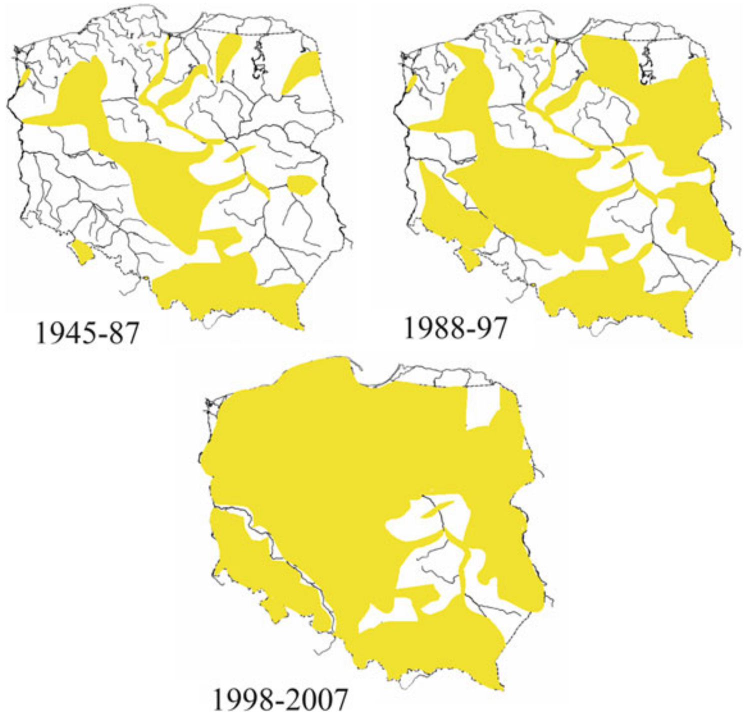

Fig. 4 The area of Poland that had been ichthyologically investigated by the end of given periods (after [84] modified)

over 400 publications containing information on fish and lamprey species recorded in Poland's rivers between 1945 and 2007 is included [86]. The result of the above described several decade long activity is a profound knowledge of the distribution of fish species in almost all Polish river systems, except for extensive fragments of two major rivers, i.e. the Vistula and the Oder (Fig. 4). Moreover, a book describing the biology of fish and lampreys recorded in inland waters of Poland, with maps of their distribution, was published in 2000 [87].

In view of the above, the state of knowledge about the distribution and abundance of fish species in Poland may be assessed as satisfactory [86]. A surprising discovery of these extensive surveys has been the determination of very abundant populations of Siberian sculpin in the basins of rivers flowing in the Eastern Pomerania region (northern Poland), i.e. the Łupawa, Reda and Łaba [88]. This bottom-dwelling rheophilic species occurs in the Carpathian tributaries of the Vistula River and in streams in the Sudetes Mountains. One isolated population in oligotrophic Lake Hańcza (NE Poland) had only been known and recognized as a postglacial relict. According to historical data, the coastal rivers of the southern Baltic basin (northern Poland) were once inhabited by another species of this genus, i.e. bullhead – *C. gobio* [87].

3.2 Riverine Fish Assemblages

Fish communities inhabiting aquatic ecosystems, particularly rivers, depend on the nature of river watercourse, which is primarily shaped by continuous changes of, e.g. water flow rate, temperature, oxygenation and type of bottom substrate. Streams and rivers have a natural longitudinal profile of habitats' distribution as a result of changes in physical and chemical variables, which is reflected in the running waters' biota [89], including fish. The fact that such profiles are mirrored by contiguous fish assemblage structures was the basis of stream classification called 'ichthyofaunal zonation', a pattern widely accepted in European fishery practice and still serving as a baseline for many studies. Although the most popular idea of the longitudinal pattern was proposed by Huet [90], the first definition was actually given by Nowicki [79], whose zones are named after those fish species that are most characteristic/abundant in particular river stretches. Consequently, the trout, grayling, barbel, bream, ruffe and flounder zones were described and accepted, although not without doubts [89] (see also [91] for a historical review). In Poland, most of rivers within the Vistula and Oder basins were classified as barbel and bream regions, while their upper Carpathian or Sudetian sections, and Pomeranian rivers, as the trout, trout and grayling, or barbel regions [92]. Consequently, a vast majority of river stretches in Poland, due to their lowland character (97%), are suitable for the predominance of cyprinids [93].

Although the longitudinal pattern of fish distribution is commonly noted [92, 94–96], field data often do not support such theoretically distinguished zones [94, 97]. In the river system, many fishes have rather wide distributions along the rivers' longitudinal profile and move from one zone to the next, e.g. for reproduction [97, 98]. The observed diversity of fish species richness and density in various zones of a river is unquestionably associated not only with the different environmental requirements of individual species but also with availability of feeding and spawning grounds as well as suitable river hydrological regimes [99].

In a more modern approach, a river is functioning as an open ecosystem in which biota, in continuously integrated series of watercourse sections, are in a dynamical equilibrium resulting from a change in the relationship between the production (P) and consumption (respiration – R) of the organic material (P/R ratio). In this theoretical approach, i.e. the River Continuum Concept [100], the longitudinal pattern of, partly overlapping, Huet's fish zones can be extended into a continuous fish community regulated by 'a continuum of abiotic and biotic factors' [101]. Consequently, abiotic factors play a more important role in shaping the fish assemblage structure in headwaters, and the importance of biotic factors gradually increases towards lower parts of a river [98]. However, this theoretical pattern is always difficult to test due to anthropogenic disturbance [95, 102].

To identify and protect the most valuable sections of rivers, and as a part of implementation of the European Union Water Framework Directive (WFD) recommendations [103], all surface watercourses and their homogeneous parts have been classified. Sections in which conditions unchanged by human activity are characterized by separate biological features were distinguished. These sections will constitute a control (standard) to determine the degree of degradation when assessing the ecological status of waters. The WFD classification was also applied to all river types and constitutes the final classification of all European river types. In Poland, 113,140.8 km of watercourses were considered (Table 1) for the classification, and 24 morphological abiotic river types were distinguished [104]. The most numerous sections were of type 17 – lowland, sandy stream, accounting for almost 40% of the total length of Polish rivers – and constituted almost 40% of homogenous river sectors [104]. The number of abiotic river types in other European countries ranges from 6 (Luxemburg) to 124 (France), with numbers similar to Poland occurring in Austria (26), The Netherlands (22) and Germany (24) [104].

Derived from abiotic river typology (Table 2), complemented by biotic components, fish-based typology indicated the dominance of river stretch type 6 – lowland sandy stream, without brown trout [105]. Consequently, in Poland the most common fish assemblage consists of (typical for abiotic river type 17) stone loach and gudgeon, with the possible presence of spined loach, spirlin, ide and chub (Table 2).

Approximately 99.7% of Poland's surface is drained by rivers flowing to the Baltic Sea (the Vistula, Oder, Pregola, Nemunas, coastal rivers), and only small fragments (0.3% in total) belong to other catchments – the Black Sea (tributaries of the Danube and Dniester Rivers) and the North Sea catchments (tributaries of the Elbe River) (Fig. 1, [105, 106]). The two largest river drainage basins in Poland, i.e. the Oder and the Vistula ones, comprise the highest morphological diversity of basins and watercourses. In both these areas, there are mountainous and foothill sectors, upland and lowland rivers. In northern Poland, on the other hand, there are rivers flowing from the terminal moraines of the latest glaciation. They are tributaries of the Vistula and Oder Rivers, which flow directly into the Baltic Sea. Among the streams and rivers not located in the Baltic Sea catchment, the Czarna Orawa River's basin is the largest one, and its ichthyofauna composition was the subject of several studies [107]. Poland's rivers of the North Sea catchment comprise small streams that are most often source sections of mountainous rivers or ones meandering among raised bogs [108]. In these streams, small rheophilic fishes mainly occur, among them trout, bullhead and rheophilous cyprinids, i.e. minnow, gudgeon, stone loach predominate, as well as Ukrainian brook lamprey. Depending on the size of the stream, two to 12 species were recorded at their sites [108].

The Pomeranian rivers and streams (in northern Poland) directly entering the Baltic Sea (the so-called coastal rivers) as well as the right-bank tributaries of the Noteć River (Oder River basin) are a separate group of water bodies. By draining an area affected by the latest glaciation, most of these rivers and streams originate from springs located in terminal moraine hills, and for this reason, the watercourses are mountainous, with a typical species composition in which rheophilic fish, both salmonids (trout, sea trout, grayling) and cyprinids, predominate. This allows to classify them as trout/grayling or barbel zone ones [92]. Due to lakes located in the basins of these watercourses, there are also limnophilous species there (tench, rudd, sunbleak) and abundant populations of three-spined stickleback and ninespine stickleback.

Table 1 Abiotic river typology applied in Poland

Abiotic river types									
		Poland's territory		Vistula catchment		Oder catchment		Others catchment[a]	
Code	Description	km	%	km	%	km	%	km	%
0	Undefined – canals and dam reservoirs	3,434.4	3.04	1,800.3	2.68	1,599.6	3.85	34.5	0.78
Highland									
1	Tatra mountain siliceous stream	104.2	0.09	104.2	0.15	–	–	–	–
2	Tatra mountain calcareous stream	36.2	0.03	36.2	0.05	–	–	–	–
3	Sudetes mountain stream	157	0.14	–	–	135.5	0.33	21.5	0.49
Mid-altitude									
4	Siliceous course substrate stream – western	2,160.2	1.91	6.2	0.01	2,017.2	4.86	136.8	3.1
5	Siliceous fine substrate stream – western	756.8	0.67	541.1	0.80	215.7	0.52	–	–
6	Calcareous fine substrate stream on loess and loess-like soil	6,817.2	6.03	5,625.8	8.37	1,191.4	2.87	–	–
7	Calcareous course substrate stream	1,151.8	1.02	968.4	1.44	183.4	0.44	–	–
8	Siliceous small river – western	668.4	0.59	199.8	0.30	468.4	1.13	–	–
9	Calcareous small river	990.2	0.88	887.1	1.32	103.1	0.25	–	–
10	Medium river – western	362.7	0.32	310.8	0.46	51.9	0.13	–	–
12	Carpathian flysch stream	6,034.9	5.33	5,664.3	8.43	40.8	0.10	329.8	7.47
14	Carpathian flysch small river	796.2	0.70	772.6	1.15	14.5	0.03	9.1	0.21
15	Medium river – eastern	509.6	0.45	509.6	0.76	–	–	–	–
Lowland									
16	Loess or clay stream	6,054.6	5.35	1,678.3	2.50	4,376.3	10.54	–	–
17	Sandy stream	43,232.5	38.21	26,757.8	39.8	15,644.6	37.70	830.1	18.81
18	Gravel-bed stream	6,610.7	5.84	1,566.3	2.33	3,503.3	8.44	1,541.1	34.93
19	Sandy-clay river	9,930.3	8.78	6,790.3	10.10	3,101.7	7.47	38.3	0.87
20	Gravel-bed river	3,419.6	3.02	1,161.5	1.73	1,629.5	3.93	628.6	14.25
21	Large river	2,976.5	2.63	1,699.1	2.53	1,277.4	3.08	–	–

(continued)

Table 1 (continued)

Abiotic river types									
		Poland's territory		Vistula catchment		Oder catchment		Others catchment[a]	
Code	Description	km	%	km	%	km	%	km	%
22	Estuary under influence of salty waters	89.5	0.08	53.7	0.08	35.8	0.09	–	–
Regardless of ecoregion									
23	Stream on peatland	7,468.6	6.60	4,652.6	6.92	2,816	6.79	–	–
24	Small and medium size river on peatland	2,419.3	2.14	1,658.1	2.47	761.2	1.83	–	–
25	Watercourse connecting lakes	6,072.3	5.37	2,896	4.31	2,333.9	5.62	842.4	19.09
26	Watercourse in the valley of large lowland river	887.1	0.78	887.1	1.32	–	–	–	–
	Total	113,140.8	100	67,227.2	100	41,501.4	100	4,412.2	100

Total length of watercourses and share in the total surface of Poland and of basin areas (after [104], modified)
[a] Sum of river basins of the Dniester, Danube, Jarftu, Elbe, Nemunas, Pregoła and Świeża

Table 2 Fish-based typology of rivers in Poland

Code	Fish-based river type	Abiotic code	Main fish species (code)	Accompanying fish species (code)
1	Highland stream with brown trout	1, 2, 3	Copoe Cogob Safar or lack of fish	Rarely, and only in lower reaches: Phpho Nebar
2	Mid-altitude course substrate stream with brown trout	4, 7, 12	Safar Phpho Nebar In lower stretches potentially spawning grounds of Sasal and/or Satru	Cogob Copoe Bapel
3	Mid-altitude fine substrate stream with brown trout	5, 6	Safar Phpho Nebar	Cogob Lapla Eumar Bapel
4	Lowland stream with brown trout	18, 17	Phpho Nebar Safar In lower stretches potentially spawning grounds of Sasal and/or Satru	Cogob Copoe Gogob Lapla Eumar Albip Leleu Peflu
5	Lowland muddy stream without brown trout	16	Gogob Lapla Gaacu	Nebar Rurut Peflu Gogob
6	Lowland sandy stream without brown trout	17	Nebar Gogob	Cotae Albip Leleu Lecep
7	Lowland organic stream without brown trout	23	Pupun and/or Gaacu	Gogob Ledel Rurut Peflu Mifos
8	Mid-altitude river with brown trout	8, 9, 14	Safar Phpho Nebar Potentially spawning grounds of Sasal and/or Satru	Ththy Cogob Gogob Lecep Leleu Bapel Albip Lapla Eumar
9	Mid-altitude river with barbel and/or grayling	10, 15	Babar Ththy Chnas Nebar Potentially spawning grounds of Sasal and/or Satru historical spawning grounds of Acoxi	Cogob Safar Phpho Lecep Leleu Gogob Bapel (Vistula catchment) Albip Lapla Eumar
10	Lowland gravel-bed river with chub	20	Lecep Babar Gogob Leleu Alalb Rurut Peflu Potentially spawning grounds of Sasal and/or Satru In lower stretches of river basins >2,000 km^2, historical spawning grounds of Acoxi	Cogob and/or Copoe Safar Ththy Nebar Lolot Esluc Leidu
11	Lowland muddy or sandy river with chub	19	Leidu Lecep Esluc Gogob Alalb Rurut Peflu In lower stretches of river basins >2,000 km^2, in stretches with high share of gravel historical spawning grounds of Acoxi	Nebar Leleu Lolot

(continued)

Table 2 (continued)

Code	Fish-based river type	Abiotic code	Main fish species (code)	Accompanying fish species (code)
12	Lowland sandy river with bream	21	Abbra Blbjo Rurut Leidu Gogob Esluc Saluc Gycer Peflu Alalb Transitional river for diadromous fish species, in stretches with gravel-bed historical spawning grounds of Acoxi	Babar Leleu Chnas Lolot Sigla Goalb Cotae Lecep
13	Lowland sandy-organic river with bream	24	Abbra Leidus Esluc Rurut Blbjo Alalb Peflu Anang	Lecep Leleu Titin Lolot Cacar Mifos Sigla Scery
14	Lowland river connecting lakes with bream	25	Rurut Peflu Alalb Abbra Blbjo Scery Esluc Anang	Different, can be several species
15	River under the influence of salty water with flounder	22	Gycer Gaacu Rurut Abbra Blbjo Abbal Anang Transitional river for diadromous fish species	Plfles and other, can be several species

For fish code see Table 3. Abiotic code – corresponding abiotic code type

An analysis of ichthyofauna structure diversity carried out on the basis of species dominance in Poland's given river basins of similar size and representing basic geographic drainage types is presented in Fig. 5. Fish and lamprey domination structure of the considered basins clearly indicates that it has been determined by environmental and not historical factors. Each of the distinguished main clusters (A and B) comprises ichthyofauna assemblages of the Baltic Sea and of the remaining sea catchments. The decisive role of environmental instead of historic factors is displayed by decisively distinguished cluster A, comprising assemblages of fishes and lampreys occurring in mountainous and submountainous rivers, irrespectively, whether they occur in the Baltic Sea (the Vistula and Oder Rivers), the Black Sea (the Dniester and Danube Rivers) or the North Sea (the Elbe River) catchments. Among the other (non-mountainous) rivers' basins, fish fauna of northern Poland (Cluster B1) and of mid-lowland Poland (Cluster B2) may be distinguished. In the latter cluster, the Pomeranian Gwda River, flowing down terminal moraines, is included. Its morphological characteristics are similar to those of the rivers of Cluster B3, formed by the fish fauna of the basins of two lowland rivers emptying directly into the Baltic Sea. The inclusion of the Gwda, whose basin adjoins the basin of the Parsęta River (Cluster B3), to the fish fauna assemblage of lowland rivers is caused by advancing degradation of the riverine environment due to dense hydrotechnical constructions related to the creation of small water power plants and thus to an increased participation of eurytopic and limnophilic species in fish assemblages [109].

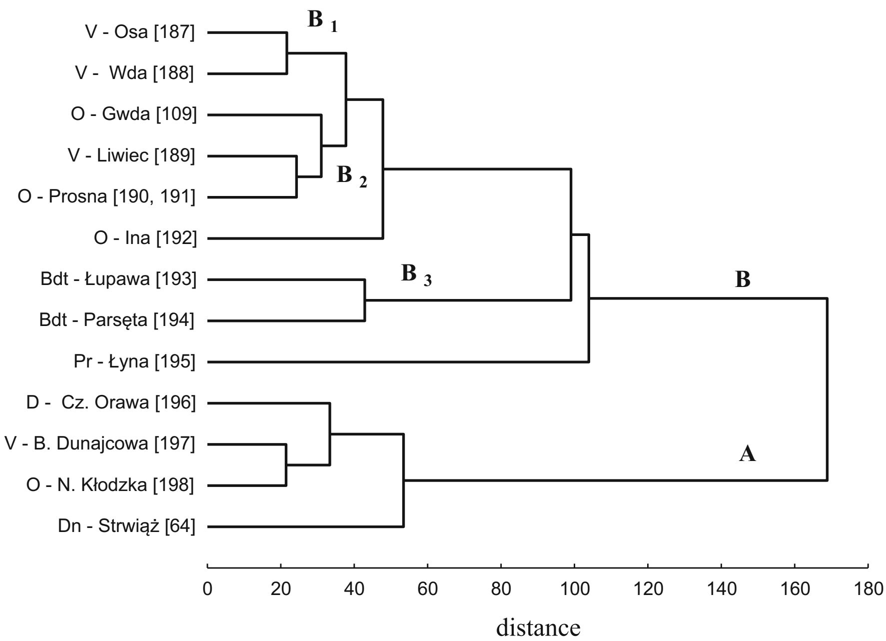

Fig. 5 Clusters and subclusters of given river basins' fish faunas obtained using the Ward method and Euclidean distances. V, Vistula basin; O, Oder basin; Bdt, rivers emptying directly into the Baltic Sea; D, Danube basin; Pr, Vistula Lagoon basin; Dn, Dniester basin. Numbers in parentheses refer to the source data in the References

3.3 Monitoring of Riverine Fish Fauna

Due to the fact that a considerable part of data on fish assemblages in Poland was collected in the second half of the twentieth century, the part needs updating. Only in the case of infrequent river basins, investigations of respective fish faunas' structure have been repeated at least twice, usually at 10-year intervals. The investigations of the Pilica River basin are unique in having been repeated six times [110]. In view of international law, i.e. Convention on Biological Diversity, Directive 92/43, the so-called Bird Directive, Directive 79/409/EEC, the so-called Habitat Directive, and also Directive 2000/60/EU, as well as national law, e.g. Law on the Protection of Nature from 16 April 2004, Polish government has been obliged to carry on monitoring of fish fauna [110]. At present two programs of monitoring running waters are conducted. Their aims are (1) assessment of the ecological state of waters on the basis of fish and lamprey assemblages [111] and (2) assessment of the ecological state of species mentioned in attachments II, IV and V to the Habitat Directive and species from the category of critically endangered (CR) and endangered (EN) [112]. The monitoring covers 12 fish species and 3 lamprey species occurring in Poland's rivers (Table 3).

Table 3 List of lampreys and fish species occurring in Polish rivers

Family/species Latin name	Common name (FishBase)	Code	Status	Habitat	Balon's rep. guild	IUCN cat.	Form of protection
A	B	C	D	E	F	G	H
Petromyzonidae							
Petromyzon marinus	Sea lamprey	Pemar	N	Ra, D	A.2.1	CR	P, HDII
Eudontomyzon mariae	Ukrainian brook lamprey	Eumar	N	Ra, D	A.2.1	VU	P, HDII
Lampetra fluviatilis	River lamprey	Laflu	N	Ra, D	A.2.1	EN	P, HDII
Lampetra planeri	European brook lamprey	Lapla	N	Rb	A.2.1	VU	P, HDII
Acipenseridae							
Acipenser oxyrinchus oxyrinchus	Atlantic (Baltic) sturgeon	Acoxy	N	Ra, D	A.1.2	Ex (P)	P, HDII, HDV
Acipenser baeri	Siberian sturgeon	Acbae	A	Ra	A.1.2		
Acipenser ruthenus	Sterlet sturgeon	Acrut	A	Ra	A.1.2		
Acipenser gueldenstaedtii	Danube sturgeon	Acgue	A	Ra	A.1.2		
Anguillidae							
Anguilla anguilla	European eel	Anang	N	E, D	A.1.1	CD	s, t, n
Clupeidae							
Alosa fallax	Twaite shad	Alfal	N			CR	P, HDII, HDV
Alosa alosa	Allis shad	Alalo	N			CR	P, HDII, HDV
Salmonidae							
Salmo salar	Atlantic salmon	Sasal	N	Ra	A.2.1	EW/CD	HDII, HDV, s, t
Salmo trutta m. trutta	Sea trout	Satru	N	Ra, D	A.2.1	CD	s
Salmo trutta m. lacustris		Salac	N	E	A.2.1	EN	s, t
Salmo trutta m. fario	Brown trout	Safar	N	Ra	A.2.1	CD	s, t, n
Oncorhynchus mykiss	Rainbow trout	Onmyk	A	Ra	A.2.1		
Salvelinus fontinalis	Brook trout	Safon	A	Ra	A.2.1		
Hucho hucho	Huchen/Danube salmon	Huchu	T	Ra	A.2.1	Ew/CD	HDII, HDV, s, t, n

A	B	C	D	E	F	G	H
Coregonidae							
Coregonus albula	Vendace	Coalb	N	L	A.1.2	VU	HDV, s, t
Coregonus lavaretus	European whitefish	Colav	N	E	A.1.3	VU	HDV, s, t
Coregonus peled	Peled	Copel	A	L	A.1.2		
Coregonus muksun	Muksun	Comuk	A	L	A.1.2		
Thymallidae							
Thymallus thymallus	Grayling	Ththy	N	Ra	A.2.1	CD	s, t, n
Osmeridae							
Osmerus eperlanus	European smelt	Osepe	N	L	A.1.2	VU	
Umbridae							
Umbra pygmaea	Eastern mudminnow	Umpyg	A	Rb	B.1.2		
Esocidae							
Esox lucius	Northern pike	Esluc	N	E	A.1.5	LC	s, t, n
Cyprinidae							
Rutilus rutilus	Roach	Rurut	N	E	A.1.4	LC	
Leuciscus leuciscus	Dace	Leleu	N	Ra	A.1.4	NT	s
Leuciscus cephalus	Chub	Lecep	N	Ra	A.1.3	LC	s
Leuciscus idus	Ide	Leidu	N	Rb	A.1.4	LC	s
Phoxinus phoxinus	European minnow	Phpho	N	Ra	A.1.3	NT	
Eupallasellas percnurus	Lake minnow	Euper	N			EN	P, HDII
Scardinius erythrophthalmus	Rudd	Scery	N	L	A.1.5	LC	s
Ctenopharyngodon idella	Grass carp	Ctide	A	E	A.1.1		
Hypophthalmichthys molitrix	Silver carp	Hymol	A	E	A.1.1		
Hypophthalmichthys nobilis	Bighead carp	Hynob	A	E	A.1.1		
Leucaspius delineatus	Sunbleak	Ledel	N	L	B.1.2	LC	
Aspius aspius	Asp	Asasp	N	Rb	A.1.3	NT	HDII, HDV, s, t, n

(continued)

Table 3 (continued)

Family/species Latin name	Common name (FishBase)	Code	Status	Habitat	Balon's rep. guild	IUCN cat.	Form of protection
A	B	C	D	E	F	G	H
Alburnoides bipunctatus	Spirlin	Albip	N	Ra	A.1.3	EN	P
Alburnus alburnus	Bleak	Alalb	N	E	A.1.4	LC	
Abramis brama	Common bream	Abbra	N	E	A.1.4	LC	
Blicca bjoerkna	White bream	Blbjo	N	E	A.1.5	LC	
Abramis ballerus	Blue bream	Abbal	N	Rb	A.1.3	LV	
Abramis sapa	White-eye bream	Absap	N	Rb	A.1.3	NT	
Chondrostoma nasus	Nase	Chnas	N	Ra	A.1.3	EN	s, t, n
Vimba vimba	Vimba bream	Vivim	N	Ra	A.1.3	CR /CD	s, t, n
Pelecus cultratus	Sichel	Pecul	N	Rb	A.1.1	CR	P, HDII, HDV
Tinca tinca	Tench	Titin	N	L	A.1.5	LC	s, n
Rhodeus amarus	Bitterling	Rhama	N	L	A.2.3	VU	P, HDII
Pseudorasbora parva	Topmouth gudgeon	Pspar	A	E	B.1.2		
Gobio gobio	Gudgeon	Gogob	N	Rb	A.1.6	LC	
Romanogobio kessleri	Kessler's gudgeon	Rokes	N	Ra	A.1.6	NT	P, HDII
Romanogobio albipinnatus	White-finned gudgeon	Roalb	N	Ra	A.1.6	VU	P, HDII
Barbus barbus	Barbel	Babar	N	Ra	A.1.3	VU	HDV, s, t, n
Barbus petenyi	Romanian barbel	Bapet	N	Ra	A.1.3	NT	HDII, HDV, t
Barbus cyclolepis waleckii	Vistula barbel	Bacyc	N	Ra	A.1.3	DD	t
Carassius carassius	Crucian carp	Cacar	N	E	A.1.5	NT	
Carassius gibelio	Prussian carp	Cagib	A	E	A.1.5		
Cyprinus carpio	Common carp	Cycar	A	E	A.1.5		s
Nemacheilidae							
Barbatula barbatula	Stone loach	Nebar	N	Ra	A.1.6	LC	P

A	B	C	D	E	F	G	H
Cobitidae							
Misgurnus fossilis	Weatherfish	Mifos	N	L	A.1.5	VU	P, HDII
Cobitis taenia	Spined loach	Cotae	N	Rb	A.1.5	LC	P
Cobitis elongatoides	Danube loach	Coelo	N	Rb	A.1.5	NT	P
Sabanejewia baltica	Golden loach	Sabal	N	Rb	A.1.5	VU	P, HDII
Siluridae							
Silurus glanis	European catfish	Sigla	N	E	B.1.2	NT	s, t
Claridae							
Clarias gariepinus	North American catfish	Clgar	A				
Ictaluridae							
Ameiurus nebulosus	Brown bullhead	Amneb	A	E	B.2.6		
Ameiurus melas	Black bullhead	Ammel	A	E	B.2.7		
Gadidae							
Lota lota	Burbot	Lolot	N	Rb	A.1.2	VU	s, t
Gasterosteidae							
Pungitius pungitius	Ninespine stickleback	Pupun	N	L	B.2.7	LC	
Gasterosteus aculeatus	Three-spined stickleback	Gaacu	N	L	B.2.7	LC	
Odontobutidae							
Perccottus glenii	Amur sleeper	Pegle	A	E	B.2.5		
Gobiidae							
Babka gymnotrachelus	Racer goby	Bagym	A	E	B.2.5		
Neogobius fluviatilis	Monkey goby	Neflu	A	E	B.2.5		
Neogobius melanostomus	Round goby	Nemel	A	E	B.2.5		
Proterorhinus semilunaris	Western tubenose goby	Prsem	A	E	B.2.5		
Cottidae							
Cottus gobio	European bullhead	Cogob	N	Ra	B.2.5	VU	P, HDII

(continued)

Table 3 (continued)

Family/species Latin name	Common name (FishBase)	Code	Status	Habitat	Balon's rep. guild	IUCN cat.	Form of protection
A	B	C	D	E	F	G	H
Cottus poecilopus	Siberian sculpin	Copoe	N	Ra	B.2.5	VU	P
Centrarchidae							
Lepomis gibbosus	Pumpkinseed	Legib	A		B.2.6		
Percidae							
Perca fluviatilis	Eurasian perch	Peflu	N	E	A.1.4	LC	s, n
Gymnocephalus cernuus	Ruffe	Gycer	N	E	A.1.4	LC	
Sander lucioperca	Pikeperch	Saluc	N	E	B.2.2	LC	s, t, n
Cichlidae							
Oreochromis niloticus	Nile tilapia	Ornil	A				
Pleuronectidae							
Platichthys flesus	European flesus	Plfle					

Status: N, native; A, alien. Habitat [113]: Ra, rheophilic, all stages of life history confined to the main river; Rb, rheophilic, some stages confined to backwaters or tributaries; E, eurytopic; L, limnophilic; D, diadromous; T, translocated. Balon's reproductive guild [114]: A.1, non-guarders, open substratum spawners; A.1.1, pelagophils, pelagic spawners; A.1.2., lithopelagophils, rock and gravel spawners with pelagic larvae; A.1.3, lithophils, rock and gravel spawners with benthic larvae; A.1.4, phytolithophils, nonobligatory plant spawners; A.1.5, phytophils, obligatory plant spawners; A.1.6, psammophils, sand spawners; A.2., non-guarders, brood hiders; A.2.2, lithophils, rock and gravel hiders; A.2.3, ostracophils, hiders in live invertebrates; B.2., substratum chosen, nest spawners; B.2.2, phytophils, plant material nesters; B.2.5, speleophils, cave hiders; B.2.6, polyphils, miscellaneous substrate nesters; B.2.7, ariadnophils, gluemaking nesters. IUCN categories [115]: EX (P), extinct (in Poland); EW, extinct in the wild; CR, critically endangered; CD, conservation-dependent; EN, endangered; VU, vulnerable; NT, near threatened; LC, least concern; DD, data deficient; NE, not evaluated. Form of protection: P, protected by Polish law; HDII, HDV, included in Habitat Directive Appendix II and V, respectively; s, protective size; t, protective season; n, protective number

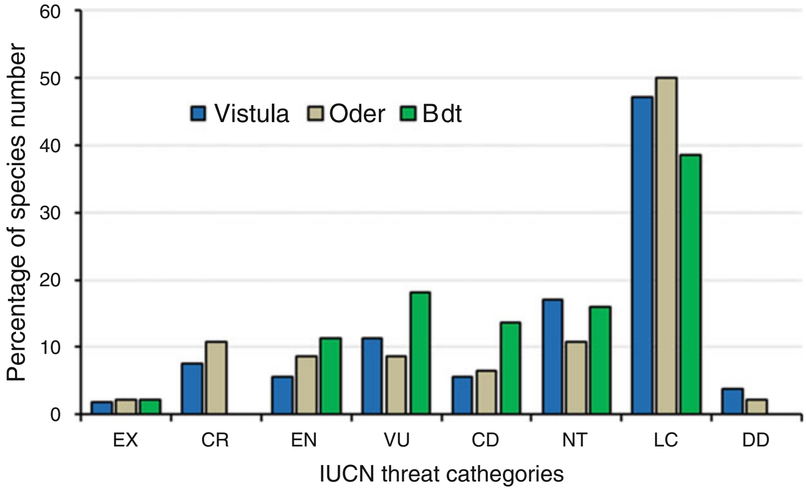

Fig. 6 Numbers of riverine native fish species in Poland that are attributed a given category of threat in the drainage basins of the Vistula and Oder Rivers and in drainage basins of rivers emptying directly into the Baltic Sea (Bdt)

3.4 Threats

The threat categorisation of Poland's freshwater fish fauna was carried out twice, the latter time in 2009 [115]. The analysis of the distribution and threats to fish and lamprey species in given areas of Poland was the subject of a conference in 2000, and the results of the meeting are presented in a supplement to an issue of the Scientific Annual of the Polish Angling Association [116].

Out of 53 native fish and lamprey species inhabiting Poland's rivers and streams, only half were determined as species of lower concern (LC), while the others were included in any of seven groups of threatened species (Table 3, Fig. 6). In Poland's freshwater fish fauna, only Atlantic (Baltic) sturgeon, a local form/subspecies of Atlantic sturgeon that developed in the Baltic Sea catchment after the latest glaciation [117], is considered extinct in the wild (EW). The status of Atlantic (Baltic) sturgeon and chances of its restitution in European waters were the subject of several publications (review in [118]). Species that are extinct in Poland's waters and whose occurrence depends on protection (EW/CD) are Atlantic salmon and huchen/Danube salmon, an endemic species of Danubian salmon inhabiting the rivers of the Danube River basin, i.e. the Czarna Orawa River and Czadeczka Stream [119]. As regards Atlantic salmon, projects to restitute it on historical spawning grounds of the Drawa River and Płociczna River have failed in attempts to establish self-sustained populations [120].

Among the threatened species, the most abundant groups are formed by vulnerable (VU) and near threatened (NT) ones. Both these groups comprise species whose status has deteriorated, e.g. nase or spirlin [121], and species expanding their occurrence ranges, i.e. Ukrainian brook lamprey and bitterling [122]. The situation of these and other species inhabiting Poland's rivers necessitates a revision of the degree of threat to fish and lampreys. Four species, European eel, grayling, sea trout and brown trout, are under strong angling pressure, and they are conservation-dependent (CD) ones, which would certainly be classified into more severe threat categories if they were not regularly stocked. Despite differences between the basins of the Vistula River, of Oder River and of rivers directly emptying into the Baltic Sea, the degree of threat to their fish faunas is very similar (Fig. 6). A slightly greater participation of species that are vulnerable (VU) and conservation-dependent (CD) in rivers that directly empty into the Baltic Sea results from a smaller number of species occurring in these rivers. Besides, certain species are classified to different categories of threat in different parts of the Baltic Sea catchment. An example of such a fish is Siberian bullhead, classified as a near threatened species in the Vistula River basin, as a vulnerable (VU) one in the Oder River basin, and as endangered (EN) one in rivers directly emptying into the Baltic Sea [107].

4 Invasive and Non-native Species and Histories of Their Establishment

The list of Polish freshwater fish fauna has been increasing due to introductions of alien species since the early Middle Ages. A total of 37 new freshwater fish species have been recorded in Poland since that time [123–125], but only 19 of them live permanently in open waters, including rivers [124]. The majority of the introductions of those 37 species were deliberate, and only 14 were accidental [125, 126]. As in other regions of the world, the main motives for the former type of introduction were aquaculture, 'improvement' of natural species composition, angling (sport, recreation), biomanipulation and ornamental aquaculture (aquarial purpose). The latter type, accidental introductions, includes escapes from aquaculture, contamination of stocking material with undesired species and spread through canals connecting different river basins [123–125].

The intensity of these introductions varied over time, e.g. till the nineteenth century, only one species, common carp, had been successfully introduced, probably brought from the Czech and Morava regions in the twelfth–thirteenth century, where it was already farmed by Cistercian monks at that time [124]. In a later period, lasting from the 1860s to 1960s, six species, rainbow trout, brown and black bullhead, brook trout, pumpkinseed and Prussian carp, were adapted and are still encountered in open waters [125, 127]. The number of exotic species has rapidly increased since World War II, when 24 fish species appeared in inland waters, i.e. 65% of all heretofore introduced freshwater fish species in Poland [125]. Between the 1960s and 1990s, there were mainly deliberate introductions, for aquaculture purposes, of species reaching considerable size [126]. These species include (1) Asian cyprinids (grass carp, silver carp, bighead carp), all of which are used both for farming and for stocking open waters, like eutrophic lakes and dam reservoirs; (2) acipenserids (Sterlet, Siberian, Danube sturgeon and their hybrids), farmed in ponds but due to escapes sporadically noted in the wild [126]; and (3) coregonids (*C. peled* and *C. muksun*) introduced to lakes in northern Poland [126].

In the 1990s the rising worldwide awareness of alien species' negative impact on biodiversity as well as the need for conservation of native ecosystems' biotic integrity resulted in a reduction of introductions of new nonindigenous fish species to the wild [124]. Poland, like other countries, became then obliged to control new introductions and the number of already present alien species in the wild and prevent expansion of potentially invasive species, i.e. those having a negative impact on native species [126]. Despite these measures 12 new fish species have been recorded since the 1990s. These are mainly small-bodied species of no commercial value, introduced mostly accidentally as a contamination of stocking material (topmouth gudgeon, Amur sleeper) or dispersing actively or passively through canal systems connecting rivers of the Black Sea catchment with the Vistula River system (the four Ponto-Caspian gobies: racer goby, monkey goby, round goby and tubenose goby) [128, 129]. All these newly arrived species have established self-sustained populations in several water bodies, and their expansion is in progress [125, 128]. Single specimens of pirapitinga (*Piaractus brachypomus*), panga (*Pangasianodon hypophthalmus*) and pleco (*Pterygoplichthys gibbiceps*) were also occasionally noted in the wild in that period because of aquarists' activities, but the individuals of these species have no chance of surviving winter [126].

Currently a total of 12 non-native species successfully reproduce in Polish waters. Prussian carp is the most widespread and abundant alien fish species occurring in almost all types of water bodies, including rivers in the whole territory of Poland. Others, like brown bullhead, peled, Ponto-Caspian gobies, Amur sleeper and topmouth gudgeon, are very abundant in some regions, while pumpkinseed, eastern mudminnow or brook trout have formed stable but local populations [125, 126]. The Ponto-Caspian gobies are the most common alien fish species in European rivers, but the Vistula River basin is till now the most westward limit of their invaded range in Poland [129]. Amur sleepers prefer rather stagnant waters, and they frequently occur in oxbow lakes of the Vistula River. However during floods and high water levels, rivers are considered as highways for rapid dispersal downstream of this fish species [125, 129]. Moreover, their occurrence in rivers is highly probable if fish farms are in vicinity, similarly as in the case of topmouth gudgeon, as both species spread as contaminants of stocking material of ponds [125].

Species like Asian cyprinids and rainbow trout do not breed naturally in the wild in Poland, but their presence in inland waters is maintained by stocking. Common carp used to be the most common stocked fish species in Poland, cultivated mainly in fish farms but kept also in inland waters for recreational fishing, including 'put and take' fishery. It is occasionally recorded in rivers, particularly in the vicinity of fish farms. Being relatively easy to cultivate, Siberian and Russian sturgeons become more popular in fish ponds, and because of this, the frequency of their accidental penetration to open waters can increase, which may cause some problems considering the Atlantic (Baltic) sturgeon reintroduction project [130]. As regards warm water fish species' production in Poland, an increasing tendency is observed only in the case of North African catfish *Clarias gariepinus* [126].

5 Human Impact

5.1 Water Pollution

After World War II, many rivers in Poland became polluted because they received a high load of raw industrial or municipal sewage. The amount of non-treated sewage discharged into waters or into the ground had been increasing until the mid-1970s and then stabilized. Due to an economic recession, the amount started decreasing since the early 1990s and within two decades dropped to approximately one tenth of the peaks observed in 1975–1985 [131].

Deteriorating water purity in the second half of the twentieth century caused changes in fish assemblages in rivers. Although the degradation of the aquatic environment often did not entail a decrease in the total biomass of fish [102], it almost always resulted in fundamental modifications of the structure of their assemblages, which was manifested in two overlapping trends. Firstly, rheophilic species which are riverine specialists and have high environmental requirements were replaced by eurytopic species (mainly roach and perch) which are more tolerant to human-induced changes [107, 132, 133]. Secondly, large-sized long-living species declined, and those of short life span and small sizes became dominant [83, 134]. Moreover, reduction in body size was reported even for dominant eurytopic species [135].

Among the most striking examples of polluted rivers in Poland are the Ner (right-bank tributary of the Warta) and Bzura (left-bank tributary of the Vistula). For decades they received untreated sewage from the Łódź Agglomeration in volumes many times exceeding their natural discharges, and the concentration of dissolved oxygen often dropped to zero [136, 137]. Along most of both rivers' courses, not only fish but also invertebrates and vascular plants were absent [138].

The improvement in water quality that has continued since the early 1990s enabled the recovery of fish communities. Fish have begun recolonizing even the Ner and Bzura Rivers, starting from limnophilous (stagnophils, including weatherfish and giebel, which are very resistant to oxygen deficits) then eurytopic, and finally rheophilic species [139–141]. It should be emphasized that a recovery may become noticeable after 10–20 years since the weakening of a long-term degradation [142, 143]. A good example is the fish fauna of the middle Warta, which recovers at a very slow pace compared to the river's upper and lower courses, because of the paucity of recolonisers [143]. Their insufficient number results from the existence of migration barriers along the river and from degraded tributaries. As it was shown in the Pilica River, the main source of recolonisers for a degraded large river are its main tributaries; small streams are too unstable in terms of abiotic factors to maintain rich fish assemblages for a long time [144]. The recolonisers may also come from the recipient. However, the beneficial impact of migrants from the Oder (for the Warta) and the Vistula (for the Pilica) was noticeable mainly in the lower reaches of the rivers [142, 143]. Another cause of delayed recovery of riverine biota may be toxic substances released from bottom sediments or from a valley previously polluted during floods and irrigation [145, 146].

Despite consistent improvement of wastewater management in Poland, raw sewage is still released to streams and rivers on a regular basis. This happens in cities with a combined sewer system [147, 148]. Its characteristic feature is transport of sewage and rainwater through the same system of canals. The system has been implemented in many city centres and, due to reducing investment costs, its size was designed for 'standard' discharge. Consequently, during heavy rains the canals are too small to transport water from street inlets to the sewage treatment plant [149]. The excess of the mixture of sewage and rainwater gravitationally pour out into the nearest stream through storm overflows (relief openings). This may happen even more than dozen times per year, which makes many urban streams fishless [150]. For example, during 51 such events in 2013–2014, just one of 18 storm overflows operating in the city of Łódź emitted 142,080 m^3 of untreated wastewater to a stream [148].

5.2 *River Regulation*

In Poland, intensive regulatory works along rivers have been performed in several recent decades in order to facilitate flow, control floods and/or increase the navigable capabilities [151, 152]. In 2016 57.7% of rivers and canals (43,442 of the total length of 75,297 km) were regulated, and the length of flood embankments was 8,451 km [153]. River regulation adversely affects riverine biota. Firstly, flood embankments are often built too close to the river so that floodplain areas are reduced and oxbow lakes become separated from the river (they are not located in the inter-embankment zone) [151]. These modifications affect fish recruitment because floodplains and oxbow lakes are used as spawning grounds and rearing places for juveniles [154–156]. Secondly, rivers are being straightened, narrowed and deprived of side channels in order to concentrate the flow and/or maintain navigable depth. Consequently, they become not only shorter but also less diversified in their cross-section in terms of water depth and speed and bottom substrate [134, 151, 157]. Thirdly, rivers are protected against erosion (e.g. with groynes, concrete slabs, riprap, fascine, wire mesh and geotextiles). This is usually accompanied by devastation of the ecotone zone and destruction of vegetation and other fish shelters, including removal of fallen trees, submersed branches and uncovered tree roots, which makes fish more susceptible to predation and deprives them of foraging areas [102, 158, 159]. However, in regulated rivers, fish often use structurally complex engineering constructions (e.g. fascine, riprap and groynes) as alternative habitats. This refers even to rheophilic species, which have been observed at the groyne heads, where water current is strongest [160, 161]. It should be emphasized that groynes and other hydraulic structures cannot effectively replace natural habitats, and their presence in the river is an ecological threat [162].

Examples of the most structurally transformed streams come from cities. Urban streams often flow in concreted canals of which some, especially in city centres, are roofed (i.e. without access to sunlight). Quite often they are fishless, or their ichthyofauna is dominated by small-sized species, like stickleback, stone loach, gudgeon, sunbleak or non-native species [150, 163].

5.3 Impoundment

Rivers have frequently been impounded in order to control floods and to meet human water, energy and transportation needs. In Poland, there are 40 and in the world over 45,000 dams of height >15 m and many more smaller ones; those in Poland were built mostly in the 1920s, 1930s and 1960s–1980s [164, 165].

River impoundment usually impacts the reproductive potential of fish. Dams preclude bedload transport from the upper part of the river system and contribute to reduced availability of gravel, pebbles and cobbles, which lithophilous fish lay eggs on [166, 167]. Additionally, flow balancing results in fewer floods in springs and reduced likelihood of removing fine material from the bottom and unveiling the coarse substrate [168, 169]. Eggs overlain with silt or mud are deprived of access to oxygenated water, and resulting hypoxia brings about reduced survival of eggs or delayed emergence and lower swimming activity of embryos [170]. Also phytophils suffer from fewer floods as they have lesser chance to spawn on inundated terrestrial vegetation of the floodplain [157].

It should be emphasized that dams exist locally, but they often make fish species extinct on a regional or global scale. This is because dams preclude migrations of diadromous fish, which live both in the sea and the river at different stages of their life cycle. They enter rivers either (1) to reach their spawning grounds and their juveniles come back to the sea (anadromous species, e.g. Atlantic salmon, sea trout or vimba bream) or (2) to feed and grow and then return to the sea for reproduction (catadromous species, which are represented in Poland only by European eel). In Poland, anadromous fish gathering downstream of dams during upstream migration towards their spawning grounds were observed downstream of dams in Poland, where they were additionally threatened by poaching [171–174].

Also during the downstream migration, diadromous fish encounter considerable difficulties because they become disoriented by the lack of water current in dam reservoirs [175]. Additionally, when a hydropower plant operates in a given dam, fish are injured or killed by turbines. In experiments made in Pomeranian rivers, up to 60% of 18 cm trout were killed by hydropower turbines [176, 177]. Downstream from the Jeziorsko dam, wounds were observed in about 45% of roach and 30% of perch at the age of 2+ and older [173, 178].

Diadromous fish achieve large sizes compared to other freshwater fish, their meat is highly valued by consumers, and this is why their populations have always been commercially exploited in Polish rivers. For example, in 1953–1965 in the Vistula, the yearly total catches made by fishery cooperatives were up to 110.5 tonnes of salmon, 186.3 tonnes of vimba bream and 31.7 tonnes of European eel. However, migration barriers made the species not only severely decline or become extinct in the parts of particular river systems located upstream from dams but also critically endangered or extinct in general [83, 171, 174]. By 1978 commercial catches of sea trout, vimba bream and European eel in the largest Polish rivers decreased by one or even two orders of magnitude, which resulted in bankruptcy of inland fishery cooperatives [171, 174].

Non-migratory species are also impacted by dams. They become divided by such dispersal barriers into separate subpopulations with limited exchange of individuals and genetic material, which may impair the long-term metapopulation persistence [179]. The extinction risk may be associated with genetic drift, inbreeding and diminished availability of suitable habitats; it may additionally increase due to the impossibility of recolonization of those parts of the river system in which a given species died out because of unstable abiotic conditions (e.g. during hot and dry summers, harsh winters or pollution incidents) [179, 180].

6 Exploitation and Restauration

Fish resources in freshwaters are exploited on the basis of the Inland Fishery Act from 18 April 1985. In contrary to standing waters, no running water can be owned by a private person or business entity in Poland. All rivers and reservoirs are owned by the Polish State Treasury, and since 2018 contracts for fishery managers are issued by a new institution called the National Water Management Holding – Polish Waters (thereafter NWMH). Freshwaters are divided into about 2,300 fishing grounds that cover ca. 480,000 ha of surface waters. According to the most recent countrywide survey (from 2016), the number of river fishing grounds was 422, with a total area of 86,336 ha. The whole area considered in that study was 395,913 ha, with 1,806 fishery grounds, which constituted 82.5% of all registered fishing grounds in Poland [181]. Fishery management contracts are usually granted for 10 year periods and have to be implemented according to strict fishery plans that are submitted to the Regional Boards of NWMH. These contracts may be won at fishing ground auctions. In general, management assessment is focused on stocking, while the height of stocking rate is the basic criterion taken under consideration during fishing ground auctions. The stocking rates depend not only on the declared value of the stocking material but also on its composition; stocking rivers with diadromous and rheophilic species is highly promoted. Much less attention is paid to restoring connectivity and habitat heterogeneity of the river systems.

The principal fishery stakeholder of running waters is the Polish Angling Association (PAA), the largest anglers association in Poland. It is responsible for fish exploitation in 86,615 ha of rivers and small reservoirs (<20 ha surface). Commercial fishery is presently rare in rivers. If it occurs, it is mostly run by the anglers associations and mostly limited to the collection of spawners for artificial reproduction. In contrary, the recreational fishery is a growing industry, involving up to 1.5 million of anglers, among whom 600,000 are affiliated presently as PAA members. Recreational and commercial fishery exploit more than 35 species, among which at least 31 species are targeted as game or food items. Additional 4–6 species constitute a by-catch (e.g. ruffe, dace) or are used as bait fish (e.g. bleak, sunbleak). According to the most recent freshwater fishery assessment report, for the year 2015, 2,391 tons and 3,331 tons of landed fish were documented by commercial and recreational fishery, respectively [181]. The most important species in terms of landed biomass are common bream (37% and 24% in commercial and recreational catch, respectively), roach (10% and 21% in commercial and recreational catch, respectively), pike (10% and 15% in commercial and recreational catch, respectively) and perch (6% and 9% in commercial and recreational catch, respectively). Among rheophilic fishes, the most abundant were ide (0.2% and 1.0% in commercial and recreational catch, respectively) and chub (0.1% and 0.7% in commercial and recreational catch, respectively) in cyprinids, brown trout (0.02% and 0.2% in commercial and recreational catch, respectively) and rainbow trout (0.1% in recreational catch, respectively) in salmonids and eel (3.3% and 1.2% in commercial and recreational catch, respectively) and sea trout (0.3% and 0.1% in commercial and recreational catch, respectively) in diadromous fish. However, according to the authors of the report, the total yearly catch of the recreational fishery sector should be estimated at 8,300 tonnes, with the average yield of 17.32 kg/ha.

Stocking is considered as a conservation activity but mostly reflects the interest of fishermen and anglers, with pike as the most dominant among all stocked species. It was reported [182] that 25.6% of all funds spent on stocking in 2016 (10 million USD) was consumed by pike stocking, giving an average value of stocking rate of 28.7 PLN/ha. Rheophilic and diadromous fishes are 2–3 times more expensive in terms of cost per unit of stocked surface (e.g. 76 PLN/ha or 48 PLN/ha for brown trout and sea trout, respectively) but of minor meaning in terms of the contribution to the total stocking budget (6.6% and 3.5%, respectively). The relatively small share of rheophilic fishes in the stocking budget reflects a high share of lakes and large reservoirs (over 75%) in the total surface of freshwater bodies and domination of potamon habitats in the majority of running waters as a consequence of the lowland character of the country. Accordingly, the most frequently stocked rheophilic species are those most tolerant or adapted to lowland rivers, like ide, asp, barbel or chub, which were stocked to 26.2%, 19.4%, 15.6% and 14.7% of fishing grounds (total evaluated surface of 395,914 ha). Stocking with species adapted to mountain or submountain rivers was more limited in space, usually covering below 12% of the surface of surveyed fishing grounds, e.g. nase (11.7%), brown trout (8.6%) or grayling (4.8%). Most of the stocking costs is covered by organizations, institutions or companies involved in fishery management, with exception of the few diadromous species (Atlantic sturgeon, Atlantic salmon, vimba bream and sea trout) that are supported by state funds in the framework of the reintroduction projects [183, 184]. Most of the restitution projects are impaired by lack of sufficient investments in restoring connectivity of rivers, as recently presented for sea trout, the catches of which are declining despite large quantities of young individuals being released annually since the 1990s [185]. Removing dams or proper designing, constructing and managing of fish passes is the compulsory activity for successful restoration projects of diadromous fishes, as was recently demonstrated at the Włoclawek dam on the Vistula River [186] and is postulated in the framework of the EU AMBER project (https://amber.international/).

References

1. Czarnecka H (ed) (2005) Atlas podziału hydrograficznego Polski. Część 2. Zestawienia zlewni (Atlas of Poland's hydrographic partitions. Part 2. Specification of catchments). IMGW (Institute of Meteorology and Water Management) Press, Warszawa
2. Dynowska I (1994) Regime of river flow, plate 32.3 River flow. In: Atlas of Polish Republic, IG PZ PAN PPWK im E. Romera SA, Warszawa
3. Jankowski AT (1994) Geneza współczesnej sieci dolinno-rzecznej na obszarze Polski (Origin of contemporaty valley-river network in the area of Poland). In: Kułtuniak J (ed) Rzeki: kultura-cywilizacja-historia t. 3 (Rivers: culture-civilisation-history vol. 3), Wyd "Śląsk" Sp z o.o., Katowice, pp 135–156
4. Uścinowicz SZ (1995) Quaternary. In: Atlas Geologiczny Południowego Bałtyku (Dewon, karbon, dolny czerwony spągowiec) (Geological Atlas of the Southern Baltic Sea (Devonian, Carboniferous, Lower Rotliegend), Państwowy Instytut Geologiczny (Governmental Geological Institute), Sopot-Warszawa, pp 22–30
5. Brykczyński M (1986) O głównych kierunkach rozwoju sieci rzecznej Niżu Polskiego w czwartorzędzie (On the main directions of river network development in the Polish Lowlands in the Quaternary). Przegl Geograf LVIII 3:411–437
6. Starkel L (1996) Rzeki w przestrzeni i w czasie (Rivers in area and in time). In: Kułtuniak J (ed) Rzeki: kultura-cywilizacja-historia, t. 5 (Rivers: culture-civilisation-history, v. 5). Wyd. "Śląsk" Sp. z o.o., Katowice. "Śląsk" Publishers Ltd., Katowice, Poland, pp 43–62
7. Starkel L (1997) The evolution of fluvial system in the Upper Vistulian and Holocene in the territory of Poland. Landform Anal 1:7–18
8. Lewandowski J, Kaziuk H (1982) Ewolucja kopalnej sieci rzecznej regionu śląsko-krakowskiego (Evolution of connate river network of the Silesian-Cracovian region). Kwartalnik Geol 26:177–190
9. Gębica P (2004) Przebieg akumulacji rzecznej w górnym vistulianie w Kotlinie Sandomierskiej (The course of fluvial accumulation during upper vistulian in Sandomierz basin). IGiPZ PAN Kraków, Prace Geograficzne, p 193
10. Michniewicz M, Czerski M, Kiełczawa J, Wojtkowiak A (1996) Staroplejstoceńska sieć dolin kopalnych Sudetów Zachodnich i ich przedpola (Old Pleistocene network of connate valleys in the Western Sudetes Mountains and in their foothills). Przegl Geol 44:1232–1238
11. Pożaryski W, Maruszczak H, Lindner L (1994) Chronostratygrafia osadów plejstoceńskich i rozwój doliny Wisły Środkowej ze szczególnym uwzględnieniem przełomu przez Wyżyny Południowopolskie (Chronostratigraphy of Pleistocene sediments and development of the valley of the middle Vistula, with particular attention paid to its gorge through South-Poland Uplands). Prace PIG 59:5–58
12. Weckwerth P (2007) Późnovistuliański rozwój sieci rzecznej w rejonie Kotliny Toruńskiej na tle struktur starszego podłoża (Late vistulian development of the river network in the area of Toruń Dale against the background of structures of older substrate). Słupskie Prace Geograficzne 4:143–156

13. Wojewoda J (2015) W poszukiwaniu dawnej Odry (In search of the old Oder River). In: Studium Generale tom XIX, pp 69–85
14. Kostrzewski A, Zwoliński Z, Andrzejewski L et al (2008) Współczesny morfosystem strefy młodoglacjalnej (Present-day morphosystem of young glacial zone). Landform Anal 7:7–11
15. Hewitt GM (1999) Postglacial recolonization of European biota. Biol J Linnean Soc Lond B Biol Sci 68:87–112
16. Hewitt GM (2000) The genetic legacy of the Quaternary ice ages. Nature 405:907–913
17. Bernatchez L, Wilson CC (1998) Comparative phylogeography of Nearctic and Palearctic fishes. Mol Ecol 7:431–452
18. Taberlet P, Fumagalli L, Wust-Saucy AG, Cosson JF (1998) Comparative phylogeography and postglacial colonization routes in Europe. Mol Ecol 7:453–464
19. Kotlík P, Bogutskaya NG, Ekmekçi FG (2004) Circum Black Sea phylogeography of *Barbus* freshwater fishes: divergence in the Pontic glacial refugium. Mol Ecol 13:87–96
20. Stewart JR, Lister AM (2001) Cryptic northern refugia and the origins of modern biota. Trends Ecol Evol 16:608–613
21. Hewitt GM (2004) Genetic consequences of climatic oscillations in the Quaternary. Philos Trans R Soc B 359:183–195
22. Durand JD, Persat H, Bouvet Y (1999) Phylogeography and postglacial dispersion of the chub (*Leuciscus cephalus*) in Europe. Mol Ecol 8:989–997
23. Englbrecht CC, Freyhof J, Nolte A et al (2000) Phylogeography of the bullhead *Cottus gobio* (Pisces: Teleostei: Cottidae) suggests a pre-Pleistocene origin of the major central European populations. Mol Ecol 9:709–722
24. Kotlík P, Berrebi P (2001) Phylogeography of the barbel (*Barbus barbus*) assessed by mitochondrial DNA variation. Mol Ecol 10:2177–2185
25. Weiss S, Persat H, Eppe R et al (2002) Complex patterns of colonization and refugia revealed for European grayling *Thymallus thymallus*, based on complete sequencing of the mitochondrial DNA control region. Mol Ecol 11:1393–1407
26. Van Houdt JK, De Cleyn L, Perretti A, Volckaert FAM (2005) A mitogenic view on the evolutionary history of the holarctic freshwater gadoid, burbot (*Lota lota*). Mol Ecol 14:2445–2457
27. Gum B, Gross R, Kuehn R (2005) Mitochondrial and nuclear DNA phylogeography of European grayling (*Thymallus thymallus*): evidence for secondary contact zones in central Europe. Mol Ecol 14:1707–1725
28. Barluenga M, Sanetra M, Meyer A (2006) Genetic admixture of burbot (Teleostei: *Lota lota*) in Lake Constance from two European glacial refugia. Mol Ecol 15:3583–3600
29. Bănărescu P (1992) Zoogeography of fresh waters. Volume 2. Distribution and dispersal of freshwater animals in North America and Eurasia. AULA-Verlag, Wiesbaden
30. Reyjol Y, Hugueny B, Pont D et al (2007) Pattern of species richness and endemism of European freshwater fish. Glob Ecol Biogeogr 16:65–75
31. Winfield IJ, Nelson JS (eds) (1991) Cyprinid fishes: systematics, biology and exploitation. Chapman and Hall, Suffolk, 667 pp
32. Griffiths D (2006) Pattern and process in the ecological biogeography of European freshwater fish. J Anim Ecol 75:734–751
33. Bănărescu P (1990) Zoogeography of fresh waters. Volume 1. General distribution and dispersal of freshwater animals. AULA-Verlag, Wiesbaden
34. Perdices A, Doadrio I (2001) The molecular systematics and biogeography of the European cobitids based on mitochondrial DNA sequences. Mol Phylogenet Evol 19:468–478
35. Levy A, Doadrio I, Almada VC (2009) Historical biogeography of European leuciscins (Cyprinidae): evaluating the Lago Mare dispersal hypothesis. J Biogeogr 36:55–65
36. Seifertová M, Bryja J, Vyskočilová M et al (2012) Multiple Pleistocene refugia and post-glacial colonization in the European chub (*Squalius cephalus*) revealed by combined use of nuclear and mitochondrial markers. J Biogeogr 39:1024–1040

37. Culling MA, Janko K, Boroń A et al (2006) European colonization by the spined loach (*Cobitis taenia*) from Ponto-Caspian refugia based on mitochondrial DNA variation. Mol Ecol 15:173–190
38. Jaarola M, Tegelstrom H, Fredga K (1999) Colonization history in Fennoscandian rodents. Biol J Linnean Soc Lond B Biol Sci 68:113–127
39. Durand JD, Unlu E, Doadrio I et al (2000) Origin, radiation, dispersion and allopatric hybridization in the chub *Leuciscus cephalus*. Proc R Soc Lond B 267:1687–1697
40. Nesbø CL, Fossheim T, Vøllestad LA, Jakobsen KS (1999) Genetic divergence and phylogeographic relationships among European perch (*Perca fluviatilis*) populations reflect glacial refugia and postglacial colonization. Mol Ecol 8:1387–1404
41. Kontula T, Väinölä R (2001) Postglacial colonization of Northern Europe by distinct phylogeographic lineages of the bullhead, *Cottus gobio*. Mol Ecol 10:1983–2002
42. Bernatchez L (2001) The evolutionary history of brown trout (*Salmo trutta* L.) inferred from phylogeographic, nested clade, and mismatch analyses of mitochondrial DNA variation. Evolution 55:351–379
43. Volckaert FAM, Hänfling B, Hellemans B, Carvalho GR (2002) Timing of the population dynamics of bullhead *Cottus gobio* (Teleostei: Cottidae) during the Pleistocene. J Evol Biol 15:930–944
44. Koskinen MT, Ranta E, Piironen J et al (2000) Genetic lineages and postglacial colonisation of grayling (*Thymallus thymallus*, Salmonidae) in Europe, as revealed by mitochondrial DNA analyses. Mol Ecol 9:1609–1624
45. Bohlen J, Šlechtová V, Bogutskaya N, Freyhof J (2006) Across Siberia and over Europe: phylogenetic relationships of the freshwater fish genus *Rhodeus* in Europe and the phylogenetic position of *R. sericeus* from the River Amur. Mol Phylogenet Evol 40:846–865
46. Bryja J, Smith C, Konečný A, Reichard M (2010) Range-wide population genetic structure of the European bitterling (*Rhodeus amarus*) based on microsatellite and mitochondrial DNA analysis. Mol Ecol 19:4708–4722
47. Hänfling B, Hellemans B, Volckaert FAM, Carvalho GR (2002) Late glacial history of the cold-adapted freshwater fish *Cottus gobio*, revealed by microsatellites. Mol Ecol 11:1717–1729
48. Bernatchez L, Osinov A (1995) Genetic diversity of trout (genus *Salmo*) from its most eastern range based on mitochondrial DNA and nuclear gene variation. Mol Ecol 4:17–26
49. Krieg F, Triantafyllidis A, Guymard R (2000) Mitochondrial DNA variation in European populations of *Silurus glanis*. J Fish Biol 56:713–724
50. Triantafyllidis A, Krieg F, Cottin C et al (2002) Genetic structure and phylogeography of European catfish (*Silurus glanis*) populations. Mol Ecol 11:1039–1055
51. Hänfling B, Dumpelmann C, Bogutskaya NG et al (2009) Shallow phylogeographic structuring of *Vimba vimba* across Europe suggests two distinct refugia during the last glaciation. J Fish Biol 75:2269–2286
52. Bartáková V, Bryja J, Reichard M (2018) Fine-scale genetic structure of the European bitterling at the intersection of three major European watersheds. BMC Evol Biol 18:105–120
53. Šlechtová V, Bohlen J, Freyhof J et al (2004) The Alps as barrier to dispersal in cold-adapted freshwater fishes? Phylogeographic history and taxonomic status of the bullhead in the Adriatic freshwater drainage. Mol Phylogenet Evol 33:225–239
54. Berg LS (1949) Freshwater fishes of the USSR and adjacent countries. Part 2, 4th edn. Izvestiia Akademii Nauk SSSR, Moscow, pp 469–925 [In Russian]
55. Bănărescu P (1989) Zoogeography and history of the freshwater fish fauna of Europe. In: Holčik J (ed) The freshwater fishes of Europe, vol 1. AULA-Verlag, Wiesbaden, pp 88–107
56. Witkowski A, Kotusz J, Przybylski M et al (2004) Pochodzenie, skład gatunkowy i aktualny stopień zagrożenia ichtiofauny w dorzeczu Wisły i Odry (Origin, species composition, and present degree of threat to fish fauna in the Vistula and Oder River systems). Arch Pol Fish 12 (Suppl.2):7–20

57. Rolik H, Rembiszewski JM (1987) Pisces et Cyclostomata. Freshwater fauna of Poland. PWN, Warszawa, 314 pp
58. Grabowska J, Przybylski M (2015) Life-history traits of non-native freshwater fish invaders differentiate them from natives in the Central European bioregion. Rev Fish Biol Fisher 25:165–178
59. Zardoya R, Doadrio I (1999) Molecular evidence on the evolutionary and biogeographical patterns of European cyprinids. J Mol Evol 49:227–237
60. Witkowski A (1994) Morphological characteristics of *Sabanejewia aurata* (De Filippi, 1865) from the Oder River Basin, with description of a new subspecies (Teleostei: Cypriniformes: Cobitidae). Zool Abhandl Staatl Mus Tierkunde Dresden 48:23–51
61. Witkowski A (1995) Phenotypic variability of *Cottus gobio* Linnaeus, 1758 in Polish waters (Teleostei: Scorpaeniformes: Cottidae). Zool Abhandl Staatl Mus Tierkunde Dresden 48:177–183
62. Perdices A, Doadrio I, Economidis PS et al (2003) Pleistocene effects on the European freshwater fish fauna: double origin of the cobitid genus *Sabanejewia* in the Danube basin (Osteichthyes: Cobitidae). Mol Phylogenet Evol 26:289–299
63. Economidis PS, Nalbant TT (1996) A study of the loaches of the genera *Cobitis* and *Sabanejewia* (Pisces, Cobitidae) of Greece, with description of six new taxa. Trav Mus Natl Hist Nat Grigore Antipa 36:297–347
64. Bylak A, Kukuła K (2018) Importance of peripheral basins: implications for the conservation of fish assemblages. Aquat Conserv Mar Freshwat Ecosyst 2018:1–12
65. Bohlen J, Ráb P (2001) Species and hybrid richness in spined loaches of the genus Cobitis L. (Teleostei: Cobitidae), with a checklist of European forms and suggestions for their conservation. J Fish Biol 59(Suppl A):79–85
66. Janko K, Culling MA, Ráb P, Kotlík P (2005) Ice age cloning – comparison of the quaternary evolutionary histories of sexual and clonal forms of spiny loaches (Cobitis; Teleostei) using the analysis of mitochondrial DNA variation. Mol Ecol 14:2991–3004
67. Janko K, Flajšhans M, Choleva L et al (2007) Diversity of European spined loaches (genus *Cobitis L.*): an update of the geographic distribution of the *Cobitis taenia* hybrid complex with a description of new molecular tools for species and hybrid determination. J Fish Biol 71:387–408
68. Boroń A, Kotusz J (2000) The preliminary data on diploid-polyploid complexes of *Cobitis* in the Oder River basin, Poland (Pisces, Cobitidae). Folia Zool 49(Suppl 1):79–84
69. Kotusz J (2000) Intra- and interpopulation morphological variability in diploid and varied-ploidy *Cobitis* from Poland. Folia Zool 49(Suppl. 1):219–226
70. Kotusz J (2008) Morphological relationships between polyploid hybrid spined loaches of the genus *Cobitis* (Teleostei: Cobitidae) and their parental species. Ann Zool 58:891–905
71. Janko K, Kotusz J, De Gelas K et al (2012) Dynamic formation of asexual diploid and polyploid lineages: multilocus analysis of *Cobitis* reveals the mechanisms maintaining the diversity of clones. PLoS One 7:e45384
72. Witkowski A (1979) A taxonomic study on fresh-water sculpins of genus *Cottus* Linnaeus, 1758 (*Cottus gobio* L. & *Cottus poecilopus* Heck.) in Poland. Acta Univ Wratislav 458:1–95
73. Bańbura J (1994) Lateral plate morph differentiation of freshwater and marine population of the three-spined stickleback, *Gasterosteus aculeatus*, in Poland. J Fish Biol 44:773–783
74. Rafiński J, Bańbura J, Przybylski M (1989) Genetic differentiation of freshwater and marine sticklebacks (*Gasterosteus aculeatus*) of eastern Europe. Z Zool Sys Evolu Forsch 27:33–43
75. Mäkinen HS, Cano JM, Merilä J (2006) Genetic relationships among marine and freshwater populations of the European three-spined stickleback (*Gasterosteus aculeatus*) revealed by microsatellites. Mol Ecol 15:1519–1534
76. Šedivá A, Janko K, Šlechtová V et al (2008) Around or across the Carpathians: colonization model of the Danube basin inferred from genetic diversification of stone loach (*Barbatula barbatula*) populations. Mol Ecol 17:1277–1292

77. Bohlen J, Šlechtová V, Doadrio I, Rab P (2007) Low mitochondrial divergence indicates a rapid expansion across Europe in the weather loach, *Misgurnus fossilis* (L.). J Fish Biol 71:186–194
78. Witkowski A (2001) Outline of the history of fish species conservation in Poland. In: Heese T, Witkowski A (eds) Regionalna ochrona gatunkowa ryb, stan zagrożenia, kierunki ochrony, vol 14(Suppl), Sci Ann Pol Angl Assoc, pp 45–57
79. Nowicki M (1883) Przegląd rozsiedlenia ryb w wodach Galicyi według dorzeczy i krain rybnych (Review of fish distribution in the waters of the Galicya region according to drainages and fish zones). E. Hölzl, Wiedeń, mapa (map)
80. Wałecki A (1863) Materiały do fauny ichtiologicznej Polski (Materials to investigating the fish fauna of Poland). Biblioteka Warszawska 4(11):325–364
81. Wałecki A (1864) Materyały do Fauny Ichtyologicznej Polski. II. Systematyczny przegląd ryb krajowych (Materials to investigating the fish fauna of Poland. II. Systematic review of our homeland's fishes). Druk Gazety Polskiej, Warszawa
82. Cios S (2003) Comments on the occurrence of trout, salmon and grayling in Polish waters in the past. Sci Ann Pol Angl Assoc 16:17–32
83. Radtke G, Bernaś R, Skóra M (2015) Occurrence of migratory and rheophilic fish and lamprey species in rivers of northern Poland in view of the historical data up to the beginning of the 20th century. Sci Ann Pol Angl Assoc 28:123–149
84. Cios S (2017) Historical information on the main fish species in the River San drainage area. Sci Ann Pol Angl Assoc 30:137–155
85. Rembiszewski JM, Rolik H (1975) Krągłouste i ryby. Cyclostomata et Pisces. Katalog Fauny Polski (Cz. 38 Cyclostomata et Pisces), vol 24. PWN, Warszawa
86. Witkowski A, Kotusz J (2008) State of fish fauna inventory in Polish rivers. Sci Ann Pol Angl Assoc 21:23–60
87. Brylińska M (2000) Ryby słodkowodne Polski (Freshwater fish of Poland). Wydawnictwo Naukowe PWN, Warszawa, 521 pp
88. Radtke G, Witkowski A, Grochowski A et al (2005) Discovery of the Siberian sculpin *Cottus poecilopus* Heckel, 1840 (Cottidae) in coastal rivers of the southern Baltic basin (N Poland). Przeg Zool 2-4:145–151
89. Hawks HA (1975) River zonation and classification. In: Whitton BA (ed) River ecology. Blackwell, Oxford, pp 312–374
90. Huet M (1959) Profiles and biology of western European streams as related to fish management. Trans Am Fish Soc 88:155–163
91. Holčik J, Banarescu P, Evanes D (1989) General introduction to fishes. In: Holčik J (ed) The freshwater fishes of Europe. Vol. 1. Part 2. General introduction to fishes. Acipenseriformes. Aula-Verlag GmbH, Wiesbaden
92. Starmach K (1956) Rybacka i biologiczna charakterystyka rzek (Fishing and biological characterisation of rivers). Pol Arch Hydrobiol 3:307–332
93. Man RHK (1996) Environmental requirements of European non-salmonid fish in rivers. Hydrobiologia 323:223–235
94. Błachuta J, Witkowski A (1990) The longitudinal changes of fish community, in the Nysa Kłodzka River (Sudety Mountains) in relation to stream order. Pol Arch Hydrobiol 37:235–242
95. Przybylski M (1993) Longitudinal pattern in fish assemblages in the upper Warta River, Poland. Arch Hydrobiol 126:499–512
96. Starmach J (1983/1984) Fish zones of the River Dunajec upper catchment basin. Acta Hydrobiol 25/26:415–427
97. Matthews WJ (1998) Patterns in freshwater fish ecology. Chapman and Hall, New York
98. Zalewski M, Frankiewicz P, Przybylski M et al (1990) Structure and dynamics of fish communities in temperate rivers in relation to the abiotic-biotic regulatory continuum concept. Pol Arch Hydrobiol 37:151–176

99. Jackson DA, Peres-Neto PR, Olden JD (2001) What controls who is where in freshwater fish communities – the roles of biotic, abiotic, and spatial factors. Can J Fish Aquat Sci 58:157–170
100. Vannote RL, Minshall GW, Cummins KW et al (1980) The river continuum concept. Can J Fish Aquat Sci 37:130–137
101. Zalewski M, Naiman RJ (1985) The regulation of riverine fish communities by a continuum of abiotic-biotic factors. In: Alabaster JS (ed) Habitat modification and freshwater fisheries. Butterworths, London, pp 3–9
102. Kruk A (2007) Role of habitat degradation in determining fish distribution and abundance along the lowland Warta River, Poland. J Appl Ichthyol 23:9–18
103. EC Parliament and Council (2000) Directive of the European Parliament and of the Council 2000/60/EC Establishing a Framework for Community Action in the Field of Water Policy. European Commission, Luxembourg. PE-CONS 3639/1/100 Rev 1
104. Błachuta J, Picińska-Fałtynowicz J, Czoch K, Kulesza K (2010) Abiotyczne typy wód płynących w Polsce (Abiotic types of running waters in Poland). Gospodarka Wodna 5:181–191
105. Błachuta J, Rosa J, Wiśniewolski W et al (2010) Ocena potrzeb i priorytetów udrożnienia ciągłości morfologicznej rzek w kontekście osiągnięcia dobrego stanu i potencjału części wód w Polsce (Assessment of requirements and priorities for morphological defragmentation of rivers in the context of attaining a good state and potential of water sectors in Poland). Krajowy Zarząd Gospodarki Wodnej, Warszawa, 56 pp
106. Kondracki J (1980) Geografia fizyczna Polski, 4th edn (Physical geography of Poland 4th edn). PWN, Warszawa, 463 pp
107. Augustyn L, Nowak M (2014) Long term changes of fish fauna in the Polish parts of the Czarna Orawa catchment. Sci Ann Pol Angl Assoc 27:51–78
108. Witkowski A, Kotusz J, Kusznierz J et al (2006) The ichthyofauna of the Polish tributaries of the Elbe River. Sci Ann Pol Angl Assoc 19:25–45
109. Tybulczuk S, Marszał L, Kruk A et al (2017) Fish Fauna of the Gwda River system (2013–2015). Sci Ann Pol Angl Assoc 30:59–94
110. Jażdżewski M, Rachalewska D, Zięba G et al (2014) Monitoring of ichthyofauna in rivers – aims and problems. Sci Ann Pol Angl Assoc 27:129–145
111. Prus P, Wiśniewolski W, Adamczyk M (eds) (2016) Monitoring ichtiofauny rzek. Przewodnik metodyczny (Monitoring of riverine ichthyofauna. Methodological guide). Biblioteka Monitoringu Środowiska, Warszawa, 89 pp
112. Makomaska-Juchiewicz M, Baran P (2012) Monitoring gatunków zwierząt. Przewodnik metodyczny. Część trzecia (Monitoring of animal species. Methodological guide. Part three). Biblioteka Monitoringu Środowiska, Warszawa, 749 pp
113. Schiemer F, Waidbacher H (1992) Strategies of conservation of a Danubian fish fauna. In: Boon PJ, Calow P, Petts GE (eds) River conservation and management. Wiley, London, pp 365–382
114. Balon EK (1975) Reproductive guilds of fishes: a proposal and definition. J Fish Res Can 32:821–864
115. Witkowski A, Kotusz J, Przybylski M (2009) The degree of threat to the freshwater ichthyofauna of Poland: Red list of fishes and lampreys – situation in 2009. Chrońmy Przyr Ojcz 65:33–52
116. Heese T, Witkowski A (eds) (2001) Regionalna ochrona gatunkowa ryb, stan zagrożenia, kierunki ochrony (Regional protection of fish species, state of threat, directions of protection). Sci Ann Pol Angl Assoc 14(Suppl):358
117. Ludwig A, Debus L, Lieckfeld D et al (2002) When the American sea sturgeon swam east. Nature 493:447–448
118. Kolman R, Kapusta A, Duda A, Wiszniewski G (2011) Review of the current status of the Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* Mitchill 1815, in Poland: principles, previous experience, and results. J Appl Ichthyol 27:186–191

119. Witkowski A, Goryczko K, Augustyn L (2007) Głowacica (*Hucho hucho*) – sukces polskiej ichtiologii (Danube salmon (*Hucho hucho*) – success of Polish ichthyology). Kom Ryb 3:17–22
120. Dębowski P, Bernaś R (2017) Zmiany ichtiofauny rzek Drawieńskiego Parku Narodowego w okresie od 1997 do 2012 roku (Changes of the ichthyofauna of the Drawsko National Park in the period 1997-2012). In: Czerniawski R, Bliski P (eds) Funkcjonowanie i ochrona wód płynących (Functioning and protection of running waters). Wojewódzki Fundusz Ochrony Środowiska i Gospodarki Wodnej w Szczecinie (Voivodeship fund of environmental protection and water management in Szczecin), pp 403–416
121. Marszał L, Przybylski M (1996) Threatened and rare fish species in Central Poland. Zool Pol 41(Suppl):61–72
122. Głowacki ŁB, Penczak T (2013) Drivers of fish diversity, homogenization/differentiation and species range expansions at the watershed scale. Divers Distrib 19:907–918
123. Witkowski A (1996) Introduced fish species in Poland: pros and cons. Arch Polish Fish 4:101–112
124. Witkowski A (2002) Introduction of fishes into Poland: benefaction or plague? Nat Conser 59:41–52
125. Grabowska J, Kotusz J, Witkowski A (2010) Alien invasive fish species in Polish waters: an overview. Folia Zool 59:73–85
126. Witkowski A, Grabowska J (2012) The non-indigenous freshwater fishes of Poland: threats to the native ichthyofauna and consequences for the fishery: a review. Acta Ichthyol Pisc 42:77–87
127. Nowak M, Koščo J, Popek W, Epler P (2010) First record of the black bullhead *Ameiurus melas* (Teleostei: Ictaluridae) in Poland. J Fish Biol 76:1529–1532
128. Grabowska J, Pietraszewski D, Ondračkova M (2008) Tubenose goby *Proterorhinus marmoratus* (Pallas, 1814) has joined three other Ponto-Caspian gobies in the Vistula River (Poland). Aquat Invasions 3:250–254
129. Semenchenko V, Grabowska J, Grabowski M et al (2011) Non-native fish in Belarusian and Polish areas of the European central invasion corridor. Oceanol Hydrobiol Stud 40:57–67
130. Skóra ME, Arciszewski B (2013) New reports on the Russian sturgeon, *Acipenser gueldenstaedtii*, in the Polish Baltic Sea. Arch Pol Fish 21:363–366
131. Bochenek D (ed) (2010) Environment 2010. Central Statistical Office, Warsaw
132. Penczak T, Kruk A (2000) Threatened obligatory riverine fishes in human-modified Polish rivers. Ecol Freshwat Fish 9:109–117
133. Kruk A (2006) Self-organizing maps in revealing variation in non-obligatory riverine fish in long-term data. Hydrobiologia 553:43–57
134. Witkowski A, Błachuta J, Kusznierz J (1991) Ichthyofauna of the Widawa River drainage basin after regulation. Sci Ann Pol Angl Assoc 4:25–46
135. Przybylski M (1996) Variation in fish growth characteristics along a river course. Hydrobiologia 325:39–46
136. Dembiński Z, Drożdżyk A (1999) Water quality in the Ner River in the context of progress in water and sewage management in Łódź Metropolitan Area. In: Burchard J, Ziułkiewicz M (eds) Chemism of atmospheric precipitation, and surface and ground waters. University of Łódź, Łódź, pp 15–18
137. Dembiński Z, Drożdżyk A (1999) Water quality in the Bzura River in the context of technical solutions in the 1990s. In: Burchard J, Ziułkiewicz M (eds) Chemism of atmospheric precipitation, and surface and ground waters. University of Łódź, Łódź, pp 18–20
138. Penczak T (1975) Ichthyofauna of the catchment area of the River Ner and perspectives of its restitution in connection with the erection of a collective sewage treatment plant for the Agglomeration of the City of Łódź. Acta Hydrobiol 17:1–20
139. Kostrzewa J, Penczak T (2002) Stan ichtiofauny dorzecza Neru i perspektywy jej restytucji (The condition of fish fauna in the Ner River system and the prospects of its restitution). In: Raport o stanie środowiska w województwie łódzkim w 2001 roku (Report on the state of

environment in the Łódź Voivodeship in 2001). Biblioteka Monitoringu Środowiska, Łódź, pp 100–102
140. Penczak T, Kruk A, Koszaliński H, Zięba G (2000) Fish fauna of the Bzura River. Sci Ann Pol Angl Assoc 13:23–33
141. Penczak T, Kruk A, Grabowska J, Śliwińska A, Koszaliński H, Zięba G, Tybulczuk S, Galicka W, Marszał L (2010) Influence of gradual improvement in water quality on the regeneration of fish fauna in the Ner River. Sci Ann Pol Angl Assoc 23:97–117
142. Kruk A, Penczak T (2013) Natural regeneration of fish assemblages in the Pilica River after a reduction of point-source pollution. River Res Appl 29(4):502–511
143. Kruk A, Ciepłucha M, Zięba G et al (2017) Spatially diverse recovery (1986–2012) of fish fauna in the Warta River, Poland: the role of recolonizers' availability after large-area degradation. Ecol Eng 102:612–624
144. Penczak T, Kruk A, Głowacki Ł, Galicka W (2014) The role of streams of different sizes in recovery of fish fauna in the Pilica River catchment. River Res Appl 30:1034–1047
145. Trawczyńska A, Tołoczko W (2005) Content of heavy metals in ground water of the soil of Bzura River valley. Ecol Chem Eng 12:121–126
146. Mosiej J, Komorowski H, Karczmarczyk A, Suska A (2007) Effect of pollutants discharged from Łódź Conurbation on quality of water in Ner and Warta Rivers. Acta Sci Pol Form Circum 6:19–30
147. Wagner I, Breil P (2013) The role of ecohydrology in creating more resilient cities. Ecohydrol Hydrobiol 13:113–134
148. Brzezińska A, Zawilski M, Sakson G (2016) Assessment of pollutant load emission from combined sewer overflows based on the online monitoring. Environ Monit Assess 188:502
149. Kaźmierczak B, Kotowski A (2012) Verification of storm water drainage capacity in hydrodynamic modeling. Prace Naukowe Instytutu Inżynierii Ochrony Środowiska Politechniki Wrocławskiej 92:1–143
150. Kruk A, Szymczak M, Spychalski P (2003) Ichthyofauna of the City of Łódź. Part I. Systems of the Jasień and Łódka Streams. Sci Ann Pol Angl Assoc 16:79–96
151. Starkel L (ed) (1987) Evolution of the Vistula River valley during the last 15000 years. Part II. Polish Academy of Sciences, Institute of Geography and Spatial Organization, Geographical Studies, Special Issue 4, pp 1–252
152. Łajczak A (2006) River training vs. flood exposure. The example of the River Vistula between Skoczów and Puławy, Poland. Infrastruktura i Ekologia Terenów Wiejskich 4:197–215
153. Bochenek D (ed) (2017) Environment 2017. Central Statistical Office, Warsaw
154. Jurajda P (1995) Effect of channelization and regulation on fish recruitment in a flood plain river. Regul Rivers Res Manage 10:207–215
155. Penczak T, Zięba G, Koszaliński H, Kruk A (2003) The importance of oxbow lakes for fish recruitment in a river system. Arch Hydrobiol 158:267–281
156. Ivanchev VP, Ivancheva EY, Tereshchenko VG (2011) Significance of a floodplain in the formation of fish communities in small rivers of Ryazan oblast. J Ichthyol 51:618–632
157. Grift RE, Buijse AD, van Densen WLT et al (2003) Suitable habitats for 0-group fish in rehabilitated floodplains along the lower River Rhine. River Res Appl 19:353–374
158. Penczak T (2001) Populations of fish in relation to riparian ecotone development in the Narew River catchment. Ecohydrol Hydrobiol 1:163–176
159. Kruk A, Penczak T, Zięba G et al (2006) Fish fauna of the Widawka River system. Part I. Widawka. Sci Ann Pol Angl Assoc 19:85–101
160. Bischoff A, Wolter C (2001) Groyne-heads as potential summer habitats for juvenile rheophilic fishes in the lower Oder, Germany. Limnologica 31:17–26
161. Eick D, Thiel R (2013) Key environmental variables affecting the ichthyofaunal composition of groyne fields in the middle Elbe River, Germany. Limnologica 43:297–307
162. Lamouroux N, Cattanéo F (2006) Fish assemblages and stream hydraulics: consistent relations across spatial scales and regions. River Res Appl 22:727–737

163. Witkowski A, Błachuta J, Kusznierz J, Kołacz M (1992) Ichthyofauna of the Ślęża and Oława Rivers and their tributaries. Sci Ann Pol Angl Assoc 5:137–154
164. Nilsson C, Reidy CA, Dynesius M, Revenga C (2005) Fragmentation and flow regulation of the world's large river systems. Science 308(5720):405–408
165. Błachuta J, Jelonek M, Panasiuk D et al (2006) Dams and floods. Towarzystwo na rzecz Ziemi. Polska Zielona Sieć, Kraków
166. Łajczak A, Plit J, Soja R et al (2006) Changes of the Vistula River channel and floodplain in the last 200 years. Geogr Pol 79:65–87
167. Pauli M, Hunzinger L, Hitz O (2018) More bed load in rivers. Achieving a sediment balance close to the natural state. J Appl Water Eng Res. https://doi.org/10.1080/23249676.2018.1497554
168. Nelson RW, Dwyer JR, Greenberg WE (1987) Regulated flushing in a gravel-bed river for channel habitat maintenance: a Trinity River fisheries case study. Environ Manag 11:479–493
169. McDonald R, Nelson J, Paragamian V, Barton G (2010) Modeling the effect of flow and sediment transport on white sturgeon spawning habitat in the Kootenai River, Idaho. J Hydraul Eng 136:1077–1092
170. Roussel JM (2007) Carry-over effects in brown trout (*Salmo trutta*): hypoxia on embryos impairs predator avoidance by alevins in experimental channels. Can J Fish Aquat Sci 64:786–792
171. Wiśniewolski W (1987) Commercial catches of fish in the Vistula, Oder and Warta Rivers in 1953-1978. Ann Agr Sci 101:71–114
172. Backiel T, Bontemps S (1996) The recruitment success of *Vimba vimba* transferred over a dam. J Fish Biol 48:992–995
173. Mastyński J, Andrzejewski W, Czarnecki M, Iwaszkiewicz M (1997) Wstępne wyniki badań oddziaływania elektrowni wodnej Jeziorsko na ryby (Preliminary results of a study on the impact of the Jeziorsko hydropower plant on fish). Kom Ryb 4:11–13
174. Bartel R, Wiśniewolski W, Prus P (2007) Impact of the Włocławek dam on migratory fish in the Vistula River. Arch Pol Fish 15:141–156
175. Čada GF (1998) Efforts to reduce the impacts of hydroelectric power production on reservoir fisheries in the United States. Int Rev Hydrobiol 83:43–50
176. Bieniarz K, Epler P, Bartel R (1992) Passing of fish trough the turbines of the hydroelectric plants situated on some Pomeranian rivers. Sci Ann Pol Angl Assoc 5:119–124
177. Bartel R, Bieniarz K, Epler P (1993) Fish passing through the turbines of 4 hydroelectric plants situated on Słupia River. Sci Ann Pol Angl Assoc 6:137–142
178. Penczak T, Głowacki Ł, Galicka W, Koszaliński H (1998) A long-term study (1985-1995) of fish populations in the impounded Warta River, Poland. Hydrobiologia 368:157–173
179. Brinker A, Chucholl C, Behrmann-Godel J et al (2018) River damming drives population fragmentation and habitat loss of the threatened Danube streber (*Zingel streber*): implications for conservation. Aquat Conserv Mar Freshwat Ecosyst 28:587–599
180. Argentina JE, Angermeier PL, Hallerman EM, Welsh SA (2018) Spatial extent of analysis influences observed patterns of population genetic structure in a widespread darter species (Percidae). Freshwat Biol 63:1185–1198
181. Wołos A, Draszkiewicz-Mioduszewska H, Mickiewicz M et al (2018) Gospodarka rybacka w śródlądowych wodach płynących w 2016 roku. Cz. 1. Uprawnieni do rybactwa, obwody rybackie, połowy gospodarcze, zatrudnienie i połowy amatorskie (Fishing economy in fluvial inland waters in 2016. Vol. 1. Certified fishermen, fishing regions, commencial fishing, employment and amateur fishing). Kom Ryb 1:7–14
182. Mickiewicz M, Wołos A, Draszkiewicz-Mioduszewska H (2018) Gospodarka rybacka w śródlądowych wodach płynących w 2016 roku. Cz. 2. Charakterystyka zarybień (Fishing economy in fluvial inland waters in 2016. Part 2. Characteristics of stocking). Kom Ryb 2:13–19
183. Bartel R, Kardela J (2009) Realizacja programu "Restytucja certy (*Vimba vimba* L.) i jesiotra ostronosego (*Acipenser oxyrhynchus* M.) w roku 2008" (Realisation of the program

"Restitution of vimba bream (*Vimba vimba* L.) and Atlantic sturgeon (*Acipenser oxyrhynchus* M.) in 2008"). Kom Ryb 5:18–21
184. Bartel R, Kardela J (2012) Stocking Polish marine areas in 2011. Kom Ryb 3:18–40
185. Dębowski P (2018) The largest Baltic population of sea trout (*Salmo trutta* L.): its decline, restoration attempts, and current status. Fish Aquat Life 26:81–100
186. Dębowski P (2018) Fish passage through the Włocławek dam fishway in 2017 and preliminary analysis of fish runs in 2015-2017. Kom Ryb 2:7–12
187. Radtke G, Bernaś R, Dębowski P, Skóra M (2012) The fish fauna of the Osa River system. Sci Ann Pol Angl Assoc 25:31–47
188. Radtke G, Grochowski A, Woźniewski M (2003) Ichthyofauna of the Wda River system. Sci Ann Pol Angl Assoc 16:33–64
189. Marszał L, Zięba G, Przybylski M et al (2006) Fish fauna of the Liwiec River system. Sci Ann Pol Angl Assoc 19:47–70
190. Penczak T, Kruk A, Kostrzewa J et al (2003) The fish fauna of the Prosna River system. Part I. Prosna. Sci Ann Pol Angl Assoc 16:65–78
191. Penczak T, Kruk A, Kostrzewa J et al (2003) The fish fauna of the Prosna River system. Part II. Tributaries. Sci Ann Pol Angl Assoc 17:55–76
192. Keszka S, Tański A, Raczyński M et al (2013) Fish fauna of the Ina River system. Sci Ann Pol Angl Assoc 26:117–149
193. Radtke G, Dębowski P, Grochowski A (2006) The ichthyofauna of the Łupawa River system. Sci Ann Pol Angl Assoc 19:71–84
194. Dębowski P (1997) The ichthyofauna of the Parsęta River drainage basin. Sci Ann Pol Angl Assoc 10:21–60
195. Terlecki J, Kozłowski J, Dostatni D et al (2004) Fish fauna of Łyna, Guber, Dajna and Sajna Rivers. Sci Ann Pol Angl Assoc 17:35–54
196. Augustyn L, Nowak M (2014) Fish fauna of the Polish part of the Czarna Orawa catchment. Sci Ann Pol Angl Assoc 27:51–78
197. Skóra S, Włodek JM, Augustyn L, Nawrocki J (1994) Ichthyofauna of the Biała Dunajcowa River catchment area. Sci Ann Pol Angl Assoc 7:19–37
198. Kotusz J, Kusznierz J, Popiołek M, Witkowski A (2009) The ichthyofauna of the Nysa Kłodzka River system. Sci Ann Pol Angl Assoc 22:5–58

Fish and Fisheries in the Lakes of Northeastern Poland

Andrzej Kapusta, Elżbieta Bogacka-Kapusta, and Arkadiusz Wołos

Contents

Abstract In this article, we present the most comprehensive and up-to-date assessment of freshwater fish diversity in the lakes of northeastern Poland, and we use these data to characterize patterns of fish diversity and characterize the fisheries. The northeastern region of Poland is home to the country's largest complex of lakes, with the deepest (Hańcza) and largest (Śniardwy) lakes in Poland. To date, 43 species belonging to 15 families are confirmed to occur in the lakes of northeastern Poland. Among these, the cyprinids are dominant. Most of the fish species noted in this region occur commonly in Poland; however, as many as 27% of the species are classified as highly endangered and are under species conservation. A substantial part of the fish fauna of northeastern Polish lakes is comprised of alien species, of which nine are noted in the region. For centuries, this region was the center of lake fisheries in the southern Baltic Sea basin. Currently, 70 different types of enterprises conduct fisheries in the region, but, in the face of progressing lake eutrophication, changes in catch structure are occurring, and the fisheries yield is decreasing.

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Common bream (*Abramis brama*), pike (*Esox lucius*), roach (*Rutilus rutilus*), and eel (*Anguilla anguilla*) are commercially the most important fish species in commercial catches in the lakes of northeastern Poland.

Keywords Fish composition · Freshwater ecosystems · Inland fisheries · Lake fisheries

1 Introduction

The northeastern region of Poland conjures up images of forests and waters, and it is home to the largest complex of lakes in Poland. There are approximately 2,700 lakes of a surface area greater than 1 ha covering an area of 1,450 km^2, which is nearly 30% of the number of lakes and 45% of lake surface area in Poland [1, 2]. This is where the largest (over 100 km^2) and deepest (over 100 m) lakes are located. This region was also the center of lake fisheries in the southern Baltic Sea basin for centuries.

Studies of riverine fish fauna include most river systems [3, 4]. In comparison to that of rivers, the state of knowledge about the fish fauna of lakes is much worse. Fish fauna studies of lakes are rendered difficult because, in comparison to rivers, they have a significantly wider range of habitats and surface areas are usually large. The species composition of fish inhabiting lakes is most frequently determined during studies focusing on the biology of commercial species [5] or during monitoring catches [6], or it is based on commercial catches and information obtained from professional fishermen [7, 8]. Studies of fish in Polish lakes more frequently concentrate on describing biological or ecological traits [9, 10], than on assessing the entire fish assemblage qualitatively or quantitatively [11]. There are relatively few studies that describe the fish assemblages of individual lakes and even fewer studies that describe the fishes of a few or a dozen or more lakes in a single area [12, 13]. Consequently, relatively little is known about the composition of fish assemblages in lakes. Knowledge of the composition of fish assemblages in many aquatic basins is based exclusively on knowledge obtained from the fisheries that are associated with either commercial or recreational catches.

2 Historical Data Regarding Fishes

Lakes located in the region of northeastern Poland have been the subject of more or less detailed studies regarding fishes and fisheries management [7, 14–16]. Nevertheless, no publication has addressed in detail the species composition or quantitative structure of assemblages in the lakes in this area. Historical publications contain summary information on the occurrence of some species in the Masurian region [14, 17–19], while more detailed studies analyze fisheries management

[15, 16]. Historical materials and fisheries documentation permit making generalizations about the fish species composition of the area. In comparison with other regions of Poland, Masuria remains one of the least investigated areas with regard to fish assemblage species composition and quantitative structure.

The most exhaustive study of fish is the combined results of research work that focuses on the fish from lakes in the vicinity of Węgorzewo [5]. A team from the Inland Fisheries Institute conducted research focusing on the productivity of the lakes and the impact of environmental factors on fish growth. Catches of fish using nets of various mesh sizes and electro-fishing also permitted fairly precise determinations of the species composition and quantitative structure of more than a dozen lakes [5, 20].

3 Species Richness

The occurrence of at least 43 species belonging to 15 families was noted in the lakes of northeastern Poland (Table 1). The most species are from the cyprinid family, while the other families are represented by single species. Species that prefer riverine habitats occur in the littoral zone of some lakes. Among these, the rarest species is *Barbus barbus*, which was noted in Lake Śniardwy in the mid-twentieth century [23], but it has not been confirmed in this area for many years. The lake form of the species *Salmo trutta* currently occurs in lakes Hańcza and Szurpiły, while formerly it also occurred in lakes Wigry, Białe Wigierskie, Sajno, and Sajenek [24]. The lakes of northeastern Poland are the primary distribution area of *Osmerus eperlanus* in Polish inland waters. Generally, species that prefer cold water and require well-oxygenated water are noted with the least frequency in the lakes of this area.

The occurrence of some species is associated with fisheries management and particularly with deliberate or accidental stocking. Asian carp species (*Ctenopharyngodon idella* and *Hypophthalmichthys* spp.) do not reproduce in the inland waters of Poland [25], which is why their occurrence depends on stocking. Fortunately, the trend of stocking lakes with *Coregonus peled* has stopped, and the further occurrence or even hybridization with indigenous species of the genus *Coregonus* appears to be quite improbable [26]. *Umbra krameri* was introduced along with *Tinca tinca* stocking material from Hungary [27], but it did not establish local populations.

4 Conservation Status

Most fish species noted in this region occur commonly in various types of inland waters in Poland, and they are characterized by broad tolerance to environmental factors [28]. It must also be noted that species that are classified as threatened or are under species conservation are also observed. In total, 27% of the species confirmed

Table 1 List of fish species confirmed in the Masurian Landscape Park

Family	Scientific name	Reproductive guilds	Origin	IUCN category
Anguillidae	*Anguilla anguilla* (L.)	Pelagophil	N	CR
Salmonidae	*Salmo trutta* L.	Lithophil	N	CD
	Coregonus peled Gmelin	Litho-pelagophil	I	–
	Coregonus albula (L.)	Litho-pelagophil	N	VU
	Coregonus lavaretus (L.)	Lithophil	N	VU
Osmeridae	*Osmerus eperlanus* (L.)	Litho-pelagophil	N	VU
Lotidae	*Lota lota* (L.)	Litho-pelagophil	N	VU
Esocidae	*Esox lucius* L.	Phytophil	N	LC
Umbridae	*Umbra krameri* Walbaum	Phytophil	I	–
Cyprinidae	*Cyprinus carpio* L.	Phytophil	I	–
	Carassius carassius (L.)	Phytophil	N	NT
	Carassius gibelio (Bloch)	Phytophil	I	–
	Rutilus rutilus (L.)	Phyto-lithophil	N	LC
	Scardinius erythrophthalmus (L.)	Phytophil	N	LC
	Leusciscus idus (L.)	Phyto-lithophil	N	LC
	Squalius cephalus (L.)	Lithophil	N	LC
	Leucaspius delineatus Heckel	Phytophil	N	LC
	Abramis brama (L.)	Phyto-lithophil	N	LC
	Blicca bjoerkna (L.)	Phyto-lithophil	N	LC
	Leuciscus aspius (L.)	Lithophil	N	NT
	Alburnus alburnus (L.)	Phyto-lithophil	N	LC
	Tinca tinca (L.)	Phytophil	N	LC
	Barbus barbus (L.)	Lithophil	N	VU
	Vimba vimba (L.)	Lithophil	N	CR
	Rhodeus amarus (Bloch)	Ostracophil	N	VU
	Pseudorasbora parva (Temminck and Schlegel)	Phyto-lithophil	I	–
	Gobio gobio (L.)	Psammophil	N	LC
	Ctenopharyngodon idella (Val.)	Pelagophil	I	–
	Hypophthalmichthys molitrix (Val.)	Pelagophil	I	–
	Hypophthalmichthys nobilis (Richardson)	Pelagophil	I	–
Nemacheilidae	*Barbatula barbatula* (L.)	Psammophil	N	LC
Cobitidae	*Cobitis taenia* L.	Phytophil	N	LC
	Misgurnus fossilis (L.)	Phytophil	N	VU
Ictaluridae	*Ameiurus nebulosus* (Lesueur)	Speleophil	I	–
Siluridae	*Silurus glanis* L.	Phytophil	N	NT
Percidae	*Perca fluviatilis* L.	Phyto-lithophil	N	LC
	Sander lucioperca (L.)	Phytophil	N	LC
	Gymnocephalus cernuus (L.)	Phyto-lithophil	N	LC

(continued)

Table 1 (continued)

Family	Scientific name	Reproductive guilds	Origin	IUCN category
Gasterosteidae	*Gasterosteus aculeatus* L.	Ariadnophil	N	LC
	Pungitius pungitius (L.)	Ariadnophil	N	LC
Gobiidae	*Neogobius fluviatilis* (Pall.)	Lithophil	I	–
Cottidae	*Cottus gobio* (L.)	Speleophil	N	VU
	Cottus poecilopus Heckel	Speleophil	N	VU

Reproductive guilds [21], IUCN [22]: CR, critically endangered; EN, endangered; VU, vulnerable; NT, near threatened; CD, conservation dependent; LC, least concern. Origin: N, native; I, introduced

in this region are classified as the highest level of critically endangered according to the International Union for Conservation of Nature (Table 1). *Anguilla anguilla* is a critically endangered species throughout its range of occurrence, while *Vimba vimba* is a critically endangered species throughout Poland. In turn, *Coregonus albula*, *C. lavaretus*, *O. eperlanus*, *Lota lota*, *B. barbus*, *Rhodeus amarus*, *Misgurnus fossilis*, *Cottus gobio*, and *C. poecilopus* are classified as vulnerable species, and *Carassius carassius*, *Leuciscus aspius*, and *Silurus glanis* are classified as near threatened (Table 1). *R. amarus*, *M. fossilis*, *C. gobio*, *C. poecilopus*, *Barbatula barbatula*, and *Cobitis taenia* are under partial species conservation. Additionally, *R. amarus*, *C. taenia*, *C. gobio*, and *M. fossilis* are listed in Annex II of the Habitat Directive, and *C. albula*, *C. lavaretus*, *B. barbus*, and *L. aspius* are listed in Annex V of the same directive.

The majority of autochthonous fish species in Poland have been severely affected by years of human pressure and have reacted with the collapse of local populations [22]. The situation of fish assemblages in the lakes of northeastern Poland is similar. The confirmation of *V. vimba* in the food of *Phalacrocorax carbo* nesting on the island in Lake Warnołty certainly qualifies as a curiosity [29]. For many years *V. vimba* had not been registered in fisheries catches, and the specimens found probably originated from the local form of this species that occurs in Lake Roś. In Poland, *V. vimba* is classified as critically endangered. Of its former range of occurrence, which included most of the Vistula and Oder river tributaries, currently only a small fragment remains. Freshwater populations have formed in a few river systems, and within the region analyzed, this species occurs in lakes Roś (the Pisa–Vistula river system) and Maróz (Łyna–Pregoła river system).

5 Alien Species

A significant part of the fish fauna of the lakes of northeastern Poland are alien species, of which nine have been noted (Table 1). To date, approximately 40 alien fish species have been noted in Polish inland waters [30]. Most of these alien species occur as the result of intentional or accidental stocking. The process itself of the spread of alien fish species is very poorly documented. *Cyprinus carpio* and *Carassius gibelio* occur in lakes as a result of intentional stocking. While Asian carps have not been permitted to be released into lakes for a few years, they are commonly expanding their distribution naturally from ponds and are released into open waters along with stocking material of other cyprinid species. Most of the alien species confirmed in Polish waters do not reproduce in these latitudes [28]; however, they do have a disadvantageous impact on native fauna and flora and sometimes on the ecological and trophic statuses of waters. When the large numbers of *C. carpio* released into lakes feed, they release nutrients that accumulate in the sediments, and this increase the trophic status of the waters. The groups of alien fish species reproducing in the region analyzed include *C. gibelio*, *Pseudorasbora parva*, *Ameiurus nebulosus*, and *Neogobius fluviatilis*. All of these species are invasive species in Poland [25]. *C. gibelio* occurs in most lakes, and its spread is linked to stocking. In the mid-twentieth century, *A. nebulosus* occurred sporadically in the region studied [20], but in recent years, its distribution has increased [31]. This species is noted at dispersed sites throughout nearly the entire area of northeastern Poland, although the occurrence of this species was noted previously [20]. The expansion of *N. fluviatilis* is progressing from the Vistula River through the Narew and Pisa rivers. In recent years, *N. fluviatilis* has settled in the southern part of the Masurian Lakeland.

6 Fisheries Management

Fisheries in the lake region of northeastern Poland has a very long tradition. Despite ownership and structural changes, it remains the central region of lake fisheries in Poland. Approximately 70 enterprises of various types are active in the fisheries sector of this area. As a rule, these enterprises each exploit a few lakes with surface areas of a dozen or so hectares [32]. The largest fisheries enterprises exploit lakes of a combined surface area exceeding 12,000 or so hectares. Thirteen fisheries enterprises exploit over 3,000 ha, and another 4 exploit lakes with a combined surface area exceeding 1,000 ha.

Traditionally, the fisheries sector in Poland is divided into four types. The previously dominant type of commercial fisheries has been replaced by recreational fisheries, which is now the dominant type of fisheries, at least in terms of area. The second most important type of fisheries is angling, which is usually conducted in fisheries districts exploited by the Polish Anglers Association. The least common type of fisheries is specialized fisheries conducted by scientific institutions (Inland Fisheries Institute, University of Warmia and Mazury) or fisheries that specialize in fish culture. It should be underscored here that the division of the fisheries sector is often arbitrary. In the case of many lakes, fish populations are regularly exploited by anglers, while catches are also made with professional fishery, and the decided majority of lakes are stocked more or less intensely.

7 Fisheries Yield

Following World War II, almost all the lakes in northeastern Poland were exploited by state commercial lake fish farms, which were legally required to conduct rational fishery management [33]. These farms were also required to keep detailed catch statistics and to record management practices. In the early 1950s, fisheries and biological studies were initiated in over 2,000 lakes, which corresponded to most Polish lakes. In the early period, the aim was to increase catches of commercially important fish species [20]; however, as the eutrophication of lakes progressed, changes were noted in the structure of the catches, and their yield decreased [34]. Catches of commercially important fishes have also decreased in recent years (Fig. 1). In the 2004–2016 period, annual yield decreased by approximately 300–880 tons in 2016. The three most significant species in terms of biomass (*Abramis brama*, *Esox lucius*, *Rutilus rutilus*) comprise 50–60% of the total catches. The species groups that contributed a 5–10% share of the catches are *C. albula*, *Perca fluviatilis*, *Blicca bjoerkna*, *T. tinca*, *A. anguilla*, and *Sander lucioperca*. However, the economic ranking of these species is different, with *A. anguilla* accounting for about 30% of the value of professional catches followed by *E. lucius* – 20%, *C. albula* – 13%, and *S. lucioperca* – 10% [32, 35].

Decreases in fish catches were also noted in catch efficiency. In the 1950s and 1960s, the mean commercial catch efficiency was 26 kg/ha [20]. In recent years catch efficiency has decreased by nearly 13–7.2 kg/ha (Fig. 2). Considering, however, the most important of the species caught, the trend in changes of fishing

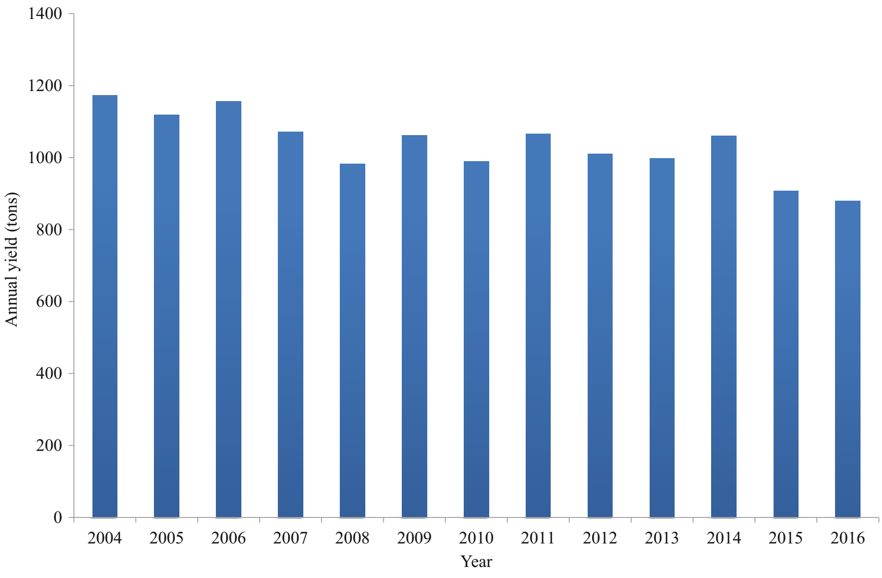

Fig. 1 Fisheries yield (tons) of professional fisheries in lakes in northeastern Poland in 2004–2016

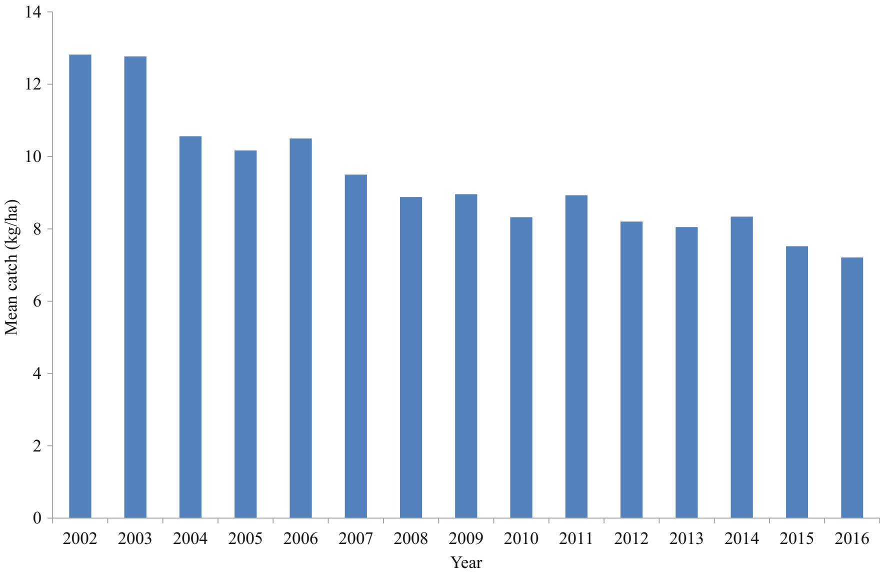

Fig. 2 Changes in mean catch efficiency (kg/ha) of catches of professional fisheries in lakes in northeastern Poland in 2002–2016

efficiency was not as unequivocal. The catch efficiency of *C. albula* exhibited substantial fluctuation (from 0.5 to 1.1 kg/ha), which is characteristic of this species. Decreases in catch efficiency were noted for *A. anguilla*, *A. brama*, and *R. rutilus*, while increases were noted for *E. lucius*, *T. tinca*, and *S. lucioperca*. It must be underscored that catch efficiency does not necessarily reflect fish biomass or species composition since only commercially valuable species are subjected to intense exploitation.

8 Conclusions

Fish inhabiting lakes form characteristic assemblages that are generally linked with lake limnological type. Similarly, in rivers the fish fauna structure depends largely on environmental conditions. The species richness and biological diversity of fish fauna is a reflection of interactions among local species groups and environmental conditions. The species composition and quantitative structure of assemblages inhabiting the lakes of northeastern Poland are subject to long-term ecological succession. The occurrence of fish in lakes is conditioned by many factors, and the natural processes that shape the structure of fish assemblages have been aggravated by anthropogenic factors [36]. Despite very clear changes caused by water pollution, habitat transformation, and fisheries management (catches, stocking), the fish fauna of the lakes of northeastern Poland is characterized by considerable species richness in comparison to other areas of the country. A large segment of the species occurring here are subject to legal conservation and are classified as threatened according to IUCN categories. The main threats to indigenous fish fauna include habitat transformation, water eutrophication, and the spread of alien fish species.

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References

1. Kondracki J (1965) A geomorphologic and hydrographic outline of the Mazurian Lakeland. In: Patalas K (ed) Mazury. Part I. XVI Limnologorum Conventus in Polonia MCMLXV. Polish Academy of Sciences, Hydrobiological Committee, Cracow, pp 21–37
2. Jokiel P, Marszalewski W, Pociask-Karteczka J (2017) Hydrology of Poland. Wydawnictwo PWN, 342 pp (in Polish)
3. Witkowski A, Kotusz J (2008) Status of ichthyofauna inventory research in Polish rivers. Rocz Nauk PZW 21:23–60. (in Polish)
4. Przybylski M, Głowacki Ł, Grabowska J, Kaczkowski Z, Kruk A, Marszał L, Zięba G, Ziułkiewicz M (2019) Riverine fish fauna in Poland. In: Korzeniewska E, Harnisz M (eds) Polish River basins and lakes – Part II: biological status and water management. Springer, Cham
5. Zawisza J (1961) The growth of fish in the lakes of the Węgorzewo district. Rocz Nauk Rol 77-B-2:681–748. (in Polish)
6. Kapusta A, Morzuch J, Bogacka-Kapusta E, Pająkowski J (2012) Ichthyofauna of Lake Mukrz in the Leon Wyczółkowski Cisy Staropolskie nature reserve in Wierzchlas. Chrońmy Przyr Ojcz 68:435–442. (in Polish)
7. Chybowski Ł, Białokoz W (1992) Analysis of fish catches from the Krutynia River lake complex. Komun Ryb 1:9–11. (in Polish)
8. Boroń A, Szlachciak J (2011) The ichthyofauna of Olsztyn. In: Jankun M, Furgała-Selezniow G, Woźniak M, Wiśniewska AM (eds) Assessment and biodiversity of waters. Wydział Ochrony Środowiska i Rybactwa UWM, Agencja Wydawnicza Argi S.C., Olsztyn and Wrocław, pp 179–189 (in Polish)
9. Czarkowski TK, Martyniak A, Kapusta A, Wójcik A, Bowszys M, Wziątek B, Szymańska U, Kozłowski J (2007) Feeding ecology of vendace, *Coregonus albula* (L.), in Lake Wigry (Northeastern Poland). Arch Pol Fish 15:117–128
10. Kozłowski K, Kozłowski J, Poczyczyński P, Martyniak A (2010) Age and growth of vendace, *Coregonus albula* (L.), from Lake Wigry (northeast Poland). Arch Pol Fish 18:239–245
11. Hutorowicz A, Kapusta A, Krzywosz T, Hutorowicz J (2005) The ichthyofauna of the dystrophic Lake Smolak (northern Poland) in light of selected physical and chemical water conditions thirty years after the conclusion of liming and fertilization. Arch Pol Fish 13:207–225
12. Białokoz W, Krzywosz T (1992) Ichthyofauna structure in the lakes of Wigry National Park. In: Zdanowski B (ed) The lakes of Wigry National Park: Eutrophication status and conservation. Polska Akademia Nauk, Komitet Naukowy przy Prezydium PAN „Człowiek i Środowisko" PAN, 3, Zeszyty Naukowe 3, Wydawnictwo Zakład Narodowy im. Ossolińskich, Wrocław, pp 153–162 (in Polish)
13. Martyniak A, Wziątek B, Szymańska U, Osewski M, Hliwa P (2004) Ichthyofauna composition and age and growth rates of fish in lakes Suchar Wielki, Sucharek, and Klonek. Rocz Augustowsko-Suwalski 4:131–136. (in Polish)
14. Benecke B (1881) Fische, Fischerei und Fischzucht in Ost- und Westpreussen. Hartungsche Verlag, Königsberg

15. Bernatowicz S (1947) On the threshold of managing the Masurian lakes. Prz Ryb 7/8:259–263. (in Polish)
16. Staff F, Rudnicki A (1949) Fisheries of Polish Prussia. Masurian Lake District. In: Staff F (ed) Fisheries of the recovered territories. Wyd. PIWR, Warsaw, pp 1–37 (in Polish)
17. Bernatowicz S (1953) Distribution and growth of vendace (*Coregonus albula* L.) in Masurian Lakes and environmental conditions. Rocz Nauk Rol 67-B-1:1–20. (in Polish)
18. Bernatowicz S (1953) The actual state of distribution of vendace (*Coregonus albula* L.) in the Masurian and Suwałki district lakes. Rocz Nauk Rol 67-B-1:61–80. (in Polish)
19. Rembiszewski JM, Rolik H (1975) Lampreys and fishes – Cyclostomata et Pisces. Catalog of Polish Fauna. 38. PWN, Warsaw, 252 pp (in Polish)
20. Zawisza J (1965) Fisheries in Mazury. In: Patalas K (ed) Mazury. Part I. XVI Limnologorum Conventus in Polonia MCMLXV. Polish Academy of Sciences, Hydrobiological Committee, Cracow, pp 53–63
21. Balon EK (1975) Reproductive guilds of fishes: a proposal and definition. J Fish Res Board Can 32:821–864
22. Witkowski A, Kotusz J, Przybylski M (2009) Degree of threat to Polish freshwater ichthyofauna: red list of lampreys and fish – state in 2009. Chrońmy Przyr Ojcz 45:33–52. (in Polish)
23. Patalas K (1965) Mazury. Part II. XVI Limnologorum Conventus in Polonia MCMLXV. Polish Academy of Sciences, Hydrobiological Committee, Cracow, p 70
24. Chybowski Ł, Białokoz W, Krzywosz T (1994) Lake trout in the waters of the Suwałki region. In: Wołos A (ed) Current issues in lake fisheries. Wydawnictwo IRS, Olsztyn, pp 141–149. (in Polish)
25. Grabowska J, Kotusz J, Witkowski A (2010) Alien invasive fish species in Polish waters: an overview. Folia Zool 59:73–85
26. Popović D, Szczepkowski M, Heese T, Weglenski P (2016) Introgression of peled (Coregonus peled) into European whitefish (*C. lavaretus*) in Poland. Conserv Genet 17:503–508
27. Ejsmont L, Słoniewski B (1969) On the accidental record of the European mudminnow (*Umbra krameri* Walbaum, 1792) in Poland (Osteichthyes, Esociformes). Przegląd Zoologiczny 13:215–219. (in Polish)
28. Brylińska M (2000) The freshwater fishes of Poland. PWN, Warsaw, 521 pp (in Polish)
29. Krzywosz T, Traczuk P (2010) Local populations of vimba in Masuria? Komun Ryb 1:29. (in Polish)
30. Witkowski A, Grabowska J (2012) The non-indigenous freshwater fishes of Poland: threats to the native ichthyofauna and consequences for the fishery: a review. Acta Ichthyol Piscat 42:77–87
31. Kapusta A, Morzuch J, Partyka K, Bogacka-Kapusta E (2010) First record of brown bullhead, *Ameiurus nebulosus* (Lesueur), in the Łyna River drainage basin (northeast Poland). Arch Pol Fish 18:261–265
32. Wołos A (2009) Diagnosis of the current state and perspectives for the development of inland and coastal fisheries areas in the Warmian-Masurian Voivodeship. Wydawnictwo IRS, Olsztyn, 352 pp (in Polish)
33. Bnińska M, Leopold M (1990) Fisheries management in Polish lakes. In: van Densen WLT, Steinmetz B, Hughes RH (eds) Management of freshwater fisheries. Proceedings of a symposium organized by the European Inland Fisheries Advisory Commission, Göteborg, Sweden, 31 May-3 June 1988. Pudoc, Wageningen, pp 406–436
34. Leopold M (1986) Status and prospects of freshwater fisheries in Poland. Expert assessment for the Ministry of Agriculture, Forestry and Food. Instytut Rybactwa Śródlądowego, Olsztyn, p 44. (in Polish)
35. Wołos A, Draszkiewicz-Mioduszewska H, Mickiewicz M (2015) Magnitude and characteristics of lake fisheries production in 2009–2014. In: Mickiewicz M, Wołos A (eds) Exploiting fisheries resources in 2009–2014. State, changes, trends. Wydawnictwo IRS, Olsztyn, pp 9–16. (in Polish)

36. Mamcarz A, Skrzypczak A (2011) Between the ichthyofauna biodiversity of Masurian waters and their management: theory, problems, and hope. In: Jankun M, Furgała-Selezniow G, Woźniak M, Wiśniewska AM (eds) Managing the ichthyofauna of diversified aquatic environments. Wydział Ochrony Środowiska i Rybactwa UWM, Agencja Wydawnicza „Argi" S.C., Olsztyn and Wrocław, pp 7–38 (in Polish)

Freshwater Habitats and Freshwater-Dependent Habitats in Poland

Mirosław Grzybowski

Contents

Abstract This paper briefly discusses the resources of habitats of importance to the European Community in Poland shaped by water relations. The types of habitat surveyed include water courses and waterbodies, wetlands and riparian vegetation (meadows, forest) and streams and bogs. The study covered 23 habitat types in freshwater habitats and freshwater-dependent habitats occurring in all 849 Special Areas of Conservation in Poland in both biogeographic regions: alpine and continental. The overall conservation value for current habitats is presented, including threats, pressures and activities, as well as their possibilities for restoration.

Keywords Biodiversity conservation · Freshwater-dependent habitats · Freshwater habitats · Natura 2000

1 Introduction

One key conservation action worldwide is the development of large-scale networks of protected areas [1]. In spite of the fact that over 200,000 protected areas cover ~14% of the world's land area [2], there are very few coordinated networks of

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protected areas aiming at continental-scale conservation. Examples of such networks stretching across national borders include the Yellowstone to Yukon Conservation Initiative in North America and the European Ecological Network – Natura 2000. The latter is the world's largest multinationally coordinated conservation infrastructure [3]. The network provides ecosystem services worth ca 200–300 billion Euro/ year [4]. Natura 2000 is also becoming an essential component of the European Green Infrastructure strategy aimed at mitigating fragmentation and increasing the spatial and functional connectivity between protected and unprotected areas [5, 6].

Protecting over 2,000 habitat types and species of community importance, the Birds and Habitats Directives are the cornerstone of EU nature conservation policy. However, the Nature Directives are still not fully implemented, and a number of listed habitat types and species are far from having a favourable conservation status. These gaps have been acknowledged by the EU Biodiversity Strategy to 2020 [7], which calls, among other targets, for the full implementation of the Nature Directives (Target 1) and for the restoration of at least 15% of degraded ecosystems (Target 2) by 2020.

Species protection under the Habitats Directive that included all in all over 1,000 animal and plant species, as well as 200 habitat types, listed in the directive's annexes are protected in various ways, such as in Special Areas of Conservation (SACs). Conservation practice should be based on the best available scientific evidence [8]. Unfortunately, implementation of conservation policies is often hampered by a lack of such knowledge, insufficient understanding of local conditions or inadequate consideration of certain taxa [9–11].

To achieve a good functionality of the network, there is a need for knowledge on the ecological conservation and management issues relevant to the Natura 2000 (e.g. status of species and habitats, ways of managing the sites) [12]. The human-induced global water crisis not only endangers human societies but also affects freshwater biodiversity [13]. Anthropogenic global climate change is not only posing challenges to humans but is also perceived as one of the most serious threats to the planet's biodiversity [14].

A relatively large part of the ecological research on the Natura 2000 network focuses on a few (or a single) species within one or a few sites (e.g. [6, 15–23]). In spite of the fact that the Natura 2000 network spans across the European continent, the majority of studies were conducted within regions at the subnational level [12]. To improve evidence-based management and conservation, more research on Natura 2000 should focus on examining the network's adaptive capacity, its coherence and the links between different Natura 2000 sites, as well as relations to conservation activities outside the network [24]. The scarcity of studies pertaining to larger spatial scales may have negative consequences for the conservation of species and habitats that are dependent on large-scale patterns and processes [25]. There is a need for biodiversity conservation actions to be tailored to biogeographic conditions [26]. This would be consistent with the conservation biogeography framework [27], which has become increasingly prominent in recent years but as yet is underutilised in Natura 2000 research [6]. Moreover, the examination of the entire biogeographical regions in ecological studies would foster more cross-scale cooperation in practical management of the network, a process that is necessary for attaining conservation goals in large-scale initiatives [26, 28]. Effective conservation requires involvement of scientists in implementing research results into practice (e.g. [29]), and inadequate distribution of research focus across Natura 2000 network could be a problem for achieving the expected conservation outcomes [30]. The aim of this work is to present the diversity of freshwater habitats and freshwater-dependent habitats in Poland as well as their condition and participation in the biogeographical regions represented in Poland.

2 Material

The study covered freshwater habitats and freshwater-dependent habitats occurring in all 849 in Special Areas of Conservation in Poland in both biogeographical regions: alpine and continental. The overall conservation value of each Natura 2000 site for a habitat includes an assessment of the degree of conservation of the structure and functions, as well as their possibilities for restoration. Data was taken into account in the analyses of Special Areas of Conservation (SACs) for the Natura 2000 network: Standard Data Forms [31], management plans [32], reporting and monitoring by EU Poland SACs [33, 34] and reports and publications by the European Topic Centre on Biological Diversity [35] and Biodiversity Information System for Europe [36].

The types of habitat surveyed include:

1. Water courses and waterbodies
2. Wetlands and riparian vegetation (meadows, forest)
3. Streams and bogs:

   (a) 3110 Oligotrophic waters (*Littorelletalia uniflorae*)
   (b) 3130 Oligo- to mesotrophic waters (*Littorelletea Isoëto-Nanojuncetea*)
   (c) 3140 Hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp.
   (d) 3150 Natural eutrophic lakes with *Magnopotamion*- or *Hydrocharition*-type vegetation
   (e) 3160 Natural dystrophic lakes and ponds
   (f) 3220 Alpine rivers and the herbaceous vegetation along their banks
   (g) 3240 Alpine rivers and their ligneous vegetation with *Salix elaeagnos*
   (h) 3260 Water courses of plain to montane levels with the *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation
   (i) 3270 Rivers with muddy banks with *Chenopodion rubri* pp and *Bidention* pp vegetation
   (j) 6410 *Molinia* meadows on calcareous, peaty or clavey silt-laden soils (*Molinion caeruleae*)
   (k) 6430 Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels
   (l) 6440 Alluvial meadows of river valleys of the *Cnidion dubii*

(m) 7110* Active raised bogs
(n) 7120 – Degraded raised bogs capable of natural regeneration
(o) 7140 Transition mires and quaking bogs
(p) 7150 Depressions on peat substrates of the *Rhynchosporion*
(q) 7210* Calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae*
(r) 7220* Petrifying springs with tufa formation (*Cratoneurion*)
(s) 7230* Alkaline fens (mountain and lowland alkaline fens of spring fen, sedge fen and sedge-moss fen characteristics)
(t) 91D0* Bog woodland
(u) 91E0* Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion*, *Alnion incanae*, *Salicion albae*)
(v) 91F0 Riparian mixed forest of *Quercus robur* and *Ulmus laevis*

Researched habitats covered 7 of 11 Europe regional biogeographic regions (Table 1).

3 Characteristic of Freshwater Habitats and Freshwater-Dependent Habitats in Poland

Overall assessment survey of habitats in biogeographical regions in Europe, occurrence of freshwater habitats and freshwater-dependent habitats in SACs in Poland and conservation status of habitat types – freshwater habitats and freshwater-dependent habitats – in SACs in Poland are presented in Tables 1, 2 and 3, respectively.

Among the pressures and threats of the identified threats to the protected habitats studied in the Natura 2000 areas in Poland [31–34], the most significant threats are human-induced changes in hydraulic conditions [landfill, land reclamation and drying out, general; removal of sediments (mud); canalisation and water deviation; flooding modifications; modification of hydrographic functioning (modifying structures of inland water courses; modification of standing waterbodies; reservoirs; small hydropower projects, weirs); water abstractions from surface waters; water abstractions from groundwater; raising the groundwater table/artificial recharge of groundwater; management of aquatic and bank vegetation for drainage purposes; siltation rate changes, dumping, depositing of dredged deposits; dykes, embankments, artificial beaches, general; abandonment of management of waterbodies; altered water quality due to anthropogenic changes in salinity], pollution [pollution to surface waters (limnic, terrestrial); pollution to groundwater (point sources and diffuse sources); air pollution, airborne pollutants; soil pollution and solid waste (excluding discharges); excess energy], invasive, other problematic species and genes (invasive non-native species; problematic native species) and climate change [changes in abiotic conditions (temperature (e.g. rise of temperature and extremes); droughts and less precipitations; flooding and rising precipitations; pH changes), changes in

Table 1 Overall assessment survey of habitats in biogeographical regions in Europe

Habitats	Biogeographical regions[a]						
	ALP	ATL	BOR	CON	MAC	MED	PAN
	Overall assessment[b]						
Water courses and waterbodies							
3110	FV	U2	U1	U1		XX	
3130	FV	U2	U1	U2	U1	U1	U1
3140	U1	U2	U1	U2		U1	XX
3150	XX	U2	U1	U2	XX	XX	U1
3160	U1	U2	U1	U2	U1	XX	U2
3220	U1	XX	FV	U2	FV	XX	
3230	U2			U2		U2	
3240	U1	XX		U2		XX	
3260	U1	U2	U2	U1		XX	U2
3270	U2	U2	XX	U2		U2	U1
Riparian meadows							
6410	U2	U2	U2	U2		XX	U2
6430	U1	U2	U1	U1		XX	U2
6440		U2		U2			U2
Bogs and bog woodland							
7110*[c]	U2	U2	U1	U2	U2	U2	U1
7120	U2	U2	U1	U2	FV		
7140	U2	U2	U1	U2	U1	U2	U2
7150	U2	U2	U1	U2		XX	XX
7210*	U1	U1	U2	U1		U1	U1
7220*	U1	U2	XX	U2	XX	XX	U1
7230	U1	U2	U1	U2		U2	U2
91D0*	U1	U2	U1	U2	U1		FV
Riparian forests							
91E0*	U2	U2	U2	U2		U2	U2
91F0	U2	U2	U1	U2		U2	U2

[a]Biogeographical regions: ALP, Alpine; ATL, Atlantic; BOR, Boreal; CON, Continental; MAC, Macaronesia; MED, Mediterranean; PAN, Pannonian
[b]Overall assessment: FV, favourable; U1, unfavourable-inadequate; U2, unfavourable-bad
[c]*, Priority feature

biotic conditions habitat shifting and alteration; desynchronisation of processes; decline or extinction of species; migration of species (natural newcomers)].

The waterbodies, to a lesser extent water courses, in Poland [31–34] are threatened mainly by agriculture [cultivation; modification of cultivation practices; grazing; livestock farming and animal breeding (without grazing); use of biocides, hormones and chemicals; fertilisation; irrigation; restructuring agricultural land holding].

The riparian meadows in Poland [31–34] are threatened mainly by agricultural activities, in addition to the already mentioned pressures: mowing/cutting of

Table 2 Occurrence of freshwater habitats and freshwater-dependent habitats in SACs in Poland

Habitats	Number of special areas of conservation in Poland			Area covered by habitat type in the biogeographic region (km^2)		Percentage of surface on the area of the habitat per MS (%)	
	Biogeographical regions[a]						
	CON	ALP	CON/ALP	CON	ALP	CON	ALP
Water courses and waterbodies							
3110	33		33	18.8		38.2	
3130	45	1	45	6		3.3	
3140	82		82	nd[b]		nd	
3150	273	3	273	4,400	0.5	70.1	0.2
3160	125	1	126	1	0.01	0.9	0
3220	17	20	27	0.5	20	0.4	0.2
3230	5	8	8		2.5		1
3240	8	13	14		5		2.4
3260	104		104	nd	–	nd	–
3270	67	1	67	nd	–	nd	–
Riparian meadows							
6410	254	3	255	nd	4	nd	1.9
6430	204	22	217	30	2	4.5	0.4
6440	28		28	31.1		36.2	
Bogs and bog woodland							
7110*[c]	131	7	137	15	4.5	5	5.5
7120	86	5	90	50	2	20.5	7.3
7140	295	10	303	100	3.2	29.1	0.1
7150	73		73				
7210*	53		53	12		16.9	
7220*	30	4	33	nd	nd	nd	nd
7230	180	18	192	nd	1	nd	0.2
91D0*	466	31	478	2,200	100	57.3	4.5
Riparian forests							
91E0*	246	10	253	780	23	58.7	1.1
91F0	164	2	166	150		18	

[a]Biogeographical regions: ALP, Alpine; CON, Continental
[b]nd, no data available
[c]*, Priority feature

grassland (intensive mowing or intensification, nonintensive mowing, abandonment/lack of mowing) and forestry (forest planting on open ground).

The bogs and bog woodland and riparian forests in Poland [31–34] are threatened mainly by sylviculture and forestry [forest and plantation management and use; forest exploitation without replanting or natural regrowth; use of biocides, hormones and chemicals (forestry); use of fertilisers (forestry); grazing in forests/woodland].

Table 3 Conservation status of habitat types: freshwater habitats and freshwater-dependent habitats in SACs in Poland

Habitat	Region[a]	Period	Overall assessment[b]			
Water courses and waterbodies						
3110	CON	2009–2017	FV (20.0%)	U1 (53.33%)	U2 (26.67%)	
3130	CON	2013–2017	FV (25.0%)	U1 (18.75%)	U2 (43.75%)	XX (12.5%)
3140	CON	2013–2017	FV (16.67%)	U1 (25.0%)	U2 (33.33%)	XX (25.0%)
3150	CON	2009–2017	FV (29.63%)	U1 (40.74%)	U2 (29.63%)	
	ALP	2010				XX (100%)
3160	CON	2011–2016	FV (64.29%)	U1 (14.29%)	U2 (14.29%)	XX (7.14%)
	ALP	2009–2016	FV (50.0%)		U2 (50.0%)	
3220	CON	2010–2016		U1 (50.0%)	U2 (50.0%)	
	ALP	2010–2016	FV (50.00%)	U1 (16.67%)	U2 (16.67%)	XX (16.67%)
3230	ALP	2009–2016	FV (33.33%)	U1 (66.67%)		
3240	ALP	2010–2016	FV (33.33%)	U1 (5.56%)	U2 (61.11%)	
3260	CON	2011–2016	FV (50.00%)	U1 (33.33%)	U2 (16.67%)	
3270	CON	2013–2016		U1 (90.91%)	U2 (9.09%)	
Riparian meadows						
6410	CON	2010–2017	FV (8.82%)	U1 (58.82%)	U2 (32.35%)	
	ALP	2010–2017			U2 (100%)	
5430	CON	2007–2014	FV (4.17%)	U1 (62.5%)	U2 (33.33%)	
	ALP	2011–2017	FV (28.57%)	U1 (71.43%)		
6440	CON	2009–2016	FV (5.88%)	U1 (41.18%)	U2 (52.94%)	
Bogs and bog woodland						
7110*[c]	CON	2007–2014	FV (20.0%)	U1 (60.0%)	U2 (20.0%)	
	ALP	2007–2014		U1 (50.0%)	U2 (50.0%)	
7120	CON	2007–2016	FV (11.76%)	U1 (35.29%)	U2 (47.06%)	XX (5.88%)
	ALP	2013–2016		U1 (50.0%)	U2 (50.0%)	
7140	CON	2010–2017	FV (19.44%)	U1 (50.0%)	U2 (25.0%)	XX (5.56%)
	ALP	2010–2016	FV (33.33%)	U1 (50.0%)	U2 (16.67%)	
7150	CON	2010–2016	FV (10.53%)	U1 (21.05%)	U2 (52.63%)	XX (15.79%)
7210*	CON	2007–2014	FV (17.65%)	U1 (32.29%)	U2 (41.18%)	XX (5.88%)
7220*	CON	2007–2017	FV (12.50%)	U1 (37.50%)	U2 (50.0%)	
	ALP	2007–2014		U1 (50.0%)		XX (50.0%)
7230*	CON	2009–2017	FV (9.38%)	U1 (37.50%)	U2 (40.63%)	XX (12.50%)
	ALP	2009–2017		U1 (85.71%)	U2 (14.29%)	
91D0*	CON	2007–2014	FV (8.82%)	U1 (76.47%)	U2 (14.71%)	
	ALP	2007–2014		U1 (100%)		
Riparian forests						
91E0*	CON	2007–2014	FV (3.07%)	U1 (55.56%)	U2 (40.74%)	
	ALP	2007–2014	FV (14.29%)	U1 (57.14%)	U2 (28.57%)	
91F0	CON	2009–2017		U1 (41.94%)	U2 (58.06%)	

[a]Biogeographical regions: ALP, Alpine; CON, Continental
[b]Overall assessment: FV, Favourable; U1, Unfavourable-inadequate; U2, Unfavourable-bad; XX, no data
[c]*, Priority feature

Other threats for freshwater habitats and freshwater-dependent habitats in Poland [31–34] were of local nature:

1. Mining, extraction of materials and energy production (mining and quarrying, exploration and extraction of oil or gas, renewable abiotic energy use)
2. Transportation and service corridors (roads, paths and railroads; utility and service lines; shipping lanes, ports)
3. Urbanisation, residential and commercial development (urbanised areas, human habitation, industrial or commercial areas; discharges, storage of materials)
4. Biological resource use other than agriculture and forestry (freshwater aquaculture, fishing and harvesting aquatic resources, taking/removal of terrestrial plants)
5. Human intrusions and disturbances (outdoor sports and leisure activities, recreational activities, sport and leisure structures)

The consequence of the described threats is reduction or loss of specific habitat features, anthropogenic reduction of habitat connectivity, biocenotic evolution and succession, species composition change (succession), accumulation of organic material, eutrophication (natural and anthropogenic), acidification (natural and anthropogenic), interspecific faunal and floral relations and reduced fecundity/genetic depression.

4 Concluding Remarks

Habitat loss has been, and still is, the greatest threat to biodiversity [37–39]. Estimated unit values of different ecosystem services provided by floodplain wetlands are the highest among terrestrial ecosystems [40]; their protection is important for sustainable development. Biodiversity is declining worldwide and at a much faster rate in aquatic than in most terrestrial systems [41].

Freshwater habitats and freshwater-dependent habitats are important centres of biodiversity and thus assume a central role for nature conservation, which, however, is subject to anthropogenic threats that can lead to biodiversity loss and consequent negative effects on nature conservation. Freshwater habitats and freshwater-dependent habitats are prone to a number of threats which mine their survival: overexploitation, habitat fragmentation, water eutrophication and pollution, human alteration of natural water cycles and alien species invasion which are only some of the main threats and pressures which influence them in Poland.

The Natura 2000 network considerably differs from previous conservation systems in Europe as it goes beyond a direct ban on damaging plants or killing animals and focuses on socially sustainable conservation harmonising the maintenance of species and habitats with economic, social and cultural human needs [42]. Biodiversity is declining worldwide and at a much faster rate in aquatic than in most terrestrial systems [41]. Political action at regional and global levels has sought to curb such trends. In Europe, the implementation of the Birds and Habitats Directives ("the Nature Directives") and the Water Framework Directive [43] aims to protect aquatic biodiversity and environments. More recently, the EU 2020 Biodiversity Strategy [44] aims to implement the Strategic Plan for Biodiversity 2011–2020 and the Aichi Targets [45, 46]. Despite progress, EU directives have been unable to halt and reverse the trend of declining biodiversity of aquatic ecosystems in Europe [47]. As a result, the EU is seeking new approaches to achieve the EU Biodiversity Targets, including improving the coherence between the Nature Directives, the WFD and MSFD [48–50] so as to provide better continuum and biodiversity protection across the freshwater.

The biodiversity of floodplain ecosystems is threatened by numerous factors, such as water regulation, drainage [51, 52], land use change [53, 54], over-exploitation [55], point source and diffuse chemical pollution from the neighbouring agricultural land [54, 56], spread of invasive species [54, 57] and climate change [51, 52]. According to the recent European Red List of Habitats [58], aquatic and wetland habitats are threatened first of all by hydrological system alteration; by climate change, pollution and invasive species; and to a lesser extent by succession, agriculture intensification, forestry, mining, urbanisation, transport and overuse of biological resources [15].

Hydroseral succession is the key natural process in floodplains [15]. High diversity is maintained in the successional series of numerous habitats [59].

Related regulatory issues, riverbed incision, lowering groundwater level, terrestrial transformation and, consequently, increasing hydro-mass succession are the main causes of the loss of biodiversity in flood areas [54, 58, 60].

Under natural conditions, biodiversity hotspots are often found near rivers, lakes, banks and floodplains, as these areas represent habitats with high levels of structural and functional dynamics, primarily induced by downstream flow [59]. Flooding areas are often characterised by a mosaic of habitats differing in humidity, sediment properties, abundance, composition and state of the succession of fauna and flora, productivity and diversity [61]. This habitat mosaic is inhabited by a multiplicity of generalist and specialist species, both terrestrial and aquatic, which often depend on the relative habitat quality and on proximity and functional connectivity of various habitat patches [16, 17].

Most floodplains in Europe are degraded, especially due to reduced hydro-morphological dynamics. This has led, among other things, to a decrease of dynamic habitat types which are an essential part of floodplains [62]. Habitat conditions in the remaining active floodplain areas have often substantially been altered by human impacts, such as river training, river damming, floodplain disconnection, aggradation, pollution by fertilisers and chemical contaminants, introduction of invasive species or intense forestry (e.g. [63–65]).

Ecosystem structure and functions do not recover fully even after 100 years, and recovery of the biological components is slower than that of hydrologic features, what Moreno-Mateos et al. [66] showed on the basis of 621 case studies from all over the world. However, as floodplains and especially wetlands are inherently dynamic systems, they can change without rehabilitation or reconstruction interventions. These spontaneous changes often go unnoticed, although they may interact with the intended results of projects in unexpected ways [18, 67]. There is a lack of studies of habitat and vegetation changes in wetlands where no large-scale hydrological alterations were implemented in the meantime; therefore, effects of landscape-level background processes are manifested [15].

Wetlands and aquatic plants and habitats are relatively tolerant to a wide range of environmental factors, assuming adequate water conditions. Hydroseral succession is accelerated by lowering of groundwater levels, which is the primary factor determining the water level of oxbow lakes [68]. Reduction of groundwater levels, in turn, is a consequence of two other main threats: hydrological modifications and climate change. Climate change, riverbed incision and floodplain aggradation are causes of degradation of floodplain system [19]. Water replenishment implementations can slow down or reset succession, but even in this case, there are other threatening factors.

In order to slow down climate change, reducing greenhouse gas emissions is of high importance. However, it has been recognised in policy responses that adaptation measures are also needed to cope with the already unavoidable impacts of climate change (see [69] for an overview of policy responses). Even assuming the most optimistic projections of the level of mitigation that can be achieved, we are still going to experience a significant degree of climate change [70]. The European Union published a White Paper on climate change adaptation [71] in which a framework is set out to enhance the EU’s resilience to the impacts of climate change. It is recognised that, although ecosystems are threatened by climate change, they are also part of the adaptation solution as they perform important services for the society such as climate regulation, carbon sequestration, flood protection and soil erosion prevention. To safeguard these services for society, resilient ecosystems are needed that are able to cope with impacts of climate change, such as the increased dynamics caused by weather extremes and the shifting of suitable climate zones.

Pollution was represented by excess nutrient load, originated from fertilisers in the neighbouring fields. Pollution and agriculture are ranked among the more dangerous threats by Janssen et al. [58], as of medium importance by Romanowski et al. [16]. Hefting et al. [72] concluded that plant and animal diversity is not threatened as long as the nutrient load does not surpass critical limits. The periodical flooding can slow down the eutrophication process of the floodplains [20]. Excess nutrient supply can also facilitate the spread of invasive plants [57]. Invasive species are considered as one of the main conservation challenges globally [57] specifically for Europe [58]. According to global trends, the spread of invasive species will accelerate and presents unpredictable danger with regard to altering species composition [73].

Succession following restoration of peat ponds has been studied in a landscape context by Beltman et al. [74]. Mitsch et al. [75] give a detailed description of the first 15 years of a created wetland, including vegetation, hydrology, soil development, water quality, nitrogen budget, carbon sequestration and greenhouse gas emission. The multifunctional approach to managing floodplain areas creates new opportunities for restoring degraded flood plains, as evidenced by syntheses for six European countries [76], although evidence of their positive impact on biodiversity is still weak [15].

Riparian forests and meadows belong to the habitats of freshwater-dependent habitats. Europe is dominated by highly modified landscapes consisting of a matrix of small remnants of natural areas of high conservation value (e.g. lowland old-growth forest) and many man-made or seminatural habitats [55]. Lack of sufficient research for any of these habitats in the perspective of Natura 2000 functioning may undermine the conservation effort of the whole network [6]. We need to understand that Natura 2000 is embedded in this matrix of different habitats, managed or protected, and how it is influenced by these areas.

Jantke et al. [77] used species distribution and habitat models for all 70 vertebrate wetland species of the Annexes of the EU Birds Directive and the EU Habitats Directive in 26 EU Member States (EU26). However, they concluded that there was a deficit of 3.02 million hectares of Natura 2000 sites in the EU26 to include at least one viable population of each modelled species.

The importance of biodiversity for sustainable forest management [78] is the primary cause for the inclusion of ca. 23% of all EU forests [79] into the Natura 2000 network in which they represent ca. 50% of habitats. Its inclusion (along with other goods and services) into forest management planning naturally has not been without consequences. Sustainable forest management coincides with the striving to improve the condition of surface and underground waters, increases retention, has a positive effect on the microclimate and thus has a positive effect on the state of biodiversity in the catchment [21–23]. However, intensive forest management generates a number of adverse effects on the state of natural forest habitats. Closely associated with forest dynamics is the concept of forest naturalness, which is defined as the degree of similarity between the present and natural ecosystem conditions [80–82]. Accordingly, an ecosystem in a natural state should be free of any long-lasting human impact. It is worth noting, however, that science argues whether natural forests presently exist at all [83, 84]. Because all forests on Earth are exposed to direct or indirect human impacts, their naturalness continues to decrease. Investigations show that all forest uses and management regimes affect forest biodiversity, either positively or negatively [85–87]. Notwithstanding their long-lasting effects that may blur the differences between the natural and managed ecosystems, the gradient of naturalness, spanning from the most simplified forests such as plantations to the most natural ones as possible under current conditions, is still evident and is clearly linked to the forest biodiversity formation [82]. According to Alluvial forests shown in Table 3, only 8.82% (91E0) of woodland and forest ecosystems have a favourable conservation status, 76.47% (91E0) and 41.94% (91F0) are of unfavourable-inadequate status and 14.71% (91E0) and 58.05% (91F0) are of unfavourable-bad status at Poland level. According to bog woodland (91D0) shown in Table 3, only 8.82% of woodland and forest ecosystems have a favourable conservation status, 76.47% are of unfavourable-inadequate status and 14.71% are of unfavourable-bad status at Poland level. This may be associated with the impacts of resource management, since forestry activities affect the majority of the Natura 2000 sites (59% of all sites) across the EU [88].

Many protected areas in Europe consist of intensively managed private lands, and many conservation actions are related to biodiversity of agricultural areas [89]. Also, in many regions of Europe, agricultural land is a dominant landscape component.

Agricultural crops as a habitat type is clearly underrepresented in Natura 2000 network research across all the biogeographical regions in the EU [6]. This may be partially explained by the fact that agricultural crops are not well represented as Habitat Directive Annex I habitats. Moreover, Habitat Directive Annex II and Bird Directive Annex I contain relatively few taxa associated with these habitats [6]. Past human activities have resulted in a high level of biodiversity of hay meadows. These "secondary" habitats are more diverse than the local potential forest vegetation [90–94]. These meadows are excellent indicators of the social, economic and ecological changes inflicting complex cultural landscapes. In those East-Central-European landscapes, where traditional grassland management has already vanished (such as large areas in Poland, Slovakia, the Czech Republic, Hungary), the intention to preserve biological and cultural diversity faces different challenges [6]. Those elements of traditional management must be identified which can be transplanted into the current conditions, practices and regulatory frameworks, and these elements of traditional management must be given high priority in subsidisation. If we manage to conserve the key elements of traditional grassland management, or replace them with new elements which are nevertheless successfully embedded in local landscape and culture, the preservation of the biological diversity of the cultural landscape could be secured, while the socio-economic system safeguarding local characteristics could be rendered more stable and resilient. Through this move, both key segments of biocultural diversity could be reinforced.

This is particularly important because 69% of the Natura 2000 network sites are affected by agricultural activities [88] and in many regions agriculture is considered a major threat to biodiversity [95, 96]. On one hand, some traditional agricultural activities, such as grazing and mowing, may be essential for maintaining and restoring some habitats and associated species and thus should be incorporated into management plans of Natura 2000 network sites [97–100]. On the other hand, some practices such as intensification of cultivation or use of pesticides may pose threats to biodiversity in the conservation network [47]. There is, however, little research that would support the design of proper management regimes for agricultural crops within Natura 2000 network [6].

Alluvial meadows are an important element of the wetlands for shaping biodiversity. According to *Molinia* meadows (6410) shown in Table 3, only 4.17% (Continental)/28.57% (Alpine) of floodplains ecosystems have a favourable conservation status, 62.5% (Continental)/71.43% (Alpine) are of unfavourable-inadequate status and 33.33% (Continental) are of unfavourable-bad status at Poland level. According to Alluvial meadows of river valleys of the *Cnidion dubii* (6440) shown in Table 3, only 5.88% of floodplains ecosystems have a favourable conservation status, 41.18% are of unfavourable-inadequate status and 52.94% are of unfavourable-bad status at Poland level. As with 6430 Hydrophilous tall herb fringe comm. of plains and montane, this habitat occurs along the margins of watercourses and is dominated by herbs and is not well described in Poland. The main threats and pressures to this habitat are identified in the conservation status assessment as drainage and river channel maintenance works, dykes and embankments which limit the occurrence of the habitat and invasion by alien species especially *Echinocystis lobata* and *Impatiens glandulifera*.

Lack of a scientific basis for conservation and management of agricultural and forest habitats may undermine conservation of its biodiversity. Poor management may undermine conservation of its biodiversity.

Green infrastructure has been increasingly proposed as a multifunctional solution for halting loss and fragmentation of habitats, and thus biodiversity, and for maintaining and restoring ecosystems and their services [44]. Multifunctional approaches are necessary in the management and use of rivers, lakes and floodplains, integrating all disciplines and stakeholders over extended areas, such as the floodplains of a whole catchment or at least significant functional parts of it.

Freshwater habitats and freshwater-dependent habitats require careful sustainable management of their natural resources, taking into account all their functions: natural, landscape, social and economic.

References

1. Rodrigues ASL, Andelman SJ, Bakarr MI, Boitani L, Brooks TM, Cowling RM, Fishpool LDC, da Fonseca GAB, Gaston KJ, Hoffmann M, Long JS, Marquet PA, Pilgrim JD, Pressey RL, Schipper J, Sechrest W, Stuart SN, Underhill LG, Waller RW, Watts MEJ, Yan X (2004) Effectiveness of the global protected area network in representing species diversity. Nature 428:640–643
2. Deguignet M, Juffe-Bignoli D, Harrison J, MacSharry B, Burgess N, Kingston N (2014) 2014 United Nations list of protected areas. UNEP-WCMC, Cambridge, p 44
3. Blicharska M, Orlikowska EH, Roberge JM, Grodzińska-Jurczak M (2016) What can social science tell us about conservation? Review of the Natura 2000 research. Biol Conserv 199:110–122
4. EC (2013) The economic benefits of the Natura 2000 network. Synthesis report. Publications Office of the European Union, Luxembourg, p 76. http://ec.europa.eu/environment/nature/natura2000/financing/docs/ENV-12-018_LR_Final1.pdf. Accessed June 2018
5. Maes J, Paracchini ML, Zulian G, Dunbar MB, Alkemade R (2012) Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biol Conserv 155:1–12
6. Orlikowska EH, Roberge JM, Blicharska M, Mikusiński G (2016) Gaps in ecological research on the world's largest internationally coordinated network of protected areas: a review of Natura 2000. Biol Conserv 200:216–227
7. EU (2011) Biodiversity strategy to 2020. Publications Office of the European Union, Luxembourg
8. Pullin AS, Knight TM, Stone DA, Charman K (2004) Do conservation managers use scientific evidence to support their decision-making? Biol Conserv 119:245–252
9. Maiorano L, Falcucci A, Garton EO, Boitani L (2007) Contribution of the Natura 2000 network to biodiversity conservation in Italy. Conserv Biol 21:1433–1444
10. D'Amen M, Bombi P, Campanaro A, Zapponi L, Bologna MA, Mason F (2013) Protected areas and insect conservation: questioning the effectiveness of Natura 2000 network for saproxylic beetles in Italy. Anim Conserv 16:370–378

11. Kati V, Hovardas T, Dieterich M, Ibisch PL, Mihok B, Selva N (2014) The challenge of implementing the European network of protected areas Natura 2000. Conserv Biol 29:260–270
12. Popescu VD, Rozylowicz L, Niculae IM, Cucu AL, Hartel T (2014) Species, habitats, society: an evaluation of research supporting EU's Natura 2000 network. PLoS One 9:13–22
13. Vörösmarty CJ, McIntyre P, Gessner MO, Dudgeon D, Prusevich A, Green P, Glidden S, Bunn SE, Sullivan CA, Liermann CR (2010) Global threats to human water security and river biodiversity. Nature 467:555–561
14. Malcolm JR, Liu C, Neilson RP, Hansen L, Hannah LEE (2006) Global warming and extinctions of endemic species from biodiversity hotspots. Conserv Biol 20:538–548
15. Ortmann-Ajkai A, Csicsek G, Hollós R, Magyaros V, Wágner L, Lóczy D (2018) Twenty-years' changes of wetland vegetation: effects of floodplain-level threats. Wetlands. https://doi.org/10.1007/s13157-018-1002-0
16. Romanowski J, Matuszkiewiecz J, Franz KW, Kowalska A, Kozłowska A (2005) Evaluation of ecological consequences of development scenarios for the Vistula River. Vistula Econet Development and Implementation (VEDI). Institute of Biology CBE/PAN-IGiPZ/DLG/ALTERRA, Warsaw
17. Scholz M, Mehl D, Schulz-Zunkel C (2012) Okosystemfunktionen von Flussauen. Analyse und Bewertung von Hochwasserretention, Nahrstoffruckhalt, Kohlenstoffvorrat, Treibhausgasemissionen und Habitatfunktion. Naturschutz Biol Vielfalt 124:2
18. Johnson WC, Dixon MD, Scott ML, Raabe L, Larson G, Volke M, Werner B (2012) Forty years of vegetation change on the Missouri River floodplain. Bioscience 62:123–135
19. Pataki B, Zsuffa I, Hunyady A (2013) Vulnerability assessment for supporting the revitalization of river floodplains. Environ Sci 34:69–78
20. Peršić V, Horvatić J, Has-Schön E, Bogut I (2009) Changes in N and P limitation induced by water level fluctuations in Nature Park Kopacki Rit (Croatia): nutrient enrichment bioassay. Aquat Ecol 43:27–36
21. Grzybowski M (2013) Factors affecting the pattern of macrophyte distribution in natural lakes. Fresenius Environ Bull 22(11):3199–3209
22. Grzybowski M (2014a) Determinants of the diversity of macrophytes in natural lakes affected by land use in the catchment, water chemistry and morphometry lakes. J Elem 19(2):401–422
23. Grzybowski M (2014b) Natural dimictic and polymictic lakes: similarities and differences in relationships among chlorophyll, nutrients, Secchi depth, and aquatic macrophytes. J Freshw Ecol 29(1):53–69
24. Davis M, Naumann S, McFarland K, Graf A, Evans D (2014) Literature review: the ecological effectiveness of the Natura 2000 network. ETC/BD report to the EEA, p 30
25. Angelstam P, Roberge J-M, Lõhmus A, Bergmanis M, Brazaitis G, Dönz-Breuss M, Edenius L, Kosiński Z, Kurlavičius P, Lārmanis V, Lūkins V, Mikusiński G, Račinskis E, Strazds M, Tryjanowski P (2004) Habitat modelling as a tool for landscape-scale conservation – a review of parameters for focal forest birds. Ecol Bull 51:427–453
26. Gustafsson L, Felton A, Felton AM, Brunet J, Caruso A, Hjältén J, Lindbladh M, Ranius T, Roberge J-M, Weslien J (2015) Natural versus national boundaries: the importance of considering biogeographical patterns in forest conservation policy. Conserv Lett 8:50–57
27. Whittaker RJ, Araújo MB, Jepson P, Ladle RJ, Watson JEM, Willis KJ (2005) Conservation biogeography: assessment and prospect. Divers Distrib 11:3–23
28. Guerrero AM, Mcallister RR, Wilson KA (2015) Achieving cross-scale collaboration for large scale conservation initiatives. Conserv Lett 8:107–117
29. Arlettaz R, Schaub M, Fournier J, Reichlin TS, Sierro A, Watson JEM, Braunisch V (2010) From publications to public actions: when conservation biologists bridge the gap between research and implementation. Bioscience 60:835–842
30. Hermoso V, Clavero M, Villero D, Brotons L (2016) EU's conservation efforts need more strategic investment to meet continental commitments. Conserv Lett 10:231–237

31. SDF Standard data form Natura 2000 (2018.) https://danepubliczne.gov.pl/dataset/baza-danych-obszarow-natura-2000/. Accessed June 2018
32. GDEP General Directorate for Environmental Protection in Poland (2018) Information and communication platform. http://pzo.gdos.gov.pl/. Accessed June 2018
33. GIEP General Inspectorate for Environmental Protection in Poland (2018) Monitoring of species and natural habitats. State Environmental Monitoring. http://siedliska.gios.gov.pl/pl/projekt-raportow-do-ke/projekt-raportow. Accessed June 2018
34. INC Institute of Nature Conservation of the Polish Academy of Sciences (2018) Monitoring of species and natural habitats with special regard to special areas of Natura 2000 habitat protection. http://www.iop.krakow.pl/cn2000/monitoring/ZestawienieWynikow.aspx. Accessed June 2018
35. Eionet (2018) European Topic Centre on Biological Diversity. https://bd.eionet.europa.eu/. Accessed June 2018
36. Biodiversity Information System for Europe (BISE) (2018.) https://biodiversity.europa.eu/info. Accessed June 2018
37. Brooks TM, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Rylands AB, Konstant WR, Flick P, Pilgrim J (2002) Habitat loss and extinction in the hotspots of biodiversity. Conserv Biol 16:909–923
38. Hanski I (2011) Habitat loss, the dynamics of biodiversity, and a perspective on conservation. Ambio 40:248–255
39. Groom MJ, Meffe GK, Carroll CR (2006) Principles of conservation biology. Sinauer, Sunderland
40. Costanza R, Sutton P, van der Ploeg S, Anderson SJ, Kubiszewski I, Farber S, Turner EK (2014) Changes in the global value of ecosystem services. Glob Environ Chang 26:152–158
41. Vaughn C (2010) Biodiversity losses and ecosystem function in freshwaters: emerging conclusions and research directions. Bioscience 60:25–35
42. Grodzińska-Jurczak M (2008) Rethinking of nature conservation policy in Poland: the need of human dimensions approach. Hum Dimens Wildl 13:380–381
43. EC (2000) Council Directive 2000/60/EC of 23 October 2000 establishing a framework for community action in the field of water policy. Office for Official Publications of the European Communities, Luxembourg. https://eur-lex.europa.eu/legal-content/En/TXT/?uri=CELEX:32000L0060. Accessed June 2018
44. EC (2016) Natura 2000. Nature and Biodiversity Newsletter No. 40, July 2016. European Commission, Brussels. http://ec.europa.eu/environment/nature/info/pubs/docs/nat2000newsl/nat40_en.pdf. Accessed June 2018
45. CBD-UNEP (2010) Strategic plan for biodiversity 2011–2020 and the Aichi targets "living in harmony with nature". Secretariat of the Convention on Biological Diversity, Montreal
46. CBD-UNEP (2013) Quick guides to the Aichi biodiversity targets. Secretariat of the Convention on Biological Diversity, Montreal
47. EEA (2015) State of the environment report 2015. European Environment Agency, Copenhagen. http://www.eea.europa.eu/soer. Accessed June 2018
48. CIS (2013) Strengthening consistency, coordination and cooperation between nature, biodiversity, water and marine policy. In: Joint meeting of the Nature, Marine and Water Directors meeting of 4 December 2013, Vilnius. Outcome of the 1st joint directors' meeting (final version)
49. CIS (2013) Informal meeting of water and marine directors of the European Union, candidate and EFTA countries. Vilnius, 4th and 5th of May 2013. Final synthesis
50. CIS (2015) Common implementation strategy for the Water Framework Directive (2000/60/EC) and the Floods Directive (2007/60/EC): Work Programme 2016–2018
51. Tockner K, Stanford JA (2002) Riverine floodplains: present state and future trends. Environ Conserv 29:308–330
52. Čížková H, Květ J, Comín FA, Laiho R, Pokorný J, Pitchart D (2013) Actual state of European wetlands and their possible future in the context of global climate change. Aquat Sci 75:3–26

53. Zorilla-Miras P, Palomo I, Gomes-Baggenthun E, Martin-Lopez B, Lomas PL, Montes P (2014) Effects of land-use change on wetland ecosystem services: a case study in the Doñana marshes (SW Spain). Landsc Urban Plan 122:160–174
54. Hein T, Schwarz U, Habersack H, Nichersu J, Preiner S, Willby N, Weigelhofer G (2016) Current status and restoration options for floodplains along the Danube River. Sci Total Environ 543:778–790
55. Harrison PA, Vandewalle M, Sykes MT, Berry PM, Bugter R, de Bello F, Feld CK, Grandin U, Harrington R, Haslett JR, Jongman RHG, Luck GW, da Silva PM, Moora M, Settele J, Sousa JP, Zobel M (2010) Identifying and prioritizing services in European terrestrial and freshwater ecosystems. Biodivers Conserv 19:2791–2821
56. Blackwell MSA, Pilgrim ES (2011) Ecosystem services delivered by small-scale wetlands. Hydrobiol Sci J 56:1467–1484
57. Mölder A, Schneider E (2011) On the beautiful diverse Danube? Danubian floodplain forest vegetation and flora under the influence of river eutrophication. River Res Appl 27:881–894
58. Janssen JAM, Rodwell JS, Garcia Criado M, Gubbay S, Haynes T, Nieto A, Sanders N, Landucci F, Loidi J, Ssymank A, Tahvanainen T, Valderrabano M, Acosta A, Aronsson M, Arts G, Attorre F, Bergmeier E, Bijlsma R-J, Bioret F, Bita-Nicole C, Biurrun I, Calix M, Capelo J, Carni M, Chytry M, Dengler J, Dimopoulos P, Essl F, Gardjfell H, Gigante D, Giusso del Galdo G, Hajek M, Jansen F, Kapfer J, Mickolajczak A, Molina JA, Molnár Z, Paternoster D, Piernik A, Poulin B, Renaux B, Schaminee JHJ, Sumberova K, Toivonen H, Tonteri T, Tsiripidis I, Tzonev R, Valachovic M (2016) European red list of habitats, part 2: terrestrial and freshwater habitats. EU, p 44
59. Ward JV, Stanford JA (1995) Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. Regul Rivers Res Manag 11:105–119
60. Schindler S, O'Neill FH, Biró M, Damm C, Gasso V, Kanka R, van der Sluis T, Krug A, Lauwaars S, Sebesvari Z, Pusch M, Baranovsky B, Ehlert T, Neukirchen B, Martin JR, Euller K, Mauerhofer V, Wrbka T (2016) Multifunctional floodplain management and biodiversity effects: a knowledge synthesis for six European countries. Biodivers Conserv 25:1349–1382
61. Geilen N, Jochems H, Krebs L (2004) Integration of ecological aspects in flood protection strategies: defining an ecological minimum. River Res Appl 20:269–283
62. Klimo E, Hager H, Matic S, Anic I, Kulhavy J (2008) Floodplain forests of the temperate zone of Europe. Lesnicka Prace, Kostelec nad Cernymi lesy
63. Nijland HJ, Cals MJR (eds) (2001) River restoration in Europe: practical approaches. In: Proceedings of the conference. RIZA report nr 2001.023, Wageningen
64. Leyer I (2005) Predicting plant species responses to river regulation: the role of water level fluctuations. J Appl Ecol 42(2):239–250
65. Rinaldi M, Wyzga B, Dufour S, Bertoldi W, Gurnell A (2013) River processes and implications for fluvial ecogeomorphology: a European perspective. In: Shroder J, Butler DR, Hupp CR (eds) Treatise on geomorphology. Ecogeomorphology, vol 12. Academic Press, San Diego, pp 37–52
66. Moreno-Mateos D, Power ME, Comin FA, Yockteng R (2012) Structural and functional loss in restored wetland ecosystems. PLoS Biol 10:1–8
67. Toth L (2010) Unrealized expectations for restoration of a floodplain community. Restor Ecol 18:810–819
68. van Geest GJ, Wolters H, Roozen FCJM, Coops H, Roijackers RMM, Buijse AD, Scheffer M (2005) Water-level fluctuations affect macrophyte richness in floodplain lakes. Hydrobiologia 539:239–248
69. Brooker R, Young JC, Watt AD (2007) Climate change and biodiversity: impacts and policy development challenges – a European case study. Int J Biodivers Sci Manag 3:12–30
70. Pallemearts M, Parker CN, Shukla PR, Van Schaik LG (2005) The Greenland dialogue on climate change: a policy discussion paper

71. EU (2013) Framework for adaptation to climate change, leading to a comprehensive EU adaptation strategy by 2013. https://ec.europa.eu/clima/sites/clima/files/docs/eu_strategy_en.pdf. Accessed June 2018
72. Hefting MM, van den Heuvel RN, Verhoeven JTA (2013) Wetlands in agricultural landscapes for nitrogen attenuation and biodiversity enhancement: opportunities and limitations. Ecol Eng 56:5–13
73. Schnitzler AE, Hale BW, Alsum E (2005) Biodiversity of floodplain forests in Europe and eastern North America: a comparative study of the Rhine and Mississippi Valleys. Biodivers Conserv 14:97–117
74. Beltman B, Omtzigt NQA, Vermaat JE (2011) Turbary restoration meets variable success: does landscape structure force colonization success of wetland plants? Restor Ecol 19:185–193
75. Mitsch WJ, Zhang L, Stefanik KC, Nahlik AM, Anderson CJ, Bernal B, Hernandez M, Song K (2012) Creating wetlands: primary succession, water quality changes, and self-design over 15 years. Bioscience 62:237–250
76. Schindler S, Kropik M, Euller K, Bunting SW, Schulz-Zunkel C, Hermann A, Hainz-Renetzeder C, Kanka R, Mauerhofer V, Gasso V, Krug A, Lauwaars S, Zulka KP, Henle K, Hoffmann M, Biró M, Essl F, Jaquier S, Balázs L, Borics G, Hudin S, Damm C, Pusch M, van der Sluis T, Sebesvári Z, Wrbka T (2013) Floodplain management in temperate regions: is multifunctionality enhancing biodiversity? Environ Evid 2:10
77. Jantke K, Schleupner C, Schneider UA (2011) Gap analysis of European wetland species: priority regions for expanding the Natura 2000 network. Biodivers Conserv 20:581–605
78. Forest Europe (2017) Forest Europe home page. http://foresteurope.org/. Accessed June 2018
79. EC (2017) EU forests and forest related policies. European Commission, DG Environment. http://ec.europa.eu/environment/forests/index_en.htm/. Accessed June 2018
80. Winter S (2012) Forest naturalness assessment as a component of biodiversity monitoring and conservation management. Forestry 85:293–304
81. Winter S, Vrska T, Begehold H (2013) Forest Naturalness as a key to forest biodiversity preservation. In: Kraus D, Krumm F (eds) Integrative approaches as an opportunity for the conservation of forest biodiversity. European Forest Institute, Joensuu, pp 52–63
82. Winter S, Fischer HS, Fischer A (2010) Relative quantitative reference approach for naturalness assessments of forests. For Ecol Manag 259:1624–1632
83. Peterken GF (1996) Natural woodland: ecology and conservation in northern temperate regions. Cambridge University Press, Cambridge
84. Remmert H (2013) Ökologie: Ein Lehrbuch. Springer, Berlin
85. Gibson L, Lee TM, Koh LP, Brook BW, Gardner TA, Barlow J, Carlos A, Peres A, Corey JA, Bradshaw WF, Laurance T, Lovejoy E, Sodhi NJ (2011) Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478:378–381
86. Götmark F (2013) Habitat management alternatives for conservation forests in the temperate zone: review, synthesis, and implications. For Ecol Manag 306:292–307
87. Paillet Y, Bergès L, Hjältén J, Ódor P, Avon C, Bernhardt-Römermann M, Bijlsma RJ, De Bruyn L, Fuhr M, Grandin U, Kanka R, Lundin L, Luque S, Magura T, Matesanz S, Mészáros I, Sebastià MT, Schmidt W, Standovár T, Tóthmérész B, Uotila A, Valladares F, Vellak K, Virtanen R (2010) Biodiversity differences between managed and unmanaged forests: meta-analysis of species richness in Europe. Conserv Biol 24:101–112
88. Tsiafouli MA, Apostolopoulou E, Mazaris AD, Kallimanis AS, Drakou EG, Pantis JD (2013) Human activities in Natura 2000 sites: a highly diversified conservation network. Environ Manag 51:1025–1033
89. Boitani L, Sutherland WJ (2015) Special section: conservation in Europe as a model for emerging conservation issues globally. Conserv Biol 29:975–977
90. Poschlod P, Wallis DeVries MF (2002) The historical and socioeconomic perspective of calcareous grasslands – lessons from the distant and recent past. Biol Conserv 104:361–376

91. Meilleur B (2010) The structure and role of folk ecological knowledge in Les Allues, Savoie (France). In: Johnson LM, Hunn ES (eds) Landscape ethnoecology. Concepts of biotic and physical space. Berghahn, New York, pp 159–174
92. Merunkova K, Preislerova Z, Chytry M (2012) White Carpathian grasslands: can local ecological factors explain their extraordinary species richness? Preslia 84:311–325
93. Wilson JB, Peet RK, Dengler J, Partel M (2012) Plant species richness: the world records. J Veg Sci 23:796–802
94. Babai D, Molnar A, Molnar Z (2014) Traditional ecological knowledge and land use in Gyimes (Eastern Carpathians). MTA Centre for the Humanities and MTA Centre for Ecological Research, Budapest and Vacratot
95. Young J, Watt A, Nowicki P, Alard D, Clitherow J, Henle K, Johnson R, Laczko E, McCracken D, Matouch S, Niemela J, Richards C (2005) Towards sustainable land use: identifying and managing the conflicts between human activities and biodiversity conservation in Europe. Biodivers Conserv 14:1641–1661
96. Henle K, Alard D, Clitherow J, Cobb P, Firbank L, Kull T, McCracken D, Moritz RFA, Niemelä J, Rebane M, Wascher D, Watt A, Young J (2008) Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe – a review. Agric Ecosyst Environ 124:60–71
97. Halada L, Evans D, Romão C, Petersen JE (2011) Which habitats of European importance depend on agricultural practices? Biodivers Conserv 20:2365–2378
98. Papanikolaou AD, Fyllas NM, Mazaris AD, Dimitrakopoulos PG, Kallimanis AS, Pantis JD (2011) Grazing effects on plant functional group diversity in Mediterranean shrublands. Biodivers Conserv 20:2831–2843
99. Faria N, Rabaça JE, Morales MB (2012) The importance of grazing regime in the provision of breeding habitat for grassland birds: the case of the endangered little bustard (Tetrax tetrax). J Nat Conserv 20:211–218
100. Arponen A, Heikkinen RK, Paloniemi R, Pöyry J, Similä J, Kuussaari M (2013) Improving conservation planning for semi-natural grasslands: integrating connectivity into agri-environment schemes. Biol Conserv 160:234–241

Communities of Freshwater Macroinvertebrate and Fish in Mountain Streams and Rivers of the Upper Dunajec Catchment (Western Carpathians) Including Long-Term Human Impact

Andrzej Kownacki, Ewa Szarek-Gwiazda, Maciej Ligaszewski, and Jan Urban

Contents

Abstract The study aimed to determine natural communities of invertebrates and ichthyofauna on the background of physiographic and geological conditions and water chemistry and to evaluate the influence of human pressure on these communities on the basis of the results of long-term studies carried out since the 1960s in the Upper Dunajec River and its Tatra tributaries (West Carpathians). Here, for the Dunajec River, we present a pattern of human impact on freshwater fauna communities in mountain streams and rivers with stony bottoms and rapid currents.

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The Tatra streams (tributaries of the Dunajec River) have mostly retained their natural character. The source of water pollution was mainly from the tourist shelter and tourism centre nearby. Discharge of untreated or partially treated sewage effluent affected invertebrate communities; however, this impact was only observed over a short river section. On the other hand, dam reservoirs completely changed the character of the Dunajec River. In the Czorsztyn Reservoir, communities of macroinvertebrates and fish developed, which are not typical of mountain rivers, replacing previous taxa with those characteristic of slow-flowing waters.

Keywords Dunajec River · Fish · Geology · Human impact · Macroinvertebrates communities · Pollution · Reservoirs · Tatra streams · Water chemistry

1 Introduction

Southern parts of Poland comprise a series of mountain massifs: the Sudetes, the Western Carpathians and parts of the Eastern Carpathians. The northern slopes of these mountains are drained by rivers and streams belonging to the basins of the Vistula River and the Odra River. Within the Western Carpathians, the alpine-type range of the Tatra Mountains, the highest Carpathian mountain range, is situated. The northern slopes of the Tatra Mts. are drained by the Dunajec River and its tributaries which are typical mountain rivers. The bedrock of the largest part of the catchment is formed by the Carpathian flysch, but the highest part of the catchment, situated in the Tatra Mts., is built of crystalline rocks. Out of the Tatra Mts., the Dunajec River crosses the range of the Pieniny Mountains, built of carbonate rocks. Geological diversity affects water chemistry and indirectly invertebrate and fish communities. Streams and rivers in the Dunajec basin have largely retained their natural character. However, in recent years there has been a growing impact of human activity (sewage from towns and villages and the development of tourism infrastructure) on the biocenosis of this river catchment. Along the river, dam reservoirs: Czorsztyn, Sromowce Wyżne, Rożnów and Czchów were constructed.

Studies of fauna of the Upper Dunajec River date back to the nineteenth century. The first reports were on fish [1–3]. In the 1920s, comprehensive ichthyological and hydrobiological studies of the Dunajec catchment were conducted [4]; however, they were interrupted by the outbreak of the Second World War. In the 1960s, Professor Karol Starmach [5] launched an initiative to conduct complex hydrochemical, hydrobiological (algology, macrofauna) and ichthyological studies in the Dunajec catchment and dam reservoirs located in the Dunajec River. Research initiated by Starmach [5] has been continued up to the present [6–7].

The study aimed (1) to determine natural communities of invertebrates and ichthyofauna on the background of physiographic and geological conditions and water chemistry and (2) to evaluate the influence of human pressure on these communities on the basis of the results of long-term studies carried out since the 1960s in the Dunajec River and its Tatra tributaries. For the Dunajec River, we present a pattern of human impact on freshwater fauna communities in mountain streams and rivers with stony bottoms and rapid currents. The study was based on our own results and rich literature.

2 Characteristics of the Dunajec Catchment Area

2.1 General Characteristics of the Catchment

The catchment of the upper and middle sections of the Dunajec River (length ~175 km) and its largest tributary, the Poprad River (165 km), covers about 5,300 km^2 and is situated within the Western Carpathians (on both sides of Polish-Slovak boundary). Springs of both rivers are located in the Tatra Mts., the highest mountain group in the Carpathians ranging 2,000–2,600 m a.s.l. The upper sections of the Dunajec River and tributaries, characterised by high stream gradients, are no longer than 10–12 km. The lower sections of the river and its tributaries flow through the intermediate mountain groups and intra-mountainous depressions. Consequently, the stream gradient for the upper and middle sections of the Dunajec River out of the Tatra Mts. ranges 4.13 per thousand.

Only the uppermost sections of stream valleys are V-shaped in cross-section, whereas most of the river valleys comprise flat-bottoms type that are partly filled with gravel of the lithological composition, similar to the surrounding rocks and rock occurring directly upstream. Nevertheless, in the sections where the Dunajec River crosses the strongest rocks, such as thick-bedded sandstones and limestones, narrow river gaps have developed, e.g. in the Pieniny Mountains, as well as on the Magura Formation outcrops in the southern zone of the Beskidy Mts. (Beskid Sądecki Mountains) [8–9].

Two pairs of dam reservoirs are located along the Dunajec River: Czorsztyn and Sromowce Wyżne as well as Rożnów and Czchów. The upper pair comprises the Czorsztyn and Sromowce Wyżne, which were constructed in 1970–1997.

Although the mountainous area of the Carpathians is predominately forested, the population density is pretty large in stream and river valleys and their vicinities. The deforested areas have been cultivated or used as pasture for several hundred years. Several relatively large urban agglomerations are situated along the upper and middle sections of the Dunajec river or close to it: Zakopane, a touristic centre and resort in the Tatra foothills; Nowy Targ, an administration-industrial centre in the Podhale Basin, Szczawnica and Krościenko resorts situated directly north of the Pieniny Mts.; and Nowy Sącz, the largest administration-industrial centre in this part of the Beskidy Mts., situated in the Sącz Basin (Kotlina Sądecka).

2.2 Geological Characteristics

In geological terms, the Western Carpathians are divided into two mega-regions: Inner and Outer Carpathians separated by the Pieniny Klippen Belt (Fig. 1a) [8, 10]. In Polish part of the area of the Dunajec River catchment, the Inner Carpathians are represented by the Tatra Mountains and Podhale Basin (in Polish: Obniżenie Podhalańskie).

The crystalline core of the Tatra Mts. is formed of granitoids, as well as schists and other metamorphic rocks, which are overlain on the northern side by several tectonic units (nappes), built of Mesozoic sedimentary rocks with predominant limestones and dolomites and subordinate marls, sandstones, quartzites and shales [8, 11]. In carbonate rocks, a mountainous type of karst has developed [12]. The surrounding depressions are formed of Paleogene siliciclastic-clayey rocks [8].

Flowing generally towards the north, the Dunajec River crosses the narrow geological unit of the Pieniny Klippen Belt, composed of various carbonate rocks: limestones, dolomites and marls with clayey and siliciclastic inserts, which are marked in relief by specific the Pieniny Mountains [8, 11]. The Outer Carpathians, situated to the north of the Pieniny Klippen Belt and in physiographic terms called the Beskidy Mts. (with several smaller mountain groups – [9]), are built of folded and faulted Upper Jurassic to Lower Miocene flysch, siliciclastic-clayey rocks, which form several nappes, thrust towards the north (Fig. 1a). These units principally comprise sequences of thick-bedded sandstone and sandstone-conglomerate series, thin-bedded sandstone-shale series and clayey-silty shale series with a thickness ranging from several tens of metres up to several hundred metres. Thick-bedded sandstones and conglomeratic sandstones of the Magura Formation (Magura Sandstones) within the Magura Unit form mountain massifs and ranges in the southern zone of the Outer Carpathians [8, 10].

2.3 Description of Studied Streams and Rivers

Studies were carried out in the Dunajec River, including a section between Nowy Targ and Nowy Sącz, and its Tatra tributaries – streams: Mnichowy Potok, Rybi Potok, Sucha Woda, as well as Białka Tatrzańska River (Table 1, Figs 1b and 2). In Mnichowy Potok five sites (M1–M5) were located in the alpine zone. In Sucha Woda, an alpine zone above and below Czarny Staw lake (SW1–SW2), the dwarf pine zone (SW3–SW4), the montane forest zone (SW5–SW7) and the submontane zone (SW8) were studied. In Rybi Potok stream, a polluted section below the discharge of domestic sewage from Morskie Oko Shelter (RP1–RP6) was studied. In the Białka Tatrzańska River, a section situated on the boundary between the montane forest and submontane zones (B1), as well as the section in submontane zone (B2–B4), was studied. The study section of the Dunajec River between Nowy Targ and Nowy Sącz (D1–D5) is of sub-mountainous character, largely urbanised.

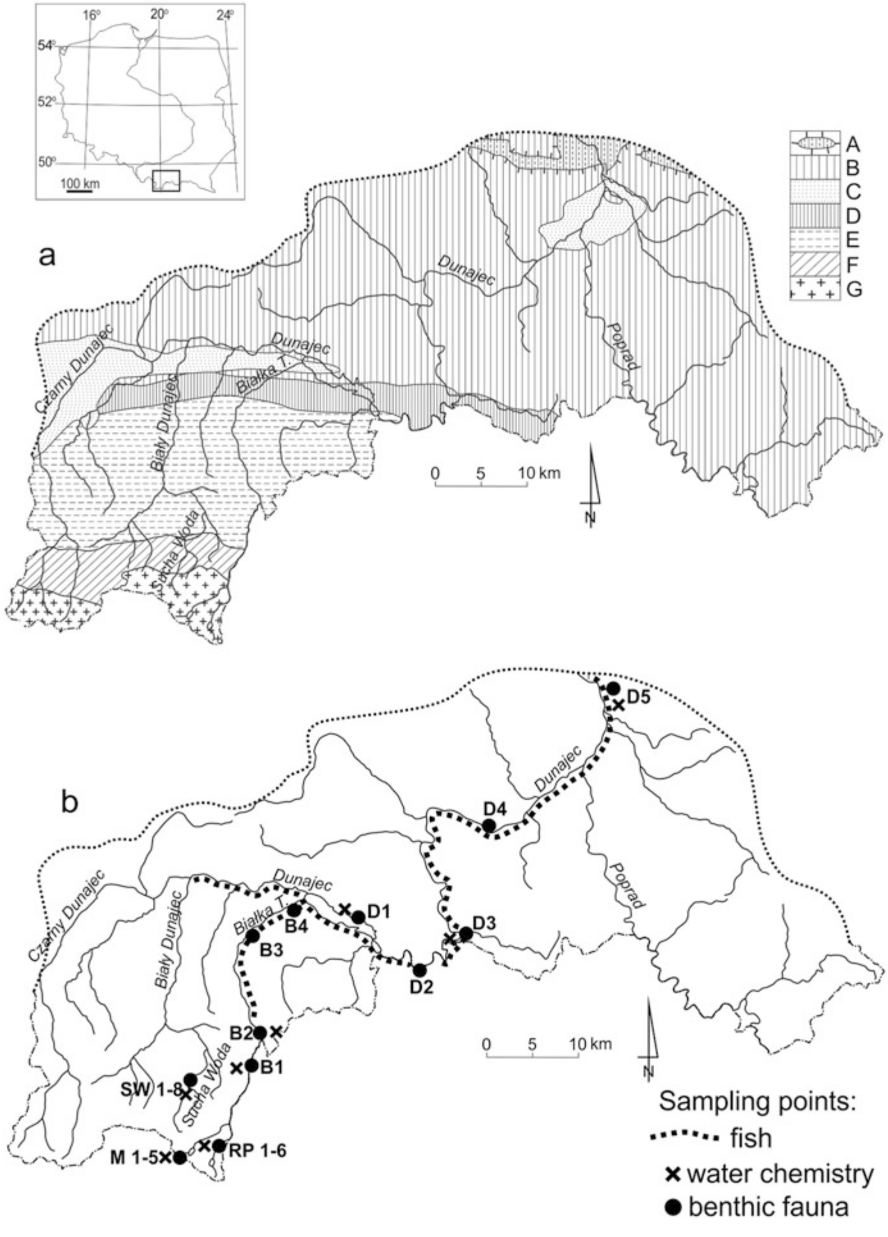


Fig. 1 (**a**) Geology of the Upper Dunajec catchment [A = Outer Carpathians (Beskidy Mts.), Grybów Unit: sandstone, siltstone and claystone flysch (Cretaceous, Palaeogene); with margins of overthrust; B = Outer Carpathians (Beskidy Mts.), Magura Unit: conglomerate, sandstone, siltstone and claystone flysch (Cretaceous, Palaeogene); C = Outer Carpathians (intra-mountainous basins), siltstones and claystones (Neogene); D = Pieniny Clippen Belt (Pieniny, Podhale Basin), limestones, marls, with radiolarites, claystones and siltstones (Jurassic, Cretaceous); E = Inner Carpathians (Podhale Basin), sandstone, siltstone and claystone flysch (Palaeogene); F = Inner Carpathians (Tatra Mts.), sedimentary cover of High-Tatric and Sub-Tatric Nappes: limestones, dolomites, marls with sandstones and claystones (Permian, Triassic, Jurassic, Cretaceous); G = Inner

Table 1 Characteristics of the studied sections of Carpathian streams and rivers

Stream/ rivers	Altitude of springs (m a.s.l.)	Altitude of mouth (m a.s.l.)	Length (km)	Gradient (‰)	Catchment area (km^2)
Mnichowy	2,070	1,900	0.5	400	–
Sucha Woda	1,787	750	15.5	25–105	68
Rybi Potok	1,393	1,100	4.0	10	–
Białka Tatrzańska	1,557	530	40.4	11–67	232
Dunajec	550	283	274	2.9–4.0	in Poland 4,854.1

3 Physico-Chemical Characteristics of Stream and River Waters

Water temperatures of the studied Tatra streams (Rybi Potok, Mnichowy, Sucha Woda) and a mountain river (Białka Tatrzańska) were usually low throughout summer (1.1–6.5°C in the alpine zone, 7–10°C and montane forest zone and 16°C in the submontane zone). They were well-oxygenated (dissolved oxygen usually exceeds 9 mg dm^{-3}, oxygen saturation >90%, frequently >100%) with pH varying from slightly acidic to slightly alkaline (Table 2) [13–17]. Waters belong to the carbonate-calcium type (Table 2). Those flowing through the granite zone were extremely poor in electrolytes (conductivity <10 μS/cm, Ca^{2+} usually <5 mg/L). Water mineralisation (ions SO_4^{2-}, Ca^{2+} and Mg^{2+}) markedly increased in the montane forest zone and submontane zone of the streams (4–6 times in Sucha Woda stream) and rivers as they flow on a substratum built of sedimentary rocks with a high content of carbonate rocks. Water mineralisation was closely related to the geological background, climate and dynamics of water flow [13–14].

In general, stream and river waters were characterized by low contents of nutrients and chemical oxygen demand (COD_{Mn}) (<2 mg O_2/L) values (Table 2) [13–14, 17]. Low nitrate content is associated with the weak washing out of poor mountain soils, while phosphate content is associated with deficiency in the substratum. Elevated contents of nutrients and biochemical oxygen demand (BOD5) and COD_{Mn} values in Rybi Potok stream were a result of human impact (tourist shelter, tourism). Below the sewage effluent outlet from the Morskie Oko tourist shelter, the contents of N–NH_4, PO_4^{3-} and BOD5 values in the stream water increased 6–20 times [15].

Fig. 1 (continued) Carpathians (Tatra Mts.), crystalline core: granitoids, schists, gneisses, with other metamorphic rocks] (according to [8]), (**b**) Locations of the sampling sites in streams and rivers: *M* Mnichowy stream, *RP* Rybi Potok stream, *SW* Sucha Woda stream, *B* Białka Tatrzańska River, *D* Dunajec River

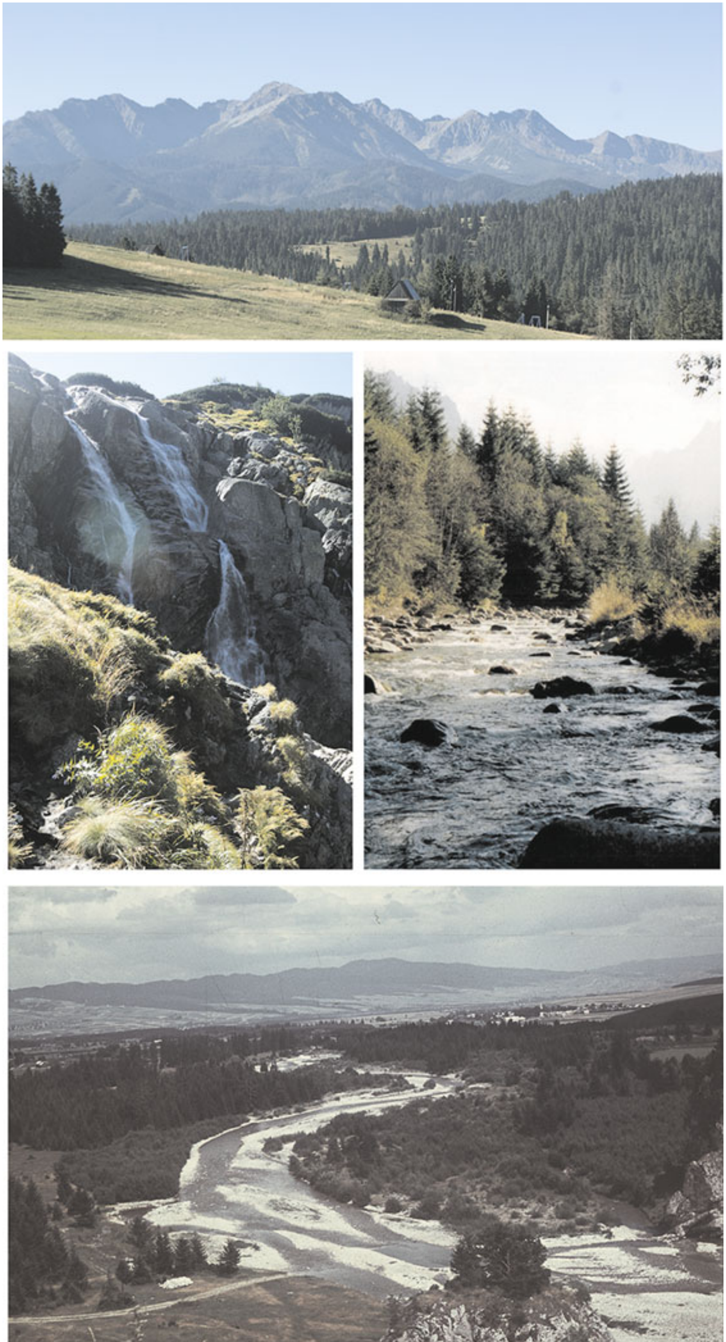


Fig. 2 Photos of Tatra Mts. and type of streams and rivers

Table 2 Water chemistry of the Carpathian streams and river

Parameter	Unit	Streams			River
		Mnichowy	Sucha Woda	Rybi Potok	Białka Tatrzańska
Temperature	°C	1.1–6.5	4.1–8.3	0.8–12.7	2.6–16.5
pH		6.2	6.2–8.2	6.4–7.8	7.1–7.8
Dissolved oxygen	mg/L	10.4	10.9–11.5	6.7–12.3	9.6–12.5
Oxygen saturation	%	–	85–96	64.3–114.5	88–102
Conductivity	μS/cm	–	8.7–62.3	22.4–51.8	–
Chlorides	mg/L	–	0.8–4.5	0.5–2.5	1.4–3.0
Sulphates	mg/L	–	1.5–9.9	1.0–4.32	14.9–20.5
Calcium	mg/L	–	2.5–22.2	4.2–6.4	17.2–32.9
Magnesium	mg/L	–	0.6–4.8	0.5–1.73	3.9–7.8
Potassium	mg/L	–	0.1–0.3	0.05–0.63	0.2–1.1
Sodium	mg/L	–	0.2–0.8	0.08–1.74	0.8–3.0
COD_{Mn}	mg O_2/L	–	0.68–1.92	0.36–2.88	0.64–1.98
$N–NH_4$	mg/L	–	nd–0.08	nd–1.192	nd–0.26
$N–NO_3$	mg/L	–	0.01–0.08	0.21–0.51	0.31–1.75
Orthophosphate	mg/L	–	0.01–0.04	nd–0.27	nd–0.10

Sampling sites were located along the streams and river course. Adapted from [13–16]
nd below detection limit

The Dunajec River was characterised by clean water until 1964 and yielded low BOD5 (<3.5 mg O_2/L) values [18]. We analysed changes in water chemistry of the Dunajec River in years 1977, 1980, 1990, 2000, 2010 and 2015 (Fig. 3) on the basis of data received from the Provincial Inspectorate for Environmental Protection in Krakow. Due to a highly dynamic flow of the river, the suspended matter content in the waters varied over a wide range (from <5 mg/L to 198 mg/L). The waters had temperatures between 10 and 20°C (mean 16°C) during summer, pH 7.2–8.7 and mean annual content of dissolved oxygen usually above 8 mg dm^{-3} (oxygen saturation above 80%).

Conductivity values and contents of major ions (Ca^{2+}, Mg^{2+}, Cl^-) and nutrients in the sub-mountain waters of the Dunajec River (sites B1, B3, B5) exceeded a few times those in the streams of alpine, dwarf pine and montane forest zones (Fig. 3). Additionally, water mineralisation (conductivity, Cl^-, SO_4^{2-}) was slightly increased along the Dunajec River. The contents of nutrients ($N–NH_4$ and PO_4^{3-}) and BOD5 values showed fluctuations along the river (Fig. 3). They were high below towns situated along the river (Nowy Targ, site D1; Nowy Sącz, site D5). Kownacki et al. [19] also indicated an increase in the values of BOD5, COD_{Cr}, COD_{Mn} and contents of $N–NH_4$, N_{org}, $P–PO_4$ below Nowy Sącz (site B5) associated with sewage effluent discharge from the treatment plant. The maximum content of nutrients ($N–NH_4$, PO_4^{3-}) and BOD5 values were found in years 1980 or 1990 and after that their gradual decrease was observed. Dam reservoirs located in the Dunajec River have influenced the river water chemistry. Downstream of the reservoirs Czorsztyn and

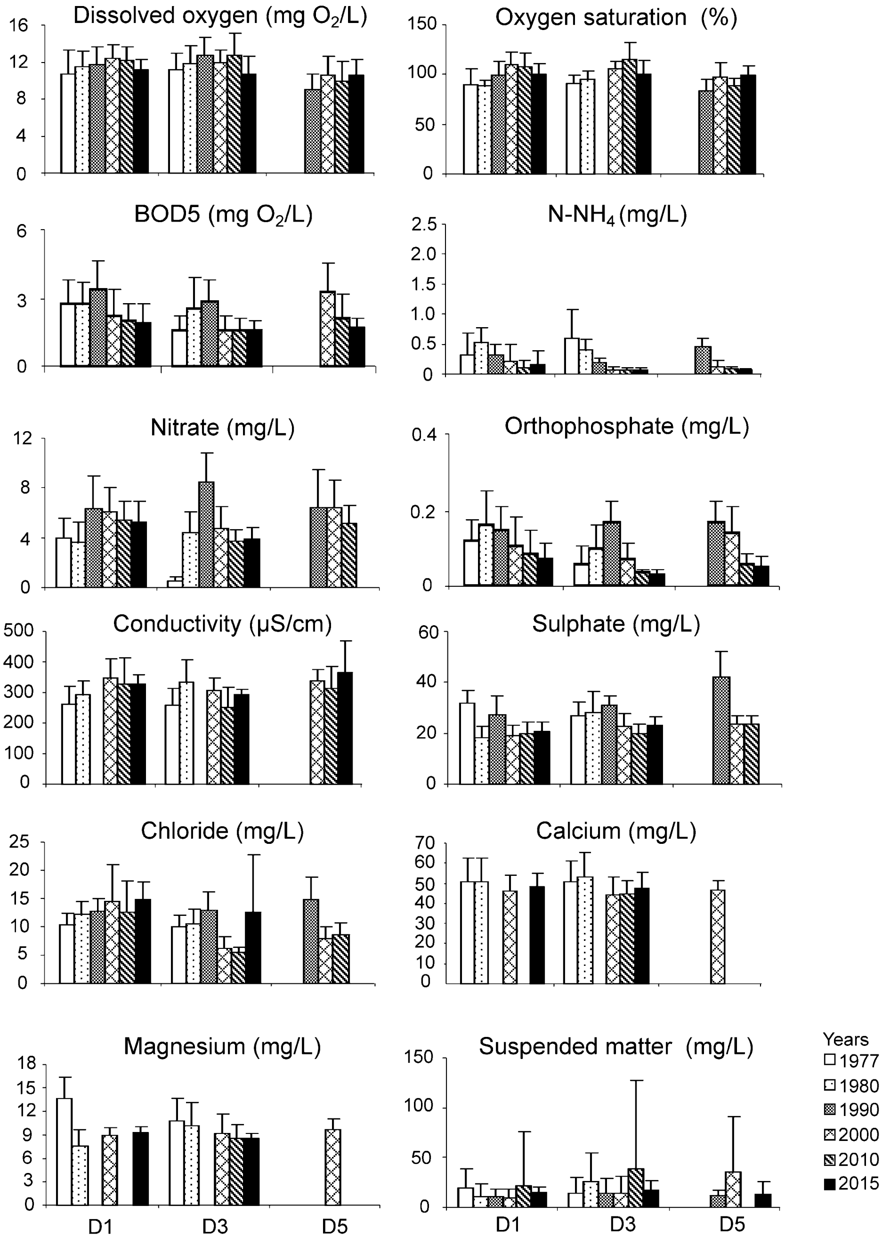


Fig. 3 Changes in the values of physico-chemical parameters (mean ± SD) in the water of the Dunajec River in the period 1977–2015. Adopted from Provincial Inspectorate for Environmental Protection in Krakow

Sromowce Wyżne, a decrease in content of suspended solids, Cl^{-}, PO_4^{3-}, and SO_4^{2-} and changes in water temperatures (decreased in spring and summer but increased in autumn and winter) were observed [20].

4 Macroinvertebrate Communities in Streams and Rivers

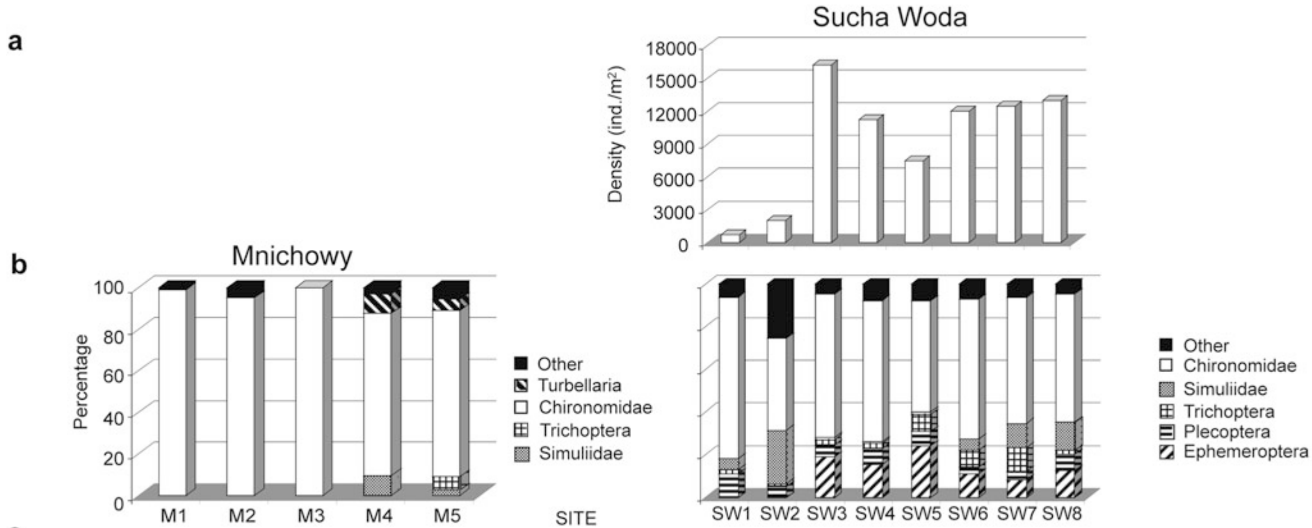
4.1 Macroinvertebrate of the Tatra Mts. Streams

Insect larvae were the most abundant in the fauna of the high Tatra streams. The most numerous group was Chironomidae (Diptera) represented by the largest number of taxa and specimens. Simuliidae (Diptera) was also important, especially in the alpine and submontane zones. In rivers of the montane forest and submontane zones, most numerous were Ephemeroptera, Plecoptera and Trichoptera. The other groups of insects (Colembola, Coleoptera, Hemiptera and Diptera: Blephariceridae, Empididae, Thaumaleidae, Ceratopogonidae, Tipulidae, Limonidae, Psychodidae) were rare. The percentage of Oligochaeta and Turbellaria usually totalled less than 3%. Single specimens of Hirudinea and Mollusca played a slight role in the whole bottom fauna. Gammaridae was absent from the study streams, despite being abundant in other Carpathian streams.

Mnichowy Stream In the upper part (2,000–2,070 m a.s.l.), below the firn glacieret, *Diamesa steinboecki*, *D. nowickiana* and *D. latitarsis* were predominant (Fig. 4b, c). It is a community typical of glacial streams. In the lower section (1,900 m a.s.l.) of this stream, apart from species of the genus *Diamesa*, *Parorthocladius nudipennis* was prevailing. Some specimens of Diptera from the families of Simuliidae (*Prosimulium* sp.), Blephariceridae, Tipulidae and Plecoptera (*Leuctra* sp.), Trichoptera (*Drusus monticola*) and Turbellaria (*Planaria alpina*) were found. Ephemeroptera was absent [16, 21].

Sucha Woda Stream The fauna of Sucha Woda stream was represented mainly by insect larvae, especially Chironomidae (Diptera) (Fig. 4a, b, c). Above 1,550 m a.s.l., the density was very low and did not exceed 3,000 ind/m^2. There, the *Diamesa latitarsis* group, and, at the outlet of Czarny Staw lake, also Simuliidae, *Prosimulium*, were the predominant taxa. At an altitude of 1,550–1,000 m a.s.l., in the upper forest zone, the density increased rapidly (up to ~12,000–18,000 ind/m^2). *Eukiefferiella minor* and *Parorthocladius nudipennis* (Chironomidae), as well as *Baetis alpinus* and *Rhithrogena loyolaea* (Ephemeroptera), predominated. Below 1,000 m a.s.l. the predominant taxa were larvae of the *Orthocladius (Euorthocladius) rivicola* group and Simuliidae [16, 22–24].

Polluted Stream: Rybi Potok In the 1970s there were comprehensive studies in Rybi Potok stream below the discharge of sewage from the Morskie Oko Shelter [25]. At site RP2 (0–10 m below sewage effluent discharge), faunal density was slightly increased in relation to a control site (RP1) (Fig. 5a). A group of



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Taxa	Site (zone)				
	M1	M2	M3	M4	M5
	A	A	A	A	A
Diamesa steinboecki	**25**	**15**	**27**	**29**	+
Diamesa latitarsis	**11**	**27**	**18**	**31**	**22**
Diamesa nowickiana	**23**	**21**	**32**	3.2	0
Parorthocladius nudipennis	0	0	0	+	**31**
Diamesa sp.	**17**	4.5	5.2	6.2	4.7
Pseudokiefferiella parva	9.7	**19**	**14**	1.6	1.0

Taxa	Site (zone)							
	SW1	SW2	SW3	SW4	SW5	SW6	SW7	SW8
	A	A	DP	DP	MF	MF	MF	S
Diamesa latitarsis	**48**	**12**	+	+	1.5	4.5	+	+
Protonemura sp.	**10**	+	3.0	3.1	1.4	1.2	+	1.0
Prosimulium sp.	8.7	**27**	+	1.0	1.8	5.6	**12**	**15**
Eukiefferiella minor	1.5	4.8	**27**	**10**	1.3	3.6	4.5	3.2
Diamesa spp. (juv.)	1.4	+	+	+	3.8	3.2	4.0	2.9
Tvetenia bavarica gr.	+	**15**	3.8	8.9	1.7	1.8	3.1	4.0
Nais variabilis	+	**14**	+	+	+	2.0	0	+
Baetis alpinus gr.	0	+	**17**	9.5	13	**10**	3.4	3.1
Parorthocladius nudipennis	3.1	+	**13**	**12**	**30**	**16**	2.0	1.4
Orthocladius (*Euorthocladius*) gr. *rivicola*	+	+	+	1.1	2.2	9.5	**27**	**31**
Eukiefferiella cyanea	0	0	0	0	0	+	+	+

Fig. 4 Macroinvertebrate of the Tatra streams: Mnichowy stream and Sucha Woda stream. (**a**) density, (**b**) percentage share, (**c**) dominant taxa. *A* alpine zone, *DP* dwarf pine zone, *MF* montane forest zone, *S* submontane zone. Adapted from [16, 22–24]

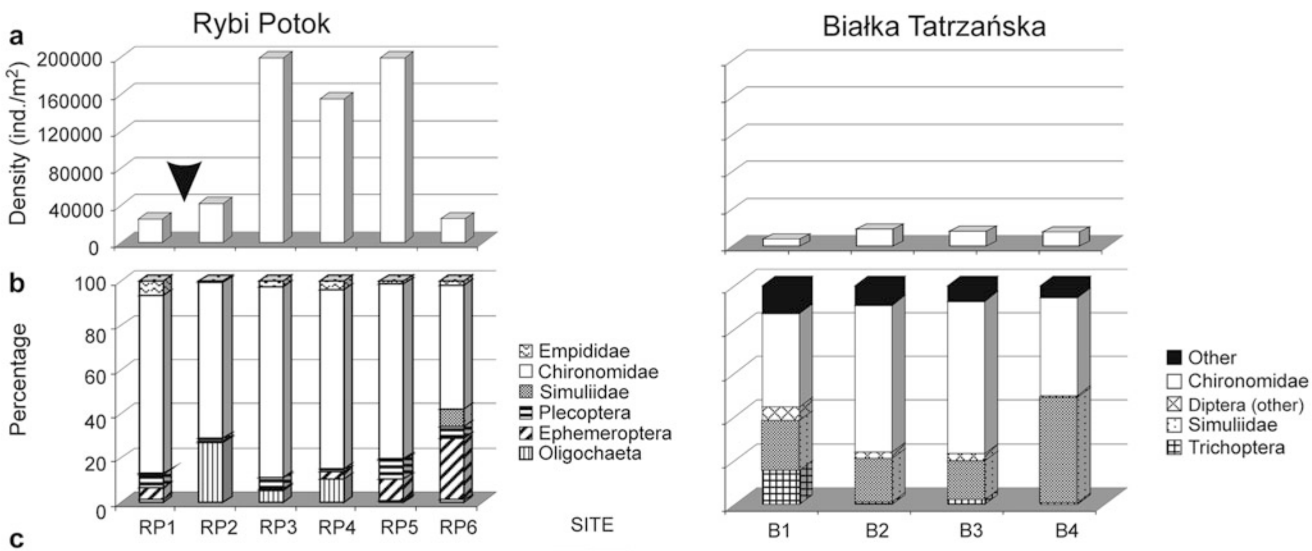
Chironomidae, especially species of genera *Polypedilum* and *Paratanytarsus*, predominated (Fig. 5b, c). The percentage of Oligochaeta increased to 27%, whereas the share of Ephemeroptera decreased to 0.3%, while Trichoptera was absent. Faunal density at RP3 (15–30 m) rapidly increased, reaching the highest values (229,125 ind./m^2). At a distance further from the sewage effluent discharge, the lower fauna density was observed. There Chironomidae predominated, while the share of Oligochaeta decreased. Downstream, about 3 km from the contamination source (site RP6), a community typical of the forest montane zone had developed. Density was lower (25,950 ind./m^2). Larvae of *Baetis alpinus,* an indicator species for this zone, were predominant (25%), and Plecoptera and Trichoptera were more numerous. Yet, the share of Chironomidae (53%) and Oligochaeta (0.8%) was reduced [25].

4.2 Macroinvertebrates of the Mountain Rivers

Białka Tatrzańska River Studies were carried out in the middle and lower course of the river [22–23, 26]. Faunal density at sites B2, B3 and B4 was less than 18,000 ind/m^2 (Fig. 5a). It was only at site B1, on the boundary between the forest montane zone and submontane zone, that density was lower (6,000 ind./m^2). The most dominant group at all sites was Chironomidae (40–60%). Simuliidae was also an important group (20–48%). Apart from Trichoptera at the site B1 (15%), the other faunal groups, for example, Ephemeroptera and Plecoptera, did not exceed a few percent (Fig. 5b). Indicator taxa for this river were Chironomidae larvae of genera *Cricotopus* and *Orthocladius* and *Orthocladius (Euorthocladius)* gr. *rivicola* and Simuliidae (Fig. 5c).

Dunajec River Until 1964, the Dunajec River section between the towns of Nowy Targ and Nowy Sącz was of class I quality (three-class water quality classification of rivers was in force). In 1963, the first study on the Dunajec river bottom fauna was conducted [27]. Faunal density on the river current and stone bottom was low (from approx. 6,600 ind./m^2 at the site D3 to 4,000 ind./m^2 at the site D5). The lowest values (3,300 ind./m^2) were found below Nowy Sącz (D5). Higher density was observed in the silt pools, especially below Nowy Sącz (26,700 ind./m^2) (Fig. 6a). In macroinvertebrate communities at the sites D1, D2, D3 and D4, Chironomidae (35–45%), Trichoptera (12–40%), Ephemeroptera (10–25%) and, at the site D3, Simuliidae (32%) were predominant. The share of Oligochaeta at these sites averaged less than 1%. At the site below Nowy Sącz, the faunal composition changed. Chironomidae (75%) were the most abundant, the share of Oligochaeta rose to 6%, while the share of Trichoptera was reduced to 3% and Ephemeroptera to 8% (Fig. 6b). It clearly indicated that pollution from Nowy Sącz had some influence on the bottom fauna as early as in the 1960s.

Further studies conducted in the 1970s at the sites below Nowy Targ (D1, D2 < D3) showed a rapid deterioration in water quality in the Dunajec River [28]. Below



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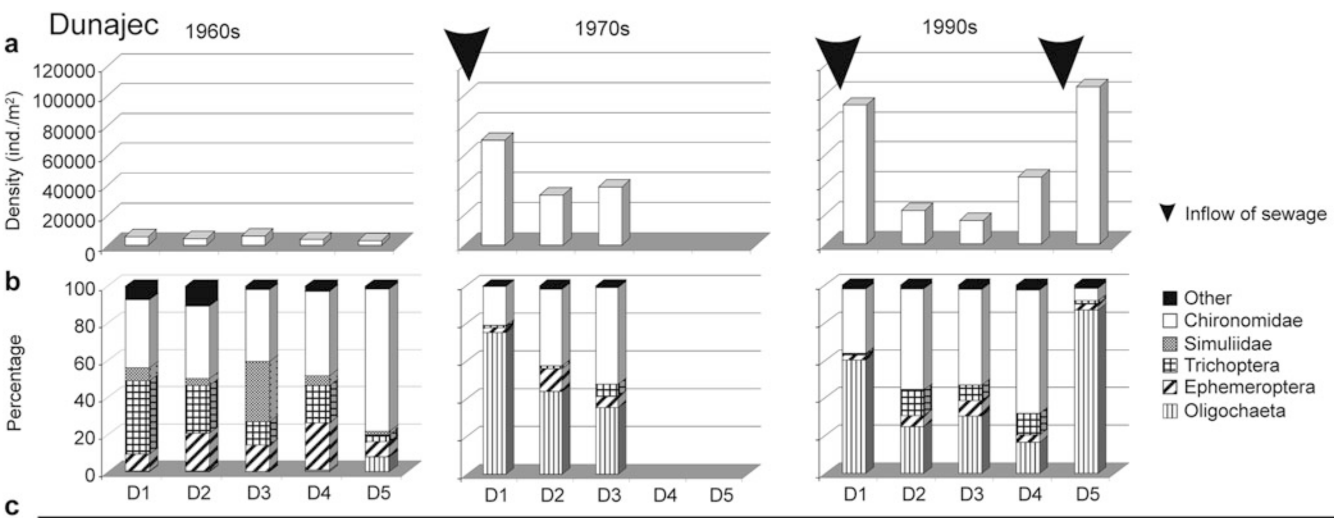
Taxa	RP1	RP2	RP3	RP4	RP5	RP6
	MF	MF	MF	MF	MF	MF
Micropsectra sp.	**19,6**	1,4	2,2	2,2	0,6	0,0
Oligochaeta	0,9	**26,6**	4,8	**10,0**	0,4	0,8
Polypedilum sp.	0,0	**11,6**	1,2	1,6	0,4	0,0
Diamesa spp.	0,6	**11,2**	3,9	1,3	**10,7**	**11,9**
Microcricotopus sp.	1,5	0	3,1	**26,8**	4,2	0,1
Baetis alpinus	0,0	0	0	0,2	0,4	**25,5**
Baetis spp.(other)	5,4	0,4	0,9	3,6	9,4	0,4
Synorthocladius semivirens	7,5	1,8	1,7	1,9	0,6	0,3
Empididae	6,2	0,5	2,2	4,1	1,2	1,4
Paratanytarsus sp.	0,1	9,2	1,0	0,3	0,1	0
Thienemanniella sp.	0,9	0,1	6,4	3,4	7,1	0,4
Tvetenia bavarica	1,1	0	0,7	6,8	1,5	1,3
Eukiefferiella spp.	2,7	0,3	1,1	5,0	1,5	6,5
Simuliidae	0,3	0,5	0,0	0,0	0,9	8,1

Taxa	B1	B2	B3	B4
	S	S	S	S
Cricotopus spp. + *Orthocladius* spp.	**43.5**	**28.9**	**27.0**	9.2
Orthocladius (*Euorthocladius*) gr. *rivicola*	**20.8**	**31.3**	**43.3**	**41.3**
Simuliidae	**22.8**	**19.8**	**17.7**	**48.3**

Fig. 5 Macroinvertebrate of the Białka Tatrzańska catchment: Rybi Potok stream and Białka Tatrzańska River. (**a**) density, (**b**) percentage share, (**c**) dominant taxa. *A* alpine zone, *DP* dwarf pine zone, *MF* montane forest zone, *S* submontane zone. Adapted from [25–26]



Taxa	1960s					1970s			1990s				
	D1	D2	D3	D4	D5	D1	D2	D3	D1	D2	D3	D4	D5
	S	S	S	S	S	S	S	S	S	S	S	S	S
Hydropsyche spp.	**18,7**	**15,6**	7,5	**12,0**	5,1	1,0	1,2	1,2	0,3	1,8	0,6	3,1	0,9
Psychomyia pusilla	**17,6**	5,7	3,8	5,8	1,6	2,7	1,5	3,9	0,0	3,3	2,3	1,0	0,1
Cricotopus spp. + *Orthoclaius* spp.	**16,0**	**12,4**	6,3	**17,6**	**35,4**	6,5	**10,7**	**18,5**	**10,7**	**18,8**	**10,3**	**13,4**	4,7
Simulidae	6,9	3,6	**32,3**	4,5	2,5	0,2	1,4	0,3	1,9	0,8	1,1	0,4	1,1
Baetis spp.	4,9	**10,9**	4,9	**14,4**	5,7	1,4	6,8	3,7	2,8	5,3	7,2	2,5	2,3
Nais elinguis	+	+	+	+	+++	**42,9**	**31,1**	**15,5**	**10,8**	4,1	**11,2**	0,0	**27,1**
Nais bretscheri	+	+	+	+	+	**25,0**	6,4	**11,0**	**10,6**	8,4	7,4	**10,1**	0,4
Orthocladius (*Euorthocladius*) gr. *rivicola*	+++	+++	+++	+++	+++	5,6	**12,6**	4,9	4,2	**12,4**	**11,2**	1,1	0,5
Nais barbata	0	0	0	0	0	0,9	0,0	0,1	**28,6**	2,0	1,7	0,6	**37,9**
Polypedilum spp.	+++	+++	+++	+++	+++	0,4	1,5	1,3	0,3	4,8	7,3	**16,9**	3,7
Eukiefferiella spp.	+++	+++	+++	+++	+++	1,8	4,5	9,1	2,3	3,8	5,1	1,6	0,8
Nais alpina	0	0	0	0	0	3,6	3,1	6,5	0,2	3,4	0,6	1,5	0,0

Fig. 6 Macroinvertebrate of the Dunajec River in the years 1960s, 1970s and 1990s. (**a**) density, (**b**) percentage share, (**c**) dominant taxa. *A* alpine zone, *DP* dwarf pine zone, *MF* montane forest zone, *S* submontane zone. Adapted from [19, 27–29]

Nowy Targ (D1), density rose to 70,000 ind./m^2. Oligochaeta: *Nais elinguis* (43%), *Nais bretscheri* (25%) and Chironomidae (20%) dominated (Fig. 6c). Apart from Ephemera (2.5%) and Trichoptera (1%), the share of the other groups ranged less than 1%. The situation at the subsequent sites (D2, D3) improved, nevertheless, high abundance (above 30,000 ind./m^2), and the share of Oligochaeta (35–45%) indicated a strong human pressure.

Studies carried out in the 1990s [19, 29] showed considerable contamination of the Dunajec River in some sections. At the sites D1 (below Nowy Targ) and D5 (below Nowy Sącz), the density of macroinvertebrates was very high, 93,000 and 105,000 ind./m^2, respectively (Fig. 6a, b, c), and Oligochaeta, *Nais barbata*, *Nais elinguis* and *Nais bretscheri*, predominated. At sites D2 and D3, density was lower, 22,000 and 16,000 ind./m^2, respectively. The predominant group was Chironomidae, approx. 60% at both sites, and mayflies (Ephemeroptera), stoneflies (Plecoptera) and caddisflies (Trichoptera) together accounted for 20% and 15%, respectively, and yet the percentage of Oligochaeta dropped to 20%. This indicated a significant improvement in water quality of the Dunajec River. At site D4, zoobenthos density rose (45,000 ind./m^2), whereas its dominance structure was similar to that at sites D2 and D3.

5 Communities of Fish in Streams and Rivers

The Tatra Streams Ichthyofauna of the Tatra streams was represented mainly by two rheophilic species: *Cottus poecilopus* and *Salmo trutta* m. *fario* [30–31]. These species usually do not exceed an altitude of 1,000 m a.s.l., but *Salmo trutta* m. *fario* was found at 1,395 m a.s.l. (Rybi Potok stream) [32] and *Cottus poecilopus* at 1,077 m a.s.l. (Chochołowski Stream) [31]. The irregular occurrence of *Thymallus thymallus* and *Phoxinus phoxinus* were observed sporadically in the lower sections of some Tatra streams. Sea trout *Salmo trutta* m. *trutta*, which had migrated for spawning from the Baltic Sea to the upper part of Rybi Potok stream reaching Morskie Oko lake (1,372 m a.s.l.) [1], was no longer found in the Tatra Mts. Alien species that were introduced into the Tatra streams are *Salvelinus fontinalis* and *Oncorhynchus mykiss*.

Białka Tatrzańska River In the 1960s, in the middle and lower sections of the Białka Tatrzańska River, eight species of rheophilic fish occurred [33]. At two sites, two species were recorded, *Salmo trutta* m. *fario* and *Cottus poecilopus*, and except for the first site, at the boundary between the forest montane zone and submontane zone, *Barbatula barbatula*, *Thymallus thymallus* and *Phoxinus phoxinus* were found. Only at the estuary site, *Squalius cephalus* occurred, whereas *Barbus carpathicus* was absent (Table 3). After 20 years, at the same river section, only five species out of those found there earlier occurred: *Salmo trutta* m. *fario*, *Cottus poecilopus*, *Phoxinus phoxinus*, *Barbatula barbatula* and *Barbus carpathicus* [30]. In fish harvesting taking place in 2002, after the construction of the Czorsztyn

Table 3 Ichthyofauna of the Białka Tatrzańska River before (1962) and after (2002) the creation of the Czorsztyn Reservoir

Name of site (km downwards from the river source)	1962 (Solewski 1965)	2002 (Augustyn 2006)
Jurgów (17.7 km)	*Salmo trutta* m. *fario*	*Salmo trutta* m. *fario*
	Cottus poecilopus	*Cottus poecilopus*
	Barbus carpathicus	–
Trybsz (28.45 km)	*Salmo trutta* m. *fario*	*Salmo trutta m. fario*
	Cottus poecilopus	*Cottus poecilopus*
	Barbus carpathicus	–
	Barbus waleckii	–
	Phoxinus phoxinus	–
	Thymallus thymallus	–
	Barbatula barbatula	–
Nowa Biała (30.45 km)	*Salmo trutta* m. *fario*	*Salmo trutta* m. *fario*
	Cottus poecilopus	*Cottus poecilopus*
	Barbus carpathicus	*Barbus carpathicus*
	Barbus waleckii	–
	Thymallus thymallus	*Thymallus thymallus*
	Barbatula barbatula	*Barbatula barbatula*
	Squalius cephalus	–
Krempachy (32.45 km)	*Salmo trutta* m. *fario*	*Salmo trutta m. fario*
	Cottus poecilopus	*Cottus poecilopus*
	Barbus carpathicus	*Barbus carpathicus*
	Barbus waleckii	–
	Phoxinus phoxinus	–
	Thymallus thymallus	*Thymallus thymallus*
	Barbatula barbatula	*Barbatula barbatula*
Frydman (40 km)	*Salmo trutta* m. *fario*	*Salmo trutta* m. *fario*
	Cottus poecilopus	*Cottus poecilopus*
	Phoxinus phoxinus	*Phoxinus phoxinus*
	Thymallus thymallus	*Thymallus thymallus*
	Barbatula barbatula	–
	Squalius cephalus	*Squalius cephalus*
	–	*Rutilus rutilus*
	–	*Perca fluviatilis*

Adapted from [7, 33]

and Sromowce reservoirs, nine fish species were present [7]. Brown trout *Salmo trutta* m. *fario* and *Cottus poecilopus* still occurred at all sites, and at the first two sites, they were the only fish species. At the other sites, the following species have been recorded: *Thymallus thymallus*, *Barbus carpaticus* and *Barbatula barbatula*. At the estuary section of the river, apart from *Squalius cephalus*, some limnophilic species from the Czorsztyn Reservoir, *Rutilus rutilus* and *Perca fluviatilis*, were present, whereas they had not been observed there before the dam construction.

Dunajec River A 97-km long section of river, between Nowy Targ and Nowy Sącz, had been divided into two fish zones: a trout zone and a barbel zone [2]. According to Kołder [34] (as cited in [30]), this border existed until the 1960s. However, changes caused by human impact affected the structure of fish species considerably. Fish that are typical of the barbel zone moved markedly upstream. Consequently, there was a sharp decrease in the trout zone of fish such as *Salmo trutta* m. *fario* and *Thymallus thymallus.* At the same time, there was an increase in the fish taxa typical for the barbel zone: *Barbus waleckii* and *Chondrostoma nasus*. A vast area is cohabited by two species characteristic of both trout and barbel zones.

The biggest changes in ichthyofauna took place after the construction of the Czorsztyn and Sromowce Wyżne reservoirs (Table 4). In fish harvesting between 2001 and 2002 in the Dunajec River above the Czorsztyn Reservoir, 18 species were found. The predominant species were as follows: *Rutilus rutilus* (21%), *Squalius cephalus* (16%), *Barbus carpathicus* (10%) and *Alburnus alburnus* (11%). The percentage of other fish species was less than 5%, including those of the salmonids: *Salmo trutta* m. *fario*, *Thymallus thymallus* and cyprinids (*Phoxinus phoxinus*) and cottids (*Cottus poecilopus)*, species typical of the trout zone. Below the reservoir, in the Pieniny segment of the Dunajec River, 17 fish species occurred. The dominant ones were *Thymallus thymallus* (27%), *Alburnus alburnus* (23%) and *Salmo trutta* m. *fario* (20%). Yet, the share of fish dominating above the reservoir was each lower than 10%: *Squalius cephalus* (8%), *Barbus* carpathicus (7%) and *Rutilus rutilus* (2%). The lower course was again dominated by cyprinids species: *Alburnus alburnus* (28%), *Squalius cephalus* (24%) and those of the family Percidae (*Perca fluviatilis* 12%). In an electrofishing study, 20 fish species were found; however, this number may be higher because in an angling fishing study, nine other species were harvested, for example, *Vimba vimba*, *Silurus glanis* and *Tinca tinca* [7, 30].

6 Ecological Characteristics of Macroinvertebrates and Fish Communities

Analysing the distribution of fauna along the natural mountain streams and rivers, a number of faunal communities characteristic of each zone associated with altitude, gradient (slope), velocity and substratum could be distinguished (Fig. 7).

In the uppermost sections of the streams, flowing in the alpine zone, above 2,000 m a.s.l., a characteristic community composed mainly of Chironomidae larvae with the dominant *Diamesa steinboecki* was developed, a species typical of glacial streams. At an altitude of 1,700–2,000 m a.s.l., in the alpine zone, a community of different species was developed with the dominant larvae of *Diamesa* gr. *latitarsis*, but also other species of Chironomidae occurred, for example, *Parorthocladius nudipennis*, *Chaetocladius*, *Eukiefferiella* and larvae of Simuliidae.

In the streams flowing from high mountain lakes and periodical springs in the dwarf pine zone (1,550–1,700 m a.s.l.), larvae of *Diamesa* gr. *latitarsis* still predominated. However, there was an increase in fauna diversity. Apart from larvae of Chironomidae and Simuliidae, some other larvae of Plecoptera, mainly of genera *Protonemura*; *Amphinemura*; *Leuctra*; Trichoptera, *Rhyacophila*; and *Drusus*, occurred. Pools were inhabited by numerous Oligochaeta – *Nais variabilis.*

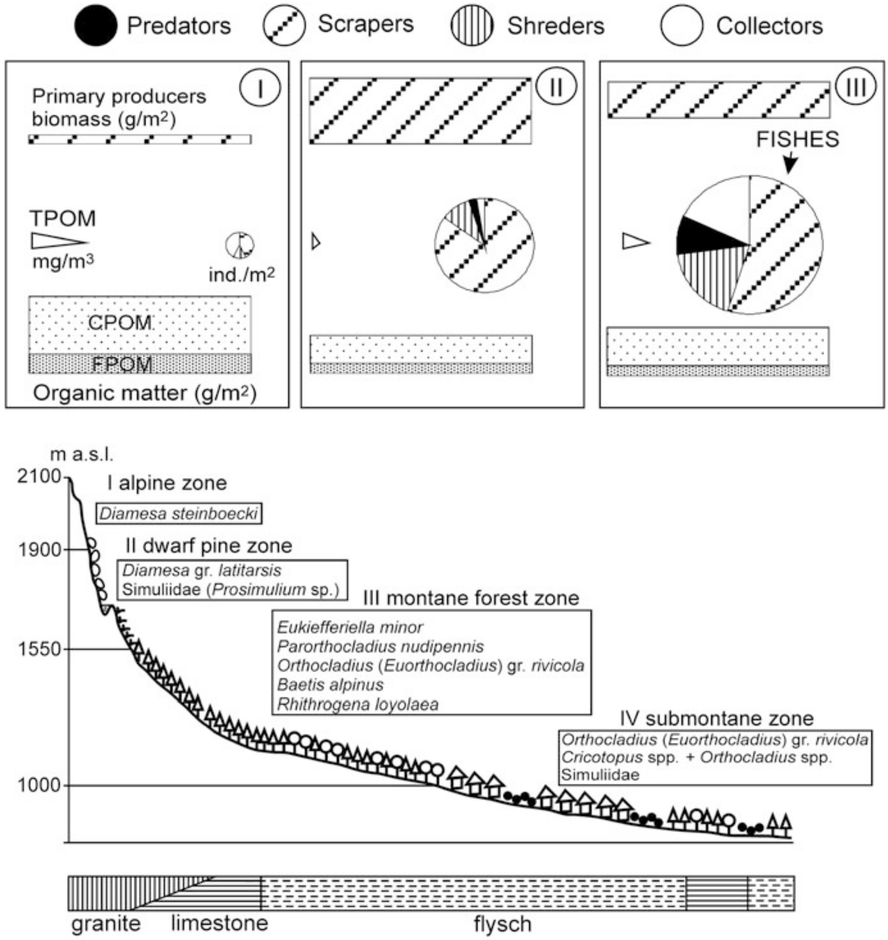
Table 4 Species structure (%) of ichthyofauna of the Upper Dunajec in 2002

Species	Sampling sites (km)													
	0–3	8	11	12	13–22	23	35	45	49	74	82	87	95	97
Salmo trutta m. *fario*	6.2	[a]	[a]	[a]	Reservoirs Czorsztyn and Sromowce Wyżne	24.0	29.0	8.8	13.0	6.4	[a]	11.8	[a]	[a]
Thymallus thymallus	8.8	[a]	[a]	[a]		32.7	36.2	14.3	16.0	22.0	28.2	[a]	[a]	[a]
Phoxinus phoxinus	18.3	[a]		[a]		[a]	[a]	8.8	[a]	[a]		[a]	[a]	
Cottus gobio	6.5	[a]	[a]	[a]		[a]	[a]							
Barbatula barbatula	42	16.3	[a]	9.5		[a]	[a]	[a]	[a]	[a]		[a]	[a]	[a]
Barbus carpathicus		27.7	7.7	8.9						22.4	14.7	[a]	[a]	[a]
Barbus waleckii		5.3		[a]				[a]	[a]	14.8	27.4	17.9	[a]	[a]
Squalius cephalus	15.0	15.3	17.7	17.2		[a]	[a]	19.0	13.3	21.7	9.8	24.3	8.1	36.0
Rutilus rutilus		14.0	39.9	17.7			[a]	[a]	[a]		[a]		[a]	7.1
Alburnus alburnus	[a]	[a]	21.7	13.4		[a]	[a]			6.9	5.1	16.3	51.9	36.3
Perca fluviatilis		[a]	[a]	21.4		[a]	[a]	[a]		[a]	11.7	13.7	17.6	13.4
Leuciscus leuciscus		[a]	[a]	[a]		[a]	6.3	8.3	16.0	[a]				[a]
Abramis brama		[a]	[a]			24.3	10.9	32.9	36.5				[a]	[a]
Esox lucius		[a]	[a]			5.3	[a]							
Other[b]	3.2	21.4	13.0	11.8		13.7	16.7	7.9	5.1	5.7	3.2	14.8	20.5	7.2
Number of species	8	17	14	14		14	15	10	9	10	12	10	16	12
Number of specimens	759	812	1,445	797		395	431	216	293	419	632	263	655	2,618

Adapted from [7], modified

[a]Value below 5%

[b]Species below 5% (km): *Cottus poecilopus* (0–3, 23–35), *Hucho hucho* (8–35, 82, 95–97), *Gobio gobio* (8, 12, 95), *Oncorhynchus mykiss* (95), *Sander lucioperca* (82, 95), *Anguilla anguilla* (74), *Lota lota* (82), *Chondrostoma nasus* (23, 35)



Site	Unit	I	II	III	IV
Altitude	m a.s.l.	1700-2100	1550-1700	1000-1550	<1000
Gradient	‰	200-600	100-200	20-100	usually < 10
Wide	m	1-2.5	2-5	4.5-8.5	6-100
Current velocity	m3/sek	0.5-0.9	0.4-1.3	0.2-1.5	0.4-1.5
Temperature	°C	0.9-9.5	2.4-12.3	1.0-9.1	0.2-12.8
pH		5.3-6.4	6.0-6.5	6.3-6.8	7.2-8.5
Conductivity	μS/cm	16-19	18-26	21-69	69-215
Oxygen saturation		79-85	82-89	84-89	78.8-94.1
Ca^{2+}	mg/L	2.4-4.4	2.6-4.9	3.3-8.6	12.9-41.0
$N\text{-}NO_3$	mg/L	0.31-1.06	0.6-1.13	0.56-1.09	0.51-1.06
PO_4^{3-}	mg/L	0.02-0.14	0.01-0.04	0.01-0.07	0.01-0.04

Fig. 7 The model of structure and function of the mountain stream and rivers in the light of the "River Continuum Concept". Adapted from [35, 39], modified

Ephemeroptera was still absent. These communities were characterised by very low biodiversity and abundance.

At an altitude of 1,500–1,550 m a.s.l., where the streams are fed by a number of springs of the moraine type, faunal abundance and biodiversity sharply rose. The predominant species were Chironomidae, *Eukiefferiella minor* and *Parorthocladius nudipennis* and Ephemeroptera, *Baetis alpinus* and *Rhithrogena loyolaea.* Also Plecoptera, *Protonemura*, *Capnia* and *Isoperla*; Trichoptera, *Rhyacophila*; and Tricladida, *Crenobia alpina* were numerous there.

In the streams of the forest montane zone (1,000–1,500 m a.s.l.), fauna composition displayed small changes in relation to the above described section. Chironomidae, *Parorthocladius nudipennis* or *Orthocladius* (*Euorthocladius*) *rivicola*, and Ephemeroptera, *Baetis alpinus* and *Rhithrogena loyolaea*, were still predominant. Also Plecoptera, especially species of genera *Leuctra* and *Protonemura*, and Trichoptera, *Rhacophila*, *Druzus* and *Allogamus*, were numerous. Oligochaeta occurred sporadically, mostly in pools. Fish (*Salmo trutta* m. *fario*, *Cottus poecilopus*) occurred in the lower sections of this zone.

Below 1,000 m a.s.l., in streams and rivers flowing at the foot of the Tatra Mts., Chironomidae – *Orthocladius* (*Euorthocladius*) gr. *rivicola* – and larvae of the genera *Cricotopus* and *Orthocladius* were still predominant. Larvae of Simuliidae were also abundant. A number of new species of Ephemeroptera, Plecoptera and Trichoptera occurred. Fish, mainly *Salmo trutta* m. *fario*, *Cottus poecilopus*, *Phoxinus phoxinus* and *Thymallus thymallus*, were regularly found in this zone.

Along the study rivers and streams, a few schemes of functioning of ecosystems can be distinguished [35] (Fig. 7).

In the upper montane zone (zones: alpine and dwarf pine) of the streams, the flow is periodical. The content of major cations and anions, as well as nitrogen and phosphorus compounds, was very low. This resulted in a limited development of algae. The amount of bottom particulate organic matter (BPOM) and transported particulate organic matter (TPOM) were relatively high. However, coarse particulate organic matter (CPOM) prevailed, which is weakly available to the juvenile stages of shredders and collectors that appear when the stream starts to flow. The abundance of invertebrate fauna was very low. Larvae of Chironomidae of the genus *Diamesa* predominated and feed on BPOM and algae. Simuliidae, which are collectors, live on TPOM from lake outlets. There were no predators: fish and insects. In this zone, an element creating the development of biocenosis is a periodic water flow which excludes a lot of species with 1-year life cycles.

In the montane forest zone, where streams flow the whole year, plant cover composed of mosses and algae developed abundantly. However, the content of organic matter BPOM and TPOM was low. This situation caused the strong development of scrapers, while the number of collectors feeding on organic matter was relatively low. In this zone fish were absent, but insect predators were present.

In the submontane zone, concentrations of most chemical elements, especially of Ca and Mg, were increased, but the contents of phosphorous and nitrogen compounds were still low. The content of organic matter BPOM and TPOM increased, yet the development of algae cover was lower than in the montane forest zone. The zoocenosis was composed mainly of scrapers; the share of shredders and collectors reached up to 40%. The first-order consumers were controlled by fish, which were the top predators controlling the whole zoocenosis.

7 Influence of Human Impact on Streams and Rivers Fauna Communities

7.1 *Effect of Pollution*

The above-presented scheme of structural and functional diversity of faunal communities along the streams and rivers was disturbed by human impact. The Upper Dunajec River, which retained its natural character until the 1960s, has been contaminated by sewage from villages and the towns of Nowy Targ and Nowy Sącz and developing tourist infrastructure in this area. Most of the Tatra streams have retained their natural character. The only source of pollution of these streams comes from tourist shelters and tourism (Table 2 and Fig. 5).

The inflow of sewage caused a drastic increase in nutrient content, as well as BOD5 and COD values (see Chap. 3, Table 2, Fig. 3 [15, 19]) in the same range, and affected invertebrate communities in streams below the tourist shelters in the forest zone and in the river below towns in the submontane zone (Figs. 5 and 6). Below the sewage effluent discharge, the faunal density grew from a few or several thousand per square metre to a few dozen thousand or even 200,000 ind./m^2 [19, 25, 28–29]. Oligochaeta, whose percentage in mountain streams and rivers ranged less than 3% of the total faunal abundance, below the sewage effluent inflow rose to 20% in the Tatra Mts. and to 80% in the Dunajec River, below the towns. Naididae (*Nais elinguis*, *Nais bretscheri and Nais barbata*), the species that feed on algae and fine detritus, were dominant there. Although the percentage of other faunal groups decreased, their density rose. For example, the density of *Ephemeroptera* in Rybi Potok at the first control site was 1,475 ind./m^2; 30 m below the sewage outflow, 2,100 ind./m^2; and 100 m below, 5,800 ind./m^2. Also biodiversity at the site above and below the discharge of sewage effluent was similar. In the Dunajec River above Nowy Sącz, 118 taxa of invertebrates were recorded, and below the discharge, there were 137. Processes of self-purification in mountain streams and rivers proceeded rapidly, in the Tatra streams over the distance of a few hundred metres and in the Dunajec River – a few kilometres. As a result of these processes, the further the distance from the sewage source, the lower faunal density and the percentage of Oligochaeta were observed, but the percentage of Chironomidae increased. Some differences in this phenomenon were observed in the Dunajec River. In the 1970s at sites D1 and D2, density was over 30,000 ind./m^2, and the percentage of Oligochaeta was from

35 to 40%, whereas in the 1990s, density was lower than 20,000 ind./m^2, and the percentage of Oligochaeta was 20%.

In the study section of the Dunajec River, Nowicki [2] distinguished two fish zones: the trout zone where rheophilic fish require well-oxygenated water and the barbel zone, where fish occur in warmer water with lower dissolved oxygen content and a higher amount of suspended matter. The boundary between these two zones in the nineteenth century was at the site of the Poprad River inflow into the Dunajec River. Due to the increase in water contamination of the Dunajec River after 1964, the boundary shifted upstream, and the fish characteristics of the barbel zone moved upstream. The barbel and nase grew in abundance in this section [30].

7.2 Reservoirs

Constructed in 1997, reservoirs of Czorsztyn and Sromowce Wyżne on the Dunajec River considerable changed the hydrological character of the river and the communities of macroinvertebrates and fish. Before the reservoirs construction, the macroinvertebrate community was characterised by high biodiversity, and the rheophilic species Ephemeroptera, Plecoptera and Trichoptera were very abundant. Inversely, the invertebrate community in the Czorsztyn Reservoir was composed of 13 species of Oligochaeta and two species of Chironomidae: *Chironomus plumosus* and *Chironomus bernensis* [36, 37]. Water dammed up in the reservoir made the river flow more slowly above the reservoir, and consequently, there was an increase in the amount of sediment deposited in the river bed. Muddy sediments were inhabited by larvae of *Prodiamesa olivacea* and taxa of the subfamily Chironominae, not typical of rivers with a stone bottom and high velocity.

The structure of ichthyofauna in the Dunajec segment of the Czorsztyn reservoir had been investigated immediately before it was filled [38]. In this segment, 16 fish species occurred. The dominant species were *Phoxinus phoxinus*, *Barbus carpathicus*, *Barbus waleckii*, *Squalius cephalus*, *Leuciscus leuciscus* and *Alburnus alburnus*. The percentage of the other species ranged from 0.5 to 5% of the total number of all fish recorded (Fig. 8).

After the Czorsztyn Reservoir was filled with water, there was a considerable change in the ichthyofauna community. Studies carried out in 2002 showed the occurrence of eight fish species [36]. The dominant species were *Rutilus rutilus* (50%), *Abramis brama* (20%) and *Perca fluviatilis* (10%). The other species, such as *Alburnus alburnus*, *Leuciscus leuciscus*, *Carasius carasius*, *Gymnocephalus cernua* and *Esox lucius*, were less than 5% (Fig. 8). Four species, *Abramis brama*, *Carassius carassius*, *Gymnocephalus cernua*, *Esox lucius*, had not been recorded before dam construction in this Dunajec section.

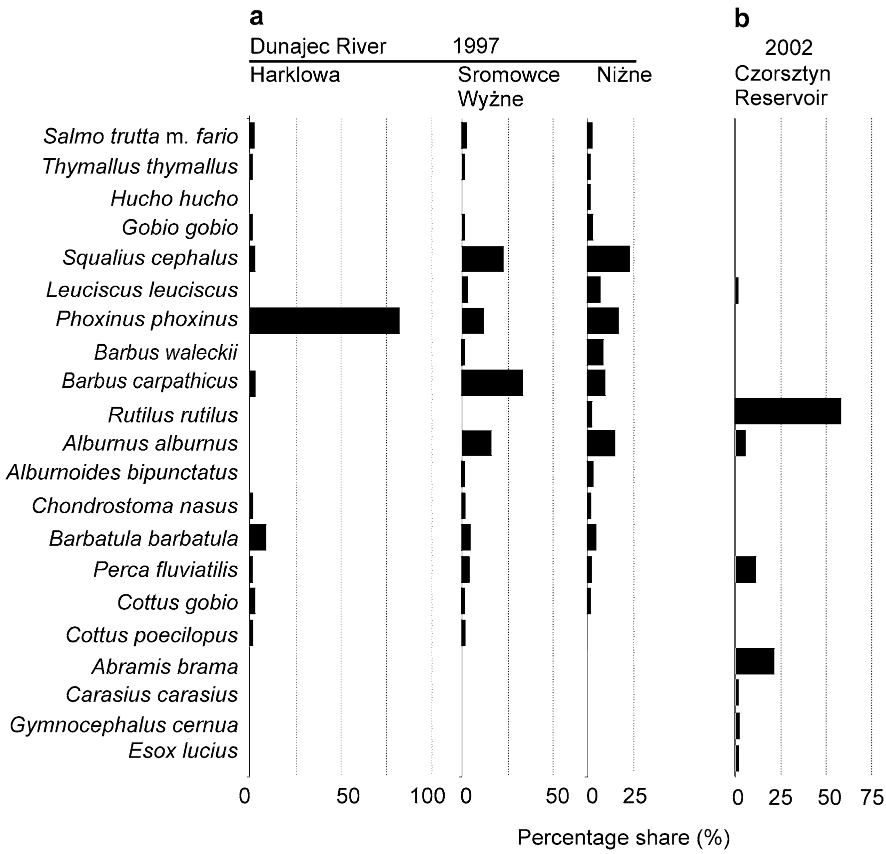


Fig. 8 Change of ichthyofauna of the Dunajec River before (**a**) and after (**b**) the creation of the Czorsztyn Reservoir [36, 38]

8 Conclusions

The Tatra streams (tributaries of the Dunajec River) have mostly retained their natural character and developing faunal communities are characteristic of these types of biotopes. The source of water pollution comes mainly from the tourist shelter and tourism centre around them. Discharge of untreated or partially treated sewage effluent affected invertebrate communities; however, this impact was only observed over a short river section.

Until 1964, the water of the Dunajec River, between the towns of Nowy Targ and Nowy Sącz, preserved its natural character with little human impact. An increase in sewage effluent inflow, mostly untreated, from Nowy Targ, Nowy Sącz and numerous recreational centres and villages, observed in the 1970s, resulted in the increasing contamination of the river and consequently changed natural communities of macroinvertebrates and fish. Since the beginning of the 1990s, there has been a decrease in nutrient contents and BOD5, which has been reflected in changes in invertebrate communities. However, the course of these changes is different from that in the lowland rivers [40].

The waters of mountain rivers with stone bottoms and rapid, turbulent flows were well oxygenated below the sewage effluent inflow. The inflow of nutrients resulted in the development of algae mats on the surface of stones, where Oligochaeta of the family *Naididae* have developed abundantly. On bottom stone surfaces, Ephemeroptera, Plecoptera and Trichoptera, characteristic of mountain rivers occur. Taxa characteristic of polluted waters may develop under stones, where water flow is slow or very slow, which favours the deposition of sediments, as well as in pools. However, this habitat is very rare.

Dam reservoirs completely changed the character of the Dunajec River. In the Czorsztyn Reservoir, communities of macroinvertebrates and fish developed, which are not typical of mountain rivers. Rheophilic communities were replaced with taxa characteristic of slow-flowing waters. Also below and above the reservoir, changes in communities were observed.

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References

1. Staszic S (1815) O ziemiorództwie Karpatów i innych gór i równin Polski. Rozprawa trzecia o Wołoszyni, o Pięciu Stawach i Morskiem Oku. Rocznik Towarzystwa Warszawskiego Przyjaciół Nauk, T. VII, Warszawa, pp 73. (in Polish)
2. Nowicki M (1883) A survey of fish distribution in the rivers of Galicia according to catchment basins and fish region (Map). E. Holzl, Vienna. (in Polish)
3. Nowicki M (1889) On fish in the Rivers Vistula, Styr, Dniester, and Prut catchment Bains. F. Kulczycki and Ska, Krakow. (in Polish)
4. Spiczakow T (1927) The study of the Dunajec basin in the summer of 1926. Okólnik Rybacki 1. in Polish
5. Starmach K (1965) Limnological investigations in the Tatra Mountains and Dunajec River basin. Kom Zagosp Ziem Górskich 11:294
6. Kawecka B, Szczęsny B (1984) Dunajec. In: Whitton BA (ed) Ecology of European rivers. Blackwell Scientific Publications, Oxford, pp 499–525
7. Augustyn L (ed) (2006) Ichthyofauna of the Dunajec River basin in the beginning of the XXI century, vol 18. Państwowa Wyższa Szkoła Zawodowa, Nowy Sącz, pp 1–84. (in Polish with English summary)
8. Lexa J, Bezák V, Elečko M, Mello J, Polák M, Potfaj M, Vozár J (eds) (2000) Geological map of Western Carpathians and adjacent areas 1: 500,000. Geological Survey of Slovak Republic, Bratislava
9. Solon J, Borzyszkowski J, Bidłasik M, Richling A, Badora K, Balon J, Brzezińska-Wójcik T, Chabudziński Ł, Dobrowolski R, Grzegorczyk I, Jodłowski M, Kistowski M, Kot R, Krąż P, Lechnio J, Macias A, Majchrowska A, Malinowska E, Migoń P, Myga-Piątek U, Nita J, Papińska E, Rodzik J, Strzyż M, Terpiłowski S, Ziaja W (2018) Physico-geographical mesoregions of Poland: verification and adjustment of boundaries on the basis of contemporary spatial data. Geogr Pol 91:143–170

10. Oszczypko N (2006) Late Jurasic-Miocene evolution of the outer Carpathian fold – and – thrust belt and its foredeep basin (Western Carpathians, Poland). Geol Q 50:169–194
11. Jurewicz W (2005) Geodynamic evolution of the Tatra Mts. and the Pieniny Klippen Belt (Western Carpathians): problems and comments. Acta Geol Pol 55:295–338
12. Gradziński M, Hercman H, Kicińska D, Barczyk G, Bella P, Holúbek P (2009) Karst in the Tatra Mountains – development of knowledge in the last thirty years. Przegl Geol 57:674–684
13. Bombówna M (1968) Hydrochemical characteristics of the Białka Tatrzańska stream. Acta Hydrobiol 10:27–37
14. Bombówna M (1971) The chemical composition of the water of streams of the Polish High Tatra, particularly with regards to the Stream Sucha Woda. Acta Hydrobiol 13:379–391
15. Bombówna M (1977) Biocenosis of high mountain stream under the influence of tourism. 1. Chemical of the Rybi Potok waters and chlorophyll content in attached algae and seston in relation of the pollution. Acta Hydrobiol 19:243–255
16. Kownacki A (2008) Kryon – communities of high mountain streams. Annales Universitatis Mariae Curie Skłodowska, Section C – Biologia 63:59–70
17. Szarek E (1994) The effect of abiotic factors on chlorophyll a in attached algae and mosses in the Sucha Woda stream (High Tatra Mts, southern Poland). Acta Hydrobiol 36:309–322
18. Florczyk H, Jermolińska K, Blezel H, Grabska I, Fila H (1972) Atlas of river pollution in Poland – 1970. Instytut Gospodarki Wodnej, Warszawa
19. Kownacki A, Flejtuch T, Damnicka E (2002) The effect of treated wastes on benthic invertebrate communities in the Mountain zone of the Dunajec River (Southern Poland). In: Kownacki A, Soszka H, Fleituch T, Kudelska D (eds) River biomonitoring and benthic invertebrate communities. Institute of Environmental Protection, Warszawa, pp 22–49
20. Szalińska E, Dominik J (2006) Water quality changes in the Upper Dunajec Watershed, Southern Poland. Pol J Environ Stud 15:327–334
21. Kawecka B, Kownacki A, Kownacka M (1978) Food relation between algae and bottom fauna communities in glacial streams. Verh Internat Verein Limnol 20:1527–1530
22. Kownacka M (1971) The bottom fauna of the stream Sucha Woda (High Tatra Mts) in the annual cycle. Acta Hydobiol 13:415–438
23. Kownacki A (1971) Taxocens of Chironomidae in streams of the Polish High Tatra Mts. Acta Hydrobiol 13:439–464
24. Kownacki A, Dumnicka E, Galas J, Kawecka B, Wojtan K (1997) Ecological characteristics of a high mountain lake-outlet stream (Tatra Mts, Poland). Archiv f Hydrobiologie 139:113–128
25. Kownacki A (1977) Biocenosis of high mountain stream under the influence of tourism. 4. The bottom fauna of the stream Rybi Potok (the High Tatra Mts). Acta Hydrobiol 19:293–312
26. Kownacka M, Kownacki A (1965) The bottom fauna of the River Białka and its Tatra tributaries the Rybi Potok and Potok Roztoka. Kom Zagosp Ziem Górskich 11:130–152. (in Polish)
27. Dratnal E, Szczęsny B (eds) (1965) Benthic Fauna of the Dunajec River. Kom Zagosp Ziem Górskich 11:162–214. (in Polish)
28. Dratnal E, Sowa R, Szczęsny B (1979) Benthic invertebrate communities in the Dunajec River between Harklowa and Sromowce Nizne. Ochr Przyr 42:183–215. (in Polish with Englisch summary)
29. Szczęsny B (ed) (1995) Degradation of the benthic invertebrate fauna of the Dunajec River in the neighborhood of Pieniny National Park (Southern Poland). Ochr Przyr 52:207–224. (in Polish with English summary)
30. Starmach J (1983/1984) Fish zones of the River Dunajec upper catchment basin. Acta Hydrobiol 25(26):415–427
31. Kozłowski K, Dynowski P, Kozłowski J, Źróbek-Sokolnik A, Wolter K, Vilmos J (2017) Vertical distribution of *Cottus poecilopus* Heckel, 1837 in streams of Tatra National Park in Poland. In: "Environmental Engineering" 10th international conference. Vilnius Gediminas Technical University, Lithuania, 27–28 April 2017. doi:https://doi.org/10.3846/enviro.2017.032
32. Żarnecki S, Bieniarz K (1967) The age and growth ration of the brown trout from the Rybi Potok in the Tatra Mountains. Acta Agraria et Silvestria, ser. Zootechnica 7:83–95. (in Polish with English summary)

33. Solewski W (1965) The ichthyofauna of the Białka Tatrzańska stream with special respect to the characteristics of brown trout (*Salmo trutta* morpha *fario* L.). Acta Hydrobiol 7:197–224. (in Polish)
34. Kołder W (1964) Ichtyofauna of the Dunajec basin. (typescript)
35. Kownacki A, Galas J, Kawecka B, Szarek E, Wojtan K (1993) Structure and functioning of stream ecosystems in the Tatra National Park. In: Radwan S, Karbowski Z, Sołtys M (eds.) Water and peat ecosystems in protected areas. Lublin, pp. 40–43
36. Starmach J, Jelonek M (2003) Evaluation of the condition of ichtyofauna and water environment of the Czorsztyn reservoirs. Suppl Acta Hydrobiol 6:65–87. (in Polish with English summary)
37. Michailova P, Szarek-Gwiazda E, Kownacki A (2009) Effect of contaminants on the genome of some species of genus Chironomus (Chironomidae, Diptera) live in sediments of Dunajec River and Czorsztyn reservoir. Water Air Soil Poll 202:245–258
38. Starmach J (1998) Ichthyofauna of the river Dunajec in the region of the Czorsztyn-Niedzica and Sromowce Wyżne dam reservoir (southern Poland). Acta Hydrobiol 40:199–205
39. Kawecka B, Kownacka M, Kownacki A (1971) General characteristics of the biocenosis in the streams of the polish high Tatras. Acta Hydrobiol 13:465–476
40. Starmach K, Wróbel S, Pasternak K (1976) Hydrbiologia, Limnologia. Państwowe Wydawnictwo Naukowe, Warsaw

Assessment Criteria and Ecological Classification of Polish Lakes and Rivers: Limitations and Current State

Agnieszka Napiórkowska-Krzebietke, Łucjan Chybowski, Paweł Prus, and Mikołaj Adamczyk

Contents

Abstract An assessment of the ecological status or potential is based mainly on examination of the aquatic ecosystem functioning regarding organisms that inhabit them. These include phytoplankton, macrophytes and phytobenthos, benthic macroinvertebrates and ichthyofauna. However, hydromorphological and physico-chemical elements are also of importance as supporting biological features.

The results of ecological status assessment, made for data obtained from the Polish-Norwegian project "DeWELopment", indicate that ichthyofauna and the hydromorphological elements in rivers as well as phytoplankton and benthic macroinvertebrates in lakes were decided primarily on final ecological classification.

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According to the accepted in Poland integration of partial assessments based on the *one out all out* principle, the lakes in the Wel River catchment were classified as not meeting the WFD-required at least good ecological status, and rivers, in turn, only a half of them. The latest analysis of the risk of a failure to achieve the environmental objectives (i.e. at least good ecological status/potential of water bodies in the two largest catchment areas in Poland) has shown that 74% of water bodies in the Vistula River Basin and 67% of water bodies in the Oder River Basin have been classified as being in danger of failing to achieve these objectives. Moreover, assessments have not been made for approx. 30% of water bodies covered by the assessment in both of these catchment areas.

Keywords Biological quality elements · Ecological potential · Ecological status · Lakes · Rivers

1 Introduction

The Water Framework Directive (WFD), i.e. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 [1], establishes the framework of community measures in regard to water policy. The aim of the directive is to maintain and ensure good water quality and to improve the degraded aquatic environment in the European Community. Therefore, common definitions of the water quality status and environmental goals have been developed in each Community country to ensure at least good status of surface waters and groundwater and to prevent further deterioration of the water status. Implementation of the WFD has resulted, among others, in establishing the principles of water control, including criteria for water quality assessment to achieve (at least) good water quality by 2015 and currently (after shifting the deadline) by 2027. There are now over 300 methods of ecological assessments of aquatic ecosystems, 260 of which have been intercalibrated [2].

The approach to water quality status assessment is based on an examination of the functioning of aquatic ecosystems, including the main organisms that inhabit them. For this reason, this paper focuses mainly on the biological methods of assessing the ecological status/potential, developed in line with the WFD guidelines, as well as on limitations of their applicability and the current results of such an assessment. Among the biological elements, most methods have already been developed, verified and – after intercalibration – used to make assessments. For the "ichthyofauna" element, the limiting values have changed considerably, and an assessment based on the "macroinvertebrates" element is temporarily not taken into account in the state monitoring.

2 Criteria and Ecological Classification of Surface Water Bodies in Poland

Currently, pursuant to the Regulation of the Minister of Environment of 21 July 2016 [3], the classification of quality/status of water bodies is based on biological elements (Figs. 1, 2 and 3) and hydromorphological and physicochemical elements as those supporting biological elements. Each of these elements is assessed on the basis of quality indices, taking into account the category of water bodies and various types of surface waters. The assessment of the ecological status is made for natural water bodies, i.e. rivers, lakes, transitional and coastal waters, whereas an assessment of the ecological potential of artificial and heavily modified water bodies is based on the elements mentioned above. There are five classes of ecological status or potential: very good/maximum, good, moderate, poor and bad.

The first stage of an assessment covers biological elements:

– Composition, abundance and biomass of phytoplankton
– Composition and abundance of other aquatic flora
– Composition and abundance of benthic invertebrate fauna
– Composition, abundance and age structure of fish fauna

Classification of biological elements in rivers is based on several indices:

– Multimetric phytoplankton index – IFPL
– Diatom index for rivers – IO
– FLORA (class of total rating IFPL and IO)
– Macrophyte Index for Rivers – MIR
– Macroinvertebrates index for rivers – MMI_PL – and macroinvertebrates index for dam reservoirs, MZB
– Fish indices for rivers – EFI+PL and IBI_PL – and auxiliary index concerning diadromous fish, D

However, different indices are used for assessment in lakes:

– Phytoplankton Metric for Polish Lakes – PMPL
– Multimetric diatom index for lakes – IOJ
– Ecological Status Macrophyte Index – ESMI
– Lake Macroinvertebrate Index – LMI
– Ichthyofauna indices LFI+ and LFI-EN

The second stage of an assessment covers hydromorphological elements supporting the biological elements. Classification of hydromorphological elements in rivers is based on:

– Hydrological regime (quantity and dynamics of water flow and connection to groundwater bodies)
– River and streams continuity (i.e. the number and type of barriers and ensuring passage for aquatic organisms)

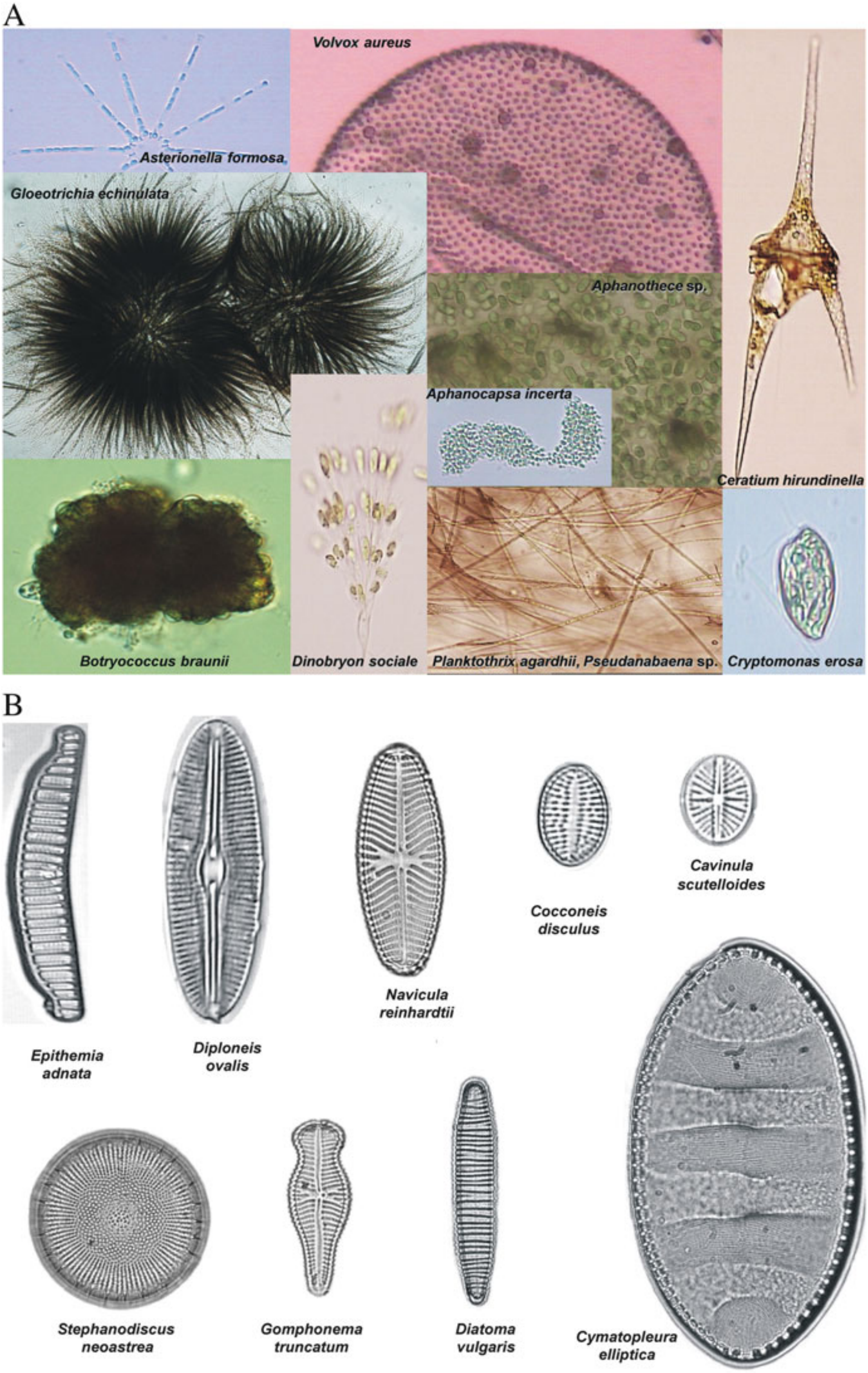

Fig. 1 Representative species of phytoplankton (**a**), phytobenthos (**b**)

A B

Fig. 2 Representative species of benthic macroinvertebrates plecopteran larvae (**a**) and *Unio pictorum* with attached small invasive *Dreissena polymorpha* (**b**)

– Morphological conditions (river and stream depth and width variation, structure and substrate of the river and stream bed, structure of the riparian zone and flow speed)

For lakes, classification of hydromorphological elements covers only one index: LHMS_PL, i.e. Lake Habitat Modification Score, but it includes several characteristics (i.e. hydrological regime: quantity and dynamics of water flow, residence time, connection to the groundwater body and morphological conditions – lake depth variation, quantity, structure and substrate of the lake bed, structure of the lake shore).

The state hydrological and meteorological service examines surface waters in regard to hydrological and morphological elements and submits their results to competent regional environmental protection inspectors for status assessment of surface water bodies – rivers. On the other hand, the regional environmental protection inspector conducts observations of hydromorphological elements for assessment of the ecological status of lakes.

The third stage of an assessment covers physicochemical elements (which support the biological elements). For lake and river water bodies, these elements include five groups of indices:

– A group of indices which characterise the physical condition, including thermal conditions
– A group of indices which characterise the oxygen-related conditions (aeration) and organic impurities
– A group of indices which characterise salinity
– A group of indices which characterise acidity (only river water bodies)
– A group of indices which characterise biogenic conditions (biogenic substances)

At this stage, water bodies are assessed in terms of concentrations of substances (specific synthetic and non-synthetic) particularly harmful to the aquatic environment.

Fig. 3 Representative species of fish *Sabanajewia aurata* (**a**), *Chondrostoma nasus* (**b**), *Barbatula barbatula* (**c**), *Romanogobio valdykovi* (**d**), *Phoxinus phoxinus* (**e**)

All three stages of classification allow the ecological status of natural surface waters to be assessed and assign to it one of the five classes of water quality: class I (high status), class II (good status), class III (moderate status), class IV (poor status) and class V (bad status). For heavily modified and artificial water bodies, their ecological potential is assessed, and the principle of classification is similar (class I, maximum potential; class II, good potential; class III, moderate potential; class IV, poor potential; class V, bad potential).

An assessment of a water body based on biological elements is not always consistent; the highest consistency occurs for very good status or the maximum ecological potential. The lack of consistency usually occurs for the status or potential below good, i.e. moderate, poor or bad. Among the biological elements, the phytoplankton index is the strictest due to a rapid response of phytoplankton to any changes in the environment. The following principle is applied in assessments of the ecological status or potential: the worst score for an individual index determines the final score – *one out all out*. Detailed guidelines for classification and interpretation of the scores are laid down in Appendices Nos. 7 and 8 of the Regulation of the Minister of Environment [3].

The overall assessment of water bodies quality includes – apart from an assessment of the ecological status or potential – determination and classification of the chemical status of waters, i.e. determination of concentrations of chemical substances included in a group of substances particularly harmful to the environment (priority substances and other impurities, which include heavy metals and their compounds as well as phenol, benzene, dioxins and their compounds). If the chemical indices are below the limit values, then the chemical status is good, and when they are above, then the chemical status is below good. The priority substances included in group 4.1 and other impurities of group 4.2 are mentioned in Appendix No. 9 of the Regulation of the Minister of Environment [3].

Additionally, the meeting of requirements for a protected area is assessed. These types of areas include:

- Protected areas intended for conservation of aquatic animals of economic importance (waters as fish habitats) and protected areas intended for conservation of habitats or species, for which maintaining or improvement of the water status is an important factor in its conservation
- Protected areas which are water bodies for recreational purposes, including those used as bathing sites
- Protected areas susceptible to eutrophication caused by pollutants of household origin
- Protected areas exposed to pollution with nitrogen compounds from agricultural sources

An overall assessment of the status of a water body is based on an assessment of the ecological status or potential and chemical status of waters and on an assessment of the meeting of requirements for protected areas. The "good status" is adopted for waters with a very good/maximum and good ecological status/potential and good chemical status, which meet the requirements for protected areas. In the other cases, the status of a water body is described as bad [3].

3 Limitations of the Use

The assessment method was developed in stages, and it included developing indices which took into account the taxonomic composition and abundance, establishing the typology, establishing the reference conditions specific to each type of waters, establishing the boundaries of classes of ecological status/potential for the biological indices, performing pan-European intercalibration of methods and estimating the risk of misclassification.

Due to unestablished reference conditions among the biological elements, the classification based on two groups of indicator organisms (benthic macroinvertebrates and ichthyofauna) was not taken into account for some time in an assessment of lakes and other natural water bodies. Currently, only the reference conditions for benthic macroinvertebrates are being established (this element was temporarily not taken into account) for such water bodies as lakes or other natural water bodies, including lakes and other natural water bodies established as heavily modified water bodies or an artificial water body. Regarding the ichthyofauna element, the intercalibration process is completed.

Limitations are also part of the classification of lakes by size – 0.5–1 km^2, 1–10 km^2, 10–100 km^2 and >100 km^2. Over 7,000 lakes in Poland have an area of over 1 ha, but there are only 1,045 lakes, belonging to 13 abiotic types [4], with an area of over 0.5 km^2, whose ecological status or potential should be assessed. Currently, when the number of abiotic types was verified and reduced to 7 [5], the number of lakes with an area exceeding 0.5 km^2 also decreased to 1,017 in accordance with a new hydrographic classification of Poland [6]. The verified typology of Polish lakes includes:

- K_a – siliceous stratified
- K_b – siliceous polymictic
- Kond – high conductivity
- WSm_a – calcareous with Schindler's ratio ≤ 2, stratified
- WSm_b – calcareous with Schindler's ratio ≤ 2, polymictic
- WSd_a – calcareous with Schindler's ratio >2, stratified
- WSd_b – calcareous with Schindler's ratio >2, polymictic

Similarly, work is still underway on verification of the types determined so far and a list of the river water bodies. Verification of boundaries of water bodies and the social debate on the issue resulted in a list of river water bodies containing 3,468 river water bodies and 45 water bodies defined as dammed reservoirs. The continuity of river networks was maintained in the new list, the classification was adjusted and the course of watercourses was harmonised.

The method of integrating an assessment of the status/potential and the possibility of misclassification determined the final outcome of the assessment. The integration in Poland took into account the *one out all out* principle. There are a number of factors which affect the risk of misclassification; the main ones include uncertainty of the assessment based on individual biological elements, position of the metric values relative to the limiting values of classes, uncertainty of the limiting values of classes or a large diversity of assessment based on its individual elements [7, 8]. An analysis of all the biological elements showed that misclassification when the *one out all out* principle was applied concerned mainly a wrong classification as worse status/potential. The risk of misclassification in the final integrated assessment of the ecological status/potential can be great when the values of the indices approach the limiting values of the classes. Considering two multimetrics based on phytoplankton and macrophytes, the risk can be 41–45% [9, 10]. Further analyses have shown that the uncertainty of the assessment decreases to several percent when the values of multimetrics are close to half of the range typical of a given class.

4 Biological Elements of Ecological Status/Potential Assessment in Poland

4.1 Rivers

Multimetric Phytoplankton Index (IFPL) For an assessment of the ecological status of the rivers of Poland, IFPL is an arithmetic average of two modules, i.e. the trophy index IT and the chlorophyll index CH [11] (Tables 1 and 2). The trophy index is based on indicator taxa, each of which has a trophic and weight value assigned to them, for a relevant level of the river water fertility. The assessment specificity for a river type, required by the WFD in Poland for rivers with a catchment area of above 5 thousand km^2 and large rivers, is taken into account at the level of the chlorophyll index. The IFPL received positive marks for rivers during the pan-European intercalibration exercise for river R-L2 (very large medium- to high-alkalinity rivers, catchment area of stretch $>$10,000 km^2, alkalinity $>$0.5 meq dm^{-3}) with the final settings of the boundaries between high and good status (1.080) and between good and moderate status (0.920) [12] (Table 3).

Diatom Index for Rivers (IO) For rivers IO is calculated as an arithmetic average of three modules, i.e. trophic TI, saprobic SI and the abundance of representative species GR indices [13] (Tables 1 and 2). The first module describes the level of water fertility [14, in modification], the second describes the level of organic impurities in water [15, in modification], whereas the third module describes the deviation of the community under assessment from a reference community [16, in modification]. Each reference taxon has the value of 1 assigned to it as an index of low trophy and/or saprobity. IO was positively intercalibrated with the final setting of the boundaries between high and good status (0.800) and between good and

Table 1 Description of biological metrics used in the ecological status/potential assessment in Poland

Metric	Formulas	Description	References
Rivers			
Multimetric phytoplankton index, IFPL	$\mathrm{IFPL} = (\mathrm{Z_{IT}} + \mathrm{Z_{CH}})/2$ $\mathrm{Z_{IT}} = 1 - ((\mathrm{IT} - 1) \times 0.25)$ $\mathrm{IT} = \sum(D_i \times wT_i \times T_i)/\sum(D_i \times wT_i)$ $\mathrm{Z_{CH}} = 1 - (\mathrm{CH} \times 0.25)$	D_i, the seasonal average of the percentage share of the *i*th species in the biovolume of indicator phytoplankton taxa; wT_i, the weight value (tolerance) of the *i*th species; T_i, the indicator value of *i*th species; CH, chlorophyll indicator	[11]
Diatom index for rivers, IO	$\mathrm{IO} = (\mathrm{Z_{TI}} + \mathrm{Z_{SI}} + \mathrm{GR})/3$ $\mathrm{Z_{TI}} = 1–(\mathrm{TI} \times 0.25)$ $\mathrm{TI} = \sum(\mathrm{TI}_i \times wT_i \times L_i)/\sum wT_i \times L_i$ $\mathrm{Z_{SI}} = 1–((\mathrm{SI} - 1) \times 0.33)$ $\mathrm{SI} = \sum(S_i \times wS_i \times L_i)/\sum wS_i \times L_i$ $\mathrm{GR} = \sum tR_i$	S_i, the sensitivity value of the *i*th species on organic pollutions; wS_i, the weight value (tolerance) of the *i*th species; L_i, the weight value of the *i*th species (the number of individuals of the *i*th species divided by the number of all counted individuals); tR_i, the relative number of the *i*th reference species (the number of individuals of the *i*th reference species divided by the number of all counted individuals)	[13]
Macrophyte Index for Rivers, MIR	$\mathrm{MIR} = [\sum(L_i \times W_i \times P_i)/\sum(W_i \times P_i)] \times 10$	L_i, indicator value of *i*th species; W_i, weighting factor weight (ecological tolerance) of *i*th species; P_i, ratio of coverage for *i*th species in a site in 9-point scale	[17, 18]
Macroinvertebrates index for rivers, MMI_PL Macroinvertebrates index for dam reservoirs, MZB	$\mathrm{MMI\_PL} = \mathrm{ICMi(0–1)} = 0.334\mathrm{ASPT} + 0.266 \times \log10(\mathrm{sel\_EPTD} + 1) + 0.067 \times (1\text{-}\mathrm{GOLD}) + 0.167 \times S + 0.083 \times EPT + 0.083 \times \mathrm{H'}$ $\mathrm{MZB} = \mathrm{ASPT}/10$	ASPT (average score per taxa), the sum of points of the simplified BMWP_PL index divided by the total number of scored families found in the sampling site; log10 (sel_EPTD +1), decimal logarithm of the total number of selected families (the sum of individuals from families: Heptageniidae, Ephemeridae, Leptophlebiidae, Brachycentridae, Goeridae, Polycentropodidae, Limnephilidae,	[11, 20]

		Odontoceridae, Dolichopodidae, Stratiomyidae, Dixidae, Empididae, Athericidae, Nemouridae +1); 1-GOLD, the share of Gastropoda, Oligochaeta and Diptera deducted from 1; *S*, the numbers of all families; *EPT*, the numbers of families from orders: Ephemeroptera, Plecoptera and Trichoptera; H′, biodiversity index of Shannon-Weaver (Shannon-Wiener)	
Fish indices for rivers: EFI+PL IBI_PL Auxiliary index concerning diadromous fish, D	EFI + $PL_{Salmonid}$ = (Ni.O2.Intol+Ni.Hab.Intol.150)/2 – for types 0–20, 22 and 26 salmonid dominated EFI + $PL_{Cyprinid}$ = (Ric.RH.Par + Ni.LITHO)/2 – for types 0–20, 22 and 26 cyprinid dominated $IBI\_PL_{21a} = \sum_{score\ (1\text{-}3\text{-}5)}$(*TR* + RWC + RB + RLR + IS% + Rr% + Pis% + Inv% + Omn-Rr% + DisHyb% + CPUE+Alien%)/60, for type 21a, sandy; $IBI\_PL_{21b} = \sum_{score\ (1\text{-}3\text{-}5)}$(*TR* + RWC + RB + E + Lith% + ES + RPis% + Inv% + Omn-Rr% + DisHyb% + CPUE +Alien%)/60, for type 21, gravel; $IBI\_PL_{23,24,25a} = \sum_{score\ (1\text{-}3\text{-}5)}$(*TR* + RLith+RWC + RB + El% + Rr% + Pis% + RPisInv% + Omn-Rr% + DisHyb% + CPUE+Alien%)/60, for types 23, 24 and 25a (no salmonids); $IBI\_PL_{25b} = \sum_{score\ (1\text{-}3\text{-}5)}$(*TR* + RWC + RB + Lith% + El% + Sal% + Pis% + RPisInv% + Omn% + DisHyb% + CPUE+Alien%)/60, for type 25b (with salmonids); *D* = PDR/HDR, for types 0–22 and 26	Ni.O2.Intol, density of species intolerant to oxygen depletion (requiring more than 6 mg dm^{-3} O_2); Ni.Hab.Intol.150, density of species intolerant to habitat degradation – specimens ≤150 mm (total length); Ric.RH.Par, richness (number of rheopar species) requiring a rheophilic reproduction habitat; Ni.LITHO, density of species requiring lithophilic reproduction habitat; *TR*, total number of fish species; RLith, richness of lithophilous species; RWC, richness of water column species; RB, richness of benthic species; RLR, richness of fish species typical for large rivers; E, species evenness index E; Lith%, percent of lithophilous species; IS%, percent of indicator species; El%, percent of *Esox lucius*; Rr%, percent of *Rutilus rutilus*; ES%, percent of eutrophic species; Sal%, percent of salmonid species; Pis%, percent of piscivores; RPis%, percent of relative piscivores; Inv%, percent of invertivores; RPisInv%, percent of relative piscivores and invertivores; Omn%, percent of omnivores; Omn-Rr%, percent of	[21, 22]

(continued)

Table 1 (continued)

Metric	Formulas	Description	References
		omnivores except *Rutilus rutilus*; DisHyb%, percent of diseased fish or fish hybrids; CPUE, abundance of fish expressed as a catch per unit effort; Alien%, percent of alien species; PDR, present diadromous species richness; HDR, historic diadromous species richness	
Lakes			
Phytoplankton Metric for Polish Lakes, PMPL	$PMPL_S = [Y_{Ch} + Y_{Bm} + Y_{CY}]/3$ $Y_{Ch2a,5a} = -3.2698 + 2.6081 \times \ln(Ch)$ $Y_{Ch3a,6a} = -1.8555 + 0.0369\ Ch + 1.3293 \times \ln(Ch)$ $Y_{Bm2a,5a} = 1.2900 \times \ln(TB) + 0.8727$ $Y_{Bm3a,6a} = 1.0325 \times \ln(TB) + 0.8135$ $Y_{CY2a,5a} = 1.4113 \times \ln[(B_{CY} + B_{CY} \times (B_{CY}/STB))/2] + 1.8112$ $Y_{CY3a,6a} = 1.0898 \times \ln[(B_{CY} + B_{CY} \times (B_{CY}/STB))/2] + 1.2835$ $PMPL_{NS} = [Y_{Ch} + Y_{Bm} + (0.5 \times Y_{CY})]/2.5$ $Y_{Ch2b,5b} = -1.1252 + 0.0649Ch + 0.6414 \times \ln(Ch)$ $Y_{Ch3b,6b} = -0.3334 + 0.2147Ch–0.0357X_{Ch} \times \ln(Ch)$ $Y_{Bm2b,5b} = 1.0720 \times \ln(TB) + 0.3778$ $Y_{Bm3b,6b} = 29{,}511 + 0.0541\ TB–28344/\sqrt{TB}$ $Y_{CYallb} = 1.1072 \times \ln B_{CY} + 1.0803$	$PMPL_S$, PMPL for stratified lakes; $PMPL_{NS}$, PMPL for nonstratified lakes; Y_{Ch}, metric "chlorophyll *a*"; Y_{Bm}, metric "total biomass"; Y_{CY}, metric "biomass of Cyanobacteria"; Ch, the average seasonal chlorophyll *a* content; TB, the average seasonal total biomass of phytoplankton; STB, total phytoplankton biomass in summer; B_{CY}, cyanobacteria biomass in summer	[23, 24]
Multimetric diatom index for lakes, IOJ	$IOJ = 0.6 \times Z_{TJ} + 0.4 \times GR_J$ $Z_{TJ} = 1–((TJ \times 0.1)$ $TJ = \sum(T_i \times wT_i \times L_i)/\sum wT_i \times L_i$; $GR = \sum tR_i$	TJ, trophic index calculated by a weighted formula of Zelinka and Marvan; tR_i, relative abundance of reference taxa according to the taxa list for soft-water or hard-water lakes	[25, 26]
Ecological Status Macrophyte Index, ESMI	$ESMI = 1 - \exp[-J \times Z \times \exp(N/P)]$ $Z = N/(P\text{-isob.}2.5)$	J, Pielou index of evenness; Z, colonisation index; *N*, total area of phytolittoral in ha or km^2; *P*, total lake area in ha or km^2	[27, 28]

Lake Macroinvertebrate Index, LMI	$LMI = EQR_{ASPT\_PL} + EQR_{Sh\text{-}W} + EQR_{Diptera\%} + EQR_{Trichoptera\%} + EQR_{EPT/Dipter}/5$ $EQR_{ASPT\_PL} = -3.597 + 0.815 \times x$ $EQR_{Sh\text{-}W} = -0.770 + 0.692 \times x$ $EQR_{Diptera\%} = 1.028–0.014 \times x$ $EQR_{Trichoptera\%} = -0.674 + 0.772 \times x$ $EQR_{EPT/Dipter} = -0.228 + 0.175 \times x$	Partial indices are converted to EQR ASPT (average score per taxa), the sum of points of the simplified BMWP_PL index divided by the total number of scored families found in the sampling site; Sh-W, Shannon-Weaver biodiversity index; EPT, the numbers of families from orders: Ephemeroptera, Plecoptera and Trichoptera; Diptera%, relative abundance of Diptera; Trichoptera%, relative abundance of Trichoptera	[29]
Ichthyofauna indices LFI+ and LFI-EN	$LFI = b0 + b1\ X1 + b2\ X2 + \ldots\ bn\ Xn$	X1 … Xn, the weight shares (%) in the catch of fish considered as metric X1, X2, Xn; b0 … bn, the regression coefficients for each weight share	[30, 31]

Table 2 Ecological classification including limit values according to the Regulation of the Minister of the Environment [3]

Metric[a]	Ecological status/potential					Type
	High/ maximum	Good	Moderate	Poor	Bad	
Rivers – natural water bodies						
IFPL	≥0.80	≥0.60	≥0.40	≥0.20	<0.20	19, 20, 21, 24, 25
IO	>0.75	≥0.55	≥0.35	≥0.15	<0.15	1, 2, 3
	>0.69	≥0.50	≥0.35	≥0.15	<0.15	4, 5, 8, 10
	>0.66	≥0.48	≥0.30	≥0.15	<0.15	6, 7, 9, 12, 14, 15
	>0.61	≥0.44	≥0.30	≥0.15	<0.15	19, 20, 24, 25
MIR	≥65.6	≥50.7	≥38.8	≥24.0	<24.0	1
	≥61.8	≥48.1	≥37.0	≥23.3	<23.3	1, 3, 4, 8, 11, 13
	≥55.4	≥42.0	≥31.4	≥18.0	<18.0	2, 7, 9, 12, 14
	≥48.3	≥37.7	≥27.0	≥16.4	<16.4	5, 6
	≥46.8	≥36.6	≥26.4	≥16.1	<16.1	16, 17, 19, 22, 25, 26
	≥47.1	≥36.8	≥26.5	≥16.2	<16.2	18, 20
	≥44.5	≥35.0	≥25.4	≥15.8	<15.8	23, 23, 25, 26
	≥44.7	≥36.5	≥28.1	≥20.0	<20.0	19, 20, 22
MMI_PL	≥0.674	≥0.614	≥0.409	≥0.205	<0.205	1, 2
	≥0.860	≥0.667	≥0.445	≥0.222	<0.222	3, 4, 5, 8, 10
	≥0.891	≥0.698	≥0.465	≥0.233	<0.233	6, 7, 9, 11, 12, 13, 14, 15
	≥0.908	≥0.716	≥0.477	≥0.239	<0.239	17
	≥0.903	≥0.717	≥0.478	≥0.239	<0.239	16, 18, 19, 20, 21, 22, 26
	≥0.893	≥0.687	≥0.458	≥0.229	<0.229	23, 24, 25
EFI + IBI_PL	≥0.911	≥0.755	≥0.503	≥0.252	<0.252	1–20, 22 (salmonid, index D applicable)
	≥0.939	≥0.655	≥0.437	≥0.218	<0.218	1–20, 22 (Cyprinid wadable, index D applicable)
	≥0.917	≥0.562	≥0.375	≥0.187	<0.187	1–20, 22 (Cyprinid boat, index D applicable)
	≥0.883	≥0.750	≥0.600	≥0.400	<0.400	21 (index D applicable)
	≥0.883	≥0.750	≥0.600	≥0.400	<0.400	23, 24, 25
Rivers – artificial and heavily modified water bodies						
IFPL	≥0.80	≥0.60	≥0.40	≥0.20	<0.20	0, 19, 20, 21, 24, 25
IO	>0.75	≥0.55	≥0.35	≥0.15	<0.15	1, 2, 3
	>0.69	≥0.50	≥0.35	≥0.15	<0.15	4, 5, 8, 10
	>0.66	≥0.48	≥0.30	≥0.15	<0.15	6, 7, 9, 12, 14, 15
	>0.61	≥0.44	≥0.30	≥0.15	<0.15	0, 16, 17, 18, 23, 24
	>0.54	≥0.39	≥0.30	≥0.15	<0.15	0, 19, 20, 24, 25
	>0.75	≥0.65	≥0.45	≥0.20	<0.20	0

(continued)

Table 2 (continued)

Metric[a]	Ecological status/potential High/ maximum	Good	Moderate	Poor	Bad	Type
MIR	≥61.8	≥48.1	≥37.0	≥23.3	<23.3	1, 3, 4, 8, 11, 13
	≥55.4	≥42.0	≥31.4	≥18.0	<18.0	2, 7, 9, 12, 14
	≥48.3	≥37.7	≥27.0	≥16.4	<16.4	5, 6
	≥46.8	≥36.6	≥26.4	≥16.1	<16.1	0, 16, 17, 19, 22, 25, 26
	≥47.1	≥36.8	≥26.5	≥16.2	<16.2	18, 20
	≥44.5	≥35.0	≥25.4	≥15.8	<15.8	23, 23, 25, 26
	≥44.7	≥36.5	≥28.1	≥20.0	<20.0	0, 19, 20, 22
MMI_PL	≥0.674	≥0.614	≥0.409	≥0.205	<0.205	1, 2
	≥0.860	≥0.667	≥0.445	≥0.222	<0.222	3, 4, 5, 8, 10
	≥0.891	≥0.698	≥0.465	≥0.233	<0.233	6, 7, 9, 11, 12, 13, 14, 15
	≥0.908	≥0.716	≥0.477	≥0.239	<0.239	17
	≥0.903	≥0.717	≥0.478	≥0.239	<0.239	16, 18, 19, 20, 21, 22, 26
	≥0.893	≥0.687	≥0.458	≥0.229	<0.229	23, 24, 25
MZB	>0.6	≥0.5	≥0.4	≥0.2	<0.2	–
EFI + IBI_PL	≥0.911	≥0.755	≥0.503	≥0.252	<0.252	1–20, 22 (salmonid, index D applicable)
	≥0.939	≥0.655	≥0.437	≥0.218	<0.218	1–20, 22 (Cyprinid wadable, index D applicable)
	≥0.917	≥0.562	≥0.375	≥0.187	<0.187	1–20, 22 (Cyprinid boat, index D applicable)
	≥0.883	≥0.750	≥0.600	≥0.400	<0.400	21 (index D applicable)
	≥0.883	≥0.750	≥0.600	≥0.400	<0.400	23, 24, 25
Lakes – all water bodies						
PMPL	≤1.00	≤2.00	≤3.00	≤4.00	>4.00	1a-7b
IOJ	>0.705	≥0.590	≥0.400	≥0.150	<0.150	1a-7b
ESMI	≥0.680	≥0.410	≥0.205	≥0.070	<0.070	1a-7b
LMI[b]	≥0.764	≥0.573	≥0.382	≥0.191	<0.191	1a-7b
LFI+	≥0.71	≥0.46	≥0.26	≥0.11	<0.11	1a-7b
LFI-EN	≥0.71	≥0.46	≥0.26	≥0.11	<0.11	1a-7b
LFI+ [c]	≥0.866	≥0.595	≥0.250	≥0.100	<0.100	1a-7b
LFI-EN[c]	≥0.804	≥0.557	≥0.250	≥0.100	<0.100	1a-7b

[a]Explanation of abbreviations was given in Table 1
[b]Proposed by Soszka and Koprowska [29]
[c]After intercalibration process according to Ritterbusch et al. [56]

Table 3 The latest monitoring system classifications (class boundary) in Poland as a result of the intercalibration process [12]

Geographical intercalibration group, GIG	Biological quality element, BQE	National metric[a]	Type[b]	Ecological quality ratios	
				High/ good	Good/ moderate
Central-Baltic rivers	Benthic invertebrate fauna	RIVECOmacro – MMI_PL	R-C1	0.910	0.720
	Macrophytes	MIR – macro-phyte index for rivers	R-C1 R-C3 R-C4	0.900 0.910 0.900	0.650 0.684 0.650
	Phytobenthos	IO	All types	0.800	0.580
Lowland-midland group (rivers)	Fish fauna	EFI+PL	All types	0.800	0.600
All – very large rivers	Benthic invertebrate fauna	RIVECOmacro – MMI_PL	R-L2	0.910	0.710
	Phytoplankton	IFPL	R-L2	1.080	0.920
Central-Baltic lakes	Phytoplankton	PMPL	L-CB1 L-CB2	0.800	0.600
	Macrophytes	ESMI	All types	0.680	0.410
	Fish fauna	LFI+ LFI-EN	All types	0.866 0.804	0.595 0.557
Cross-GIG phytobenthos	Phytobenthos	IOJ	HA	0.910	0.760

[a]Explanation of abbreviations was given in Table 1
[b]*River types*: R-C1, small lowland siliceous sandy river, catchment area 10–100 km^2, alkalinity >0.4 meq dm^{-3}; R-C3, small mid-altitude siliceous with rock and gravel, catchment area 10–100 km^2, alkalinity <0.4 meq dm^{-3}; R-C4, medium lowland mixed with sandy and gravel, catchment area 100–1000 km^2, alkalinity >0.4 meq dm^{-3}; R-C5, large lowland mixed, catchment area 1000–10,000 km^2, alkalinity >0.4 meq dm^{-3}; R-C6, small, lowland calcareous, with gravel, catchment area 10–300 km^2, alkalinity >2 meq dm^{-3}; R-L2, very large medium- to high-alkalinity rivers, catchment area of stretch >10,000 km^2, alkalinity >0.5 meq dm^{-3}. *Lake types*: L-CB1, lowland, shallow and calcareous, altitude <200 m a.s.l., mean depth 3–15 m, alkalinity >1 meq dm^{-3}, residence time 1–10 years; L-CB2, lowland, very shallow and calcareous, altitude <200 m a.s.l., mean depth < 3 m, alkalinity >1 meq dm^{-3}, residence time 0.1–1 year; L-CB3, lowland, shallow, small and siliceous, altitude <200 m a.s.l., mean depth 3–15 m, alkalinity 0.2–1 meq dm^{-3}, residence time 1–10 years; HA, high-alkalinity lakes, alkalinity >1 meq dm^{-3}, ecoregions: Alpine, Central-Baltic, Eastern Continental, Mediterranean

moderate status (0.580) for all river types used within Central-Baltic rivers GIG phytobenthos intercalibration exercise [12] (Table 3).

The integrated index – *FLORA* – is referred to as a class of joint assessment of IFPL and IO. It is applied in classification of the ecological potential of artificial and heavily modified water bodies. The limit values are determined according to IFPL and/or IO or from the arithmetic average of both indices.

Macrophyte Index for Rivers (MIR) The macrophyte-based method of river assessment was developed in 2006 and included in the national monitoring system on the basis of the macrophyte index for rivers [17, 18] (Tables 1 and 2). It is based on the qualitative and quantitative stocktaking of aquatic and rush vegetation occurring on a 100-m study section of the river. It takes into account the index values, the weight index and the coverage index for a species. It allows for an assessment of the ecological status as per the requirements of the Water Framework Directive. The limiting values for classes high/good and good/moderate for three river types, from small lowland and mid-altitude to medium lowland (R-C1, R-C3 and R-C4) determined during the pan-European intercalibration exercise, were given in Table 3. The statistically significant correlations were fund with TP ($r = -0.59$) [19].

Macroinvertebrates Index for Rivers (MMI_PL) and Macroinvertebrates Index for Dam Reservoirs (MZB) Under the Regulation of the Minister of Environment [3], two multimetric indices of the ecological status using the composition of macrozoobenthos are applied in the State Environment Monitoring: MMI_PL – for rivers – and MZB, for dam reservoirs. Due to a huge species diversity of invertebrates, as well as considerable methodological difficulties in their identification, the level of identification precision down to the family or species has been adopted in both indices. The same level has been adopted in determination of the biological diversity indices that comprise the MMI_PL index (Table 1). The methodological principles are given by Bis and Mikulec [20].

The method of assessment of the ecological status/potential which uses macrozoobenthic communities, MMI_PL, is based on the values of the multimetric index ICMi (Intercalibration Common Metrics index) [20] calculated as a combination of six partial indices given in Table 1. Specified class limits were adopted for each abiotic type of rivers [3] (Table 2). For macroinvertebrate-based ecological status assessment, all rivers were additionally included into six biocenotic types:

– Type I – Tatra mountain streams
– Type II – Sudety mountain streams and siliceous western highland rivers
– Type III – eastern highland rivers, calcareous and siliceous
– Type IV – small lowland rivers
– Type V – lowland rivers and estuarine rivers
– Type V – lowland rivers of the organic substrate and lowland rivers connecting the lakes

Due to much lower taxonomic diversity and smaller sizes of benthos populations observed in dam reservoirs compared with rivers, a simplified index – MZB – was used to assess the ecological potential. It is based on the BMWP scale with a Polish modification, BMWP_PL (Table 1), which is used to calculate the value of ASPT (average score per taxa). The class limits for the ecological potential of dam reservoirs based on the macrozoobenthos are presented in Table 2. Three types of dam reservoirs were identified in the course of work on the index: R, rheolimnic; P, transitional; and L, limnic, but – ultimately – identical class limits were adopted, regardless of its type [10]. The intercalibration of macroinvertebrate-based methods with other European methods was carried out successfully for small lowland rivers (biocenotic type IV, i.e. R-C1) and for large rivers (biocenotic type V, i.e. R-L2) [12] (Table 3).

Fish Indices for Rivers (EFI+PL and IBI_PL) According to the standard methodology adopted for monitoring of the ecological status of waters based on ichthyofauna [21, 22], studies of the composition of fish communities are conducted by electrofishing, in accordance with the CEN standard of CEN EN 14011 and PN-EN 14011. The results can be used to calculate the EFI+PL and IBI_PL indices applied in the State Environment Monitoring [3] (Table 1).

In order to calculate the EFI+PL or IBI_PL index, the electrofishing catch (the number and weight of fish of individual species, broken down to classes of total length of $\leq$150 mm and $>$150 mm) is entered into the EFI+IBI_PL computer programme. At the same time, data that characterise the method of fishing are entered (length of the site, fishing area, fishing method, etc.) as well as the parameters describing the site, collected during the study and field work (such as the ecoregion, abiotic type of the river, catchment area, distance from the source, depth and width of the river, presence of a flood plain, average air temperature: annual, in January and July). The literature data and available information (gained, e.g. from fishery managers) are used to supplement data on the occurrence of diadromous species, both in the past and at present, in order to calculate the diadromous index D. It is a supplement for fish-based assessment of the ecological status or potential for most abiotic types of rivers except for organic and inter-lake river types 23, 24 and 25. Subsequently, the programme calculates the values of EFI+PL or IBI_PL and gives the ecological status assessment or potential class, according to the class ranges for individual river types and fishing methods [3, 21], adopted in the SEM. Uniform class boundaries were adopted for the ecological status and potential of rivers (Table 2).

The EFI+PL index uses two metrics for each of the identified categories of rivers: with dominant salmonids (Ni.O2.Intol, Ni.Hab.Intol.150) and with dominant cyprinids (Ric.RH.Par, Ni.LITHO).

1. The metric Ni.O2.Intol – density of fish (the number of specimens per 100 m^2 of the fishing area) of species sensitive to oxygen deficit; there are 20 native species in the ichthyofauna of Poland.
2. The metric Ni.Hab.Intol.150 – density of fish in the class of the total length under 150 mm, intolerant of habitats degradation; it combines the features of a quantitative metric with information about the size and – indirectly – age structure of the ichthyofauna community; there are 25 native fish species in the ichthyofauna of Poland.
3. The metric Ric.RH.Par – number of fish species which require the lotic environment for reproduction among the number of fish species recorded in electrofishing; there are 31 native fish species in Poland.

4. The metric Ni.LITHO – lithophilic species which require coarse substrate for spawning: gravel, rock, pebbles and boulders of various fractions; the number of fish recorded in electrofishing is taken into account; it includes 31 native fish species.

Therefore, the index EFI+PL is sensitive to modifications of the riverine environment, particularly those that are associated with a change of the bottom substrate, especially in rivers with a gravel bottom (lithophilic species are included in the Ni. LITHO metric), changes in the flow regime and temperature of water (rheopar species in metric Ric.RH.Par) and species sensitive to oxygen deficit.

Ranges of classes of the EFI+PL index are diverse depending on the category of the river being assessed (salmonid or cyprinid) as well as in rivers with dominant cyprinids – depending on the method of fishing (wading or boat fishing). The class boundaries for salmonid rivers and wadable rivers are stricter due to higher effectiveness of fishing than in boat-fished rivers (Table 2).

For the group of abiotic rivers, to which the EFI+PL cannot be applied, the index of biotic integrity (IBI), in a version adapted to the conditions of Poland (IBI_PL), is applied. The index of biotic integrity (IBI) assesses fish communities through a system of 12 metrics grouped in 3 categories describing (1) species richness and composition, (2) trophic structure and (3) abundance and condition of fish community [32, 33]. Fish and lamprey species in rivers of Poland are classified into functional ecological groups. The qualitative metrics include the first four metrics mentioned in the group "Richness of fish assemblages and proportion of species". The metric "Total species richness" reflects the overall diversity of the ichthyofauna community; the next metrics, "number of species of the water column" and "number of benthic species", concern the ecological groups associated with habitat preferences, whereas the metric used to assess the environmental status of large lowland rivers covers the richness of species typical of large rivers. Another quality metric, used in the IBI_PL method for great lowland rivers with the gravel bottom (type 21b), is the Pielou evenness index E [34]. This metric reflects the proportionality of the ichthyofauna community and has lower values in cases of a significant dominance of any individual species.

The IBI_PL method uses a range of metrics based on quantitative proportions, i.e. a share of individuals of different indicator species or species typical of the river type. On the other hand, the metric based on a share of eutrophic species was applied to great lowland rivers.

All metrics of the IBI_PL index in categories "Proportions of trophic groups" and "Abundance and health of fish" are based on the proportions of the animals of individual species or their groups. The category "Proportions of trophic groups" concerns the trophic structure of the ichthyofauna community, with the increase in the IBI_PL index being caused by a greater proportion of predators, relative predators and species feeding on invertebrates. An increased share of omnivorous species conversely results in a decrease in the index value. The last metric category concerns the health status of a fish population, the proportion of alien species and the overall population size of the ichthyofauna community as referred to the catch per unit effort (CPUE). In this group, only an increase in the CPUE translates into a higher value of IBI_PL, while both an increase in the proportion of hybrids and individuals with anomalies and the proportion of alien species result in a decrease in the value of IBI_PL.

Index D is based on a comparison of the number of diadromous species noted in the river under examination in the past and at present. Both of these values are determined from the available sources, which include not only the catch but also literature data [35, 36]. It was assumed that the presence of at least half of the species noted in the river under examination in the past would not affect the result of an assessment by the EFI+PL or IBI_PL method, while for the sites with fewer than 50% of the diadromous species occurring in it in the past, an assessment result is adjusted with the basic index by decreasing the class of the ecological status/potential by one. It must be stressed that restoring the continuity of the migration routes of diadromous species in the catchment area of the river under examination will result in an increase in the result by one class in a fish-based re-examination of the water body. This effect will be observed assuming that the condition of local populations of fish assessed by the basic index does not deteriorate.

The Polish method of fish-based assessment of the ecological status of rivers was subjected to the procedure of auto-intercalibration [37]. Since the conditions of class compliance with the indices applied in other EU countries are met, the EFI+PL is regarded as intercalibrated [12] (Table 3), whereas the IBI_PL method requires collecting a larger dataset of data from the sites with a wider gradient of pressure [21]. No ichthyologic index of assessment has been developed for dam reservoirs. The work on this methodology is in the concept phase [38].

4.2 Integrated Ecological Status Assessment of Rivers: A Case Study

An assessment of the ecological status of rivers and the method of the assessment integration is presented with the research conducted in the catchment area of the Wel River as part of a Polish-Norwegian project "DeWELopment" lasting in 2008–2011 as an example (Fig. 4). This project aimed to develop and validate methods for integrated ecological status assessment of rivers and lakes to support the river basin management plans, and results were given in methodological guideline [39]. The results of an assessment of the ecological status of river water bodies indicate that their classification into an ecological status in accordance with the *one out all out* principle was decided mainly by the "ichthyofauna" element and the hydromorphological elements. A half of studied rivers met the WFD-required at least good ecological status.

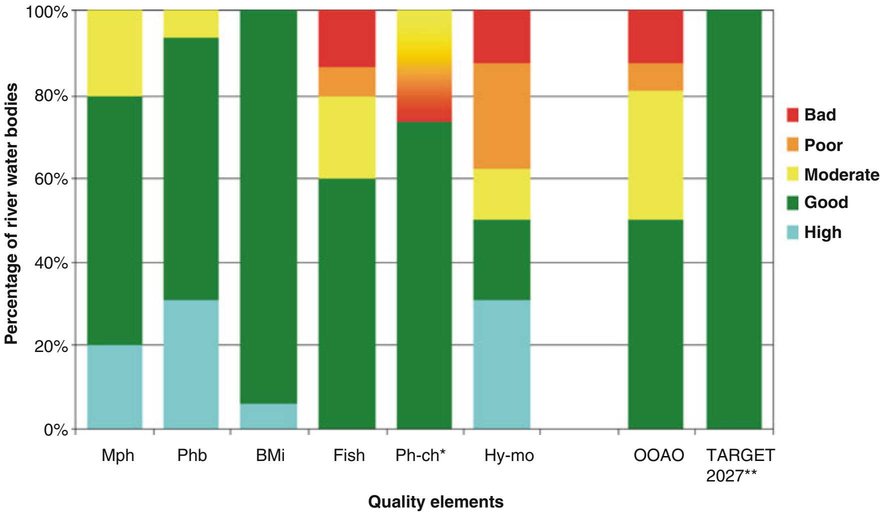

Fig. 4 Ecological status assessment of surface water bodies – rivers in the Wel River catchment (based on data included in Soszka (Ed.) [39], modified) according to quality elements: Mph, macrophytes; Phb, phytobenthos; BMi, benthic macroinvertebrates; Fish, fish; Ph-ch, physico-chemical; Hy-mo, hydromorphological; OOAO, *one out all out*; *classification: at least good and worse than good, **at least good

4.3 Lakes

Phytoplankton Metric for Polish Lakes (PMPL) The multimetric Phytoplankton Metric for Polish Lakes (PMPL) allows for an integrated assessment based on three partial indices: metric "Chlorophyll *a*", metric "Total biomass" and metric "Biomass of cyanobacteria". During the period 2009–2011, work was underway aimed at developing the PMPL and its verification based on data from the research conducted as part of the Polish-Norwegian project [9] and testing with other European methods, i.e. with the German and Hungarian ones [40]. The test results confirmed strong or moderate correlations between the PMPL and PSI ($r = 0.822$) and the Q index ($r = 0.650$) as well as between the main pressure parameters, i.e. TP ($r = -0.629$), TN ($r = -0.435$) and Secchi disc depth ($r = 0.695$). The PMPL was subjected to intercalibration for two lake types of the Central-Baltic Geographic Intercalibration Group (GIG): L-CB1 (lowland, shallow, calcareous) and L-CB2 (lowland, very shallow, calcareous). After a positive intercalibration of the Polish phytoplankton method, described in detail in the final report [23], the PMPL was approved as sufficient for lake ecological status assessment according to the WFD [41]. Ultimately, the valid methodology of the Polish intercalibrated phytoplankton method was described [24] (Tables 1, 2 and 3). In the testing of the partial metrics and PMPL response to eutrophication pressure (primarily TP and TN), the strongest correlations were found for nonstratified lakes. For example, in case of PMPL, significant correlation coefficients were with TP up to $r = -0.70$

and with TN up to $r = -0.58$ [24]. The class boundaries according to the values of PMPL are given as numbers from 0 to 5, after being converted to EQR (ecological quality ratio; $EQR_{PMPL} = -0.2 \times PMPL+1$) from 0 to 1.

Multimetric Diatom Index for Lakes (IOJ) The diatom index for lakes (IOJ) covers two partial modules, i.e. the trophy index (TJ) and the reference species module (GRJ), which are calculated separately for soft-water bodies, hard-water bodies and, jointly, all types of lakes [13] (Tables 1 and 2). The values of sensitivity of indicator diatoms to the trophic status of lakes were taken from the index developed for German lowland lakes T_{NORD} [16]. The index values were significantly correlated with the concentration of biogenic substances. In the Central-Baltic GIG phytobenthos intercalibration exercise, positive intercalibration result was obtained for lake type HA – high-alkalinity lakes, alkalinity >1 meq dm^{-3} (relationship with intercalibration metric IT_EQR $R^2 = 0.644$ and with $\log_{10}$ TP $R^2 = 0.133$ at significance level $p = <0.001$) [25, 26].

Ecological Status Macrophyte Index (ESMI) The macrophytic method of lakes assessment was initially proposed by Rejewski [42], on the basis of structural-spatial vegetation systems of the whole littoral, taking into account the main ecological relationships of macrophytes. This method was subsequently applied to develop the assessment index ESMI in accordance with the WFD requirements [27]. The index was then applied in routine monitoring in Poland [3] with positive result of the intercalibration process [28]. The ESMI method comprises partial metrics to evaluate primarily two aspects, the first of taxonomic composition, i.e. evenness index J, and the second of abundance, i.e. colonisation index Z [27] (Tables 1 and 2). The calculation of partial metrics is based on data on the total lake area, total phytolittoral area, number of the plant communities (including only association area of the minimum >1 m^2 and the cover >25% but not a single plant) identified in phytolittoral, percentage share of particular plant communities in the total phytolittoral area and area determined by the isobath 2.5 m with the minimum potential phytolittoral. This method includes identification of submerged and emergent macrophytes, i.e. hydrophytes (charophytes, potamids and mosses), floating-leaves nympheids, non-rooting limneids and helophytes (rush and sedge rush). In the testing of the partial metrics and ESMI response to the pressure gradient expressed primarily as total phosphorus, total nitrogen, chlorophyll *a* and water transparency expressed as Secchi disc depth, the significant correlations were found with water transparency (up to $r = 0.62$ and $r = 0.79$ in stratified and nonstratified lakes, respectively) and slightly weaker with phosphorus and nitrogen concentrations (with TP up to $r = -0.50$ and TN up to $r = -0.57$, in stratified and nonstratified lakes, respectively).

Lake Macroinvertebrate Index (LMI) According to the Water Framework Directive, an assessment of the ecological status of lakes based on benthic macroinvertebrates should take into account the metrics which describe the composition and abundance of fauna, its diversity and the occurrence of sensitive and tolerant taxa. The index for benthic macroinvertebrates in lakes is not taken into account at present because of the incomplete intercalibration process; the methods of such an assessment of the ecological status were not intercalibrated until the second round of the exercise. Various indices were tested in Poland, including the biotic index *MBI_CPET* (*Multimetrics Biotic Index Chironomid Pupal Exuvial Technique*) [43, 44]. The multimetric *MBI_CPET* covers two quantitative metrics: *CPET* and the proportion (%) of Chironomidae. The *CPET* index in Poland ranges from −0.08 to 0.30 and is strongly correlated with the water eutrophication indices, i.e. the concentration of chlorophyll *a* and the content of total phosphorus in water. Subsequently, a group of 25 metrics was tested. Ultimately, in Poland a multimetric index LMI was proposed, which is the arithmetic average of the component metrics: EQR_{ASPT_PL}, $EQR_{Sh\text{-}W}$, $EQR_{Diptera\%}$, $EQR_{Trichoptera\%}$ and $EQR_{EPT/Dipter}$ [29] (Tables 1 and 2). The correlations between the pressure factors and the multimetric LMI were moderate and statistically significant, i.e. with a cumulative pressure index, % of agricultural land in the catchment area ($r = -0.48$), TP ($r = -0.43$) and Chl *a* ($r = -0.71$).

Ichthyofauna Indices: Lake Fish Index (LFI+ and LFI-EN) Poland has two assessment methods of the ecological status/potential of lakes based on ichthyofauna: the LFI-EN method based on the results of one-off fishing with Nordic gillnets, as per EN 14757, and the LFI+ method, based on the outcome of multi-year commercial fishing [30].

Both Polish assessment methods of the status/potential of lakes (LFI+ and LFI-EN) were developed assuming that changes in the environment status translate directly into the composition and structure of ichthyofauna. The opposite is also true: the composition and structure of ichthyofauna are direct indicators of the environment status. The theoretical foundations of such assumptions are provided in a few papers [45–47] (http://www.gios.gov.pl/images/dokumenty/pms/monitoring_wod/PRZEWODNIK_DO_OCENY_STANU_EKOLOGICZNEGO_RZEK_MAKROBENTOS_2013.pdf). The patterns demonstrated in these papers, which involve mainly the succession of specific fish species and groups of species in the course of environmental changes, have been used to assess the transformations of lakes in Poland [48–54].

The choice of metrics useful both for determination of models of reference lakes and for assessment of lakes was based on an analysis of the matrix of variables correlation. The variables were the following: proportions (w/w) of species in the total catch with Nordic gillnets as per EN 14757 or proportions (w/w) of species or functional groups of fish [55] (LFI+) as well as variables which characterise the pressure on the lake environment: visibility of the Secchi disc, total phosphorus content and chlorophyll content, as well as the trophic state index (TSI) [31] calculated from these values TSI_{SD}, TSI_{TP} and TSI_{Chl}.

The mean shares of the index variables (metrics) from the reference lakes were used as the reference values in an assessment of lakes with data on fishing with a Nordic gillnet (LFI-EN), for which the pressure indices TSI were available, or with contemporary fishing data (average for the last 10 years in LFI+). The range of the index variables was limited by the 10th and 90th percentile. For those variables, separately for each type of lakes, a normalised partial score was calculated based on deviations from the reference values, and the total score was calculated from them which is at the same time the temporary Y in the regression analysis. After these scores were given a mathematical form and after it was made sure that the selected variables are statistically significant, formulas were obtained in the form of multiple regression equations, which were used to calculate LFI in different types of lakes.

The formulas contain independent variables (metrics), which fully correspond to the theoretical assumptions of the method. In the LFI-EN formula, in stratified lakes, they included percentage shares of bream, white bream, roach, bleak and ruffe, whose shares increased with increasing pressure indices, and the shares of tench, rudd and perch, whose shares decreased with increasing pressure indices. In nonstratified lakes, they included percentage shares of bream, white bream, roach, bleak, ruffe and zander, whose shares increased with increasing pressure indices, and the shares of rudd and perch, whose shares decreased with increasing pressure indices. In the LFI+ formula, in deep stratified lakes, they included the percentage shares of zander and bream P (S, M, N), whose shares increased considerably with increasing pressure indices, and the shares of pike, tench, perch and bream D in the total catch of bream, whose shares decreased with increasing pressure indices. In shallow stratified lakes, they included the percentage shares of zander, crucian and white bream, whose shares increased with increasing pressure indices, and the shares of pike, tench, perch, bream D and bream D in the total catch of bream and roach S, whose shares decreased with increasing pressure indices. In nonstratified lakes, they included the percentage shares of zander and crucian, whose shares increased with increasing pressure indices, and the shares of pike, tench, perch and roach S, whose shares decreased with increasing pressure indices.

Lakes are assessed by comparing the ichthyofauna of individual lakes to the ichthyofauna of undisturbed lakes, regarded as reference lakes. Therefore, a fish-based assessment of the ecological status/potential concerns the degree of disturbance of ichthyofauna in lakes rather than their trophic status.

The intercalibration exercise was completed in 2015. Both Polish assessment methods of the ecological status/potential of lakes based on ichthyofauna were intercalibrated [56] based on direct links of EQR scores (LFI) with pressure factors (TAPI) [57]. In the intercalibration, a division into two types of lakes (stratified and nonstratified) was used in both Polish methods. This division corresponds to the abiotic typology of lakes. The new classification of lakes in the LFI+ changed the metrics. Currently, in stratified lakes, those were percentage (w/w) shares of zander, crucian, bream P (bream S, M and N combined) and white bream, whose shares increased with increasing pressure indices, and the shares of tench, perch, bream D, and bream D in the total catch of bream and roach S, whose shares decreased with increasing pressure indices. The metrics were not changed in the LFI-EN method.

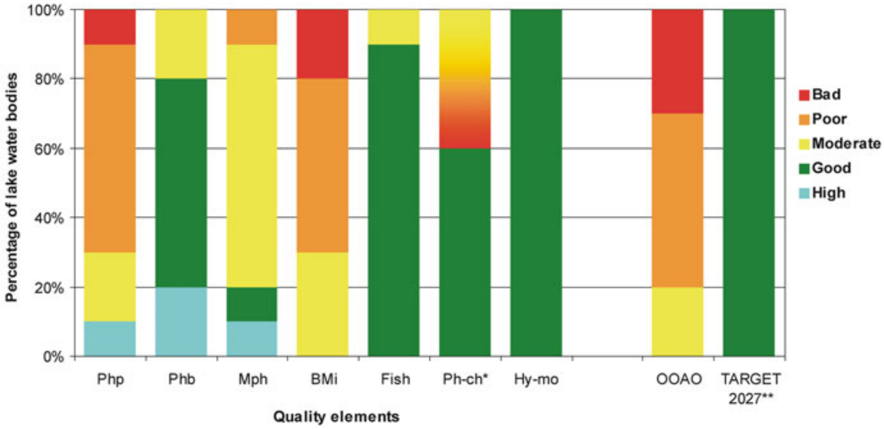

Fig. 5 Ecological status assessment of lake water bodies in the Wel River catchment (based on data included in Soszka (Ed.) [39]) according to quality elements: Php, phytoplankton; Phb, phytobenthos; Mph, macrophytes; BMi, benthic macroinvertebrates; Fish, fish; Ph-ch, physico-chemical; Hy-mo – hydromorphological; OOAO, *one out all out*; **classification: at least good and worse than good, **at least good

4.4 Integrated Ecological Status Assessment of Lakes: A Case Study

Similar to rivers, an assessment of the lake ecological status and the method of the assessment integration is presented also with the research conducted in the catchment area of the Wel River as part of a Polish-Norwegian project "DeWELopment" as an example (Fig. 5). The results of an assessment of the ecological status of lake water bodies indicate that their classification into an ecological status in accordance with the *one out all out* principle was decided primarily by phytoplankton and benthic macroinvertebrates. All studied lakes did not meet the WFD-required at least good ecological status.

5 General Overview

As per the implementation of the WFD [1], the water bodies under assessment should have reached at least good status of water by 2027. Based on the studies conducted in 2007–2013 as per the valid methodology, the indices described above have been calculated, and the ecological status/potential of water bodies in Poland has been assessed.

Pursuant to the Regulation of the Council of Ministers of 18 October 2016 on the plan of water management in the Vistula River catchment area [58], the water bodies in the Vistula River catchment area were assessed, including:

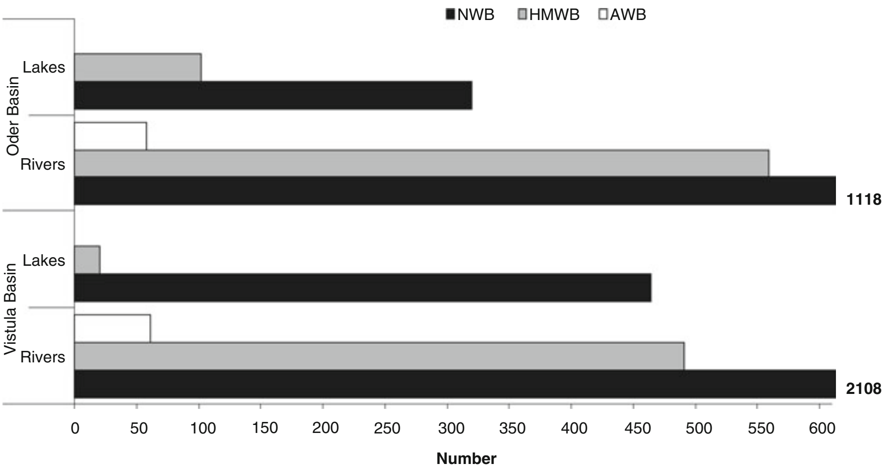

Fig. 6 River and lake water bodies in Vistula Basin and Oder Basin in Poland, AWB, artificial water body; HMWB, heavily modified water body; NWB, natural water body (data according to the Regulations of the Council of Ministers of 18 October 2016 [58, 59])

– River water bodies – 2,660 belonging to 24 types (0–2, 4–10, 12, 14–26)
– Lake water bodies – 484 belonging to 13 abiotic types (1a, 1b, 2a, 2b, 3a, 3b, 4, 5a, 5b, 6a, 6b, 7a, 7b)

Pursuant to the Regulation of the Council of Ministers of 18 October 2016 on the plan of water management in the Oder River catchment area [59], the following water bodies in the Oder River catchment area were assessed:

– River water bodies – 1,735 belonging to 21 types (0, 3–10, 12, 14, 16–25)
– Lake water bodies – 422 belonging to 7 abiotic types (1a, 1b, 2a, 2b, 3a, 3b, 4)

Considering the water body category, natural water bodies, heavily modified water bodies and artificial water bodies were identified. The largest portion of water bodies in the Oder catchment area were classified as heavily modified water bodies (Fig. 6), mainly because of alterations of their banks, flow disturbances and hydromorphological changes. On the other hand, natural water bodies dominate in the Vistula catchment area.

When analysing the risk of failure to achieve the environmental goals, i.e. at least good ecological status/potential in the Vistula catchment area, the results of lake monitoring in 2001 were used, whereas in the case of unstudied lakes, the pressure in the catchment area was analysed. The results for river water bodies were from 2010 to 2013. These results were calculated from data contained in both Regulations of the Council of Ministers [58, 59] and were presented on diagrams (Fig. 7a, b). Seventy-four percent of bodies of water in the Vistula River basin and 67% of bodies of water in the Oder River basin have been classified as threatened with a failure to achieve these objectives, i.e. good status because of the co-existence of several pressure factors. Moreover, the water bodies which

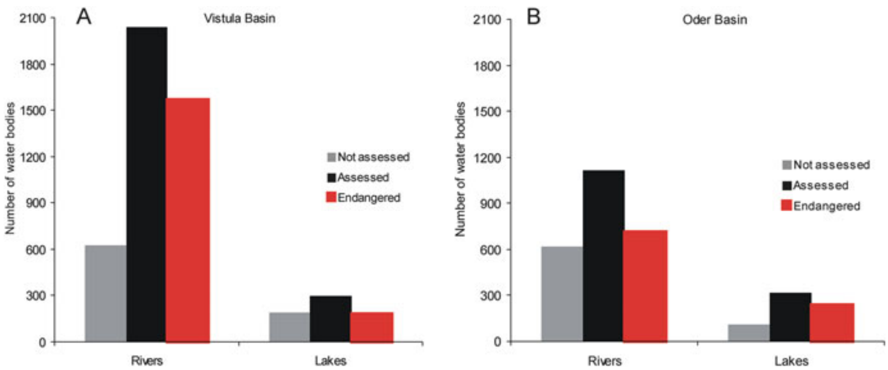

Fig. 7 Actual state of the river and lake water bodies examined including the risk of not achieving at least good ecological status/potential in Vistula Basin (**a**) and in Oder Basin (**b**) in Poland (data according to the Regulations of the Council of Ministers of 18 October 2016 [58, 59])

were not assessed were not taken into account. A minority of the water bodies in the Vistula catchment area were not assessed, whereas there are two times less non-assessed than assessed water bodies.

6 Conclusions

The Polish state monitoring system to assess the ecological status/potential includes biological elements as well as hydromorphological and physicochemical elements as those supporting biological elements. Recently, majority methods have been positively intercalibrated except for method based on benthic macroinvertebrates.

Based on studies conducted within a Polish-Norwegian project "DeWELopment", the final ecological classification according to the *one out all out* principle was definitely beneficial regarding rivers (a half water bodies with good ecological status) compared to assessment of lakes (all below good ecological status). The most rigorous partial assessments of an ecological status were primarily based on biological elements, i.e. ichthyofauna in rivers whereas phytoplankton and benthic macroinvertebrates in lakes. The latest ecological status/potential assessment of the river and lake water bodies in Poland is generally not very optimistic. Concerning the two largest catchment areas in Poland, 74% of water bodies in the Vistula River basin and 67% of water bodies in the Oder River basin have been indicated as not satisfying the WFD-required at least good ecological status.

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References

1. European Commission 2000 of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water Policy. OJEC L 327/1
2. Poikane S, Zampoukas N, Borja A, Davis SP, van de Bund W, Birk S (2014) Intercalibration of aquatic ecological assessment methods in the European Union: lessons learned and way forward. Environ Sci Pol 44:237–246. https://doi.org/10.1016/j.envsci.2014.08.006
3. Regulation of the Minister of the Environment of 21 July 2016 for the status classification of surface water bodies and environmental quality standards for priority substances. Official Journal of the Laws of 2016, item 1187
4. Kolada A, Soszka H, Cydzik D, Gołub M (2005) Abiotic typology of Polish lakes. Limnologica 35:145–150
5. Kolada A, Soszka H, Kutyła S, Pasztaleniec A (2017) The typology of Polish lakes after a decade of its use: a critical review and verification. Limnologica 67:20–26. https://doi.org/10.1016/j.limno.2017.09.003
6. Barszczyńska M, Borzuchowski J, Kubacka D, Piórkowski P, Rataj C, Walczykiewicz T, Woźniak T (2013) The hydrographic map of Poland at a scale of 1:10,000 – new thematic reference data for hydrography. Roczniki Geomatyki 9(3):15–26. (in Polish)
7. Moe J, Soszka H (2011) Integrated assessment of ecological status and estimating the risk of misclassification. In: Soszka H (ed) Ecological status assessment of the waters in the Wel River catchment. Guidelines for integrated assessment of ecological status of rivers and lakes to support river basin management plans. IFI, Olsztyn, pp 267–290. (in Polish)
8. Moe SJ, Lyche Solheim A, Soszka H, Gołub M, Hutorowicz A, Kolada A, Picińska-Fałtynowicz J, Białokoz W (2015) Integrated assessment of ecological status and misclassification of lakes: the role of uncertainty and index combination rules. Ecol Indic 48:605–615
9. Hutorowicz A, Napiórkowska-Krzebietke A, Pasztaleniec A, Hutorowicz J, Lyche Solheim A, Skjelbred B (2011) Phytoplankton. In: Soszka H (ed) Ecological status assessment of the waters in the Wel River catchment. Guidelines for integrated assessment of ecological status of rivers and lakes to support river basin management plans. IFI, Olsztyn, pp 143–168. (in Polish)
10. Kolada A, Ciecierska H, Ruszczyńska J, Mjelde M, Dziedzic J, Dynowski P, Jasnorzewski A (2011) Macrophytes. In: Soszka H (ed) Ecological status assessment of the waters in the Wel River catchment. Guidelines for integrated assessment of ecological status of rivers and lakes to support river basin management plans. IFI, Olsztyn, pp 169–185. (in Polish)
11. Picińska-Fałtynowicz J, Błachuta J (2012) Methodological guidelines for conducting phytoplankton research and assessing the ecological status of rivers on its basis. IM-GW, Warszawa. (in Polish)
12. Commission Decision (EU) 2018/229 of 12 February 2018 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise and repealing Commission Decision 2013/480/EU, Official Journal of the European Union L 47/1
13. Picińska-Fałtynowicz J, Błachuta J (2010) Methodological guidelines for conducting the ecological status assessment of rivers and lakes and ecological potential assessment of artificial and heavily modified flowing waters of Poland based on phytobenthos. GIOŚ, IMGW, O, Wrocław. (in Polish)
14. Rott E, Pfister P, van Dam H, Pipp E, Pall K, Binder N, Ortler K (1999) Indikationslisten für Aufwuchsalgen in österreichschen Flieβgewässern. Teil 2: Trophieidikation sowie geochemische Präferenz, taxonomische und toxikologische Anmerkungen. Bundesministerium für Land- und Forstwirtschaft. Wien. (in German)
15. Rott E, Hofmann G, Pall K, Pfiester P, Pipp E (1997) Indikationslisten für Aufwuchsalgen. Teil 1: Saprobielle Indikation. Bundesministerium für Land- und Forstwirtschaft, Wien. (in German)

16. Schaumburg J, Schranz C, Stelzer D, Hofmann G (2007) Action instruction for the ecological evaluation of lakes for implementation of the EU water framework directive: macrophytes and phytobenthos. Bavarian Water Management Agency, München
17. Szoszkiewicz K, Zbierska J, Jusik S, Zgola T (2010) Macrophyte method for river assessment – manual for the assessment and classification of ecological status of running waters based on water plants. Bogucki Wydawnictwo Naukowe, Poznan. (in Polish)
18. Szoszkiewicz K, Jusik S, Lawniczak AE, Zgola T (2010) Macrophyte development in unimpacted lowland rivers in Poland. Hydrobiologia 656:117–131. https://doi.org/10.1007/s10750-010-0439-9
19. Szoszkiewicz K, Ciecierska H, Kolada A, Schneider SC, Szwabińska M, Ruszczyńska J (2014) Parameters structuring macrophyte communities in rivers and lakes – results from a case study in North-Central Poland. Knowl Manag Aquat Ecosyst 415:08. https://doi.org/10.1051/kmae/2014034
20. Bis B, Mikulec A (eds) (2013) Guidelines to the ecological status assessment of rivers based on benthic macroinvertebrates. Biblioteka Monitoringu Środowiska, Warszawa. (in Polish)
21. Prus P, Wiśniewolski W, Adamczyk M (eds) (2016) Methodological guidelines for the Ichthyofauna monitoring in rivers. Biblioteka Monitoringu Środowiska, Warszawa. (in Polish)
22. Adamczyk M, Prus P, Buras P, Wiśniewolski W, Ligięza J, Szlakowski J, Borzęcka I, Parasiewicz P (2017) Development of a new tool for fish-based river ecological status assessment in Poland (EFI + IBI_Pl). Acta Ichthyol Piscat 47(2):173–184. https://doi.org/10.3750/AIEP/02001
23. Phillips G, Free G, Karottki I, Laplace-Treyture C, Maileht K, Mischke U, Ott I, Pasztaleniec A, Portielje R, Søndergaard M, Trodd W, van Wichelen J, Poikane S (2014) Water framework directive intercalibration technical report: Central Baltic Lake Phytoplankton ecological assessment methods. Office of the European Union, Luxemburg. https://doi.org/10.2788/73991
24. Hutorowicz A, Pasztaleniec A (2014) Phytoplankton metric of ecological status assessment for Polish lakes and its performance along nutrient gradients. Pol J Ecol 62:525–542
25. Kelly M Ács É, Bertrin V, Bennion H, Borics G, Burgess A, Denys L, Ecke F, Kahlert M, Karjalainen SM et al (2014) Water Framework Directive intercalibration technical report: phytobenthos ecological assessment methods. EUR 26512 EN. Office of the European Union, Luxembourg
26. Kelly M, Urbanic G, Acs E, Bertrin V, Burgess A, Denys L, Gottschalk S, Kahlert M, Karjalainen SM, Kennedy B et al (2014) Comparing aspirations: intercalibration of ecological status concepts across European lakes using littoral diatoms. Hydrobiologia 734:125–141
27. Ciecierska H, Kolada A (2014) ESMI: a macrophyte index for assessing the ecological status of lakes. Environ Monit Assess 186:5501–5517. https://doi.org/10.1007/s10661-014-3799-1
28. Portielje R, Bertrin V, Denys L, Grinberga L, Karottki I, Kolada A, Krasovskienė J, Leiputé G, Maemets H, Ott I, Phillips G, Pot R, Schaumburg J, Schranz C, Soszka H, Steller D, Søndergaard M, Willby N, Poikane S (2014) Water Framework Directive intercalibration technical report: Central Baltic Lake Macrophyte ecological assessment methods. Report EUR 26514 EN. Office of the European Union, Luxembourg. https://doi.org/10.2788/75925
29. Soszka H, Koprowska K (2016) Guidelines for the assessment of lakes based on invertebrate macrofauna. www.gios.gov.pl/images/dokumenty/pms/monitoring_wod/wytyczne_LMI.pdf
30. Chybowski Ł, Białokoz W, Wołos A, Draszkiewicz-Mioduszewska H, Szlakowski J (2016) Methodological guidelines for ichthyofauna monitoring in lakes. Biblioteka Monitoringu Środowiska, Warszawa. (in Polish)
31. Carlson RE (1977) A trophic state index for lakes. Limn Oceanogr 22:361–369
32. Karr JR (1981) Assessment of biotic integrity using fish communities. Fisheries 6:21–27
33. Karr JR, Fausch KD, Angermeier PL, Yant PR, Schlosser IJ (1986) Assessing biological integrity in running waters: a method and its rationale. Illinois National History Survey Special Publication 5, Urbana
34. Głowaciński Z (1996) Species diversity – its interpretation and calculation. In: Andrzejewski R, Wiśniewski RJ (eds) Biodiversity, concepts, evaluation, protection and development issues.

Instytut Ekologii PAN “Człowiek i Środowisko” Oficyna Wydawnicza, Dziekanów Leśny, pp 57–70. (in Polish)
35. Wiśniewolski W, Engel J (eds) (2006) Restoring migratory fish and connectivity of rivers in Poland. Wyd IRS, Olsztyn
36. Błachuta J, Rosa J, Wiśniewolski W, Zgrabczyński J, Bartel R, Białokoz W, Borzęcka I, Chybowski Ł, Depowski R, Dębowski P, Domagała J, Drożdżyński K, Hausa P, Kukuła K, Kubacka D, Kulesza K, Ligięza J, Ludwiczak M, Pawłowski M, Picińska-Fałtynowicz J, Lisiński K, Witkowski A, Zgrabczyński D, Zgrabczyńska M (2010) Assessment of the needs and priorities of improving the morphological continuity of rivers in the context of achieving the good status and potential of water bodies in Poland. Krajowy Zarząd Gospodarki Wodnej, Warszawa. (in Polish)
37. European Union (2015) Guidance Document No. 30. Autointercalibration guidance document 2015. Procedure to fi t new or updated classification methods to the results of a completed intercalibration exercise. Technical report 2015-085
38. Wiśniewolski W, Prus P (2009) Preliminary criteria for ecological status assessment index for dam reservoirs on the basis of the ichthyofauna assemblages. Ann Warsaw Univ Life Sci SGGW Land Reclamation 41(2):131–142
39. Soszka H (ed) (2011) Ecological status assessment of the waters in the Wel River catchment. Guidelines for integrated assessment of ecological status of rivers and lakes to support river basin management plans. IFI, Olsztyn, p 290. (in Polish)
40. Napiórkowska-Krzebietke A, Pasztaleniec A, Hutorowicz A (2012) Phytoplankton metrics response to the increasing phosphorus and nitrogen gradient in shallow lakes. J Elem 17 (2):289–303. https://doi.org/10.5601/jelem.2012.17.2.11
41. Commission Decision of 20 September 2013 establishing, pursuant to Directive 2000/60/ EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise and repealing Decision 2008/915/EC, OJEC
42. Rejewski M (1981) Vegetation of the lakes of the Laski region in the Tucholskie Forest. Rozprawy UMK, Toruń. (in Polish)
43. Golub M (2010) An assessment of ecological status of lakes based on benthic macroinvertebrates in accordance with the requirements of the water framework directive – overview of methodological solutions in Europe. Ochrona środowiska i zasobów naturalnych 45:30–45
44. Golub M, Prus P, Koszałka J (2011) Macrozoobenthos. In: Soszka H (ed) Ecological status assessment of the waters in the Wel River catchment. Guidelines for integrated assessment of ecological status of rivers and lakes to support river basin management plans. IFI, Olsztyn, pp 203–215. (in Polish)
45. Colby PJ, Spangler GR, Hurley DA, McCombie AM (1972) Effects of eutrophication on salmonid communities in oligotrophic lakes. J Fish Res Bd Can 29:975–983
46. Hartmann J (1977) Fischereiliche Veranderungen in kulturbedingt eutrophierenden Seen. Schweitz Z Hydrol 39(2):243–254
47. Hartmann J (1979) Unterschiedliche Adaptionsfahigkeit der Fosche an Eutrophierung. Schweitz Z Hydrol 41(2):374–382
48. Leach JH, Johnson MG, Kelso JR, Hartmann J, Numann W, Entz B (1977) Responses of percid fishes and their habitats to eutrophication. J Fish Res Bd Can 34:1964–1971
49. Bnińska M (1985) The possibilities of improving catchable fish stocks in lakes undergoing eutrophication. J Fish Biol 27(Suppl A):253–261
50. Bnińska M (1991) Fisheries. Cyprinid fishes. Systematics, biology and exploitation. Fish and Fisheries Series 3. Chapman & Hall, London, pp 572–589
51. Bnińska M, Wołos A (1998) Effectiveness of coregonid management versus environment quality. Arch Pol Fish 6(2):295–314
52. Leopold M, Bnińska M, Nowak W (1986) Commercial fish catches as an index of lake eutrophication. Arch Hydrobiol 106(4):513–524

53. Mickiewicz M, Wołos A, Leopold M (2003) Effectiveness of fisheries management in eutrophic lakes near Mrągowo (North-Eastern Poland). Arch Pol Fish 11(1):123–139
54. Wołos A, Zdanowski B, Wierzchowska M (2009) Long-term changes in commercial fish catches in Lake Mamry Północne (North-Eastern Poland) on the background of physical, chemical, and biological data. Arch Pol Fish 17:195–210
55. EFI+ Manual (2009) Manual for the application of the new European Fish Index (EFI+) with annexes. http://efi-plus.boku.ac.at/software/documentation.php
56. Ritterbusch D, Argillier C, Arle J, Białokoz W, Birzaks J, Blabolil P, Breine J, Draszkiewicz-Mioduszewska H, Jaarsma N, Karotki I, Krause T, Kubečka J, Lauridsen T, Logez M, Maire A, Palm A, Peirson G, Říha M, Szlakowski J, Virbickas T, Poikane S (2017) Water Framework Directive intercalibration: Central-Baltic Lake Fish fauna ecological assessment methods. Part A: descriptions of fish-based lake assessment methods. EUR 28022 EN. https://doi.org/10.2791/084375
57. Poikane S, Ritterbusch D, Argillier C, Białokoz W, Blabolil P, Breine J, Jaarsma NG, Krause T, Kubečka J, Lauridsenj TL, Nõges P, Peirson G, Virbickas T (2017) Response of fish communities to multiple pressures: development of a total anthropogenic pressure intensity index. Sci Total Environ 586:502–511
58. Regulation of the Council of Ministers of 18 October 2016 on the Plan of water management in the Vistula river catchment area. Official Journal of the Laws of 2016, item 1911. (in Polish)
59. Regulation of the Council of Ministers of 18 October 2016 on the Plan of water management in the Oder river catchment area. Official Journal of the Laws of 2016, item 1967. (in Polish)

Selected Aspects of Lake Restorations in Poland

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Contents

Abstract The work presents the problems of revitalization of water reservoirs in Poland. Lakes are subject to the inevitable process of aging, but as a result of anthropopression, the processes of natural eutrophication have been repeatedly accelerated. The consequence of the supply of excess allochtonic matter to water reservoirs, with the passive character of these landscape elements, is their degradation manifested by a sudden deterioration of water quality. A number of methods for the renovation of natural water reservoirs have been developed. Methods involving the removal of excess nutrients outside the ecosystem are the removal of hypolimnion waters, removal of bottom sediments, flushing the lake bowl, and removal of seston, macrophytes, and fish. The second group of methods consists in reducing the

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amount of nutrients in the matter circulation in the lake. This includes artificial aeration of lakes, inactivation of phosphorus compounds, treatment of bottom sediments, or biomanipulation. Despite such diversification of existing ways of affecting the aquatic environment, both in Poland and in the world, it is rarely possible to obtain really good effects of the restoration measures undertaken. The most common reason for defeats is the failure to take into account the basic assumption of eliminating excess loads of pollutants reaching the lake from the catchment before commencing its reclamation. Poor effects are also caused by insufficient understanding of the functioning of the complicated lake ecosystem and incorrect selection of revitalization techniques according to the environmental conditions of lakes. Properly conducted reclamation requires collecting research results about the lake and its catchment, recognizing the conditions of the eutrophication process, impartial selection of methods, securing funds for all planned operations, and substantive supervision and monitoring which provides the basis for assessing effectiveness and possible adjustments in the project implementation.

Keywords Bottom sediments · Eutrophication · Phosphorus · Restoration of lakes · Water quality

1 Introduction

> In the scale of time measured by human age, lakes seem to be permanent elements of the landscape, but on the geological scale they are temporary elements, usually resulting from disasters, later to quietly and imperceptibly mature and disappear.

That is how G.E. Hutchinson, precursor of modern limnology, wrote about lakes in 1957 [1]. In fact, today, after more than 60 years of research and observation, we confirm that lakes are elements that are particularly vulnerable to changes in the surrounding environment. By taking the lowest place in the area, they are the receivers of all kinds of pollutants introduced from the catchment area. Initially, this results in a moderate increase in trophies, but if the load of nutrients is overloaded, the eutrophication process accelerates rapidly, and, as a consequence, the waterbody completely degrades. It is extremely difficult to restore the condition from before the pollution period.

The restoration of lakes seems to be a more spectacular treatment, allowing for quick improvement of water quality, rather than tedious, long-term limitation of the nutrient supply from the catchment [2]. However, a reclamation project will not result in permanent renewal of the reservoir if the reasons for its degradation are not removed earlier. In preparing the program for the repair of the eutrophicated lake, it is first necessary to determine the current level of supply of organic and biogenic matter from outside the ecosystem, determine the size of the catchment, identify in detail the method of its management and hydrological conditions, inventorize all sources of pollution (point, dispersed, spatial, linear, atmospheric), and, finally, calculate the volume of external loading coming from these sources, specifying the possibilities of their acceptance by the lake.

The priority, and at the same time a relatively easy technical method of limiting the pollutant load, is the elimination of point sources. The optimal solution is to stop the flow of sewage, even purified sewage to the lake and direct it to the river system, which is able to absorb much larger amounts of nutrients without harming the ecosystem. Another way may be the three-stage (or higher) wastewater treatment with in-depth phosphorus removal (chemical precipitation) or both basic nutrients – phosphorus and nitrogen – by additional biological methods. An economically justified solution may also be the construction of a wastewater sewage system, intercepting the inflow of sewage from the surrounding reservoirs and recreational facilities and redirecting them to a joint treatment plant, already below the lake. This system, although expensive, has been successfully used on many European lakes and is already being introduced in Poland.

Dispersed pollutants are sewage originating from places without sanitary sewage, getting into waters through soil, groundwater, and drainage ditches due to lack or improper use of individual septic tanks. Liquidation of these sources of pollution primarily involves the construction of a sanitary sewage system parallel to water supply. It may also involve the construction and proper operation of household, local and collective sewage treatment plants, or at least provide the possibility of removing sewage from septic tanks to treatment plants.

Spatial sources of pollution in our agricultural landscape are often the main cause of the growth of trophies of lakes. Both their quantification and emission limitations are a serious challenge for water management. These pollutants travel to lakes and rivers together with surface runoff and groundwater, with an intensity depending on physiographic, geological, and hydrological conditions. Limiting their amounts requires rational farming – compliance with specific agrotechnics, adapted to the physiographic conditions of the area and the use of natural biogeo-chemical barriers (ponds, mid-field and mid-forest lakes, drains, shrubs, and shelters), appropriate structure of agricultural land, proper use of mineral fertilizers, and the elimination of large farms in direct catchments of lakes. It is worth bearing in mind the provisions of the Code of Good Agricultural Practice [3]. Introduction of methods and principles of counteracting area pollution does not have to, and should not, mean reducing the effectiveness of agricultural production, but favorable adaptation of the agricultural economy to natural conditions and the restoration of the cultural landscape [4, 5]. Another aspect is the proper development of the outskirts of water reservoirs. These areas should not be excessively urbanized or intended for recreational development. Due to the attractiveness of areas around lakes and the pressure of landowners, these demands are often not respected.

For many water reservoirs, an important source of pollution, and sometimes even decisive, are loads introduced with the waters of the tributaries, i.e., from the indirect catchment. To prevent these unfavorable phenomena, it is proposed to build preliminary buffer reservoirs, wetlands, or river water treatment plants. However, in this case, it is the most justified to eliminate the causes and not the effects and thus to limit the migration of nutritious substances valuable for agriculture to the river network supplying the lakes.

2 Hydrological, Physiographic, and Climate Conditions of the Eutrophication of Polish Lakes

Poland is a country with unevenly distributed surface water resources. The lakeland area primarily occupies the northern, lowland part of the country, which is associated with the postglacial origin. A small percentage of natural lakes (4.1%) are located in southern and eastern Poland – in mountain ranges and karst areas. The average lake presence indicator, expressed as the ratio of lakes area to total land area, is only 0.9%, but in lake districts, it reaches 3–5% [6] and even 24% (Great Masurian Lakes). Nevertheless, small and relatively shallow lakes dominate in terms of quantity. Statistical studies [7] show that out of 7,081 lakes in Poland, as many as 44% are in the range of 1–5 ha, and only 14% of lakes exceed 50 ha. Unfortunately, there is a progressive disappearance of lakes [8], especially the smallest ones, and one of the reasons is eutrophication.

Most lakes in Poland (58.3%) are fed by surface inflows. On the one hand, this facilitates the exchange of water and the removal of excess pollution, but above all it causes their easy transfer from the basin. Poland is an agricultural country, so for most lakes spatial sources of pollution are a permanent factor stimulating eutrophication.

Along with the development of water and sewage infrastructure and industrialization of the country, from the middle of the last century, the lakes underwent very strong anthropopression, commonly constituting of sewage receivers. The reversal of this situation has slowly occurred since the 1990s, along with political changes. A thorough ordering of water and sewage management along with the modernization of infrastructure (National Sewage Treatment Program) began in 2003 and continues to this day. It has clearly contributed to the improvement of rivers and partly lakes. However, some lakes in Poland are still influenced by point sources due to negligence in sewage management.

Poland lies in the temperate climate zone, which determines the dimictic type of water circulation in surface reservoirs in our country. The cycle of the lake year consists of spring circulation, summer stagnation, autumn circulation, and winter stagnation under the ice cover. In the case of the shallowest lakes, multiple water mixing is observed throughout the entire warm season (polymictic lakes). Due to the occurrence of water stagnation periods, lakes in our country commonly suffer from oxygen deficits in the near-bottom waters, caused by the lack of the possibility of replenishing oxygen in water with simultaneous significant internal charge of organic matter. Under these conditions, the phenomenon of internal loading from bottom sediments with biogenic matter (mainly phosphorous) occurs [9]. It is a process that allows the lake ecosystem to efficiently maintain high trophy regardless of external factors.

All of the above conditions cause that the problem of lake eutrophication is common in Poland. Of the lakes studied in recent years (2010–2015), only 36% of water is in good or very good ecological condition [10]. It should be emphasized that the data provided concern lakes covered by state monitoring, taking into account only lakes with an area exceeding 50 ha, while the general regularity is the higher rate of degradation of the smallest lakes. It can therefore be concluded with a high probability that at least 2/3 of the lakes in Poland require external intervention. Unfortunately, a lack of reaction to the implementation of protective measures is a common phenomenon. The reason for this scenario is the internal self-nutrition mentioned above. In such cases, only reclamation can restore their proper ecological balance.

3 Formal and Legal Conditions of Restoration Work Performing

Lake restoration, as a relatively modern field of environmental interventions, until recently was not included in the Polish legislation related to environmental protection and water management. This caused a number of difficulties in the way of formalizing reclamation works and controlling them.

Polish legislation stipulates that all flowing waters, including hydrologically open lakes (70.4% of all lakes in Poland [8]), are owned by the State Treasury. In the case of these lakes, the voivodeship marshal administered them and very often handed them over to local municipalities. At the same time, these local governments were also owners of seepage lakes located on their own land. In this way, the most frequent actual hosts of lakes in Poland were the municipalities. A small percentage of seepage lakes are private property or business entities such as industrial plants, mines, or enterprises. The problem of poorly selected or improperly conducted restoration of lakes often resulted from the lack of competence of the ordering entities to assess the correctness of the reclamation program adopted for implementation. This was particularly true of restoration offered by young and expansive private companies, lacking sufficient experience, scientific support, or knowledge about the environment.

In the years 1974–2015, Polish Water Law provided for the obtaining of a water permit for actions related to lake reclamation only in the case of "introducing substances that inhibit the growth of algae into surface waters" [11, 12]. This record historically referred to actions using poisons such as copper sulfate or dedicated biocidal preparations used to fight blooms in the 1960s–1980s. Nowadays, especially after 2000, such activities are not used. However, this provision allowed at least for the procedure of issuing a water permit for the use of preparations to inactivate phosphorus – indirectly affecting the reduction of the rate of phytoplankton development. Other methods had no reference in these laws.

In 2015, the amendment to the Water Law Act introduced the obligation to obtain a water permit for "re-cultivation of surface or underground waters," but without specifying exactly what the activities are.

Since 2018, there is a completely new Water Law in Poland [13], which has considerably expanded the process of obtaining water-legal consent. Current procedures in case of any actions affecting the achievement of environmental goals (and thus bringing lakes to a good ecological status through reclamation) provide for a two-stage process of obtaining a permit. The main permit is preceded by the issuance of the "Water-legal assessment" (paragraph 425/1), which is a substantive, in-depth expert analysis of the correctness of project assumptions. In our opinion, this is a very favorable provision, giving the opportunity to improve the effectiveness of the work undertaken for the revitalization of lakes in Poland.

Another underdeveloped element was the frequent disagreement between the lake administrator and the entity managing the fisheries (Fishing User). Especially before the year 2000, the fishing economy on Polish lakes was focused on the commercial harvesting of fish for consumption purposes. The species contributing to the eutrophication of the aquatic ecosystem such as carp, bream, or silver carp were strongly preferred. At the same time, such lakes were reclaimed – of course with a correspondingly smaller effect. Currently, however, ecological awareness of Fishing Users is growing significantly, and they are interested in improving the quality of the aquatic environment, among others, due to the need to attract lakes of new groups of their clients, i.e., for anglers and tourists. The result is more and more frequent inclusion of Fishing Users in the process of lake restoration, through pro-ecological stocking with environmentally valuable fish, predominantly predators.

4 Restoration Methods for Lakes Applied in Poland

Reclamation (renewal) of lakes is aimed at withdrawing, stopping, or even just releasing lake eutrophication processes [2]. In contrast to protective methods, reclamation methods are applied within the lake catchment.

Spontaneous de-eutrophication caused by protective procedures can occur only in those cases where the lake is not heavily contaminated, and the reduction of external loading of phosphorus exceeds 90% [14, 15]. In other cases, the improvement of the trophic state is small, is often temporary, or does not occur at all, and the only remedy for the lake is a properly constructed restoration program. However, it does not change the fact that reclamation treatments must be preceded by protective measures, and all attempts to renovate lakes without protecting them by reducing external loads of biogenic compounds and pollutants (especially sewage) mean the inevitable failure of reclamation and, in fact, unjustified spending of social funds [2].

Reclamation methods can be divided into two groups:

1. Methods involving the removal of excess nutrients from the lake ecosystem – the removal of hypolimnion waters, removal of bottom sediments, irrigation of lakes, and removal of seston, macrophyte, and fish
2. Methods of reducing the amount of nutrients in the maternal circulation in the lake – artificial aeration of lakes, inactivation of phosphorus compounds, treatment of bottom sediments, and biomanipulation.

4.1 *Hypolimnetic Withdrawal (Selective Water Removal)*

As a result of the progressive eutrophication of deep, thermally deposited lakes each year during the period of stagnation of water, the accumulation of excess biogenic substances and organic matter in the near-bottom waters occurs. When the water circulates, the pool of these pollutants feeds surface waters and stimulates primary production in the lake. The principle of the presented method consists in the selective removal of the most fertile hypolimnion layer of water beyond the waterbody – instead of the natural outflow of surface waters.

The first mention of the possibility of rescuing degraded lakes through repeated pumping of deoxygenated, loaded with biogenic elements of the hypolimnion water, was by Thomas [16], and the wider application of the method began in the second half of the twentieth century. Poland is where these solutions were first applied – on Lake Kortowskie (area, 90 ha; depth, max. 17 m) in Olsztyn [17]. In 1956, on the initiative of Prof. Przemysław Olszewski, the natural outflow from this lake was blocked, which allowed for water accumulation by about 0.5 m and independent outflow hypertrophic waters of hypolimnion through a pipeline laid on bottom sediments. It was the first implementation of the reclamation method on a technical scale in the world (Fig. 1).

Restoration by selective water removal has been applied to several dozen lakes in North America and Europe, including four lakes in Poland (Kortowskie, Pławniowice, Święte, Maróz). Bottom waters can be removed using different methods, but the most frequent method is the discharge of water to the outflow using the Olszewski method, i.e., using the siphon principle. Such a technique almost eliminates operating costs, which is a huge advantage in many years of repair programs for lake ecosystems. An important aspect of the pipeline's operation is to shorten the anaerobic period in the hypolimnion – as a result of a decrease in the thermal stability of water in the summer and accelerating the autumn circulation.

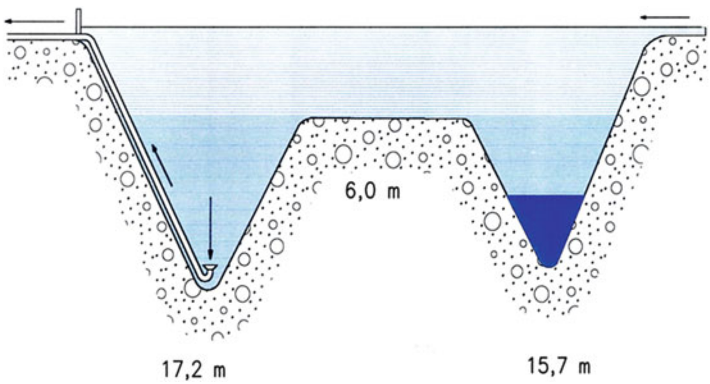

Fig. 1 Diagram of the Olszewski pipeline operation (Kortowskie Lake), according to [18]

The positive effects of this method consist in successive reduction of the nutrient pool and organic matter accumulated in the lake's basin [19, 20]. With natural surface drainage, much fewer of these compounds would be discharged than would be due to the forced outflow of the bottom waters. However, the effectiveness of the system is directly related to hydrological conditions – the ability to maintain the pipeline's operation. An insufficient amount of inflowing water limits the amount of damming up the lake and thus limits or even interrupts the outflow of water through the pipeline.

Conditions of Applying the Method

- Water balance in the lake – the method can be used only in flow-through reservoirs with a significant water exchange during the year, giving the possibility to remove at least once a season the entire volume of hypolimnion.
- Thermal stratification related to the depth of the lake – the method should not be used in polymictic lakes that mix in the summer.
- Morphometry of lake basin – the pipeline must absolutely reach the site with the greatest depth. This is a condition for the effectiveness of the method. Therefore, the location of deep water at a large distance from the outflow, although it does not cross the method, will cause technical difficulties and increase investment costs.
- The method should not be used when the next water reservoir is located on the water flowing from the lake nearby. Strongly eutrophied water drained by a pipeline would then form an additional source of pollution.
- The gradual elimination of internal loading of nutrients indicates the need to conduct reclamation activities for many years.

4.2 Removal of Bottom Sediments

Bottom sediments play an extremely important role in the lake ecosystem. In the initial periods of existence, lakes constitute a "trap," detaining excess nutrients (especially nitrogen and phosphorus). Over time, the intensity of primary production increases in the surface layers, and the decomposition of organic substances near the bottom is less and less efficient. This causes deoxidation of deep parts of water and reduction of oxidoreductive potential in the sediment-water interphase. The matter which has accumulated for over 10,000 years then begins to enter into the water. This is called internal enrichment. The amount of nutrients in this "stock" – which the bottom sediments become – is extremely high. It is estimated that the 10-cm upper layer of bottom sediments contains 90% of the total phosphorus in all components of the lake ecosystem [21]. Hence, failure to control the internal loading destroys all restoration procedures.

The removal of bottom sediments is generally considered the most effective method of re-cultivation of shallow, heavily degraded lakes, but it creates unique technical and organizational difficulties. Theoretically, the total removal of bottom sediments, up to the parent rock, would guarantee a radical "rejuvenation" of the lake. However, with their considerable thickness, up to several or even several meters, it is practically impossible. As a rule, only modern settlements are removed (Fig. 2). They are deposited during the period of increased pollution of the reservoir; they are the richest in organic matter and contain the vast majority of mobile forms of biogenic substances. In Poland, the method is not widespread, and it mostly concerns only small lake fragments [22, 23]. This type of activities on a larger scale is regularly (every 4 years) undertaken at the Malta Reservoir in Poznań [24].

It should be emphasized that it is not enough to excavate bottom sediments and leave them on the shores of the lake. It is actually harmful, because highly hydrated sediments generate leachates which contain very high concentrations of nitrogen and phosphorus, exceeding by one or even two orders of magnitude the concentration of these nutrients in the water depth. Uncontrolled runoff of these waters from the heap of bottom sediments is a very dangerous source of secondary pollution of the lake.

Therefore, the main limitations of the wider use of this method are high costs associated not only with the extraction of bottom sediments from the lake but, above all, with their utilization. This is often related to the devastation of the surrounding area, caused by the need to build sediment clarification tanks and equipment for treating seepage waters. Often, there is simply a lack of such areas in the immediate vicinity of the lake. There are also objections to excessive elimination of littoral, which is often a habitat of valuable plant species and waterfowl [5]. On the other hand, an important, positive aspect is the deepening of the lake basin, which

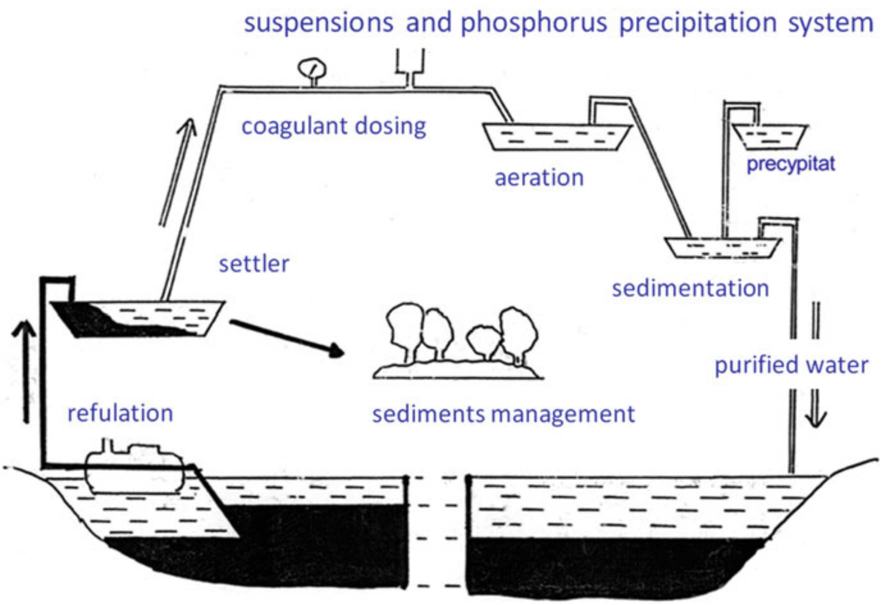

Fig. 2 Removal of the Truman Lake sediments in Sweden, according to [25]

increases the water resources of the ecosystem and helps to maintain biological balance.

Conditions of Applying the Method

- The occurrence of the internal supply phenomenon that plays an important role in the nutrient balance of the lake.
- Very well-recognized composition of sediments and spatial distribution of contaminants – which allows accurate and economically justified planning of works. It is connected with the necessity of making precise and cost-intensive research.
- Technical possibilities for storing deposits on the edge of the reservoir and developed directions for their further use.
- Available technology for the neutralization of sedimentary waters draining from the heap of bottom sediments.
- Maintenance of at least part of the littoral and sublittoral in order to ensure the continuity of the biological ecosystem.

4.3 Flushing/Dilution

The principle of the method is to rinse or dilute the waters of the degraded lake with better quality water. The improvement of the ecosystem's condition occurs as a result of the reduction of concentration of biogenic compounds due to dilution processes and the simultaneous discharge of pollutants outside the lake – as a result of increased water exchange. The irrigation water is most often obtained from underground intakes, another less eutrophicated reservoir, or nearby watercourses artificially diverted in whole or in part to a lake subjected to a reclamation treatment. The choice of delivery method depends on local conditions.

This method has been used in many lakes of the [5]. Literature data indicate, however, that lasting effects were obtained only in a few cases, mainly when the lakes were contaminated for a short time and the intensity of rinsing was significant. In Poland, the only (and unsuccessful) example was an attempt to flush Lake Długie in Olsztyn in the 1980s. Although the water of the less eutrophicated Lake Ukiel was brought to this lake, due to the improperly prepared installation, there were problems with draining excess water [26]. After a single operation, further tests were abandoned.

Conditions of Applying the Method

- Morphometry of a lake basin – the lake must be shallow and polymictic, not stratified thermally and chemically. The most advantageous are the compact shapes of lake basins, without side, stagnating bays, promontories, or large islands.

- The availability of a very large amount of water with a clearly better quality – the best results are obtained with the intensity of rinsing corresponding to several times the water exchange in a year.
- Inflow and outflow located opposite.

4.4 Artificial Aeration

The main goal of all methods of lake reclamation is to reduce the availability of nutrients for primary producers in the lake. The abovementioned superior role of bottom sediments in the distribution of these elements in water makes the improvement of oxygen conditions in the deepest layers of the lake's water one of the most important conditions for ecosystem revitalization [5, 27–29]. Artificial aeration techniques are designed to quickly improve the oxygenation of deeper layers of water, which increases the redox potential at the water-sediment interface and prevents, or at least limits, the release of biogenic compounds from bottom sediments to water [30].

Artificial aeration is carried out in two ways:

– With thermal destratification (total mixing)
– Without destroying thermal layers (oxygenation of the hypolimnion)

4.4.1 Artificial Aeration with Thermal Destratification

This group of methods aims to eliminate the thermocline separating the warm water of the epilimnion from the cold, stagnant layers of hypolimnion, which consequently unifies the physicochemical conditions in the entire water column.

Destratification of water can be obtained by various methods, but the simplest and requiring the least energy expenditure is the discharge of compressed air above the bottom at the deepest part of the lake. This method has been applied in many Polish lakes and other countries. The mechanism of destratification consists in repeatedly pushing compressed air through the bladders of the bottom waters to the surface which, in turn, causes equalization of temperature and mechanical mixing of the entire mass of water. The air here is mainly a transport factor – the efficiency of oxygen diffusion to water during rising of generated air bubbles is estimated at only about 5%. The basic installation of the aeration system consists of a compressor located in a container located on the shore of the lake, a transmission line, and an air discharge system (Fig. 3).

The reclamation carried out in this way changes the conditions both in surface water layers (temperature decrease, decrease in pH, reduction of phytoplankton biomass, reduction of oxygenation), and deeper (oxygenation of water, reduction of organic and biogenic compounds, while significantly increasing their temperature).

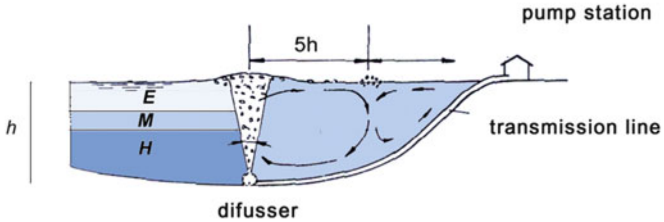

Fig. 3 Functional diagram of destratification devices (source: [28], amended)

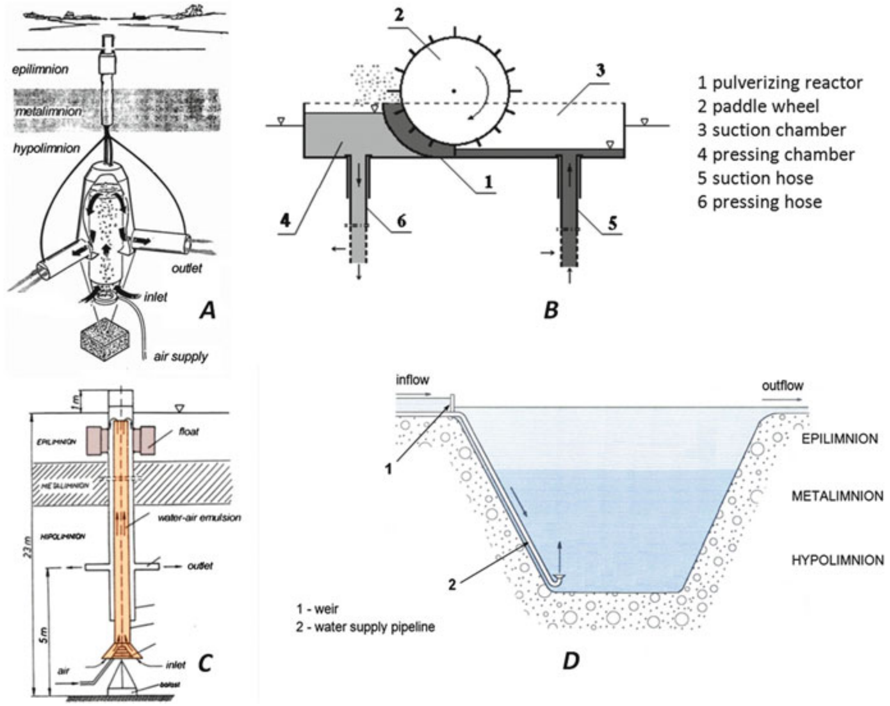

Fig. 4 Hypolimnion oxygenation techniques: (**a**) Limnox deep aerator, (**b**) pulverizing aerator, (**c**) stratiflox pipe aerator, (**d**) oxygenating pipeline (according to [5, 27, 32, 33], amended)

4.4.2 Oxygenation of Hypolimnion

Avoiding the mixing of water masses requires the use of special devices and systems preventing the destruction of thermal stratification [5, 27, 31]. This is achieved by a mechanical aeration system (a system with a pulverizing chamber located on the edge), with partial water lift (Limnox aerator) or with full water elevation (e.g., "ekoflox" and "stratiflox" aerators, pulverizer aerators – Fig. 4). Modern constructions allow the simultaneous introduction of a phosphate-binding coagulant, which significantly improves the effectiveness of the method.

The simplest aerators consist of two coaxially mounted pipes – the inner one, to which compressed air is supplied from the compressor located on the edge, and the outer water reaching over the water table. The water which is pushed out through the inner tube undergoes partial oxygenation and returns to the hypolimnion by an outer tube.

An innovative solution in lakes with a positive water balance may be the redirection of oxygenated surface inflow with a pipeline to the deepest parts of lake [32]. This type of installation has been operating on Lake Łajskie (47 ha) in the Olsztyn Lake District since 2013, and its distinctive feature is the negligible operating cost. The energy required for the water flow is obtained due to the small damming up of the watercourse (in cited case study, it is 0.2 m).

The technique of artificial aeration without destroying thermal systems is less invasive than destratification systems. However, it should be remembered that it only affects the near-bottom zone. Thus, the improvement of the lake's state is gradual – in subsequent cycles of the lake year, there is successive inhibition of the release of biogenic compounds from bottom sediments to water during periods of water stagnation [5]. In Poland, after 2000, more and more applications using pulverizing aerators have been observed [34–38]. However, in the case of larger water bodies, the impact of these devices seems to be insufficient, especially when their driving force is wind energy. Due to the limited wind force in summer, an aerator's work is discontinuous, which is manifested by the limited (space and time) positive impact of such equipment on oxygen conditions in the bottom zones of lakes [39, 40]. Therefore, additional measures such as phosphorus inactivation and biomanipulation are currently used in this method.

Conditions of Applying the Method

- Artificial aeration makes sense only for stratified lakes – it should not be used in polymictic reservoirs.
- The improvement in water quality is obtained gradually by consistently lowering the internal supply rate and mineralization of excess organic matter in water; hence this method is used in a long-term cycle.
- High investment and operating costs of works. In addition, it is necessary to ensure constant monitoring of the ecosystem, in order to adjust the intensity of equipment to the real environmental requirements.
- The destratification method – the necessity of consistent aeration throughout the period of stratification of waters in the cycle of the lake year. It is unacceptable to interrupt the operation of compressors, e.g., due to financial constraints. The consequence of the reclamation undertaken is the change of the lake's circulation type and thus strong interference in the ecosystem. A significant increase in the temperature of deeper water layers limits the possibility of using this method, e.g., in lakes used for breeding cold-water fish.
- Oxygenation of hypolimnion – since aerators usually have limited efficiency, especially those powered by wind energy, this technique works well for small bodies of water. In the case of more extensive hypolimnion, the effect of a single

aerator on the oxygen ratios of bottom waters may be too weak to achieve the effect of reducing the internal loading phenomena.

- Artificial aeration allows withdrawal of eutrophication symptoms only to a certain level resulting from changes that have been caused by the improvement of aerobic conditions. This does not always correspond to the acceptable ecological condition of the lake. Further improvement of the condition of the ecosystem, especially in lakes that are phosphorus-derived, can be achieved by additional techniques, mainly aimed at increasing the sorption capacity of bottom sediments.

4.5 Phosphorus Inactivation

The phosphorus inactivation method is recommended for lakes in which, despite lowering the external load with biogenic compounds, high fertility is maintained by release from bottom sediments [27]. It works well in shallow, polymictic lakes [28, 41, 42], as well as stratified dimictic lakes [27, 43–45].

The main objective of the method is to reduce the amount of phosphorus in the lake as a result of its removal from the water column (precipitation, coagulation, and sedimentation) and inhibition of release from bottom sediments (as a result of increasing their sorption capacity). As phosphorus binders, metal salts of mainly aluminum, iron, and calcium are used in this method. In recent years, other coagulants have also appeared on the market, based on lanthanum salts, characterized by good phosphorus binding capacity from water [46, 47].

The most common are aluminum and iron coagulants. Introduced into water, they rapidly hydrolyze, resulting in the formation of sparingly soluble sediments of hydroxides (flocs) with high sorption capacity in relation to phosphorus. The produced flocs fall to the bottom of the lake, where they form an active layer preventing migration of phosphorus from the sediments into the water. They should be characterized by a high ability to stabilize the removed pollutants and good sedimentation properties.

Due to their chemical properties, aluminum salts effectively bind phosphorus, even in the case of occurring deoxidation of the bottom waters and, thus, in highly eutrophicated water reservoirs. In turn, coagulants based on iron compounds are less suited for profundal, deepwater sediments [5]. This is due to the fact that iron is an element with variable valence, sensitive to changes in the reduction and oxidation potential. Under anaerobic conditions, it destroys the ferrophosphorus connections and releases phosphates into the water. However, these coagulants are ecologically safer.

Restoration of lakes using aluminum salts requires special care, especially when used in lakes poor in calcium, susceptible to acidification. Aluminum has amphoteric properties. In the range of pH 6–8.5, it occurs in an insoluble form, but at the lower and higher pH, the quantity of its dissolved form, toxic to organisms, increases in water. According to Cooke et al. [27], the ecologically safe level of concentration in water is 50 μgAl per liter. Hence, a great role is attached to the precise determination of the optimal dose of coagulant [48].

Coagulants can be applied by the surface method, directly into the hypolimnion water or directly on the surface of bottom sediments (Fig. 5). The method of introducing coagulants should be adapted to the assumptions of the reclamation project and environmental conditions in the lake. Dosing of the coagulant into the water is fast and the least expensive and allows the removal of phosphorus from the

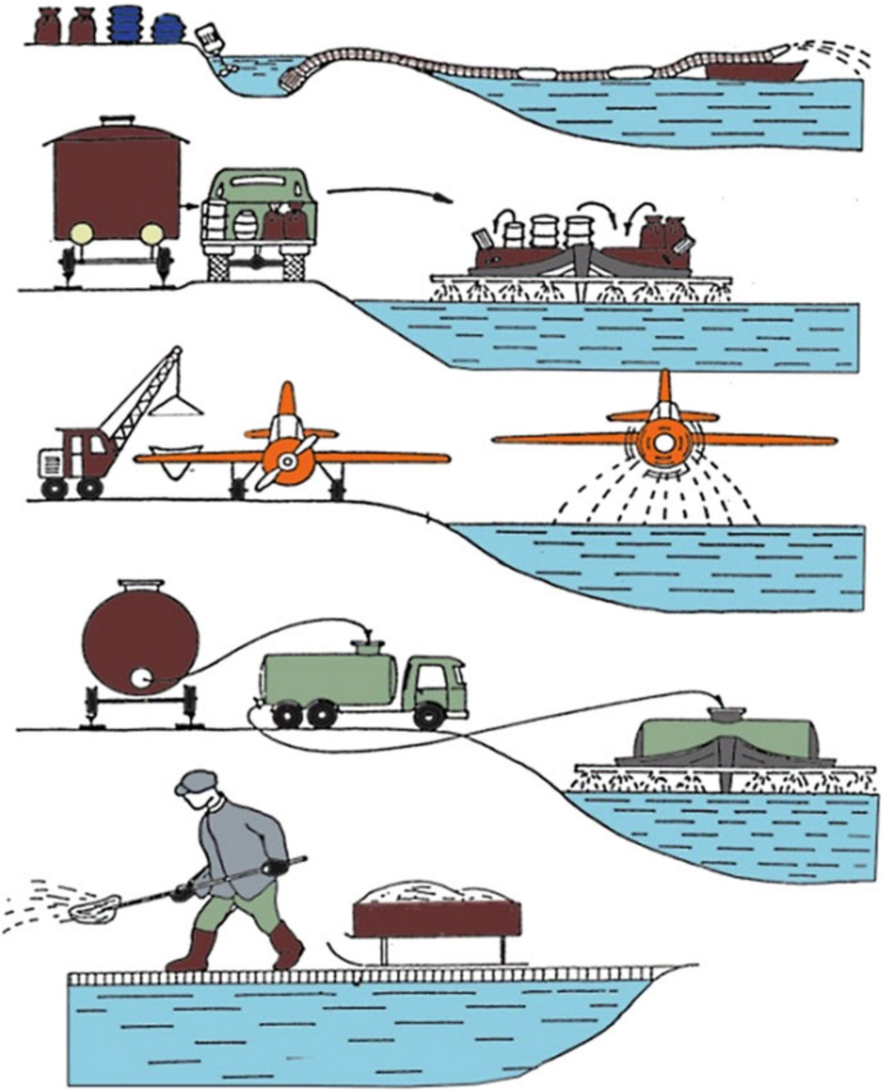

Fig. 5 Methods of coagulant dosing (source: [28], amended)

water depth [5]. However, it can cause problems with uneven settling of flocks on the bottom and possible toxic effects on living organisms living mainly in surface zones. When coagulants are added directly to the hypolimnion, the entire dose can be used to block the release of phosphorus from the bottom sediments. It is also easier to maintain the correct reaction because the alkalinity of the bottom waters is usually higher than the epilimnetic waters. However, this requires more advanced application technologies.

The deadline for conducting reclamation activities should cover the period beyond the vegetation peak – in the case of shallow polymictic lakes, it is best if it is early spring – just after the ice has descended. For deeper lakes, it may also be a period of autumn water circulation. In the case of deepwater coagulant application, the initial period of summer stagnation is also appropriate.

In Poland, this method was first applied to Lake Starodworskie in Olsztyn in 1994 [23]. After 2000, aluminum, iron, and lanthanum coagulants were applied in 25 lakes.

Conditions of Applying the Method

- Phosphorus inactivation is a relatively universal method, suitable both for shallow and deep lakes.
- As with all restoration methods, the prerequisite for beneficial changes is the maximum limitation of the external biogenic load of the lake.
- The selection of the coagulant, its dose, and timing of the procedure must be determined based on the results of thorough studies of the lake ecosystem, taking into account water parameters (mainly nutrient concentration, thermal oxygen conditions, the level of primary production intensity, and its impact on buffer properties and water reaction) and bottom sediments (chemical composition, sorption capacity in relation to phosphorus, and its form of occurrence).
- During reclamation, water quality parameters should be monitored, especially when using preparations potentially toxic to the environment. Restoration must be carried out under the supervision of an experienced team of limnologists.
- The ability to inactivate phosphorus by the formulations used is defined. After complete saturation with phosphorus, the coagulant will no longer absorb further portions of this element. Therefore, the durability of effects in the absence of excessive nutrient load from the outside is usually several, rarely over a dozen years. After this period, reclamation treatments should be repeated.

4.6 Treatment of Bottom Sediments

This method, like aeration of waters, aims to increase the oxidation-reduction potential on the border of bottom sediments and water, and similarly to phosphorus inactivation limits the enrichment of the ecosystem with this element. Both of these effects are obtained by the simultaneous direct addition to the sediments of chemical compounds, mechanical treatment, aeration, and coagulation. The consequence of these treatments is the oxidation of reduced compounds contained in the upper layers of bottom sediments and the more permanent binding of phosphorus.

The first application of this technique took place on the small shallow Lake Lillesjön in Sweden [49]. In European countries, this method has been developed and improved. Also in Poland some successful attempts have been made, using nitrates contained in natural water from catchment [50]. It is much cheaper than removing sediments and can be successfully used in small, shallow water reservoirs.

Conditions of Applying the Method

- The treatment of bottom sediments requires intensive dosing of a number of chemicals. It is therefore invasive for organisms inhabiting the bottom of the lake (benthos).
- Mechanical sludge moving contributes to their re-suspension and secondary pollution in the water. This makes it necessary to cover fragments of the lake subjected to reclamation with curtains protecting the main body of water. An alternative may also be a Polish solution [51, 52] involving the use of underwater modules equipped with a hull covering the working area of the harrow (Lake Jelonek, Winiary, Wolsztyńskie).
- Due to the need for very precise control of dosing devices, the method requires mastering advanced, expensive technologies.
- It is suitable for small lakes, because its use in large lakes is much more difficult, time-consuming, and costly.

4.7 Biological Methods

The literature describes many attempts to interfere in the aquatic environment by controlling the natural species and quantitative composition of the world of plants and animals. In the vast majority of cases, activities, such as the introduction of herbivorous fish, seston removal, and the use of so-called biostructures, have failed, indicating a lack of understanding of the complex trophic conditions in lake ecosystems.

It is worth mentioning here the microbiological bioremediation method based on effective microorganisms (EM) developing in recent years [53]. Studies show that in shallow, small water reservoirs, as a result of the application of bacterial preparations, there are rapid mineralization of pollutants and the release of mineral forms of phosphorus and nitrogen. When the phosphate-binding agent can be simultaneously introduced, it is possible to purify the reservoir from excess organic matter and reduce the bioavailability of nutrient compounds. The method is not yet widespread in Poland, but preliminary experiments indicate the possibility of further improvement.

An expedient, local factor in the fight against algal blooms is the exposure of barley straw [54, 55]. The algistats contained in it inhibit the growth of cyanobacterial cells. This method, although it cannot be treated as a lake reclamation operation, is an interesting option for interventional improvement of water quality in small areas, e.g., bays or within bathing areas.

In the light of numerous experiments and more or less successful attempts of biological reclamation, the most legitimate course of action seems to be the so-called biomanipulation – involving the creation of optimal conditions for the development of large species of planktonic crustaceans, controlling the excessive development of phytoplankton, and limiting its "blooms" [56]. To achieve this, it is necessary to reconstruct the species composition of the ichthyofauna with the restoration of adequately numerous populations of predatory fish – in Polish conditions, mainly pike, zander, perch, and catfish [57]. These predators are able to control the amount of white fish at a level low enough to allow the communities of large plankton filtration systems to develop. However, this requires very intensive, often perennial restocking with appropriate assortments of predator fry and is rarely successful in practice. However, such a direction of fishing economy is indispensable when undertaking reclamation works, regardless of the leading method.

The predominance of predatory fish, in natural conditions characteristic of lakes with low trophic levels, is still valid for another reason. The food pressure of predators also keeps the populations of benthivorous fish actively searching for food in bottom sediments (bream, carp), which significantly contributes to limiting the internal loading by bioturbation.

The phenomenon initiated by the breakdown of algae and cyanobacteria as a result of the revitalization work undertaken is the reconstruction of immersed higher vegetation – macrophytes. Under favorable conditions (increasing transparency of water, lack of mechanical destruction by fish, or as a result of recreational pressure), they can form vast underwater meadows, constituting refugia for valuable crustaceans and stabilizing bottom sediments. This is increasingly the positive effect of the reclamation measures being undertaken and, at the same time, the condition for maintaining the obtained symptoms of de-eutrophication of water reservoirs.

5 Restoration of Polish Lakes: History and Current Status

The development of the field of applied limnology, called the restoration of lakes, began in practice in 1956 when, for the first time in the world, an installation for cleaning a lake was launched in Olsztyn, Poland. The experimental site was Lake Kortowskie (90 ha) inseparably connected with the academic center of Olsztyn academies, currently constituting the University of Warmia and Mazury. The Olszewski pipeline operates to this day, and it is the oldest and the longest documented project for the renewal of the water reservoir all over the world.

Since then, there has been a rapid development of restoration methods, both in Poland and in the world. Dunst et al. [58] noted almost 200 documented treatments carried out on the lakes of Europe, Asia, and the Americas. Today, globally, examples of the reclamation of water reservoirs can be counted in thousands.

In Poland, 29 lakes had been reclaimed by 2000 [23], 7 of which had been treated with more than 1 method. The vast majority of them were reservoirs that underwent artificial aeration (Fig. 6). Due to technological limitations and the beliefs of limnologists at that time, they were carried out mainly through destratification of water masses (20 cases). Five lakes underwent hypolimnion oxygenation. In five lakes, biostructures were also exposed as an addition to the principal works. Individual cases concerned other restoration methods.

In most cases, the reclamation works in this period were carried out with only one method, generally putting technical activities above biological methods. Due to the low ecological awareness, attempts were made to reclaim without taking care of earlier implementation of protective measures. A frequent mistake was also limiting the intensity of designed projects due to economic reasons (e.g., periodically switching off aerators).

After 2000, there is noticeable progress in the development and improvement of lake reclamation in Poland. The frequency of treatments recognized in limnology as

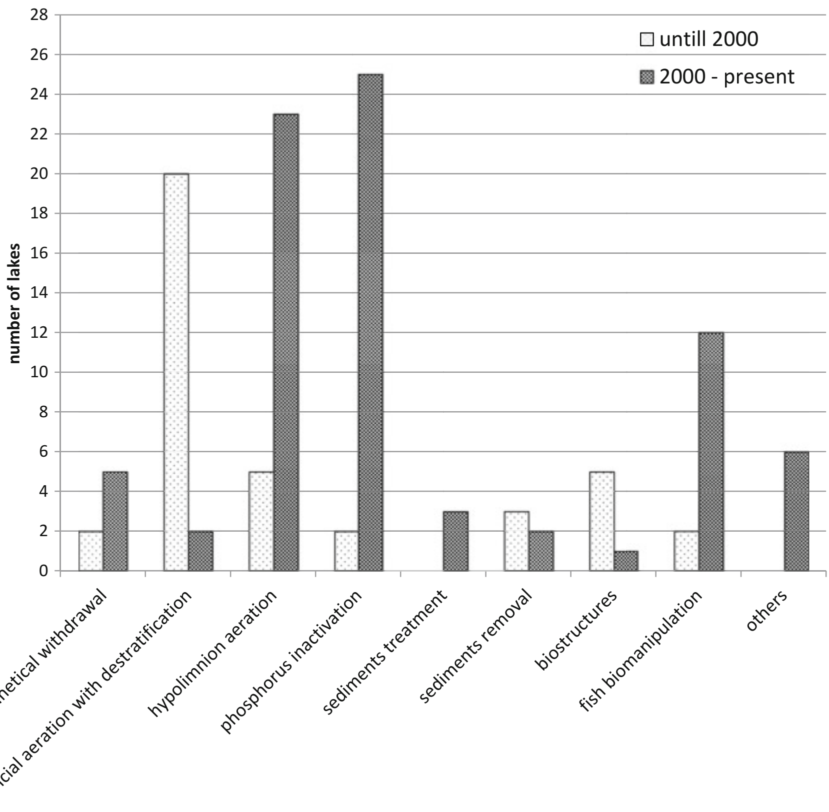

Fig. 6 The number of restoration treatments carried out in Poland in the twentieth and twenty-first centuries

advanced techniques – oxygenation of hypolimnion, phosphorus inactivation, and fish biomanipulation – has increased. At the same time, there has been a tendency to combine restoration methods in real rehabilitation programs, which are increasingly well-thought-out and more effective. In total, including reservoirs in which reclamation is carried out permanently and started in the last century, from 2000, reclamation was carried out on 46 lakes and covers 3,766 ha (1.34% of lakes in Poland). Among them, 24 lakes (52%) are reservoirs subjected to at least 2 types of treatments. In Figs. 6 and 7, statistical data on the intensity of reclamation work were presented collectively. The list does not include works of a nursing or melioration nature carried out on urban ponds and other artificial reservoirs or experimental work carried out in situ on a fractional-technical scale, which for obvious reasons cannot be considered as the restoration of lakes. Under the designation "others," there are examples of actions using microbiological methods, algistatics, and electromagnetic fields.

Numerous scientific papers describing the obtained effects provide valuable knowledge about changes in water ecosystems under the influence of the actions

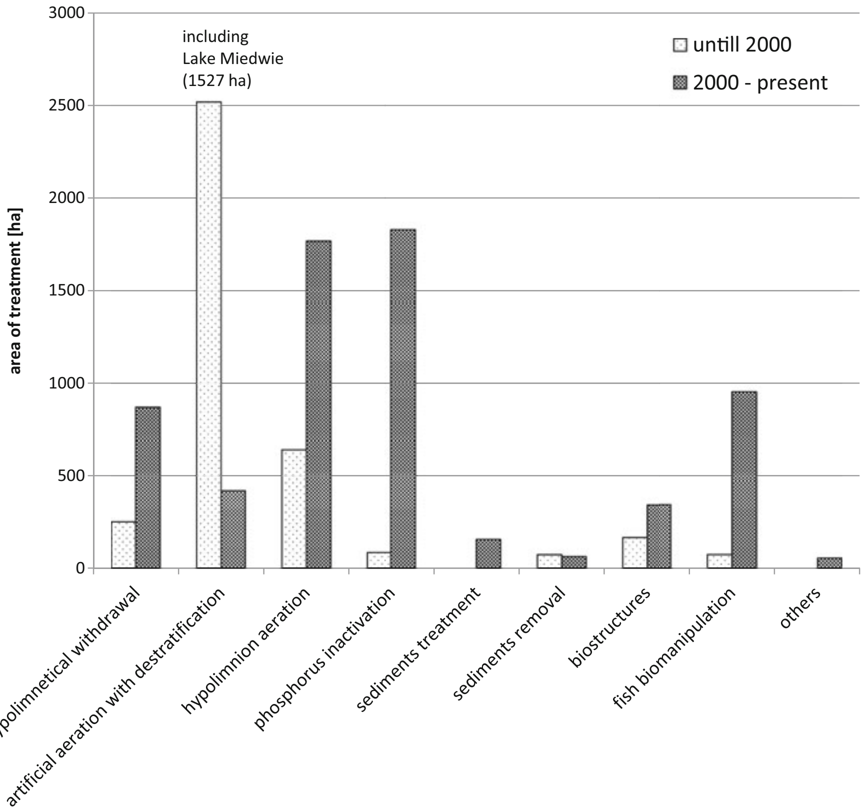

Fig. 7 Changes in the surface of lakes covered by repair programs in the twentieth and twenty-first centuries

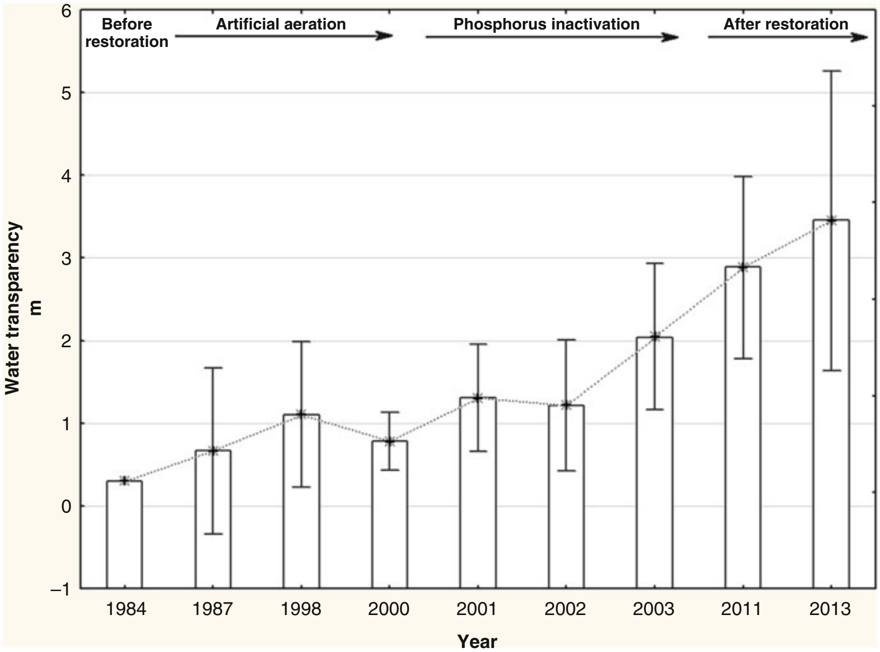

Fig. 8 Light conditions in the water of the restored Lake Długie in Olsztyn (source: [60])

taken. As a rule, however, these are still reports summarizing short-term research cycles, in which, in most cases, the authors report slightly positive reactions of the lakes. Nevertheless, the determinant of the success of the undertaken works is the durability of the effects after their completion. There are few such studies. One of the best-documented restoration projects is the renewal of the Lake Długie in Olsztyn [26, 45, 59–61] recognized as one of the most polluted lakes in the country. The measures applied here – artificial aeration, phosphorus inactivation with the use of aluminum coagulant, and change of fisheries management directions – were preceded by the ordering of water and sewage management in the basin and produced the expected results (Fig. 8). Currently, 15 years after the end of restoration, the lake remains in excellent ecological condition, and a measurable effect of its recovery is the emergence of Charophyta – widely recognized as an indicator of the cleanest and most valuable natural habitats [61, 62].

6 Conclusion

The analysis of archival and contemporary data on the effects of restoration of Polish lakes is only partially optimistic. Limnologist practitioners have long been calling for the protection of these ecosystems, understanding that lakes can easily be destroyed, but it is extremely difficult to reclaim them. The reasons for these failures are increasingly often interpreted based on contemporary theories of alternative stable states or ecological resilience [63–65]. The transition of the lake ecosystem with the so-called clean-water state to the turbid, with the predominance of phytoplankton, occurs very quickly, but the return is very slow. In practice, this means that the reclamation must cause a profound change so that the lake ecosystem at least goes to the zone of "unstable equilibrium" and preferably immediately to systems characteristic of clean water [30, 65].

In practice, it is difficult to obtain. Revitalization projects mainly include city lakes subjected to strong anthropopression, often already in an "agonal" ecological state – and therefore the most difficult to cure. We still do not know enough about the interactions of physical, chemical, and biological factors in water ecosystems, which means that the adopted methods and repair techniques are not always precisely selected for real reasons for the degradation of water bodies. It is also connected with the alarming phenomenon of commercialization of the approach to preparing programs for revitalization of water reservoirs. The correct diagnosis of the state of the complex ecosystem, such as a lake, requires the work of a team of specialists who impartially provide recommendations on the directions of repair work. Meanwhile, decisions on the commencement of restoration are often made as a result of a superficial recognition of environmental conditions and under the influence of agitation of various groups of stakeholders, including private companies striving for the execution of works. Finally, with re-cultivation already begun, there is often a lack of funds for their consistent conduct, substantive supervision, or reliable research results that give rise to possible modifications or improvements to the activities carried out. Hence, the effects of restoration works are often far from the expectations of water users.

The preparation of the correct substantive program of revitalization of water reservoirs should take into account a number of basic elements [30]. It is necessary to determine the purposefulness of the restoration of a lake, taking into account its natural, recreational, and economic value. The current trophic status of the lake should be investigated, and the causes and sources of its degradation should be diagnosed – based on the results of physicochemical and biological tests of the lake's waters, its tributaries, and the catchment situation – in at least a full 1-year research cycle. This provides the basis for calculating the current external load with the load of nutrients introduced into the reservoir. Interpretation of the level of environmental impact on the lake is possible using various models defining the so-called permissible and critical loads, including the most widespread Vollenweider criteria [66, 67]. For excessive external loads, protective measures should first be designed and performed to eliminate, or at least limit, the emission of pollutants from individual sources. At this stage, if it is not practically possible to reduce the external load to a level that provides a chance of improving water status, restoration of such a reservoir should be considered unintentional, and it should be discontinued until the problem of inflow of pollutants is resolved.

The concept of the optimal method of restoration or a combination of methods should take into account the trophic status and morphometric and hydrological conditions of the reservoir, the possibility of using the area of the shoreline, their management, ownership status, and expectations of the Fishing User. It is important to holistically approach the planning of activities by combining knowledge of technical possibilities with ecological conditions. The concept must also include the anticipated work schedule, at least the approximate date of completion of restoration, and at least an estimation of the level of necessary financial expenditures. Next, it is necessary to specify the obligatory administrative activities and to secure funds and technical means to implement the program. Finally, the restoration work should be entrusted to a well-prepared team, led by experienced limnologists. The course of restoration and its effects must be documented by the results of research carried out both during and after its completion.

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References

1. Hutchinson GE (1957) A treatise on limnology. Yale University, New Haven
2. Lossow K (1998) Protection and reclamation of lakes. Theory and practice (in Polish). Idee Ekologiczne 13:55–70
3. Code of Good Agricultural Practice (in Polish) (2004) Ministry of Agriculture and Country Development, Ministry of Environment. Warsaw, p 98
4. Cieściński J (2013) The usefulness of the Code of Good Agricultural Practice in the programs of protection and reclamation of aquatic ecosystems (in Polish). In: Proceedings of the VIII Scientific Conference “Ochrona jezior ze szczególnym uwzględnieniem metod rekultywacji”, Toruń. pp 27–31
5. Lossow K, Gawrońska H (2000) Lakes – restoration - review of methods (in Polish). Przegląd Komunalny 9(108):91–106
6. Kondracki J (1998) Regional geography of Poland (in Polish). Wyd. Nauk. PAN, Warsaw
7. Choiński A (1992) Catalog of Polish Lakes (in Polish). Wyd. Nauk. UAM, Poznań
8. Choiński A (1995) Outline of Polish physical limnology (in Polish). Wyd. Nauk. UAM, Poznań
9. Søndegaard M, Jensen JP, Jeppesen E (2003) Role of sediment and internal loading of phosphorus in shallow lakes. Hydrobiologia 506–509:135–145
10. Soszka H, Kolada A, Pasztelaniec A, Ochocka A, Kutyła S, Biełczyńska A (2016) Development of a collective assessment of the condition of lakes from 2010–2015 (in Polish). Instytut Ochrony Środowiska – Polski Instytut, Badawczy, pp 1–14
11. Water Law (1974). Pol J Acts 38:230
12. Water Law (2001). Pol J Acts 115:1229
13. Water Law (2017). Pol J Acts:1566
14. Cullen P, Forsberg C (1988) Experiences with reducing point sources of phosphorus to lakes. Hydrobiologia 170:321–336
15. Jeppesen E, Søndergaard M, Jensen JP, Havens KE, Anneville O, Carvalho L, Coveney MF, Deneke R, Dokulil MT, Foy B (2005) Lake responses to reduced nutrient loading - an analysis of contemporary long-term data from 35 case studies. Freshw Biol 50:1747–1771
16. Thomas E (1944) Biologische Untersuchungen am Greifensee. Ber Schweiz Bot Ges 54:141–196

17. Olszewski P (1961) Versuch einer Ableitung des hypolimnischen Wassers an einem See. Ergebnisse des ersten Versuchsjahres. Proc Int Assoc Limnol 18:1792–1797
18. Lossow K, Gawrońska H, Mientki C, Łopata M, Wiśniewski G (2005) Lakes of Olsztyn - trophic status, threats (in Polish). SPW "Edycja", Olsztyn, p 164
19. Dunalska J, Wiśniewski G, Mientki C (2007) Assessment of multi-year (1956–2003) hypolimnetic withdrawal from Lake Kortowskie, Poland. Lake Reserv Manage 23:377–387
20. Wiśniewski G (2007) Permanence of the effects of the Lake Kortowskie restoration, started in 1956. In: Anthropogenic and natural transformations of lakes, vol 1. Polish Limnological Society, Szczecin, pp 126–128
21. Kajak Z (2001) Hydrobiology - limnology. Inland water ecosystems (in Polish). PWN, Warsaw
22. Bańkowska A, Wasilewicz M (2008) Review of activities carried out to improve the condition of Lake Zdworskie (in Polish). Prz Nauk Inż Kszt Środ 17(1):80–90
23. Solarczyk A, Burak S (2000) Information on the state of reclamation of lakes in Poland (in Polish). In: Proceedings of the IV Scientific Conference. "Ochrona i rekultywacja jezior". Polish Association of Sanitary Engineers and Technicians, Przysiek, pp 113–122
24. Stanińska J, Szczepaniak Z, Picz K, Piotrowska-Cyplik A (2014) Assessment of the potential of Malta lake sediments in Poznan towards their use for agricultural purposes. J Res Appl Agr Eng 59(1):116–120
25. Bjork S (1972) Ecosystem studies in connection with the restoration of lakes. Verh Int Ver Limnol 18:379–387
26. Gawrońska H, Lossow K, Grochowska J (2005) Restoration of the Długie Lake in Olsztyn (in Polish). SPW "Edycja", Olsztyn, p 52
27. Cooke GD, Welch EB, Peterson SA, Nichols SA (2005) Restoration and management of lakes and reservoirs. Taylor & Francis/CRC, London
28. Klapper H (1991) Control of eutrophication in inland waters. Ellis Hornwood, New York
29. Lossow K (1994) Polish experiences in the reclamation of lakes and reservoirs (in Polish). In: Proceedings of the II Scientific Conference "Ochrona i rekultywacja jezior i zbiorników wodnych". Polish Association of Sanitary Engineers and Technicians, Międzyzdroje, pp 145–162
30. Gawrońska H, Lossow K (2008) Lake restoration - limitations and practical indications (in Polish). In: Jasser I, Robak S, Zdanowski B (eds) Ochrona i rekultywacja wód Wielkich Jezior Mazurskich narzędziem rozwoju naukowego, gospodarczego, społecznego i kulturowego regionu. Inland Fishery Institute, Olsztyn, pp 39–51
31. Konieczny R (2004) Wind devices - the possibilities of their use in water aeration (in Polish). In: Proceedings of the V Scientific Conference "Ochrona i rekultywacja jezior". Polish Association of Sanitary Engineers and Technicians, Grudziądz, pp 99–110
32. Łopata M, Wiśniewski G (2013) The use of surface water flow to improve oxygen conditions in a hypertrophic lake. Global J Adv Pure Appl Sci 1:710–715
33. Konieczny R Pieczyński L (2006) Technological solutions in lake aeration of the West Pomeranian Voivodeship (in Polish). Inżynieria Ekologiczna 14:76–82
34. Heese T, Wilk-Woźniak E, Żurek R, Kaczorkiewicz M, Szmidt R, Arciszewski M, Pikuła K, Wojcieszonek A, Chrzczonowicz H, Zakościelna A (2013) The estimation of the ecological effect for restoration Trzesiecko Lake in the period between 2005 and 2012 (in Polish). In: Proceedings of the VII Scientific Conference "Ochrona i rekultywacja jezior". Polish Association of Sanitary Engineers and Technicians, Przysiek, pp 65–80
35. Konieczny R, Pieczyński L (2006) Reclamation possibility of West pomerania Region lakes using pulverizing aeration technology. Acta Agrophysica 7:947–957
36. Osuch E, Podsiadłowski S (2012) Efficiency of pulverizing aeration on Lake Panieńskie. Limnol Rev 12(3):139–145
37. Rosińska J, Kozak A, Dondajewska R, Gołdyn R (2017) Cyanobacterial blooms before and during the restoration process of a shallow urban lake. J Environ Manag 198:340–347

38. Rosińska J, Kozak A, Dondajewska R, Kowalczewska-Madura K, Gołdyn R (2018) Water quality response to sustainable restoration measures – case study of urban Swarzędzkie Lake. Ecol Indic 84:437–449
39. Siwek H, Włodarczyk M, Czerniawski R (2018) Trophic state and oxygen conditions of waters aerated with pulverising aerator: the results from seven lakes in Poland. Water 10(219):11
40. Wesołowski P, Brysiewicz A (2015) The effect of pulverising aeration on changes in the oxygen and nitrogen concentrations in water of Lake Starzyc. J Water Land Dev 25:31–36
41. Łopata M, Gawrońska H (2006) Effectiveness of the polymictic Lake Głęboczek in Tuchola restoration by the phosphorus inactivation method. Pol J Nat Sci 21(2):859–870
42. Łopata M, Gawrońska H, Wiśniewski G, Jaworska B (2013) Restoration of two shallow, urban lakes using the phosphorus inactivation method - preliminary results. Water Sci Technol 68 (10):2127–2135
43. Gawrońska H, Łopata M (2009) Durability of restoration of the Długie Lake by phosphorus inactivation (in Polish). Gaz Woda Techn San 9(82):21–22
44. Gawrońska H, Łopata M, Jaworska B (2007) The effectiveness of the phosphorus inactivation method in reducing the trophy of lakes of different morphometrical and hydrological features. Limnol Rev 7(1):27–34
45. Grochowska J, Brzozowska R, Łopata M (2013) Durability of changes in phosphorus compounds in water of an urban lake after application of two reclamation methods. Water Sci Technol 68(1):234–239
46. Hagseresht F (2004) An overview of the properties of Phoslock and its performance in improving water quality. IMT Holdings PTY LTD, Herndon
47. Łopata M, Gawrońska H, Brzozowska R (2007) A comparison of the effectiveness of aluminum coagulant and lanthanum coagulant in phosphorus inactivation. Limnol Rev 7(4):247–253
48. Rydin E, Welch EB (1998) Aluminum dose required to inactivate phosphate in lake sediments. Wat Res 32(10):2969–2976
49. Ripl W (1986) Lake restoration methods developed and used in Sweden. Restoration of lakes and inland water. EPA 440(5):81–100
50. Dondajewska R, Kozak A, Kowalczewska-Madura K, Budzyńska A, Podsiadłowski S, Tomkowiak A (2018) The response of a shallow hypertrophic lake to innovative restoration measures – Uzarzewskie Lake case study. Ecol Eng 121:72–82. https://doi.org/10.1016/j.ecoleng.2017.07.010
51. Wiśniewski R (1999) Attempts to inactivate phosphates in bottom sediments and inhibition of cyanobacterial blooms in Lake Łasińskie as potential methods of reclamation (in Polish). In: Proceedings of the I Scientific Conference "Postęp w inżynierii środowiska", Polańczyk, pp 189–202
52. Wiśniewski R, Ślusarczyk J, Kaliszewski T, Szulczewski A, Nowacki P (2010) "Proteus", a new device for application of coagulants directly to sediment during its controlled resuspension. Verh Int Ver Limnol 30(9):1421–1424
53. Chróst R (2017) Bioremediation of microbial contaminated and eutrophicated water reservoirs: myths and facts, for and against. Proceedings of the IX Scientific Conference "Hydromicro 2017: drobnoustroje - osiągnięcia i wyzwania". UWM, Olsztyn, p 19
54. Pawlik-Skowrońska B, Skowroński T, Pirszel J, Adamczyk A (2004) Relationship between cyanobacterial bloom composition and anatoxin-a and microcystin occurrence in the eutrophic dam reservoir (SE Poland). Pol J Ecol 52(479–490):139–145
55. Wiśniewski R (2002) Attempts to eliminate cyanobacterial blooms in Lake Leasinskie. Environ Prot Eng 1:15–26
56. Shapiro J, Wright DI (1984) Lake restoration by biomanipulation. Round Lake, Minnesota – the first two years. Freshw Biol 14:371–383
57. Gołdyn R (2000) Shines and shadows of biomanipulation on the example of the Maltańskie Reservoir in Poznań (in Polish). Proceedings of the IX Scientific Conference "Ochrona i rekultywacja jezior". Polish Association of Sanitary Engineers and Technicians, Przysiek, pp 145–153

58. Dunst RC, Born SM, Uttomark PD, Smith SA, Nichols SA, Peterson JO, Knauer DR, Serns SL, Winter DR, Wirth TL (1974) Survey of lake rehabilitation techniques and experiences. Dep Nat Resour Madison Tech Bull 75:1–177
59. Grochowska J, Brzozowska R, Łopata M, Dunalska J (2015) Influence of restoration methods on the longevity of changes in the thermal and oxygen dynamics of a degraded lake. Oceanol Hydrobiol Stud 44(1):18–27
60. Grochowska J, Augustyniak R, Łopata M (2017) How durable is the improvement of environmental conditions in a lake after the termination of restoration treatments. Ecol Eng 104:23–29
61. Łopata M, Wiśniewski G, Parszuto K, Dunalska JA (2016) From restoration to revitalization: how to recover recreational values of urban lakes. A case study of Lake Domowe Duże in Szczytno. Pol J Nat Sci 31(3):471–483
62. Dunalska JA, Ciecierska H, Napiórkowska-Krzebietke A, Ruszczyńska J, Sieńska J, Szymański D (2017) Lakes of Olsztyn: the most beautiful gift of nature: trophic state and ecology (in Polish). Mantis, p 164
63. Carpenter SR, Cottingham KL (1997) Resilience and restoration of lakes. Ecol Soc 1(1):2
64. Scheffer M (1990) Multiplicity of stable states in freshwater systems. Hydrobiologia 200/201:475–486
65. Wiśniewski R (2004) Restoration of water reservoirs. From practice to theory (in Polish). Proceedings of the IX Scientific Conference "Ochrona i rekultywacja jezior". Polish Association of Sanitary Engineers and Technicians, Grudziądz, pp 239–245
66. Vollenweider RA (1968) Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference of nitrogen and phosphorus as factors in eutrophication. OECD, Directorate for Sci. Affairs, Paris, DAS (CSI) 68, 27, pp 1–182
67. Vollenweider RA (1976) Advances in defining critical loading levels for phosphorus in lake eutrophication. Mem Ist Ital Idrobiol 33:53–83

Support of the Self-purification Processes in Lakes Restored in Poland

Katarzyna Parszuto, Michał Łopata, Jolanta Grochowska, Renata Tandyrak, and Renata Augustyniak

Contents

Abstract This study analyses the dynamics of organic matter and algae pigments in water of urban lakes subjected to various restoration techniques. Most of the lakes were restored with the inactivation method using coagulant PAX 18. One of them was supported with biomanipulation. Assessments of the organic carbon form content (POC, in suspension; DOC, dissolved form) and the chlorophyll and pheophytin concentrations were conducted. The ratios of individual parameters were then evaluated, and the relationships between these indicators were tested. It was found that decreasing primary production and the associated organic matter in a suspended form can improve self-purification processes in restored lakes. The decreasing POC after restoration leads to a reduction in the amount of DOC in the lake and determines the balance of production and decomposition. This improvement may initially be able to reduce the DOC content, mainly due to easily biodegradable compounds. The lower primary production supplies a lower content of hardly biodegradable compounds (when algae decay). After that, the amount of high-molecular-weight DOC can be reduced. The improved self-purification of the

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lakes was also reflected in a reduced chlorophyll/pheophytin ratio. There was no effective reduction of the organic matter content after the first restoration in polymictic lakes with high loaded biogens and organic matter (DOC) from the catchment.

Keywords Inactivation · Organic matter · Primary production · Restoration · Self-purification

1 Introduction

Self-purification processes take place in every type of water. Regardless of the type of reservoir (river, lake, groundwater), the basis of this process is the biodegradation of organic matter, for which oxygen conditions and microorganisms are required [1, 2]. A specific type of pollution is the organic matter generated in the primary production process. In addition, it is introduced into lake waters by watercourses and as a runoff from the basin [3–5].

Overproduction of organic matter causes its accumulation in a suspended form. If the suspension is not mineralized, it may sediment to the bottom of the reservoir [6]. Mineralization of organic matter in lake water requires an appropriate number of microorganisms and conditions for their development. The physical and chemical properties of water affect this mineralization [7]. In the eutrophicated lakes (high primary production), disturbances are observed by changing pH, oxygenating waters and the presence of by-products secreted by algae (also toxins) [8]. The role of organic matter as a substrate for heterotrophic microorganisms is indisputable. Bacteria can shape the environmental conditions and affect exchange processes in the interphase water-bottom sediment [9–12].

Restoration of the lake forms new physicochemical water and sediment properties, which affect the self-purification processes of a reservoir [13–15]. The content of gases dissolved in water and the rate of sedimentation and mineralization of organic matter change depending on the technique of restoration applied (aeration with destratification; hypolimnetic water removal). Moreover, the availability of nutrients in the trophogenic layer is reduced (P inactivation – removal of phosphorus and organic matter and inhibition of its release from sediment to water; hypolimnetic water removal), and, consequently, the productivity of lake waters decreases [16–23].

Both at the stage of implementing new restoration techniques and in the process of improving already used ones, it is extremely important to study in which direction the quantitative and qualitative changes of NOM (natural organic matter) take place. An analysis of the impact of various restoration methods found that they transformed the quality and reduced the amounts of macromolecular compounds of DOC (dissolved organic carbon) in lake waters [24]. It was also found that the share of the larger POC (particulate organic carbon) fractions associated with primary production was reduced [25, 26].

The aim of this study was to assess the dynamics of self-purification processes in lake waters subjected to various restoration techniques.

The authors attempted to verify the following theses:

1. Restoration, modifying the rate of lake self-purification processes in different ways, affects the production and degradation of organic matter in lake waters.
2. The changes in environmental conditions caused by using suitable restoration techniques modify and quantitatively differentiate the resources of organic matter.

In order to evaluate the changes of self-purification process rates, the directions of organic matter changes were first identified. An evaluation of the variation for the quantity of organic carbon forms and algae pigments and their differentiation in subsequent stages of restoration was then conducted. The relationships between the studied parameters were then determined and analysed to assess how environmental changes might modify and differentiate organic matter dynamics in water.

2 Material and Methods

2.1 Characteristics of Lakes and Restoration Techniques Used

The research was conducted on lakes, which were restored using the following techniques: artificial aeration with destratification, chemical inactivation of phosphorus using PAX 18 and biomanipulation.

All restoration methods used on these lakes were elaborated and implemented by employees of the Department of Water Protection Engineering, UWM, in Olsztyn. The biomanipulation method was supported by Robert Czerniawski (University of Szczecin).

Lake Długie (Fig. 1) is a deep stratified lake in Olsztyn (bradymictic). All morphometric data are presented in Table 1. Lake Długie is a reservoir which, after a failed use of the flushing-dilution method, was subjected to multi-annual aeration with destratification (1987–2000 – with interruptions). When the sorptive capacity of sediment was exhausted and aeration was not effective, another method, which was implemented in order to inhibit nutrient internal loading, was the first restoration in Poland using polyaluminum chloride – PAX 18 (2001–2003: spring, autumn, autumn, respectively) [19].

Lake Wolsztyńskie is a shallow reservoir with a large area. All morphometric data are presented in Table 1. The water of this lake is constantly mixed (polymictic type). This reservoir is supplied with water from the Dojca River (Fig. 1). This is a threat to the lake in terms of biogens and organic matter loading. In 2005–2007, three dosages of coagulant PAX 18 were introduced [27].

Fig. 1 Localization of the investigated lakes in Poland in which the restoration was carried out: (**a**) Lake Klasztorne Górne; (**b**) Lake Wolsztyńskie; (**c**) Lake Długie [31]

Table 1 Morphometric data and catchment development of the selected lakes

Parameter [unit]	Lake Długie [32]	Lake Klasztorne Górne [28–30]	Lake Wolsztyńskie [28, 33]
Surface area [ha]	26.8	18.5	124.2
Lake basin volume [m^3]	141,480	680,000	2,522,600
Maximum depth [m]	17.3	7.4	4.2
Average depth [m]	5.3	3.3	2.0
Maximum length [m]	1,670	1,340	2,490
Maximum width [m]	240	220	860
Length of the shoreline [m]	4,080	3,000	7,475
Shoreline development	2.23	1.92	1.89
Total catchment area [km^2]	0.137	28.1	193.5
Catchment's development	Urban	Agricultural	Diversified

Lake Klasztorne Górne has a surface area of about 20 hectares – according to Jańczak [28] – and has the shape of a long narrow trough (maximum depth 7.2 m – difficult mixing) (Table 1). This reservoir is supplied with water from the Młynówka River (Fig. 1). The restoration was conducted in 2012–2014. In the spring of 2012, the first dose of coagulant PAX 18 was introduced. Subsequent doses were introduced in the springs of 2013 and 2014. The inactivation method was supported by biomanipulation (2012–2013) [29, 30].

2.2 Tested Parameters and Methods

A quantitative study of organic carbon in a suspension (POC) and its dissolved form (DOC) in the surface water (1 m) was conducted. The DOC (after filtration through a

0.45 μm membrane filter, Millipore) and the total organic carbon (TOC) concentrations were determined by a carbon analyser, where the organic compounds are subjected to catalytic combustion to CO_2, which guarantees total mineralization of organic matter (SHIMADZU TOC-5000 or HACH IL 550 TOC-TN). Carbon dioxide was removed before determination by acidifying the sample (HCl 4m) and passing oxygen through it. POC was obtained as the difference between TOC and DOC:

$$\mathrm{POC} = \mathrm{TOC} - \mathrm{DOC}\ \left[\mathrm{mgCdm}^{-3}\right]$$

To determine the algae pigments, lake water (with a defined volume) was filtered using a fibreglass filter with a pore size of 1.6 μm (GF/C Whatman). After drying, the filter was ground, and the pigments were then extracted from the obtained pulp using 90% acetone in a volume of 5 cm^3. The obtained samples were centrifuged, and the absorbance of the extracts at specific wavelengths before and after acidification with 0.12 m HCl was measured. The determination of chlorophyll was made by the spectrophotometric method with correction for phaeopigments by using a SHIMADZU UV-1601PC spectrophotometer (spectra absorption measurements). To calculate the concentration of chlorophyll and pheophytin, the absorption was used for the following wavelengths: 663, 665 and 750 nm.

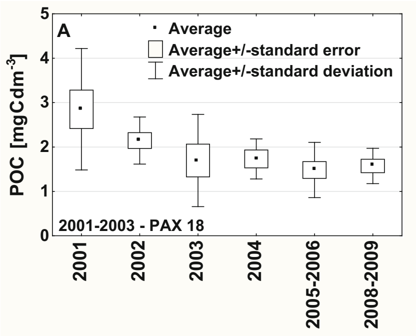
The A-coefficient was calculated as A663/A665. The value of this indicator ranges from 1 to 1.7. A high value indicates a high content of chlorophyll a in the water tested, while a low one (near 1.0) indicates a high concentration of pheophytin [34].

To determine if there were differences in parameter values between consecutive years of research (during and after restoration), a statistical analysis was performed (Statistica 13.5). Parametric (ANOVA) and non-parametric (Kruskal-Wallis) tests were used. The correlations between parameters were calculated using Spearman's Rank-Order Correlation.

3 Results

3.1 *Lake Długie*

After the introduction of the first and second doses of coagulant, a reduction in the average concentration of POC, DOC, chlorophyll, pheophytin and value [POC/DOC]*100 parameter and chlorophyll/pheophytin ratio was observed (Figs. 2, 3, 4 and 5). After restoration, in 2004, the average content of DOC again decreased. In the years 2005–2009, the concentrations of TOC, DOC, chlorophyll and pheophytin remained at levels similar to those following the end of inactivation. The A-coefficient, which indicates the advantage of chlorophyll in relation to



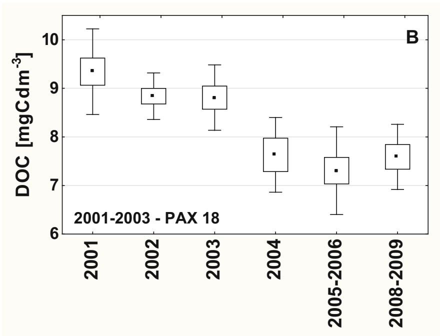


Fig. 2 The POC (**a**) and DOC (**b**) content (range, average) in surface water in Lake Długie

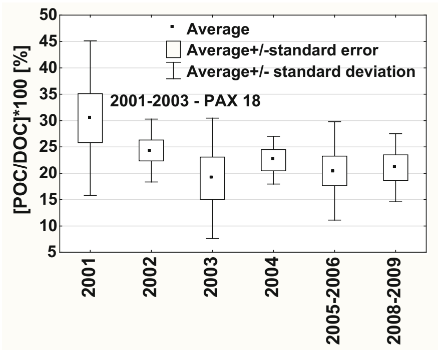
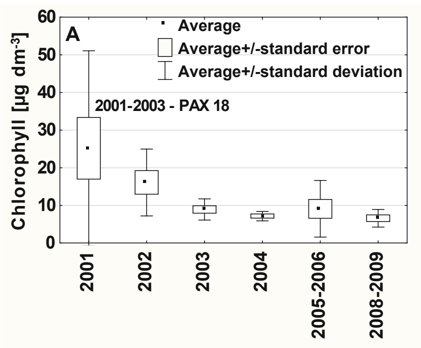


Fig. 3 The [POC/DOC]*100 [%] value (range, average) in surface water in Lake Długie



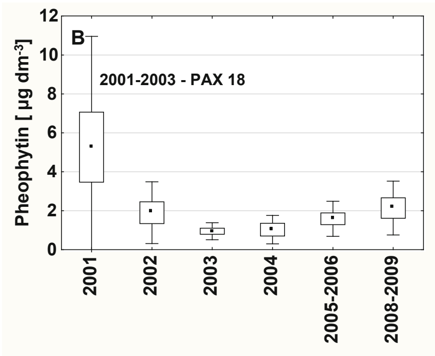


Fig. 4 The chlorophyll (**a**) and pheophytin (**b**) content (range, average) in surface water in Lake Długie

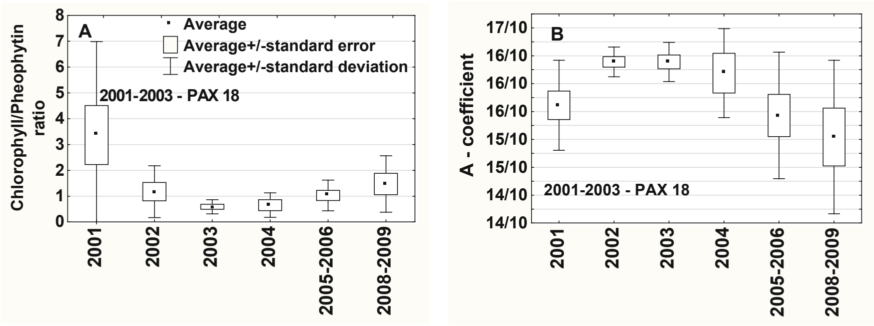


Fig. 5 The chlorophyll/pheophytin ratio (**a**) and the A-coefficient value (**b**) (range, average) in surface water in Lake Długie

pheophytin, was higher during restoration, but in the years following restoration, it decreased (Fig. 5).

Statistically significant differences were found between the following studied years:

POC – between 2001 and 2002, 2003, 2004, 2005–2006, 2008–2009 (ANOVA; $p = 0.0214$)
DOC – between 2001–2003 and 2005–2006 (Kruskal-Wallis test; $p = 0.0006$–0.0391) and next between 2001 and 2005–2006, 2008–2009 (Kruskal-Wallis test; $p = 0.0006$; $p = 0.0142$)
Chlorophyll – between 2001 and 2002, 2003, 2004, 2005–2006, 2008–2009 (ANOVA; $p = 0.0376$)
Pheophytin – between 2001 and 2003 (Kruskal-Wallis test; $p = 0.0090$)

Spearman's Rank-Order Correlation, performed on the restoration data, showed a positive relationship between POC and the following parameters: DOC, [POC/DOC]*100 and chlorophyll. There was also a high positive correlation between pheophytin and chlorophyll. Both pigments had an effect on the chlorophyll/pheophytin ratio (Table 2). Spearman's Rank-Order Correlation, carried out after the restoration, showed a high positive relationship of changes in pheophytin and the chlorophyll/pheophytin ratio. The A-coefficient values and chlorophyll were also correlated (Table 3).

3.2 *Lake Wolsztyńskie*

In the last year of the restoration, the increase of average DOC values from 7.49 to 9.03 $mgCdm^{-3}$ was observed. The average POC concentration did not decrease in 2005–2007. The POC was about 6.00 $mgCdm^{-3}$ after the third dosage of PAX 18. The average value of the [POC/DOC]*100 parameter was high over the restoration

Table 2 Spearman's Rank-Order Correlation between investigated parameters – Lake Długie (during restoration, 2001–2003 and 2004; $p < 0.0500$, $n = 31–32$; *ns* non-significant)

Parameter	POC	DOC	[POC/DOC]*100	Chlorophyll	Pheophytin	A-coefficient	Chlorophyll/pheophytin
POC	–	0.3915	0.9589	0.3766	ns	ns	ns
DOC	0.3915	–	ns	ns	ns	ns	ns
[POC/DOC]*100	0.9589	ns	–	ns	ns	ns	ns
Chlorophyll	0.3766	ns	ns	–	0.6427	ns	0.6310
Pheophytin	ns	ns	ns	0.6427	–	−0.7238	0.9992
A-coefficient	ns	ns	ns	ns	−0.7238	–	−0.7298
Chlorophyll/pheophytin	ns	ns	ns	0.6310	0.9992	−0.7298	–

Table 3 Spearman's Rank-Order Correlation between investigated parameters – Lake Długie (2005–2009, after restoration; $p < 0.0500$, $n = 16–18$; *ns* non-significant)

Parameter	POC	DOC	[POC/DOC]*100	Chlorophyll	Pheophytin	A-coefficient	Chlorophyll/pheophytin
POC	–	ns	0.9071	ns	ns	ns	ns
DOC	ns	–	ns	ns	ns	ns	ns
[POC/DOC]*100	0.9071	ns	–	ns	ns	ns	ns
Chlorophyll	ns	ns	ns	–	ns	0.8824	ns
Pheophytin	ns	ns	ns	ns	–	ns	0.9824
A-coefficient	ns	ns	ns	0.8824	ns	–	ns
Chlorophyll/pheophytin	ns	ns	ns	ns	0.9824	ns	–

Table 4 Average values and Kruskal-Wallis test for parameters in Lake Wolsztyńskie (surface water: $n = 10$–12, $n = 11$–12, $n = 9$–10 for 2005, 2006 and 2007; $p < 0.0500$)

Parameter [unit]	2005	2006	2007	Homogeneous groups with similar average
POC [mgCdm^{-3}]	4.44	5.16	5.95	2005*, 2006*, 2007*
DOC [mgCdm^{-3}]	7.45	7.04	9.00	2005*, 2006*, 2007**
[POC/DOC]*100 [%]	59.63	73.16	59.73	2005*, 2006*, 2007*
Chlorophyll [μgCdm^{-3}]	51.81	40.35	50.87	2005*, 2006*, 2007*
Pheophytin [μgCdm^{-3}]	9.63	7.49	18.25	2005*, 2006*, 2007*
Chlorophyll/pheophytin	6.86	6.54	7.30	2005*, 2006*, 2007*
A-coefficient	1.59	1.59	1.55	2005*, 2006*, 2007*

Homogeneous groups (*;**) – this is means that for every parameter the years with * have similar averages and ** have the similar average – the difference was observed only for DOC – average in 2007 is different than 2005 and 2006

and from 60.00 to 73.00%. The average value of chlorophyll and A-coefficient was also very high, reaching above 50 and 1.59, respectively. There were no statistically significant differences between the studied parameters in the following years of restoration, except for DOC (Table 4).

Spearman's Rank-Order Correlation showed a relationship between POC and DOC and the [POC/DOC]*100 parameter. Correlations between changes in pheophytin and changes in A-coefficient and the chlorophyll/pheophytin ratio were also found. A strong dependence occurred between pheophytin and the A-coefficient and the chlorophyll/pheophytin ratio. The A-coefficient values and the chlorophyll/pheophytin ratio were correlated with each other (Table 5).

3.3 Lake Klasztorne Górne

The average amount of POC decreased from above 4.00 to around 2.30 mgCdm^{-3} after the introduction of two doses of coagulant, which was statistically significant. Since the range of DOC concentrations averaged 6.25–5.31 mgCdm^{-3}, the average value of the [POC/DOC]*100 parameter decreased from 65% to 45%. There was no statistically significant difference between average values of DOC and the [POC/DOC]*100 parameter in subsequent years of restoration. A statistically significant decrease in the average chlorophyll/pheophytin ratio was observed. No changes in the average pheophytin concentration occurred, but the chlorophyll showed a decreasing tendency (Table 6).

Spearman's Rank-Order Correlation showed a relationship between POC and pheophytin and the [POC/DOC]*100 parameter. Correlations between changes in chlorophyll and changes in A-coefficient, pheophytin and the chlorophyll/pheophytin ratio were also found. A weaker dependence occurred between pheophytin and the A-coefficient and the chlorophyll/pheophytin ratio (Table 7).

Table 5 Spearman's Rank-Order Correlation between the studied parameters – Lake Wolsztyńskie (2005–2007; $p < 0.0500$, $n = 30–33$; *ns* non-significant)

Parameter	POC	DOC	[POC/DOC]*100	Chlorophyll	Pheophytin	Chlorophyll/pheophytin	A-coefficient
POC	–	0.3908	0.8428	ns	ns	ns	ns
DOC	0.3908	–	ns	ns	ns	ns	ns
[POC/DOC]*100	0.8428	ns	–	ns	ns	ns	ns
Chlorophyll	ns	ns	ns	–	ns	ns	ns
Pheophytin	ns	ns	ns	ns	–	−0.5775	−0.8278
Chlorophyll/pheophytin	ns	ns	ns	ns	−0.5775	–	0.6314
A-coefficient	ns	ns	ns	ns	−0.8278	0.6314	–

Table 6 Average values and one-way ANOVA (grouping variable – year) for parameters in Lake Klasztorne Górne (surface water: $n = 8$, $n = 10$, $n = 8$ for 2012, 2013 and 2014; $^{**}p < 0.0500$)

Parameter [unit]	2012	2013	2014	p
POC [$mgCdm^{-3}$]	4.11	2.27	2.23	0.042349**
DOC [$mgCdm^{-3}$]	6.23	6.03	5.35	0.506503
[POC/DOC]*100 [%]	64.78	44.84	45.94	0.324264
Chlorophyll [$\mu gCdm^{-3}$]	36.49	29.67	21.58	0.133410
Pheophytin [$\mu gCdm^{-3}$]	13.64	13.12	14.95	0.766915
Chlorophyll/pheophytin	3.00	2.60	1.57	0.039511**
A-coefficient	1.51	1.49	1.40	0.104428

4 Discussion

Overproduction of organic matter in the water of a highly eutrophic lake makes the conditions for its disposal unfavourable [35]. The cause of an imbalance in the ecosystem of lake is often an oversupply of organic matter from the catchment. In such cases, protective measures should be conducted in the lake catchment [36]. This can enhance the rate of self-purification, which are essentially biological and chemical processes supported by physical processes [37–39].

The research presented in this work was aimed at explaining how the change of environmental conditions in restored lakes can shape the content and ratios of individual components of organic matter in water. This is important for decreasing and maintaining the trophic state of the lake. It can be assumed that improvement of self-purification processes after restoration is applied will take place in different ways, depending on the technique used. The aim of each restoration is to restore balanced production-distribution to restore the advantages of natural water [20]. In connection with the change of physicochemical condition in restored lakes, the primary production decreases and the visibility increases. Consequently, water with a lower trophic state will be able to provide an environment for ecological biodiversity [40–42].

The most commonly used and best-known method of lake restoration, in Poland and around the world, is artificial aeration. Aeration can be carried out in two variants: without destroying the thermal stratification of water and with destratification [43, 44]. Today, an increasing number of techniques are being tested and implemented which rely on the chemical inactivation of phosphorus using different types of iron and aluminium coagulants [45, 46]. A special technique, the least intervening in the environment, is the removal of hypolimnion water via a pipeline. It is used in deep, flowing lakes with stable hydrological regime [16]. Technical methods are increasingly being replaced by biological methods of lake restoration or are combined with them [47–50].

The phosphorus inactivation method seeks to reduce the availability of phosphorus for primary producers through its precipitation from water and inhibition of its release from bottom sediments. The consequence of such treatments is the reduction of the primary production in the lake and, therefore, the reduction the amount of

Table 7 Spearman's Rank-Order Correlation between studied parameters – Lake Klasztorne Górne (2012–2014; $p < 0.0500$, $n = 26–30$; *ns* non-significant)

Parameter	POC	DOC	[POC/DOC]*100	Chlorophyll	Pheophytin	Chlorophyll/pheophytin	A-coefficient
POC	–	ns	ns	ns	0.4062	ns	ns
DOC	ns	–	ns	ns	ns	ns	ns
[POC/DOC]*100	0.9191	ns	–	ns	ns	ns	ns
Chlorophyll	ns	ns	ns	–	0.4497	0.6033	0.6033
Pheophytin	0.4062	ns	ns	0.4497	–	−0.3845	−0.3738
Chlorophyll/ pheophytin	ns	ns	ns	0.6033	−0.3845	–	0.9902
A-coefficient	ns	ns	ns	0.6111	−0.3738	0.9902	–

organic matter in the water and the reconstruction of trophic relationships [26, 51]. An important condition for success is to minimize nutrient loads from the catchment. Lake Długie is a perfect example of a reservoir in which the decrease of primary production has improved the self-purification of water (by an approx. 70% reduction of chlorophyll). This was possible due to the reduction of the POC associated with the algae production. Due to the decrease in primary production, the POC amount decreased by 40% and was maintained over the subsequent years of research. As a result, there was also a reduction in the amount of pheophytin by about 60%. A return to balanced production-distribution was visible in an over threefold reduction in the chlorophyll/pheophytin ratio after restoration. This was confirmed by the reduced values of the A-coefficient within a few years after restoration. After restoration, the relationships between parameters changed. The most important change was the lack of a positive chlorophyll/pheophytin dependence.

Polyaluminum chloride may affect the reduction in the amount of molecular weight (suspended) and the organic matter dissolved in water [52–54]. There was a 20% reduction in DOC concentration observed only after completing the Lake Długie restoration. The fraction of DOC is a mixture of organic compounds with different molecular structures and, thus, different chemical and functional properties [55, 56]. The role of the DOC fraction and its share in the biogeochemical cycles of elements affecting the production capacity of lakes and capacity for self-purification is relatively widely known [57]. As a labile fraction, DOC is responsible for transporting forms of elements connected with it. These forms include both biogenic elements and all forms of toxic heavy metals that limit the self-purification of lake waters through an inhibitory effect on the biological processes [58]. Despite the potential risk of release of undesirable biogenic elements and metals [59], no increase in the amount of DOC and primary production was observed. Research carried out in Lake Długie will analyse the changes occurring during the use of various restoration techniques in the same reservoir. The decrease in DOC was due to a decrease in primary production and better conditions for the UV degradation of DOC compounds after increasing the transparency of water (POC reduction). Preliminary studies performed to assess the quality of dissolved organic matter using UV-VIS spectrophotometry showed significant changes in the optical properties of organic matter (compounds included in the DOC). They confirmed the intensification of mineralization of organic matter in the near-bottom water during aeration with destratification [22]. Immediately after using the phosphorus inactivation method and the associated reduction in primary production, the amount of organic carbon suspended in water and the dissolved fraction decreased – mainly easy biodegradable compounds [24, 60].

Another reservoir, in which the inactivation of phosphorus affected the reduction of suspended organic matter, was Lake Klasztorne Górne. After the first dosage of PAX 18, the decrease in POC occurred by over 45%, on average. The declining [POC/DOC]*100 [%] ratio and the decreasing trend of chlorophyll indicate that a further reduction in POC was still possible. The simultaneous decrease in the average chlorophyll/pheophytin ratio indicated that in the last year of restoration, an acceleration of degradation processes occurred, although the positive relationship between chlorophyll and pheophytin did not confirm this. The biomanipulation introduced in Lake Klasztorne Górne has a significant importance for the reduction of algae biomass during restoration. The biological method used in this lake involved reconstructing the species composition of ichthyofauna (the introduction of predatory species), thus enabling control of the size and abundance of zooplankton [61–63]. This, in turn, affects the possibility of reducing the algae amount [64, 65] whose composition also changes under the influence of inactivation [40]. It is very important to reduce primary production and support the positive effects of chemical inactivation using coagulants.

A significant part of the contaminants in surface water pass into sediment as a result of the processes of sedimentation. Elements associated with the organic matter may be deposited in the sediment, but under suitable conditions, they may return to the water column [66]. In a polymictic lake (constantly mixing), there are favourable conditions for carrying suspended organic matter from sediment to water. A higher temperature at near bottom can improve the mineralization of organic carbon [67]. The research conducted in the polymictic Lake Wolsztyńskie during restoration did not show a reduction in the amount of organic matter. The results confirmed that it is difficult to achieve a balance of production and distribution of organic matter in this type of lake. However, the increase in the activity of decomposition processes may be confirmed by the increase in DOC concentration in the last year of the restoration. In addition, the increase in the concentration of pheophytin during the experiment and the negative relationship between this pigment and the chlorophyll/pheophytin ratio indicate the increasing decomposition processes of the produced organic matter. Changes in the quantity and quality of dissolved organic matter can be associated both with the direct impact of coagulant used [68] or be the result of changing conditions of biotic and abiotic factors during restoration [51]. The research showed that no changes in the quality of DOC compounds were observed in Lake Wolsztyńskie [24]. The permanent inflow of hardly decomposed DOC with the Dojca River waters and remaining high primary production inhibits these changes.

By observing the relationship between the amount of organic carbon, chlorophyll and pheophytin (a chlorophyll decay product) and observing these changes during and after restoration, the self-purification processes can be monitored. Studies carried out in lakes subjected to different catchment influences and different dynamics of water masses and restored using different methods have shown that:

1. In a bradymictic urban lake, we can decrease primary production and organic matter content and thus return back to balanced production-decomposition, using inactivation of phosphorus (PAX 18) at the right time, aided by long-term aeration.
2. An increase in the rate of self-purification processes is possible through the proper selection of the method of technical and biological restoration – by using chemical inactivation and biomanipulation methods and influencing the production rate and the biomass of algae – which is a supplier of hardly biodegradable organic matter.

3. In a polymictic lake which is constantly supplied with nutrients and organic matter from its catchment, restoration of the self-purification ability of waters is difficult using only one restoration method (inactivation).

5 Conclusion

This study complements a gap in knowledge of the properties of organic matter and the possibility of its transformation under the influence of changing environmental conditions in restored lakes and its role in the aquatic environment. This research will also expand the knowledge of the mechanisms modifying the resources of organic matter in lake water. By understanding the issues of the dynamics of organic matter in the water of restored lakes, the changes in self-purification processes can be observed. This knowledge should be used in evaluating the functioning of lake ecosystems. The study of the possibility of transforming organic matter under the influence of changing environmental conditions in the restored lakes may help to explain its role in the transport of nutrients and toxic substances in the aquatic environment. The obtained information may be used for modelling the qualitative and quantitative characteristics of organic matter, including the impact of a number of factors related to the changes in the rate of self-purification processes in lakes subjected to restoration procedures.

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References

1. Chróst RJ, Siuda W (2006) Microbial production, utilization, and enzymatic degradation of organic matter in the upper trophogenic layer in the pelagial zone of lakes along a eutrophication gradient. Limnol Oceanogr 51:749–762
2. Sitarek M, Napiórkowska-Krzebietke A, Mazur R et al (2017) Application of effective micro-organisms technology as a lake restoration tool - a case study of Muchawka Reservoir. J Elem 22(2):529–543. https://doi.org/10.5601/jelem.2016.21.2.1196
3. Górniak A (2016) Export of nutrients from the catchment of the upper Szeszupa River (drainage basin of the Neman River) and its seasonality. Limnol Rev 16(4):213–219. https://doi.org/10.1515/limre-2016-0023
4. Łopata M, Czerniejewski P, Wiśniewski G et al (2017) The use of expanded clay aggregate for the pretreatment of surface waters on the example of a tributary of Lake Klasztorne Górne in Strzelce Krajeńskie. Limnol Rev 17(1):3–9. https://doi.org/10.1515/limre-2017-0001
5. Minor E, Stephens B (2008) Dissolved organic matter characteristics within the Lake Superior watershed. Org Geochem 39:1489–1501
6. Parszuto K, Tandyrak R, Galik J (2009) Qualitative and quantitative characteristics of organic matter in the of a small reservoir. Arch Environ Prot 35(3):59–71
7. Siuda W, Chróst RJ (2002) Decomposition and utilization of particulate organic matter by bacteria in lakes of different trophic status. Pol J Environ Stud 11(1):53–65

8. Meriluoto J, Blaha L, Bojadzija G et al (2017) Toxic cyanobacteria and cyanotoxins in European waters - recent progress achieved through the CYANOCOST action and challenges for further research. Adv Oceanogr Limnol 8(1):161–178. https://doi.org/10.4081/aiol.2017.6429
9. Brzozowska R, Dunalska J, Filipkowska Z (2013) Seasonal changes of total carbon, nitrogen and phosphorus concentration in the near-bottom and interstitial water of three deep, stratified urban lakes. GJPAAS 1:736–740
10. Cotner JB, Biddanda BA (2002) Small players, large role: microbial influence on biogeochemical processes in pelagic aquatic ecosystems. Ecosystems 5:105–121
11. Ramanan R, Byung-Hyuk K, Dae-Hyun C et al (2016) Algae–bacteria interactions: evolution, ecology and emerging application. Biotechnol Adv 34:14–29
12. Mladenov N, Sommaruga R, Morales-Baquero R et al (2011) Dust inputs and bacteria influence dissolved organic matter in clear alpine lakes. Nat Commun 2:405. https://doi.org/10.1038/ncomms1411
13. Bonnett SA, Ostle N, Freeman C (2006) Seasonal variations in decomposition processes in a valley bottom riparian peatland. Sci Total Environ 370(2–3):561–573
14. Brzozowska R, Gawrońska H (2009) The influence of a long term artificial aeration on the nitrogen compounds exchange between bottom sediments and water in Lake Długie. Oceanol Hydrobiol Stud 38(1):1–7
15. Huser B, Berezonik P, Newman R (2011) Effects of alum treatment on water quality and sediment in the Minneapolis Chain of Lakes, Minnesota, USA. Lake Reserv Manage 27 (3):220–228. https://doi.org/10.1080/07438141.2011.601400
16. Dunalska JA, Wiśniewski G, Mientki C (2007) Assessment of multi-year (1956–2003) hypolimnetic withdrawal from Lake Kortowskie, Poland. Lake Reserv Manage 23:377–387
17. Dunalska JA, Górniak D, Jaworska B et al (2012) Effect of temperature on organic matter transformation in a different ambient nutrient availability. Ecol Eng 49:27–34
18. Grochowska J, Gawrońska H (2004) Restoration effectiveness of a degraded lake using multi-year artificial aeration. Pol J Environ Stud 13(6):671–681
19. Grochowska J, Brzozowska R, Łopata M (2013) Durability of changes in phosphorus compounds in water of an urban lake after application of two reclamation methods. Water Sci Technol 68(1):234–239
20. Grochowska J, Brzozowska R, Parszuto K (2014) The influence of different recultivation techniques on primary production processes in a degraded urban lake. Oceanol Hydrobiol Stud 43(3):211–218
21. Łopata M, Gawrońska H, Jaworska B et al (2013) Restoration of two shallow, urban lakes using the phosphorus inactivation method – preliminary results. Water Sci Technol 68 (10):2127–2135
22. Parszuto K, Grochowska J (2005) Influence of artificial aeration with destratification on the selected indices of organic matter content variability in Lake Długie water. Inżynieria Ekologiczna 12:219–220
23. Welch EB, Cooke GD (1999) Effectiveness and longevity of phosphorus inactivation with alum. Lake Reserv Manage 15(1):5–27. https://doi.org/10.1080/07438149909353948
24. Parszuto K, Głażewski R, Łopata M et al (2017) The use of the optical properties of dissolved organic matter to evaluation of the lakes restoration effects (in polish). In: Wiśniewski R (ed) Ochrona i rekultywacja jezior. Polskie Zrzeszenie Inżynierów i Techników Sanitarnych, Toruń, pp 99–116
25. Parszuto K (2003) Dynamics of organic carbon in lake waters subjected to various methods of restoration (in polish). Dissertation, UWM Olsztyn
26. Parszuto K, Głażewski R (2004) The relationship between the contents of particulate (POC) and dissolved (DOC) organic carbon in the waters of Lake Długie and Lake Kortowskie recultivated with different methods. Limnol Rev 4:193–200
27. Łopata M, Wiśniewski G (2010) Studies of environmental conditions of Lake Wolsztyn in terms of its protection and reclamation (in polish). Typescript, UWM Olsztyn

28. Jańczak J (ed) (1996) Atlas of Polish lakes (in polish). Instytut Meteorologii i Gospodarki Wodnej, Bogucki Wydawnictwo Naukowe, Poznań
29. Czerniejewski P, Czerniawski R (2008) Final report on hydrochemical and hydrobiological studies carried out on the lake Klasztorne Górne with the assessment of restoration opportunities. http://www.archiwum.strzelce.pl/usr_files/8dfb354dab1e6e738750c90f00cf5a5d_3/raport_koncowy.pdf. Accessed 12 Dec 2014
30. Łopata M, Wiśniewski G, Czerniawski R et al (2013) Possibilities of restoration Lake Klasztorne Górne in Strzelce Krajeńskie. In: Wiśniewski R (ed) Ochrona i rekultywacja jezior. Polskie Zrzeszenie Inżynierów i Techników Sanitarnych, Toruń, pp 135–143
31. https://www.google.com/maps/@52.9821495,17.9160747,246595m/data=!3m1!1e3. Accessed 23 Aug 2018
32. IRS Olsztyn (1958) Mapa batymetryczna i opracowanie danych morfometrycznych Jeziora Długiego (Bathymetric map and elaborating of morphometric data of Długie Lake)
33. WIOŚ Poznań (2004) Raport o stanie środowiska w Wielkopolsce w roku 2003 (Report on the state of the environment in Wielkopolska in 2003). Biblioteka Monitoringu Środowiska, Poznań
34. Standard methods for the examination of water and wastewater (1985) American Public Health Association, Washington, DC
35. Parszuto K (2008) POC, chlorophyll and pheophytin a vertical distribution in restored Kortowskie Lake (Poland). Limnol Pap 3:79–88
36. Søndergaard M, Jeppesen E (2007) Anthropogenic impacts on lake and stream ecosystems, and approaches to restoration. J Appl Ecol 44:1089–1094
37. Jarosiewicz A (2007) The self-purification process in river ecosystems (in polish). Słupskie Prace Biologiczne 4:27–41
38. Ostroumov SA (2000) The concept of aquatic biota as a labile and vulnerable component of the water self-purification system. Dokl Biol Sci 372:286–289. Translated from Dokl Akad Nauk 372(2):279–282
39. Ostroumov SA (2010) Biocontrol of water quality: multifunctional role of biota in water self-purification. Russ J Gen Chem 80(13):2754–2761
40. Jaworska B, Gawrońska H, Łopata M (2009) Changes in the phytoplankton community of a lake restored with phosphorus inactivation (Lake Głęboczek, northern Poland). Oceanol Hydrobiol Stud 38(1):93–101
41. Kangabam RD, Boominathan SD, Govindaraju M (2015) Ecology, disturbance and restoration of Loktak Lake in Indo-Burma Biodiversity Hotspot-an overview. NeBIO 6(2):9–15
42. Żelazo J (2006) Renaturalization of rivers and valleys (in polish). Infrastruktura i ekologia terenów wiejskich 4:1):11–1):31
43. Dąbrowska J (2008) Methods of reclamation of water reservoirs. Acta Sci Pol Formatio Circumiectus 7(1):63–79
44. Osuch E, Podsiadłowski S (2012) Efficiency of pulverizing aeration on Lake Panieńskie. Limnol Rev 12(3):139–145
45. Anderson MA, Berkowitz J (2010) Aluminum polymers formed following alum treatment of lake water. Chemosphere 81(7):832–836
46. Łopata M, Gawrońska H (2008) Phosphorus immobilization in Lake Głęboczek following treatment with polyaluminium chloride. Oceanol Hydrobiol Stud 37(2):99–105
47. Anttila S, Ketola M, Kuoppamäki K et al (2013) Identification of a biomanipulation - driven regime Shift in Lake Vesijärvi: implications for lake management. Freshw Biol 58:1494–1502. https://doi.org/10.1111/fwb.12150
48. Chęciński G, Czernaś K, Szczurowska A et al (2014) Changes in the dominance structure of phytoplankton in a pond exposed to biological reclamation treatments. In: Abstracts of the 33th International Conference of the Polish Phycological Society. Gdynia-Cetniewo, 19–22 May. pp 62–63
49. Górniak A, Zieliński P, Grabowska M et al (2006) Results of biomanipulation of humic reservoir after four years of study. Verh Int Ver Limnol 29:2059–2062

50. Jeppesen E, Søndergaard M, Lauridsen TL et al (2012) Biomanipulation as a restoration tool to combat eutrophication: recent advances and future challenges. Adv Ecol Res 47:411–488
51. Grochowska JK, Brzozowska R, Parszuto K et al (2017) Modifications in the trophic state of urban lake, restored by different methods. J Elem 22(1):43–53. https://doi.org/10.5601/jelem.2016.21.2.1058
52. Gumińska J (2011) The influence of transformation of aluminum forms on water treatment using polymerized coagulants (in polish). Ochrona Środowiska 33(2):17–21
53. Sperczyńska E, Dąbrowska L, Wiśniowska E (2014) Removal of turbidity, colour and organic matter from surface water by coagulation with polyaluminium chlorides and with activated carbon as coagulant aid. Desalin Water Treat 2014:1–6. https://doi.org/10.1080/19443994.2014.989634
54. Wilmański K, Trzebiatowski M (2009) Removal of organic substances from surface water in processes of coagulation and sorption on powdered active carbon (in polish). Ochrona Środowiska 31(4):39–42
55. Barreto SRG, Nozaki J, Barreto WJ (2003) Origin of dissolved organic carbon studied by UV-VIS spectroscopy. Acta Hydrochim Hydrobiol 31(6):513–518. https://doi.org/10.1002/aheh.200300510
56. Głażewski R, Wójcik I (2009) Preliminary examinations of the functional properties of dissolved organic matter in selected lakes of the district of Ełk. Limnol Rev 7(2):35–40
57. Wood CHM, Al-Reasia HA, Smith DS (2011) The two faces of DOC. Aquat Toxicol 105(3–4):3–8
58. Bogatov VV, Bogatova LV (2009) Heavy metal accumulation by freshwater hydrobionts in a mining area in the south of the Russian Far East. Russ J Ecol 40(3):187–193
59. Seekell DA, Lapierre JF, Karlsson J (2015) Trade-offs between light and nutrient availability across gradients of dissolved organic carbon concentration in Swedish lakes: implications for patterns in primary production. Can J Fish Aquat Sci 72(11):1663–1671
60. Głażewski R, Parszuto K (2002) Optical properties of dissolved organic matter (DOM) in the recultivated lakes of Olsztyn. Limnol Rev 2:137–142
61. Albright MF, Harman WH, Tibbits WT et al (2004) Biomanipulation: a classic example in a shallow eutrophic pond. Lake Reserv Manage 20(3):181–187. https://doi.org/10.1080/07438140409354242
62. Gołdyn R (2007) Biomanipulation in water reservoirs as a method of restoration (in polish). In: Wiśniewski R, Piotrowiak J (eds) Ochrona i rekultywacja jezior. Polskie Zrzeszenie Inżynierów i Techników Sanitarnych, Toruń, pp 65–75
63. Prejs A, Pijanowska J, Koperski P et al (1997) Food web manipulation in a small, eutrophic Lake Wirbel, Poland: long-term changes in fish biomass and basic measures of water quality. A case study. Hydrobiologia 342/343:383–386
64. Perrow MR, Meijer ML, Dawidowicz P et al (1997) Biomanipulation in shallow lakes: state of the art. Hydrobiologia 342/343:355–365
65. Peretyatko A, Teissier S, De Backer S et al (2012) Biomanipulation of hypereutrophic ponds: when it works and why it fails. Environ Monit Assess 184:1517–1531. https://doi.org/10.1007/s10661-011-2057-z
66. Maloney KO, Morris DP, Moses CO et al (2005) The role of iron and dissolved organic carbon in the absorption of ultraviolet radiation in humic lake water. Biogeochemistry 75:393–407
67. Gudasz C, Bastviken D, Steger K et al (2010) Temperature-controlled organic carbon mineralization in lake sediments. Nature 466:478–481. https://doi.org/10.1038/nature09186
68. Korshin G, Chow CW, Fabris R et al (2009) Absorbance spectroscopy-based examination of effects of coagulation on the reactivity of fractions of natural organic matter with varying apparent molecular weights. Water Res 43(6):1541–1548. https://doi.org/10.1016/j.watres.2008.12.041

Hypertrophic Lakes and the Results of Their Restoration in Western Poland

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Contents

Abstract In this chapter we present our research on hypertrophic lakes and the identified changes in their water quality, phytoplankton and zooplankton, under the influence of restoration activities. Most of these lakes have been used for recreation, and – as a result of the deterioration of water quality – this option has lost. The average annual concentration of total phosphorus in almost all the lakes was higher than 100 μg/L, and the mean concentration of chlorophyll a was in the range of 40–172 μg/L, thus exceeding the limits for hypertrophy. To improve the water quality, so-called sustainable restoration was initiated, combining physical, chemical, and biological methods. Currently, we are documenting changes in the lakes Maltańskie, Uzarzewskie, Swarzędzkie, Rusałka, and Konin to verify the results of in-lake treatment. In the first four lakes, restored by phosphorus

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inactivation, oxygenation of water overlying the sediments, biomanipulation, and nitrate treatment of the hypolimnion, a gradual improvement of water quality was observed. This was manifested in phytoplankton community structure changes (less *Cyanobacteria*) as well as reduction in nutrient concentrations. In the case of Lake Konin, the inefficiency of EMs in lake restoration was noted.

Keywords Hypertrophic lakes · Lake restoration · Nutrients · Phytoplankton · Western Poland · Zooplankton

1 Introduction

There is considerable nomenclature confusion in the literature concerning lakes that have been overloaded with nutrients. They are often called hypertrophic, hypereutrophic, or polytrophic, and these terms are treated synonymously [1–4]. Sometimes the term saprotrophic is added for hypertrophic lakes polluted with sewage. Olszewski [5] made a clear separation of these terms many years ago; however, because this work was published in Polish, it never appeared in the global dataset and consequently was not widely known. In Olszewski's view the sequence of harmonic lakes ends with eutrophic lakes. Lakes that have been excessively enriched with nutrients should therefore be treated as disharmonic lakes. The first level beyond eutrophy is polytrophy. Polytrophic lakes display an increased supply in nutrients from catchments, due to their human transformation. The entire external load of nutrients in the case of such lakes is involved in primary production. Hypertrophic lakes are characterized by a surplus of mineral forms of nitrogen and phosphorus in the water column, not used in primary production even though it is very high. Thus the limiting factors are microelements, light, mineral carbon, etc. The surface water layer is oversaturated with oxygen. Saprotrophic lakes have a surplus of organic matter in the water column, which is brought with untreated sewage. The process of organic matter decomposition is often incomplete and more intensive than primary production. This production is inhibited due to the excess of metabolites and is reduced until total collapse in extreme cases. Microbial decomposition of organic matter leads to a sharp decrease of oxygen content and may finish in its total depletion.

Nowadays, we no longer have saprotrophic lakes in Poland, while polytrophic and hypertrophic lakes are still very common. According to the EU Water Framework Directive, these lakes – determined by cyanobacterial water bloom, low transparency, and high concentration of phosphorus and nitrogen in the water column – are classified as belonging to poor or bad ecological status. It is necessary to implement remedial programs for them, using protective and restoration measures to achieve a good ecological status.

In the following parts of this chapter, we present our research on hypertrophic lakes and identify changes in their water quality, phytoplankton and zooplankton, under the influence of restoration activities.

Table 1 Characteristics of studied lakes

No	Name of the lake	Coordinates	Max depth (m)	Mean depth (m)	Surface area (ha)	Volume (10^3 m^3)
1	Nowowiejskie	52°12′26″N 15°53′40″E	2.0	1.4	31.0	450
2	Łekneńskie	52°50′35″N 17°17′32″E	2.8	1.6	85.2	1,300
3	Konin	52°23′06″N 15°52′27″E	4.2	3.1	87.7	2,700
4	Maltańskie	52°24′08″N 16°58′12″E	5.0	3.1	64.0	2,000
5	Raczyńskie	52°08′43″N 17°09′57″E	5.8	2.8	84.4	2,360
6	Rogoźno	52°44′09″N 17°00′58″E	5.8	3.0	139.0	4,170
7	Swarzędzkie	52°24′49″N 17°03′54″E	7.2	2.6	93.7	2,400
8	Uzarzewskie	52°26′54″N 17°08′01″E	7.3	3.4	10.6	360
9	Rusałka	52°25′36″N 16°52′42″E	9.0	1.9	36.7	697
10	Błędno	52°14′06″N 15°54′09″E	9.6	3.5	742.5	26,000
11	Budziszewskie	52°41′51″N 17°06′35″E	14.0	4.8	163.0	7,840

2 Study Site

Most of the studied hypertrophic lakes are shallow, completely mixed in summer (No 1–6, Table 1) or partly stratified with epi- and metalimnion (No 7–10). Only one of them, namely, Lake Budziszewskie, is a dimictic lake with a distinct hypolimnion.

They are located no more than 70 km from Poznań, which is the capital of the Wielkopolska Region (Fig. 1). Most of them are small, with an area of less than 1 km^2, and only three are in the range of 1–10 km^2 (Table 1). Half of them are either urban lakes or towns adjacent to a part of their shoreline (No 4–7 and 9–10). The characteristics of the lakes are presented separately for non-restored lakes and lakes subjected to restoration treatments, following changes taking place under their influence.

3 Material and Methods

Most of the studied lakes have been used for recreation. As a result of the deterioration of water quality, this function was lost. The presented research was conducted largely due to the need of local governments who wished to restore the lakes to their

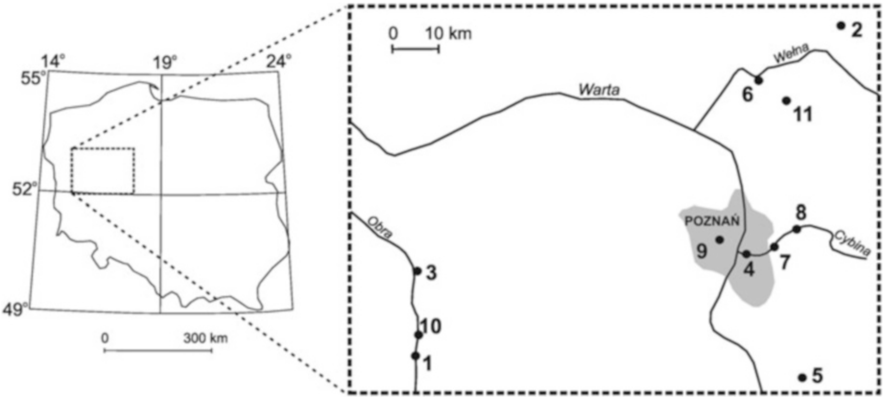


Fig. 1 The distribution of studied lakes in Western Poland

former condition. Therefore, lakes intended for recreation were examined to determine their condition and to propose possible protective and restoration measures aimed at restoring their lost recreational function. After the restoration began, the study was continued to assess whether it yielded the expected results. The studies cover different periods, depending on when particular local authorities began to take an interest. The lakes were studied from 6 to 12 times a year to determine the seasonal variation of the studied parameters. Secchi depth transparency, temperature, oxygen content, conductivity, and pH were measured in the field, using a YSI meter. Water samples were also taken from the surface water layer for laboratory analyses, to determine the concentration of nitrogen, phosphorus, chlorophyll a, phytoplankton, and zooplankton. Samples for nutrient analyses were fixed with chloroform, for biological analyses with Lugol's solution, those for chlorophyll a were not fixed. Laboratory analyses were carried out according to APHA [6]. Total nitrogen was the sum of mineral and organic nitrogen forms. The concentration of ammonium nitrogen was determined by a method with Nessler's reagent, nitrate nitrogen by a method with sodium salicylate, nitrite nitrogen with sulfanilic acid, and organic nitrogen by Kjeldahl's method. The concentration of orthophosphates was analyzed by applying a method with ascorbic acid and total phosphorus by a method with ascorbic acid after mineralization. Samples for the concentration of chlorophyll a analyses were filtered on GF/C using Coli 5 and determined by the spectrophotometric method after extraction with 90% acetone. The qualitative and quantitative compositions of phytoplankton and zooplankton were analyzed using the light microscope Olympus CX 21 LED in a Sedgewick-Rafter chamber (0.46 mL volume). Statistical analyses were performed with STATISTICA 12.5 software.

4 Results and Discussion

4.1 Characteristics of Hypertrophic Lakes

In the Wielkopolska Region, located in Western Poland, there are about 800 lakes, of which only 81 are under monitoring of the state services. Of these, in recent years 38, approximately 46%, were designated as poor and of bad ecological status. Even more lakes are unmonitored; they usually have a small area and are mainly located in agricultural areas. Diffuse pollution from agricultural areas is one of the main sources of deterioration of water quality in lakes of the Wielkopolska Region [7]. The other important source of pollution originates from point sources. Until recently, these were mainly municipal sewage works. Currently, according to the Water Law from 2001, sewage cannot be discharged into lakes, even after thorough treatment. However, it happens that treated sewage is discharged to streams that reach the lakes. Many urban lakes are polluted by stormwater runoff, which carries large loads of mineral and organic suspensions, heavy metals, and nutrients, even after treatment in settling tanks with separators. These devices tend to homogenize the quality of discharged rainwater, to a small extent reducing the incoming pollution load [8–10]. Pollutants discharged to lakes from point sources are deposited to a large extent in bottom sediments, from which they are released for many years after sewage diversion. They constitute a source of internal nutrient loading for lakes, which in some cases may be more important than external loading [11, 12].

Non-restored lakes were selected for the characterization of hypertrophic lakes as well as those currently under restoration, but results from the period before restoration were used. The lakes were examined in various years, dependent on the plans related to their restoration (Table 2). The earliest results of the initial research concern Lake Maltańskie, which suggests that restoration began there the earliest.

The average annual concentration of total phosphorus (TP) in almost all lakes was higher than 100 μg/L, which according to the OECD [13] was the limit for hypertrophy. Only Lake Rusałka had a lower value, which was also in line with the lower values of soluble reactive phosphorus (SRP). It was only in this lake that SRP was occasionally not detected. According to Olszewski [5], this indicates the affiliation of Rusałka Lake to polytrophic lakes. Differences in concentrations of both TP and SRP between individual lakes were very large, indicating a varying degree of eutrophication. The highest values were found in Lake Maltańskie, which is an urban lake, polluted by stormwater from residential and industrial areas [8, 10, 14]. The total nitrogen (TN) concentrations in all lakes were very high. The average annual value never fell below 2 mg N/L. The content of mineral nitrogen forms never reached zero, which confirms the hypertrophic nature of these lakes.

The mean concentration of chlorophyll a was in the range of 40–172 μg/L, and the highest reached 306 μg/L. The lowest concentrations were found in Lake Rusałka, which is due to relatively low concentrations of phosphorus, while the highest were found in Lake Nowowiejskie, located in an extensive agricultural catchment. The standard value of the mean concentration of chlorophyll a for

Table 2 Mean annual concentrations of TP, SRP, TN, and chlorophyll a and mean annual water transparency (SD) together with min-max values

Lake	Year	TP (μg/L)			SRP (μg/L)			TN (mg/L)			Chlorophyll-a (μg/L)			SD (m)		
		Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max
Maltańskie[a]	1997–2000	193–917	70	1,830	123–762	260	1,650	2.30–3.41	1.86	4.44	58.7–104.4	29.1	177.8	0.45–0.69	0.30	1.00
Uzarzewskie	2005	162	98	240	77	52	87	2.88	0.88	4.62	68.7	17.1	167.9	0.73	0.45	1.00
Rusałka	2005	56	21	130	8	0	11	2.15	2.00	2.30	40.6	25.5	66.3	0.71	0.60	0.80
Swarzędzkie	2011	138	39	226	83	16	131	5.24	3.21	7.19	103.0	28.2	211.8	0.75	0.50	1.20
Konin[a]	2011–2012	190–395	106	1,177	90–133	35	159	3.13–3.71	2.21	5.03	64.1–139.9	31.4	189.7	0.30–0.31	0.20	0.60
Łekneńskie	2011/2012	125	63	236	68	8	134	3.95	2.97	5.28	103.2	13.7	223.2	0.35	0.20	0.60
Błędno[a]	2013–2014	205–270	134	460	137–138	61	203	2.85–3.08	2.49	3.58	101.5–103.6	86.6	122.1	0.48–0.67	0.40	1.10
Nowowiejskie[a]	2013–2014	216–363	149	435	124–260	54	434	2.54–3.50	1.48	4.84	93.3–172.9	61.0	306.1	0.37–0.50	0.15	0.70
Raczyńskie	2015	111	44	153	33	8	61	4.21	3.40	4.93	125.5	6.4	251.9	0.58	0.25	1.10
Rogoźno	2017	146	77	233	68	16	124	3.04	2.32	3.89	89.5	52.8	148.7	0.64	0.45	1.10
Budziszewskie	2017	106	58	142	55	29	75	3.65	2.82	4.70	80.3	58.5	102.0	0.60	0.35	0.95

[a]Range of means from individual years of research

hypertrophy according to the OECD [13] is 25 µg/L, so all the studied lakes meet this condition. However, according to the maximum concentration of chlorophyll a, which should be higher than 75 µg/L, Lake Rusałka again does not qualify for hypertrophic status. Moreover, the limit value given for hypertrophy by Welch [15] amounting to 100 µg/L was not exceeded only in Rusałka Lake. The minimum Secchi depth in all studied lakes was below 0.7 m, i.e., the threshold value for hypertrophy according to the OECD [13]. The average value was also lower than 1.5 m, so all lakes qualified for hypertrophy according to this criterion.

Trophic state index (TSI) calculated according to Carlson's equations [16–18] was usually the highest for TP (Table 3). However, in some cases value calculated based on the chlorophyll a values (lakes Rusałka, Swarzędzkie, Raczyńskie) or Secchi depth values was higher (Lake Łekneńskie). It is typical that the TSI (TP) predominates in hypertrophic lakes because phosphorus is not a limiting factor there. When chlorophyll dominates, it indicates the presence of large organisms, usually colony-forming cyanobacteria [18]. Only the value of the TSI (Chl) for all lakes exceeded 70, which is typical for hypertrophy. The remaining TSI for some lakes was lower than 70, indicating strong eutrophy (polytrophy). Only in the case of Lake Rusałka was the average TSI value less than 70, confirming the polytrophic character of this lake. The highest average value of 84.4 was found in Lake Nowowiejskie, with a large agricultural catchment. According to Carlson and Simpson [18], such a high TSI, exceeding 80, is characteristic of lakes with algal scum and possible fish kills in summer.

Phytoplankton of lakes was dominated by cyanobacteria. They accounted for 56.6% in Lake Uzarzewskie from May to October and up to 98.5% of the total phytoplankton biomass in Lake Konin. Diatoms and cryptophytes accompanied cyanobacteria in spring, while dinoflagellata were common in summer, in a smaller amount (Fig. 2). The phytoplankton biomass was quite variable both throughout the year, in subsequent years of research, and above all between individual lakes. The largest average biomass was found in Lake Konin (107 mg/L) and the maximum in Lake Nowowiejskie (170 mg/L).

Table 3 The values of trophic state index (acc. to Carlson 1977, Kratzer, Brezonik 1981)

Lake	Year	TSI (SD)	TSI (TP)	TSI (TN)	TSI (Chl)	TSI
Maltańskie[a]	1997–2000	67.6–75.6	80.1–102.5	66.5–72.2	72.1–78.3	71.6–82.2
Uzarzewskie	2005	65.6	81.2	75.8	70.1	75.7
Rusałka	2005	66.2	61.2	65.5	70.8	65.9
Swarzędzkie	2011	68.6	79.5	78.4	81.5	77.0
Konin[a]	2011–2012	78.6–80.0	77.7–89.5	70.9–73.4	71.7–80.1	74.7–80.8
Łekneńskie	2011/2012	83.2	80.9	74.3	79.7	79.5
Błędno[a]	2013–2014	73.2–73.2	81.6–81.7	69.6–70.7	75.3–76.6	74.9–75.6
Nowowiejskie[a]	2013–2014	83.2–87.4	86.4–92.2	67.9–92.2	79.9–85.3	79.4–84.4
Raczyńskie	2015	78.6	70.0	75.2	79.6	75.9
Rogoźno	2017	71.5	80.3	70.5	75.8	74.5
Budziszewskie	2017	70.0	77.4	73.1	73.5	73.5

[a]Range of means from individual years of research

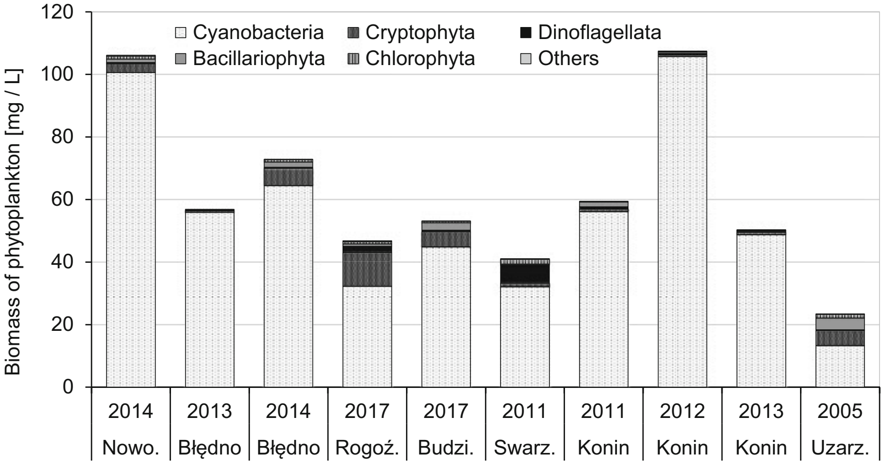


Fig. 2 Phytoplankton biomass composition of selected hypertrophic lakes (Nowo., Lake Nowowiejskie; Rogoź., Lake Rogoźno; Budzi., Lake Budziszewskie; Swarz., Lake Swarzędzkie; Uzarz., Lake Uzarzewskie)

4.2 *Restoration of Hypertrophic Lakes in Wielkopolska Region*

Hypertrophic lakes are usually shallow and polymictic, although there are also thermally stratified lakes among them. The main source of nutrients in shallow lakes in summer is bottom sediments, located within reach of the epilimnetic zone, i.e., the so-called active bottom. The low water transparency results in a complete decline of submerged macrophytes. Such lakes are in the so-called turbid state [19, 20]. The main purpose of their restoration is to obtain a clear water state, i.e., to remove cyanobacterial water blooms and to increase water transparency. There are numerous methods for the restoration of lakes, both technical and biological [21–23]. Phosphorus inactivation, oxygenation of water overlying the sediments, and biomanipulation are among the most commonly used. Many years of experience have shown that even very intensive restoration may produce only limited effects. Usually after a short period of water quality improvement, the ecosystem returns to its previous turbid state. Sometimes this is related to the omission of one of the nutrient sources in the restoration process, but most often it is related to the stability of the ecosystem, which maintains the existing structure and function due to feedback mechanisms that attempt to remove or weaken changes caused by restoration [24]. The old approach to restoration preferred the use of intense methods that very drastically interfered with the ecosystem so as to force ecological resilience [21, 22, 25]. Nowadays, the ecological approach to restoration is becoming more and more popular, paying attention to the preservation of biodiversity as an important element affecting the adaptation of the ecosystem in the face of change [24]. This approach uses less invasive methods, closer to nature. Frequently combined methods are used simultaneously, i.e., physical, chemical, and biological methods [26].

In recent years in Western Poland, so-called sustainable restoration was preferred. This involves a variety of approaches to ecological restoration. It uses combined physical, chemical, and biological methods but only to the extent that is necessary for the gradual reconstruction of the ecosystem [27]. One of the first methods was the oxygenation of water overlying the sediments with a wind-driven aerator [28, 29]. Another method is the inactivation of phosphorus in the water column using magnesium chloride or iron sulfate, i.e., compounds present in the lakes but in summer in too small quantities. The doses of these compounds were small and adjusted exactly to the conditions prevailing in the ecosystem. They inactivated phosphates, only slightly coagulating the suspensions [30, 31]. The third method is biomanipulation, treated as supportive, consisting of stocking the lakes with the fry of predatory species [30, 32]. Another innovative method is the direction of spring waters containing high concentrations of nitrates to the deoxygenated bottom of the lake. This method increased the redox potential in the sediment-water interface, preventing the release of phosphorus from the bottom to the water column [27, 33]. These methods, being innovative, still have insufficient research documentation. We are trying to fill this gap by documenting changes in the lake ecosystems under the influence of sustainable restoration methods.

Not all the lakes discussed in the previous chapter have been restored. This was usually due to financial considerations. Two of the studied lakes, namely, Lake Raczyńskie and Lake Rogoźno, began restoration treatment in the current year, 2018. They will not be discussed in this section, as results of their monitoring are still only at a preliminary stage. Lake Maltańskie has been undergoing restoration for the longest time, for 13 years. Uzarzewskie lake has been under restoration for a slightly shorter time (12 years), and the rest of the lakes (Rusałka, Swarzędzkie, and Konin) have been treated for several years.

Lake Maltańskie It is an artificial water reservoir, created in the 1950s, and newly formed during the 1980s. It has a specific hydrological regime, associated with its function for water sport, namely, for canoeing and rowing competitions. Every 4 years the water is drained in autumn and refilled in spring. In the meantime, repair and maintenance works are carried out on the regatta track equipment. Because the lake is supplied with water of the River Cybina, which is very rich in nutrients, it has been in a hypertrophic condition since its filling in 1990 [34]. The first attempt to improve the lake water quality took place in 1993, when it was stocked with fry of predatory fish. Its effects were visible mainly in the first year and to some extent in the second year. As a result of the entry of cyprinids into the reservoir with the tributary waters and high nutrient concentration, a strong cyanobacterial water bloom occurred, which prevented the development of the cladocerans [35, 36]. The second approach to restoration began in 2005 and continues to this day. It is carried out using two methods: inactivation of phosphorus in the water column and biomanipulation. Because the water of the River Cybina is still heavily loaded with stormwater, large loads of nutrients reach the lake. The inactivation of phosphorus should create unfavorable conditions for the growth of cyanobacteria, and the aim of stocking fry of predatory species is to promote the development of cladocerans, thereby improving the transparency of water. In the first years of restoration, iron sulfate in small doses (300–700 kg for the whole reservoir, i.e., ca 10 kg/ha) was used to inactivate phosphorus. Each year, the treatment was repeated three to nine times, when the concentration of phosphorus exceeded 0.1 mg/L. Once in 2011 and three times in 2012, a new product, called Sinobent, in a dose of 150 kg was spread into the reservoir [27]. From 2012, the use of magnesium chloride in small doses (300 kg for the entire reservoir) was begun. In 2012 and 2013, it was used three times alternately with iron sulfate, while in subsequent years, iron sulfate was used only once, while magnesium chloride was applied five times. As a result of the interaction of this compound with both phosphates and ammonium nitrogen, it forms struvite (magnesium ammonium phosphate), which in the form of crystals sediments to the bottom of the lake. The reservoir was stocked with fry of pike, pikeperch, and catfish, to the total amount of 600–800 fingerlings per ha almost every year with the exception of the final year of every hydrological period, i.e., in 2008, 2012, and 2016. During draining all the fish are caught and removed from the reservoir. Despite the stocking of the reservoir with predatory species, cyprinids predominate in the catches, indicating good conditions for breeding and recruitment of cyprinids. During the 2008 drainage, 577.7 kg/ha of fish were caught, in which roach accounted for 70.7% and predatory species only 20.1% [37].

Compared to the period without restoration treatments in 1997–2000 (Table 2), there is a clear improvement in water transparency from an average of 0.6 m before restoration to 1.1 m in its course in the period 2005–2017 (Fig. 3a). Distinct differences in results are visible in the individual years of restoration, resulting from a 4-year hydrological cycle in the reservoir and the amount of rainfall in a given year. Usually, the greatest transparency occurred in the first year after filling of the reservoir (2005, 2009, 2013, and 2017), because there were very few fish in the water body. It was only as a result of fry inflow with the River Cybina water and the spawning of cyprinids in the reservoir that their number and biomass increased gradually, despite the stocking of predatory fish fry. TP concentration decreased from 555 μg/L before restoration to 148 μg/L during its use. In the subsequent years of restoration, there is a gradual decrease in the concentration of phosphorus (Fig. 3a), which may be related to its inactivation not only in the water column but also in the bottom sediments, limiting the internal loading of phosphorus. Chlorophyll a also decreased by more than half, from 81.6 μg/L to 38.8 μg/L. Its changes were negatively correlated with Secchi depth ($r = -0.72$, $p < 0.05$).

The value of the average TSI during restoration remained essentially below 70, indicating the eutrophic state of the reservoir (Fig. 3b). The TSI (SD) depth was significantly lower than 70, as well as TSI (Chl) and TSI (TN), 62.5, 68.4, and 67.1, respectively. Only TSI (TP) remained high, characteristic of hypertrophy (Fig. 3b). Nevertheless, its value has clearly decreased compared to before restoration, from 91.3 to 77.6.

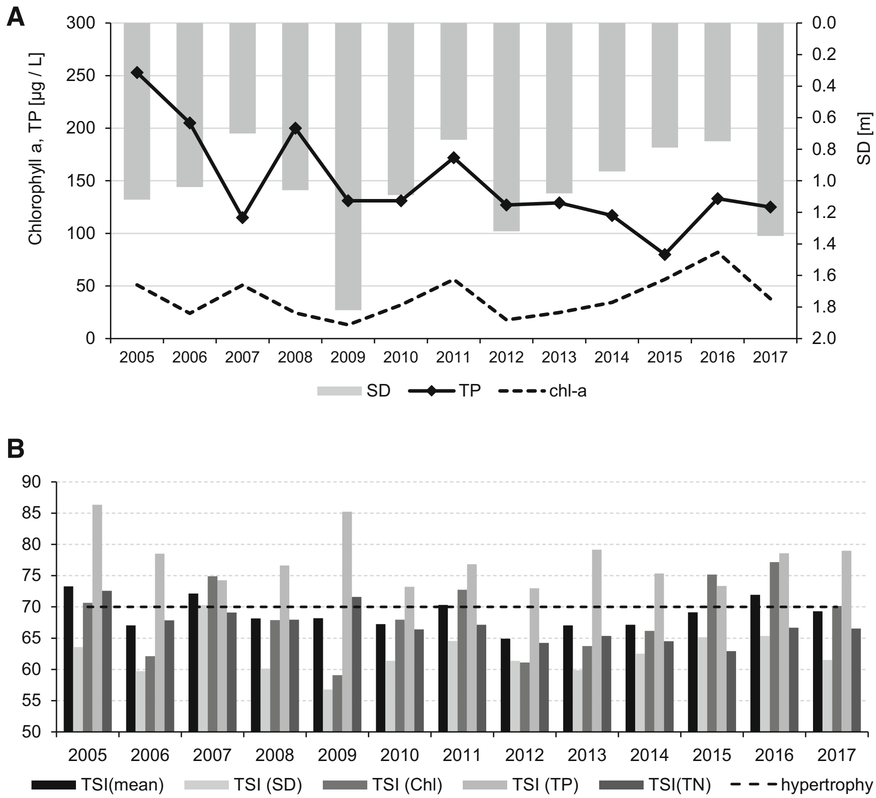


Fig. 3 Changes of (**a**) few water quality variables during restoration process of Lake Maltańskie (SD, Secchi disk transparency; TP, total phosphorus; chl-a, chlorophyll a) and (**b**) TSI values during the restoration of Lake Maltańskie

The most important change, however, concerned the composition and biomass of phytoplankton. Before restoration it was dominated by cyanobacteria, which accounted for up to 90% of the total phytoplankton. During restoration, the number and biomass of phytoplankton were significantly reduced, and cyanobacteria were replaced by green algae, diatoms, chrysophytes, and cryptophytes. Despite restoration measures, cyanobacteria were still present in some years, for example, in 2005, 2007, 2011, 2015, and 2016. In 2005, the process of phytoplankton reconstruction was just beginning, which is why the biomass was reduced by 32%, but the cyanobacteria were still dominant. In the second year of restoration, they had almost completely disappeared (Fig. 4a). Unfortunately, they reappeared in 2007. In this year the reservoir was stocked with hatchlings of pike instead of fingerlings, which were less effective. Thus the cyanobacteria returned, although their biomass was not very high. There was another reason in 2011. During the previous autumn, a large load of silt, nutrients, and fish from the two preliminary reservoirs situated in the

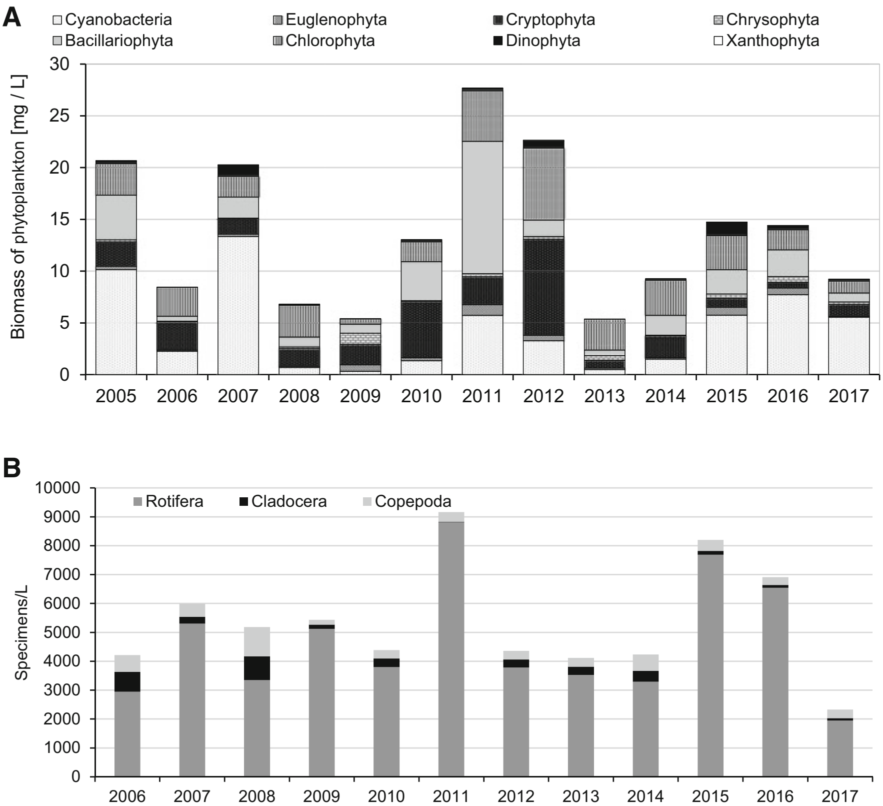


Fig. 4 Phytoplankton biomass (**a**) and zooplankton abundance (**b**) in Lake Maltańskie during its restoration

course of River Cybina drained to the reservoir. This led to a sharp intensification of eutrophication symptoms which continued into the following year. Such an event could not be overcome with sustainable restoration methods [30].

During the restoration of Maltańskie Lake, zooplankton was dominated by rotifers, which is characteristic of hypertrophic lakes (Fig. 4b). However, crustaceans also achieved significant amounts during the first years of each hydrological cycle. This is the result of biomanipulation, which was intensified in these years, due to the very small population of cyprinids. The influence of cladocerans on phytoplankton was very pronounced, and a clear negative correlation of this group of zooplankton with chlorophyll a was found ($r = -0.49$, $p < 0.05$).

However, a question remains about the sense of restoring a lake into which excess loads of nutrients continue to inflow. The classic textbooks on restoration emphasize that the external sources of nutrients should first be cut off so that restoration can succeed [21, 23, 26]. In the case of Lake Maltańskie, it can be seen that noninvasive and relatively inexpensive methods of sustainable restoration can maintain the quality of water that allows the lake to be used for sporting and recreational purposes. This is a problem faced by many cities whose urban lakes are affected by stormwater inflows. The combined approach attempts to remedy this using various nature-based solutions, primarily specially designed wetlands, retaining suspensions and part of nutrients [38–40]. In the case of Lake Maltańskie, such a method was also used, building a system of four preliminary reservoirs in which the discharged stormwater is treated [10, 41].

Lake Uzarzewskie This small postglacial lake is located in the course of River Cybina (Table 1). It is a kettle lake, with steep slopes and a flat bottom, located at the bottom of the deep valley whose slopes are overgrown by forest; hence the lake weakly mixes under the influence of wind. Despite its small depth, it shows partial thermal stratification. Restoration of this lake can be divided into three stages. The first covered 2 years of iron treatment in 2006–2007, using PIX-112 (iron sulfate) in small doses of 5–7 kg/ha, to bind phosphorus but not to coagulate the entire suspension. The lake was treated six times in 2006 and three times in 2007 [42]. The second stage began in 2008 and continues. It consists of supplying water with the use of plastic pipes to the bottom of the lake from two small tributaries, flowing out of springs at the bottom of the slope of the valley. Water from these sources is well oxygenated, cool (8–12°C), and contains high nitrate concentrations (37 and 40 mgN/L on average) [27, 33]. Its main purpose is to increase the redox potential in the sediment-water interface, which will then limit the internal phosphorus loading from bottom sediments, eliminate the hydrogen sulfide in the water overlying the sediments, and reduce nitrogen concentration, due to the denitrification process. Since the beginning of 2017, the third stage has been in progress. This involves providing an additional supply of magnesium chloride to the inflow. Every month, a sack containing 25 kg of magnesium chloride is mounted in the stream, the content of which is slowly washed out into the lake. Its purpose is to bind phosphorus and part of nitrogen in the epilimnion in a mineral called struvite, which precipitates to bottom sediments, limiting phytoplankton growth.

Secchi depth, which amounted to an average of 0.73 m before the restoration, decreased to 0.67 m in the first restoration stage, slightly increased to 0.86 m in the second stage and only exceeded 1 m in the third stage (Fig. 5a). The total phosphorus in the surface water layer only slightly decreased during the iron treatment from 162 μg/L to 149 μg/L. It remained at a similar level, with an average of 152 μg/L, in the second stage. It clearly decreased in 2017 as a result of dosing with magnesium chloride, down to 74 μg/L. Chlorophyll a also slightly decreased in the first stage of restoration, with an average ranging from 68.7 μg/L to 65.6 μg/L. In the second stage, it also decreased slightly, with an average of 57.3 μg/L, and only in the third stage was a greater decrease observed, to 37.9 μg/L. A comparison of the above values with the limits of the OECD [13] for hypertrophy indicated that the three variables only passed into the eutrophic state in the last stage of restoration in 2017. These were mean value of TP, minimum Secchi depth (0.8 m), and maximum concentration of chlorophyll a (65.7 μg/L).

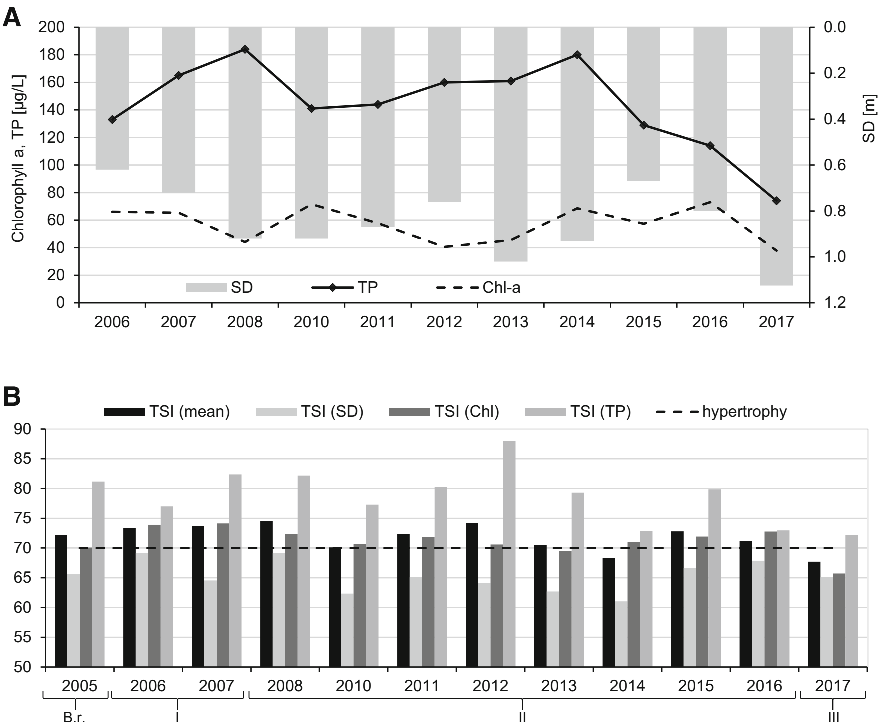


Fig. 5 Changes of (**a**) few water quality variables during restoration process of Lake Uzarzewskie (SD, Secchi disk transparency; TP, total phosphorus; Chl-a, chlorophyll a) and (**b**) TSI values before and during the restoration of Lake Uzarzewskie (B.r., before the restoration; I–III, stages of lake restoration)

The mean value of the TSI indicated that both before the restoration, during its first stage, and in principle also in the second stage, the lake remained in a hypertrophic state (Fig. 5b). It was only in the third stage of restoration that it went into the eutrophic state, although the TSI (TP) still indicated hypertrophy. An average eutrophic state was also found temporarily in 2010 and 2014, due to the low TSI-SD values.

Both before the restoration and during the first stage, cyanobacteria generally predominated in the summer phytoplankton (Fig. 6a). A partial reconstruction of the phytoplankton composition was confirmed in the second stage of restoration. Cyanobacteria still occurred but in smaller quantities. Cryptophytes, diatoms, and green algae were often dominant [33]. Only in the last stage did cyanobacteria completely disappear, being replaced by cryptophytes and green algae. The total biomass of phytoplankton decreased significantly; when compared to the year before the restoration, it was on average almost three times and, until 2012, down to ten times less.

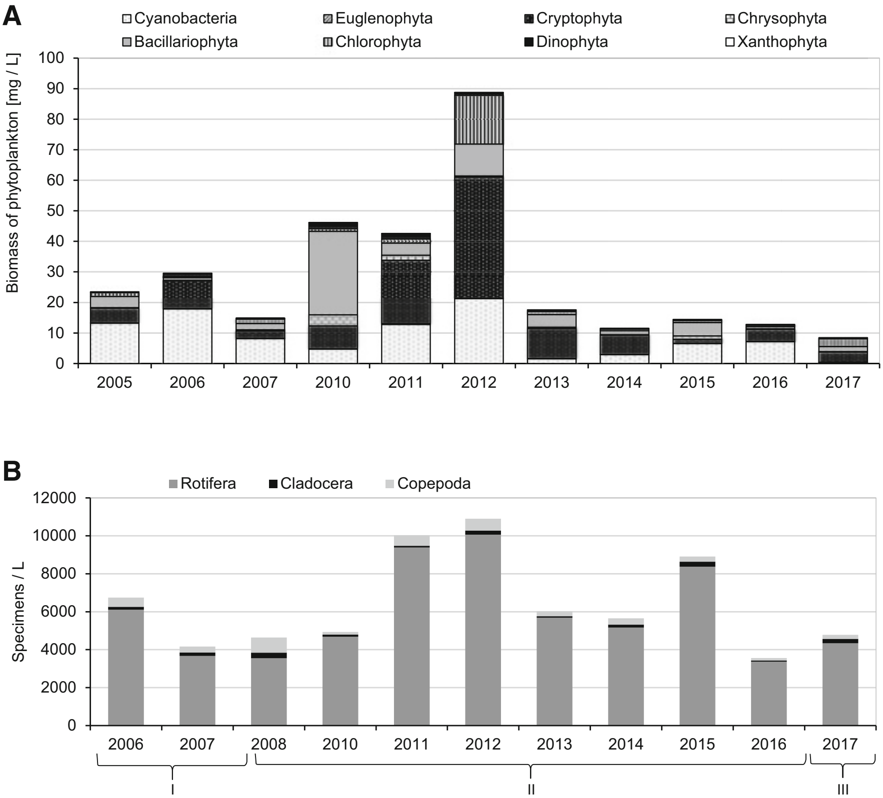


Fig. 6 Phytoplankton biomass (**a**) and zooplankton abundance (**b**) in Lake Uzarzewskie during its restoration (I–III – three stages of lake restoration)

Zooplankton throughout all the years of research was dominated by rotifers (Fig. 6b), which is a feature of strongly eutrophicated lakes [43]. Planktonic crustaceans constituted a small share in zooplankton, among which copepods were always more numerous than cladocerans. This resulted from the presence of numerous cyprinids, especially bream (oral information from anglers), because biomanipulation was not used in the lake.

In the first stage of restoration related to iron treatment, water quality was not improved but even deteriorated. Biomass of cyanobacteria and the duration of their occurrence increased. Since oxygen deficits occurred in the metalimnion, hydrogen sulfide was present there, and the redox potential was very low, sedimenting flocs of iron hydroxide with adsorbed phosphorus were dissolved, and mineral phosphorus was returned to water column stimulating the growth of cyanobacteria [44]. Bottom sediments were still an important source of phosphorus in the lake, which was associated with the oxygen depletion and very low redox potential in the water overlying the sediments within range of the metalimnion. The most important goal of the second stage of restoration was to reduce the internal loading of phosphorus by increasing the redox potential in the sediment-water interface. This reduction was very marked, as internal loading from the profundal zone decreased from 1,040.8 kg P/year to 89.1 kg P/year, i.e., more than ten times [12]. The water delivered to the bottom with a large concentration of nitrate caused a denitrification process that did not allow the reduction of iron and the release of phosphates. The concentration of nitrogen in the lake also decreased. Low oxygen concentration in the water overlying the sediments probably also enabled the anammox reaction to occur, as the concentration of ammonium nitrogen decreased significantly [33, 45]. This reduced internal nutrient loading resulted in a clear reconstruction of phytoplankton. Because the impact of the new restoration method was related to the profundal, the bottom located within the epilimnion (the so-called active bottom) still released nutrients into the water column which in turn caused the presence of cyanobacteria in summer, associated mainly with excessive concentrations of phosphorus in epilimnetic water [46]. Therefore, it was decided to lower the concentration of phosphorus in the epilimnion, by adding doses of magnesium chloride $MgCl_2 \cdot 6H_2O$ in the third stage of restoration to one of the small tributaries of the lake. The joint operation of these two sustainable reclamation methods has led to a clear improvement in water quality in 2017, as the water's transparency reached 1.5 m, the cyanobacteria have been replaced by other groups of algae, and the phytoplankton biomass has decreased significantly. It should be stressed that both methods are extremely cost-effective compared to other technical methods of restoration.

Lake Swarzędzkie It is located in the course of River Cybina, 5 km downstream of Uzarzewskie Lake. It is also a postglacial and partially thermally stratified water body. Due to pollution with untreated sewage from the city of Swarzędz in the 1980s, the lake was heavily polluted. After sewage diversion in 1991, the quality of water changed only slightly. Cyanobacteria were still dominant in the phytoplankton, and the lake was unsuitable for recreation. Therefore, since autumn 2011, the lake has been restored with three methods: (a) deep water oxygenation using a wind-driven aerator, (b) inactivation of phosphorus in the water column with the use of iron sulfate and magnesium chloride, and (c) biomanipulation [47]. The aerator oxygenated the water taken from the layer just above the bottom on the surface and carried it back to the bottom zone [27, 29]. Inactivation of phosphorus took place with small doses of iron sulfate, and magnesium chloride (200–300 kg/lake) used nine times in 2012 and five times in 2013 and in 2014 [47]. Biomanipulation consisted of catching some of the cyprinids in autumn 2011 and stocking in 4 subsequent years the autumn fry of pike (*Esox lucius* L., 70 kg/year) and once pikeperch fry (*Sander lucioperca* L.) in the amount of 7,200 fingerlings in summer 2014 [47]. After 3 years of restoration, phosphorus inactivation and biomanipulation were concluded, leaving only oxygenation of deep water using an aerator. The influence of such limited restoration was documented in 2015–2016, as the second stage of restoration [48].

The water transparency increased significantly as a result of sustainable restoration from an average of 0.75–1 m (Fig. 7a), after which, during the limited restoration period, it decreased again to 0.8 m. Similarly, chlorophyll a decreased

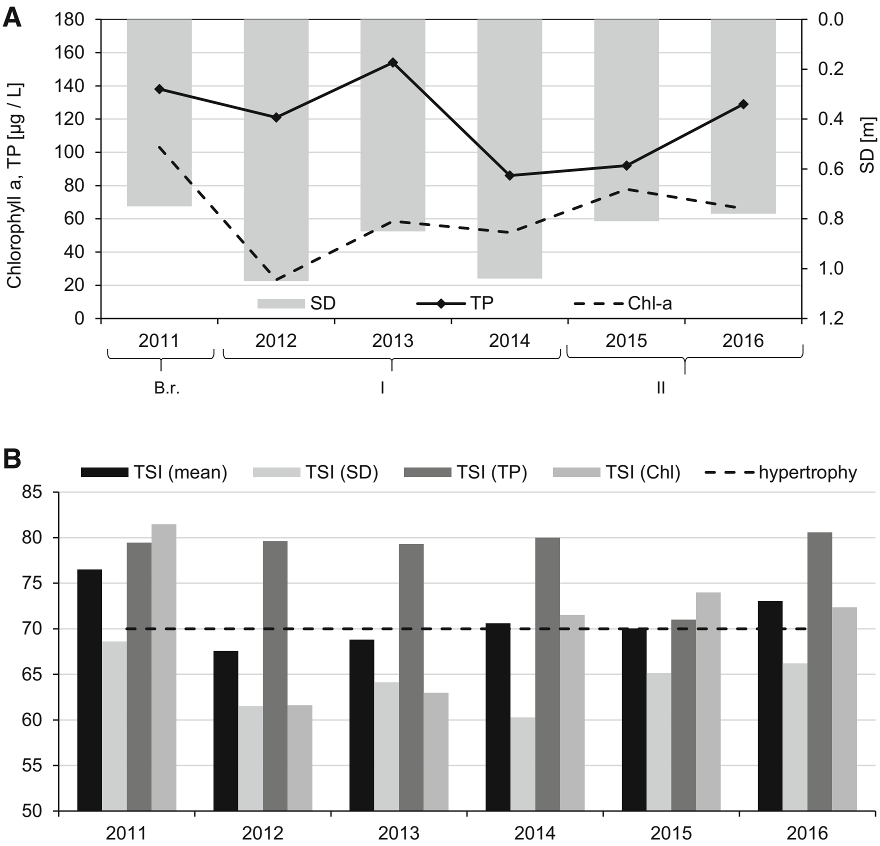


Fig. 7 Changes of (**a**) mean values of water quality variables (SD, Secchi disk transparency; TP, total phosphorus; Chl-a, chlorophyll a; B.r., before the restoration; I, first stage of restoration; II, second stage of restoration) and (**b**) TSI values before and during the restoration of Lake Swarzędzkie

from 103 µg/L to 44.5 µg/L and then increased in the second restoration period to 72 µg/L. TP concentration in the first 2-year restoration period remained at a similar level to that before restoration (138 µg/L); however, in the third year, it clearly decreased to 86 µg P/L. This concentration gradually increased again during the limited restoration in the second stage (Fig. 7a).

The calculated TSI values indicate a clear decrease in the trophy of the lake (transition to eutrophic state) as a result of sustainable restoration (Fig. 7b). During the limited restoration period, the TSI again indicated its hypertrophy. Changes within the TSI (SD) and partly also in the TSI (Chl) were responsible for the changes of TSI mean. On the other hand, changes within the TSI (TP) were almost invisible, and the values were always very high, typical for hypertrophy.

Phytoplankton reacted to sustainable restoration by clearly rebuilding the species composition. As early as the first year of restoration (2012), cyanobacteria almost disappeared and were replaced by other groups of algae. This was similar in the third year of restoration, in which cyanobacteria also occurred in small amounts. At the same time, the total biomass of phytoplankton decreased by half compared to the year before restoration. Only in the second year of restoration (2013) did cyanobacteria return to dominate phytoplankton in August. At the same time, however, in addition to cyanobacteria, there was a high diversity of algae from other groups, which in the other months of this year dominated the cyanobacteria (Fig. 8a). The abandonment of two methods of restoration in the two following years resulted in the gradual return of cyanobacteria. They dominated phytoplankton in 2016, creating a water bloom. The diversity in phytoplankton composition also clearly decreased (Fig. 8a).

Zooplankton was still dominated by rotifers during the restoration period, indicating the high trophic status of the lake and the limited impact of the biomanipulation used. It is true that the number of cladocerans was quite large

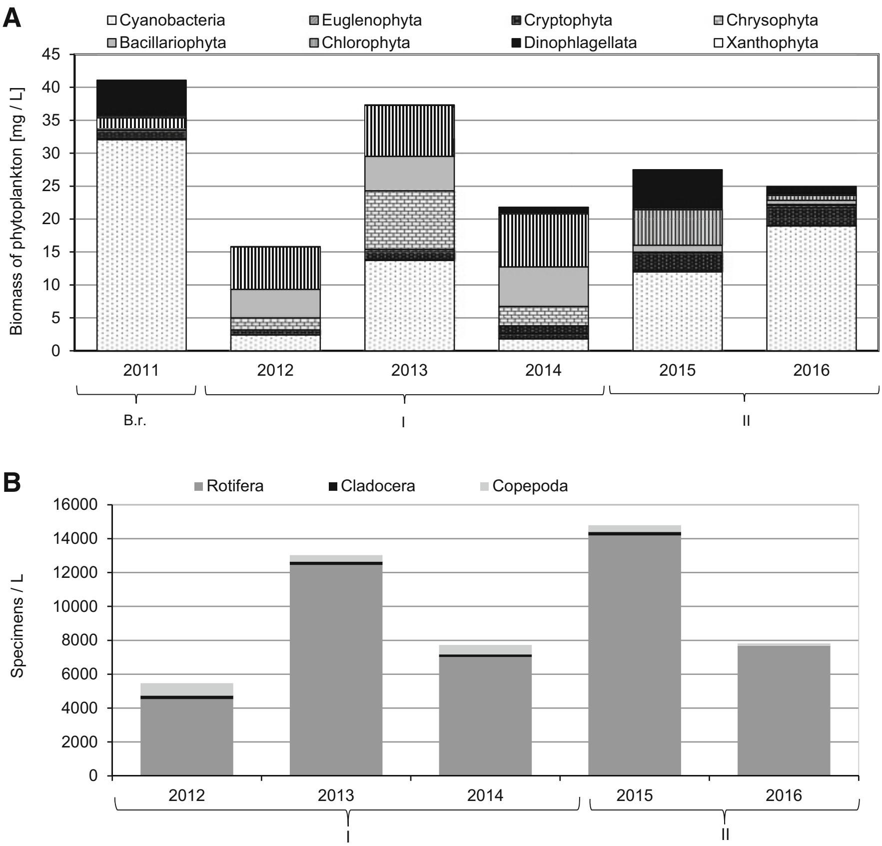


Fig. 8 Phytoplankton biomass (**a**) and zooplankton abundance (**b**) in Lake Swarzędzkie (B.r., before the restoration; I, first stage of restoration; II, second stage of restoration)

(on average almost 200 individuals/L); however, small taxa dominated. Copepods were much more numerous (over 500 individuals/L), but they were dominated by juvenile forms (nauplii and copepodites), which were not very important in reducing the number of phytoplankton (Fig. 8b). The impact of summer pikeperch fry and autumn pike fry stocking in 2014 was visible in the presence of cladocerans in the lake in the subsequent year. The lack of stocking in 2015–2016 resulted in a significant reduction in the number of crustaceans in the final year of research (Fig. 8b).

Before restoration, Lake Swarzędzkie had been in a hypertrophic state for many years, manifested by strong water blooms [49], large internal nutrient loading, and a very poor composition of water vegetation [50]. Its restoration with sustainable methods has proved to be very difficult but still possible. It required as many as nine repetitions of phosphorus inactivation with small doses of chemicals in the first year of restoration. These treatments turned out to be insufficient in the second year of restoration, as the very high water temperature resulted in increased internal phosphorus loading and repeated cyanobacterial water blooms [47]. Biomanipulation proved a helpful but not very effective method of water quality improvement. This was largely due to inadequate stocking of the lake with predators. Only small quantities of autumn pike fry were introduced, and only once, in 2014, were summer fry of pikeperch stocked. Nevertheless, the number of crustaceans has clearly increased, which must have been reflected in the increased phytoplankton grazing. The aerator was an important restoration tool, and, because it uses the wind energy to oxygenate the waters overlying the bottom sediments, it costs very little. It did not provide very large amounts of oxygen, which in turn did not intensify microbial decomposition of organic matter. As a result, the release of phosphorus to the water column did not significantly increase. Keeping low oxygen levels at the bottom, it enabled the denitrification process and anammox reaction to take place, and by increasing the redox potential, it prevented iron reduction and phosphorus release [48].

Lake Rusalka This artificial reservoir was created in the city of Poznań in 1943 on the River Bogdanka, the left tributary of River Warta. It is a shallow lake with an average depth of 1.9 m. Thermal stratification only occurs in 7.7% of the total lake area, in close proximity to a dam where there is a depth of 9 m. Two of its tributaries are polluted by stormwater discharged from the residential and industrial-residential districts of the city of Poznań. They have caused strong eutrophication of the lake, manifested by cyanobacterial water blooms, lack of submerged vegetation, and a deoxygenated hypolimnion. As a result, the recreational use of the lake was severely affected, and application was made for its restoration. In 2006–2007, iron treatment was applied, using small doses of iron sulfate. Six treatments were conducted in 2006 (930 kg altogether), while three in 2007 (350 kg altogether). As a result, the concentration of phosphorus in the water decreased, and internal phosphorus loading from the bottom sediments strongly decreased [51]; however, cyanobacterial blooms were still present. In 2006, they were even stronger than before restoration in 2005. Cyanobacterium *Raphidiopsis raciborskii* (syn. *Cylindrospermopsis raciborskii*), which is considered a tropical invasive species, appeared at that time in large quantities [52]. There followed was a 6-year break in the lake restoration. It was resumed in 2014 by treatment with small doses of magnesium chloride. This is applied approximately ten times a year, in small doses of 75 kg each time for the whole lake.

As a result of this restoration, water transparency increased from the average 0.71 m before restoration (Table 2) to 1.15 m in the first year of restoration. It constantly increased in subsequent years and in 2017 was 1.7 m on average (Fig. 9a).

Chlorophyll a clearly decreased under restoration from an average 40.6 μg/L before restoration to 20.4 μg/L in the years 2014–2017. TP, which before restoration amounted to an average of 56 μg/L, decreased to 32 μg/L. However, in the subsequent years of restoration, it successively increased, reaching an average of 67 μg/L in 2017 (Fig. 9a). As a result of improving water transparency in the reservoir, the succession of submerged vegetation began. In 2014, *Potamogeton crispus* appeared, which very quickly overgrew the bottom to a depth of 2 m.

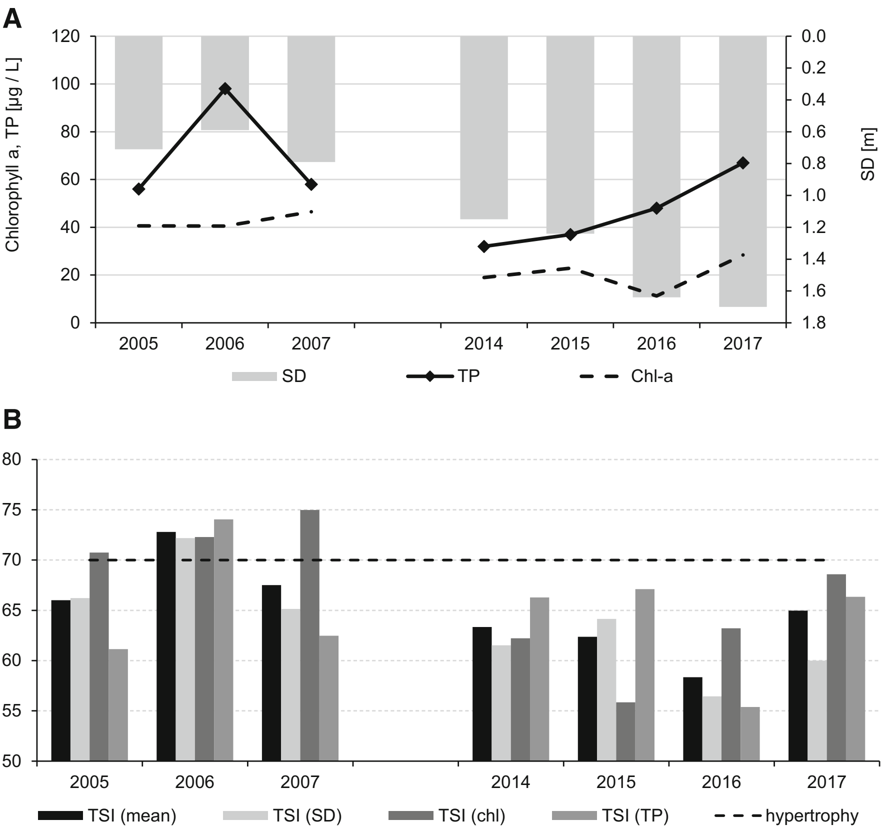


Fig. 9 Changes of (**a**) mean values of water quality variables (SD, disk transparency; TP, total phosphorus; Chl-a, chlorophyll a) and (**b**) TSI values before and during the restoration of Lake Rusałka

In the following year, its range and coverage significantly increased, overgrowing about 3/4 of the bottom area, reaching a depth of 3.5 m.

The vegetation season of *P. crispus* was quite brief, and its decay sets in at the end of June. The proliferation of phytoplankton begun from July; it was first dominated by diatoms and green algae, but in August there was the growth of cyanobacteria, which reached maximum biomass in September, at which time the highest chlorophyll a (53.5 μg/L) and TP (115 μg/L) concentration also occurred. The average biomass of phytoplankton in the period May–October was usually not lower than before restoration, but there was a clear reconstruction of its composition (Fig. 10). Cyanobacteria no longer dominated over other groups, constituting 26.2–37.6%. There were more green algae and cryptophytes, while dinoflagellates, chrysophytes, and diatoms were also of quite large biomass. The variety of phytoplankton compositions has clearly increased. The trophic state index decreased markedly, especially in relation to the values calculated on the basis of chlorophyll a and water transparency (Fig. 9b).

The example of Lake Rusalka shows that if the lake is not in a very advanced hypertrophic state, it is possible to improve the water quality with the help of one sustainable restoration method. The use of small doses of magnesium chloride led to the development of *P. crispus*, which is a competitor for phytoplankton and also releases allelopathic substances that prevent the growth of cyanobacteria [53].

Lake Konin It is quite small (88 ha) and very shallow (mean depth 3.1 m) waterbody. It is situated in the Lubuskie Lake District (Western Poland), periodically fed by the River Obra during its high water level period (usually spring). Due to the great fertility and pollution of river waters (watershed covers over 850 km^2, mainly agricultural areas), a dike was constructed in summer 2013 to prevent river water inflow to the lake. This protective action was followed by in-lake restoration with effective microorganisms (EM), i.e., a composition of photosynthetic bacteria,

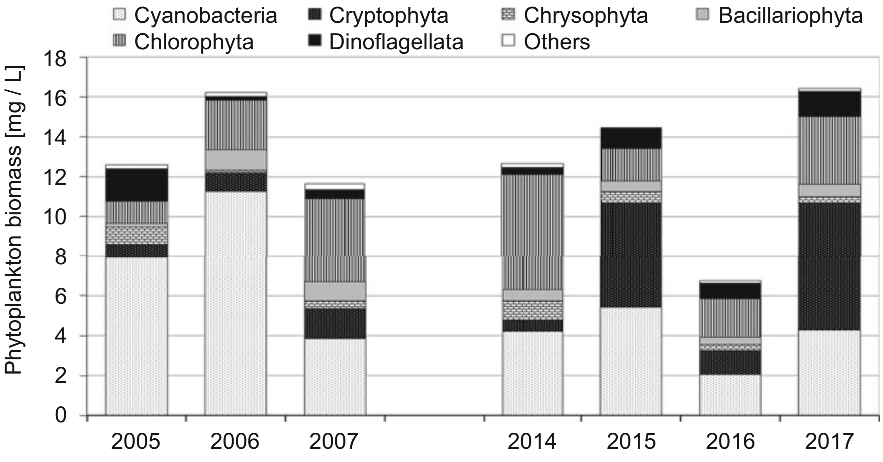


Fig. 10 Phytoplankton biomass of Lake Rusałka before and during restoration measures

lactic acid bacteria, actinomycetes, yeasts, and fermenting fungi, whose aim is to alter the lake microbial community by suppressing harmful bacteria (including cyanobacteria) and increasing the dominance of beneficial species. The concept of EMs was developed by Higa [54], and its main purpose is to accelerate organic matter decomposition; therefore there are a variety of applications such as waste and wastewater management, odor control, and agriculture. Attempts are being made to implement EMs in freshwater quality problems; however – as stressed by Lürling et al. [55] – they are not supported by scientific evidence. A case study of Lake Konin was a great opportunity to verify the effectiveness of EM in the restoration of a very shallow, hypertrophic water body. EMs were used in 2014 and 2015 in a liquid form (so-called EM activated solution) and EM mudballs. Additionally, barley straw was used as an algal control agent as it is a source of algicides (e.g., phenolics, carboxylic acids) that are secreted to water during its decomposition [56].

Prior to the restoration, phytoplankton biomass was very high, reaching over 100 mg/L and consisting mainly of *Cyanobacteria* (96%), especially potentially toxic *Planktothrix agardhii*, accompanied by other filamentous species, i.e., *Pseudanabaena limnetica*, *Limnothrix redeckei*, and *Aphanizomenon gracile.* The community structure was not strongly affected by the restoration measures; however, the biomass decreased to less than 40 mg/L (Fig. 11a). The lack of a fertile river water inflow in 2014 and 2015 due to dike construction resulted in increased Chlorophyceae biomass; nevertheless *Cyanobacteria* dominated almost throughout the year. Additionally, *R. raciborskii* – a tropical invasive species [52] – was observed in large quantities in 2015. The diminishment in phytoplankton biomass resulted from a greater variety of zooplankton, with greater abundance of Cladocera and Copepoda in comparison to 2013 (Fig. 11b), although rotifers were still most numerous due to detritus availability and lack of macrophytes, creating a refugee for crustacean zooplankton [57, 58].

Phytoplankton proliferation resulted in great chlorophyll a concentrations, increasing from 84.6 μg/L in 2013 to 146.6 μg/L in 2015, and very low water transparency – less than 0.6 m (Fig. 12). The reduction in TP concentration may be linked to EM application as well as to the elimination of an external nutrient source, which was the River Obra. Noted values were still over 100 μg/L, characteristic of hypertrophic lakes. The mean value of the TSI indicated that both before and during the restoration, the lake remained in a hypertrophic state (Fig. 12).

5 Conclusions

Hypertrophic lakes in Western Poland are characterized by severe water blooms, resulting in high chlorophyll a content and decreased transparency. Poor water status hampers recreational activities and poses a threat to human health due to

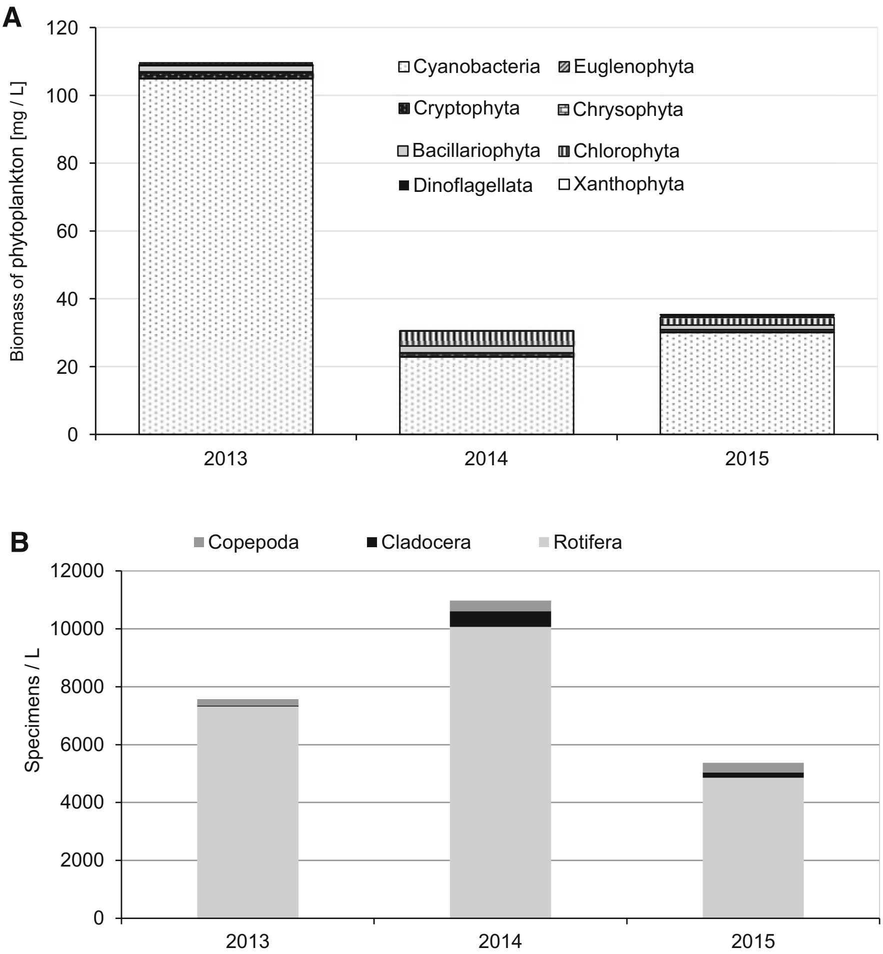
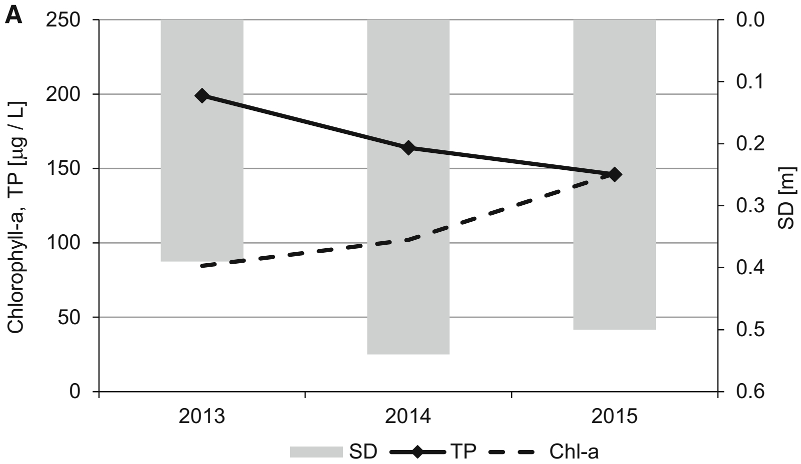


Fig. 11 Phytoplankton biomass (**a**) and zooplankton abundance (**b**) in Lake Konin

Cyanobacteria-related nuisances; therefore restoration treatment is essential. We have proposed a sustainable approach, involving a combination of physical, chemical, and biological methods, but in a more natural, less invasive manner. The treatment applied to lakes Maltańskie, Swarzędzkie, Uzarzewskie, and Rusałka fulfills these requirements and results in gradual improvement of water quality, manifested in the phytoplankton community structure (less *Cyanobacteria*) as well as nutrient concentrations. The example of Lake Konin confirms the inefficiency of EM in lake restoration due to the limitation to one method only, which seemed to be a dubious approach in the case of such a heavily polluted lake.



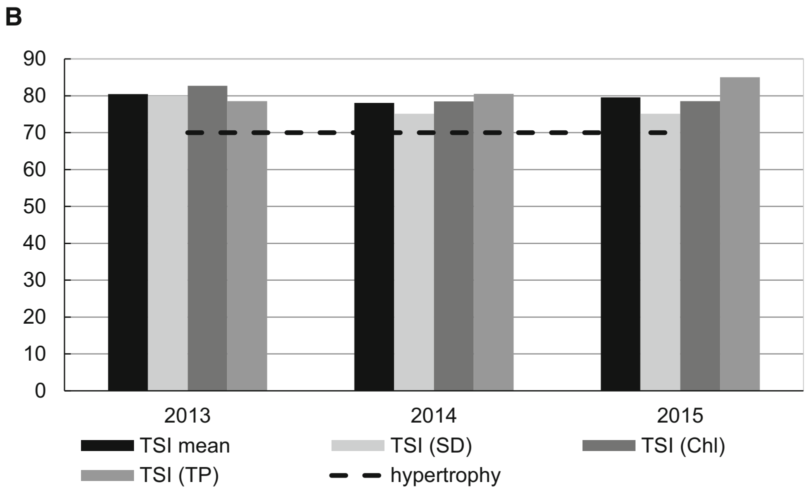


Fig. 12 Changes of (**a**) mean values of water quality variables (SD, Secchi disk transparency; TP, total phosphorus; Chl-a, chlorophyll a) and (**b**) TSI values before and during the restoration of Lake Konin

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References

1. Ramm K, Scheps V (1997) Phosphorus balance of a polytrophic shallow lake with the consideration of phosphorus release. Hydrobiologia 342/343:43–53. https://doi.org/10.1023/A:1017092618517

2. Dobrowolski Z (2001) Density, biomass, and distribution of benthic invertebrates in the mid-lake zone of the coastal Lake Gardno. Oceanol Stud 30(1–2):39–58
3. Mischke U (2003) Cyanobacteria associations in shallow polytrophic lakes: influence of environmental factors. Acta Oecol 24:S11–S23. https://doi.org/10.1016/S1146-609X(03)00003-1
4. Ciecierska H (2006) Evaluation of the status of lakes located in the City of Olsztyn (Masurian Lake District, N-E Poland) by the macrophytoindication method (MPhI). Hydrobiologia 570:141–146. https://doi.org/10.1007/s10750-006-0173-5
5. Olszewski P (1971) Trofia a saprobia [Trophy and saprobity]. Zeszyty Naukowe WSR in Olsztyn, Ser C Suppl 3:5–14
6. [APHA] American Public Health Association (1996) Standard methods for the examination of water and wastewater, 19th edn. Washington (DC)
7. Gołdyn B, Chudzińska M, Barałkiewicz D, Celewicz-Gołdyn S (2015) Heavy metal contents in the sediments of astatic ponds: influence of geomorphology, hydroperiod, water chemistry and vegetation. Ecotox Environ Saf 118:103–111. https://doi.org/10.1016/j.ecoenv.2015.04.16
8. Barałkiewicz D, Chudzińska M, Szpakowska B, Świerk D, Gołdyn R, Dondajewska R (2014) Storm water contamination and its effect on the quality of urban surface waters. Environ Monit Assess 186(10):6789–6803. https://doi.org/10.1007/s10661-0014-3889-0
9. Zębek E, Napiórkowska-Krzebietke A (2016) Response of phytoplankton to protective-restoration treatments enhancing water quality in a shallow urban lake. Environ Monit Assess 188(623):1–16. https://doi.org/10.1007/s10661-016-5633-4
10. Gołdyn R, Szpakowska B, Świerk D, Domek P, Buxakowski J, Dondajewska R, Barałkiewicz D, Sajnóg A (2018) Influence of stormwater runoff on macroinvertebrates in a small urban river and a reservoir. Sci Total Environ 625:743–751. https://doi.org/10.1016/j.scitotenv.2017.12.234
11. Kowalczewska-Madura K, Gołdyn R, Dondajewska R (2010) The bottom sediments of Lake Uzarzewskie – a phosphorus source or sink? Oceanol Hydrobiol St 39(3):81–91. https://doi.org/10.2478/v10009-010-0042-4
12. Kowalczewska-Madura K, Dondajewska R, Gołdyn R, Podsiadłowski S (2017) The influence of restoration measures on phosphorus internal loading from the sediments of a hypereutrophic lake. Environ Sci Pollut R 24(16):14417–14429. https://doi.org/10.1007/s11356-017-8997-2
13. OECD (1982) Eutrophication of waters: monitoring, assessments and control. Organisation for Economic Co-operation and Development, Paris
14. Szpakowska B, Świerk D, Gołdyn R, Barałkiewicz D (2014) Contents of Cu, Zn, Cd, Pb and Fe in rainwater effluents discharged to surface waters in the city of Poznań. J Elem 3:779–794. https://doi.org/10.5601/jelem.2014.19.2.522
15. Welch EB (1980) Ecological effects of waste water. Cambridge University Press, Cambridge
16. Carlson RE (1977) A trophic state index for lakes. Limnol Oceanogr 22:361–369. https://doi.org/10.4319/lo.1977.22.2.0361
17. Kratzer CR, Brezonik PL (1981) A Carlson-type trophic state index for nitrogen in Florida lakes. Water Resour Bull 17:713–715. https://doi.org/10.1111/j/1752-1688.1981.tb011282.x
18. Carlson RE, Simpson J (1996) A coordinator's guide to volunteer lake monitoring methods. North American Lake Management Society
19. Scheffer M, Hosper SH, Meijer M-L, Moss B, Jeppesen E (1993) Alternative equilibria in shallow lakes. Trends Ecol Evol 8(8):275–279. https://doi.org/10.1016/0169-5347(93)90254-M
20. Rosińska J, Gołdyn R (2018) Response of vegetation to growing recreational pressure in the shallow Raczyńskie Lake. Knowl Manag Aquat Ecosyst (419):1. https://doi.org/10.1051/kmae/2017050
21. Ryding S-O, Rast W (eds) (1989) The control of eutrophication of lakes and reservoirs. MAB Series, vol 1. UNESCO/The Parthenon Publishing Group, Paris/Carnforth
22. Klapper H (2003) Technologies for lake restoration. J Limnol 62(Suppl.1):73–90. https://doi.org/10.4081/jlimnol.2003.s1.73

23. O'Sullivan PE, Reynolds CS (2005) Lakes handbook: lake restoration and rehabilitation, vol 2. Blackwell Publishing, Malden
24. Angeler DG, Allen CR, Birgé HE, Drakare S, McKie BG, Johnson RK (2014) Assessing and managing freshwater ecosystems vulnerable to environmental change. Ambio 43(Suppl.1):113–125. https://doi.org/10.1007/s13280-014-0566-z
25. Peterson SA (ed) (1979) Limnological and socioeconomic evaluation of lake restoration projects: approaches and preliminary results. US EPA-600/3-79-005, Freshwater System Division, Corvallis
26. Jeppesen E, Søndergaard M, Liu Z (2017) Lake restoration. In: Allison SK, Murphy SD (eds) Routledge handbook of ecological and environmental restoration. Routledge, London, pp 226–242
27. Gołdyn R, Podsiadłowski S, Dondajewska R, Kozak A (2014) The sustainable restoration of lakes – towards the challenges of the Water Framework Directive. Ecohydrology&Hydrobiology 14(1):67–74. https://doi.org/10.1016/j.ecohyd.2013.12.001
28. Osuch E, Podsiadłowski S (2012) Efficiency of pulverizing aeration on Lake Panieńskie. Limnol Rev 12(3):139–145. https://doi.org/10.2478/v10194-012-0054-4
29. Podsiadłowski S, Osuch E, Przybył J, Osuch A, Buchwald T (2018) Pulverizing aerator in the process of lake restoration. Ecol Eng. https://doi.org/10.1016/j.ecoleng.2017.06.032
30. Kozak A, Gołdyn R, Dondajewska R (2015) Phytoplankton composition and abundance in restored Maltański Reservoir under the influence of physico-chemical variables and zooplankton grazing pressure. PLoS One 10(4):e0124738. https://doi.org/10.1371/journal.pone.0124738
31. Rosińska J, Kozak A, Dondajewska R, Kowalczewska-Madura K, Gołdyn R (2018) Water quality response to sustainable restoration measures – case study of urban Swarzędzkie Lake. Ecol Indic 84:437–449. https://doi.org/10.1016/j.ecolind.2017.09.009
32. Kozak A, Gołdyn R, Dondajewska R, Kowalczewska-Madura K, Holona T (2017) Changes in phytoplankton and water quality during sustainable restoration of an urban lake used for recreation and water supply. Water 9(9):713. https://doi.org/10.3390/w9090713
33. Dondajewska R, Kozak A, Kowalczewska-Madura K, Budzyńska A, Gołdyn R, Podsiadłowski S, Tomkowiak A (2018) The response of a shallow hypertrophic lake to innovative restoration measures – Uzarzewskie Lake case study. Ecol Eng. https://doi.org/10.1016/j.ecoleng.2017.07.010
34. Gołdyn R, Joniak T, Kowalczewska-Madura K, Kozak A (2003) Trophic state of a lowland reservoir during 10 years after restoration. Hydrobiologia 506–509:759–765. https://doi.org/10.1023/b:hydr.0000008585
35. Gołdyn R, Kozak A, Romanowicz W (1997) Food-web manipulation in the Maltański Reservoir. Hydrobiologia 342–343:327–333. https://doi.org/10.1023/a:1017036717550
36. Gołdyn R, Mastyński J (1998) Biomanipulation in the Maltański Reservoir. Int Rev Hydrobiol 83:393–400
37. Andrzejewski W, Szlakowski J, Mastyński J, Mazurkiewicz J, Godlewska M (2010) Fish biomass and species composition in the Malta Reservoir, Poland. J Water Land Dev 14:67–82
38. Campbell CS, Ogden MH (1999) Constructed wetlands in the sustainable landscape. Wiley, New York
39. Dondajewska R, Kozak A, Budzyńska A, Kowalczewska-Madura K, Gołdyn R (2018) Nature-based solutions for protection and restoration of degraded Bielsko Lake. Ecohydrology&Hydrobiology 18:401–411. https://doi.org/10.1016/j.ecohyd.2018.04.001
40. Jurczak T, Wagner I, Kaczkowski Z, Szklarek S, Zalewski M (2018) Hybrid system for the purification of street stormwater runoff supplying urban recreation reservoirs. Ecol Eng 110:67–77. https://doi.org/10.1016/j.ecoleng.2017.09.019
41. Gołdyn R, Dondajewska R, Szeląg-Wasielewska E, Szyper H (2009) An appraisal of changes in seasonal water quality during passage through a shallow reservoir in Western Poland. Environ Monit Assess 151:181–188. https://doi.org/10.1007/s10661-008-0259-9

42. Kowalczewska-Madura K, Gołdyn R, Dera M (2015) Spatial and seasonal changes of phosphorus internal loading in two lakes with different trophy. Ecol Eng 74:187–195. https://doi.org/10.1016/j.ecoleng.2014.10.033
43. Ejsmont-Karabin J (2012) The usefulness of zooplankton as lake ecosystem indicators: rotifer trophic state index. Pol J Ecol 60(2):339–350
44. Budzyńska A, Gołdyn R, Zagajewski P, Dondajewska R, Kowalczewska-Madura K (2009) The dynamics of *Planktothrix agardhii* population in shallow dimictic lake. Oceanol Hydrobiol Stud 33(2):7–12
45. Lu X, Bade DL, Leff LG, Mou X (2018) The relative importance of anammox and denitrification to total N_2 production in Lake Erie. J Great Lakes Res 44(3):428–435. https://doi.org/10.1016/j.glr.2018.03.008
46. Kozak A, Gołdyn R (2014) Variation in phyto- and zooplankton of restored Lake Uzarzewskie. Polish J Environ Stud 23(4):1201–1209
47. Rosińska J, Kozak A, Dondajewska R, Gołdyn R (2017) Cyanobacteria blooms before and during the restoration process of shallow urban lake. J Environ Manag 198:340–347. https://doi.org/10.1016/j.envman.2017.04.091
48. Kozak A, Rosińska J, Gołdyn R (2018) Changes in phytoplankton structure due to prematurely limited restoration treatments. Pol J Environ Stud 27(3):1097–1103
49. Kozak A, Kowalczewska-Madura K, Gołdyn R, Czart A (2014) Phytoplankton composition and physicochemical properties in Lake Swarz�dzkie (midwestern Poland) during restoration: preliminary results. Arch Polish Fish 22(1):17–28. https://doi.org/10.2478/aopf-2014-0003
50. Rosińska J, Gołdyn R (2015) Changes in macrophyte communities in Lake Swarzędzkie after the first year of restoration. Arch Polish Fish 23(1):43–52. https://doi.org/10.1515/aopf-2015-0005
51. Kowalczewska-Madura K, Dondajewska R, Gołdyn R (2011) Seasonal changes of phosphorus release from the bottom sediments of Rusalka Lake during the restoration process. Ecol Chem Eng A 18(2):219–224
52. Budzyńska A, Gołdyn R (2017) Domination of invasive Nostocales (Cyanoprocaryota) at 52°N latitude. Phycol Res 65(4):322–332. https://doi.org/10.1111/pre.12188
53. Haroon AM, Abdel-Aal EI (2016) Chemical composition and in vitro anti-algal activity of *Potamogeton crispus* and *Myriophyllum spicatum* extracts. Egypt J Aquat Res 42:393–404. https://doi.org/10.1016/j.ejar.2016.12.002
54. Higa T (1995) What is EM technology. College of Agriculture, University of Ryukyus, Okinawa
55. Lürling M, Waajen G, de Senerpont Domis LN (2016) Evaluation of several end-of-pipe measures proposed to control cyanobacteria. Aquat Ecol 50(3):499–516. https://doi.org/10.1007/s10452-015-9563-y
56. Pęczuła W (2012) Methods applied in cyanobacterial bloom control in shallow lakes and reservoirs. Ecol Chem Eng A 19(7):795–806
57. Kuczyńska-Kippen N, Joniak T (2016) Zooplankton diversity and macrophyte biometry in shallow water bodies of various trophic state. Hydrobiologia 774:39–51. https://doi.org/10.1007/s10750-015-2595-4
58. Dondajewska R, Kozak A, Rosińska J, Gołdyn R (2019) Water quality and phytoplankton structure changes under the influence of effective microorganisms (EM) and barley straw - Lake restoration case study. Sci Total Environ 660:1355–1366

Transformations of Wetlands in N-E Poland Postglacial Landscape and Its Relation to Lake Water Quality

Marek Kruk

Contents

Abstract Using an input-output ecosystem model, it was demonstrated that wetlands of the various stages of anthropogenic transformations typical for the postglacial landscape of N-E Poland vary in nitrogen (N) and phosphorus (P) forms retention. It concerned particularly nitrates (NO_3-N) and phosphates (PO_4-P), the most responsible for eutrophication effect. The calculation of retention effectiveness for studied wetland ecosystems showed that undrained minerotrophic peatland and transition bog retained about 70–85% of total N, including 95% of nitrates, but at the same time, they lost more than 50% of incoming phosphates. Wetlands with a shallow lake or transition bog crossed by a watercourse were considered as typical through-flowing systems with at most a moderate tendency to retained phosphates. Drained peatlands were considered dangerous in exporting N and P forms into recipient lakes and rivers. A directly drained minerotrophic fen lost even 70% of nitrates in the first year after melioration. Formerly drained peatlands can also lose nitrates, but first of all, they lost even 80–100% of total phosphorus

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input. All drained peatlands examined did not retain or lose total nitrogen and phosphates.

Keywords Drainage · Ecosystem budget · Nutrients · Retention · Water quality · Wetlands

1 Introduction

The ecosystem budget approach in landscape and watershed scale is widely used in the research and management evaluation of forest [1, 2] and agricultural [3] areas. These types of ecosystems, situated in uplands, are very suitable for system delimitation and nutrient budget calculation. This approach allows assessment of land-water interactions, in particular the impact of runoff from watershed on water quality. We should be aware also of several uncertainties using this difficult methodology [4].

Temperate wetlands, including riparian and inland, constitute a constant component connecting upland and water systems in many landscapes. They are especially distributed throughout postglacial areas in northern Europe and North America. The intensive research in recent decades of the biogeochemical role of these ecosystems, transitional in space and time, has been mostly motivated by freshwater bodies' protection [5]. The major interest of these studies was to calculate retention potential of wetlands for nitrogen (N) and phosphorus (P) inflowing from watershed [6]. The ecosystem budget method calculation was the dominant approach to assess nutrient removal effectiveness [6, 7]. It should be pointed out that wetlands and northern peatlands, in particular, had been subjected to various anthropogenic transformations of global and local origin [5, 8]. Hydrological transformations or modifications changed the ecosystem structure and internal nutrient cycling causing even a contrasting turn in retention and buffer role of N behavior [9]. Drained peatlands lost their capacity to intercept N and P [10] but also such environmentally dangerous elements as trace metals [11].

The article presents the aims on evaluation and, in sustainable management and water quality context, discusses the benefits and losses caused by maintaining and transforming temperate wetlands represented by peatlands in the Masurian Lakeland Region in N-E Poland. We would like to propose an ecosystem budget method as an effective tool for evaluating a buffer or threats to the range of Masurian wetlands in the context of water body protection. Wetlands cover nearly 5–15% of postglacial areas in Europe; however, the most of the catchment outflow in Lakeland regions passes through peatland areas [12]. Its role in maintaining the water quality of lakes and rivers should not be omitted [13].

For developing the practical aspects of the biogeochemical function of wetlands, it is important to discuss the usefulness of ecosystem budget method for evaluating the benefits of retention of nutrients in various types of wetlands. What is the role of anthropogenic changes of wetland ecosystems in the context of maintaining or improving the water quality of recipients as lakes and rivers? Finally, how can we introduce environmental and socioeconomic benefits of the retention effect of wetlands as sustainable management practice?

2 Material and Methods

2.1 Study Area

The study includes various kinds of undrained and managed peatlands characteristic for the present-day Lakeland landscape of N-E Poland. The study area is located in the Masurian Lakeland Region, west of the town of Mikołajki, and comprises the forest landscape of Puszcza Piska, as well as agricultural areas along the chain of lakes Inulec – Jorzec (Fig. 1). The geographical coordinates of the examined area are approximately 53°50′ N and 21°30′ E. A moderate temperate climate mostly influences the region, yet a continental climate also affects it, which lowers the mean annual temperature (6.5°C) and shortens the vegetative season [14]. The long-term level of precipitation amounts to 650 mm. However dry years do occur. The anthropogenic impact on Masurian wetlands concerns peat mining, drainage and agricultural cultivation, vegetation disturbance, and water pollution from fertilized fields.

2.2 Types of Wetlands and Its Transformations

The three stages of anthropogenic hydrological transformations were examined including seven types of wetland differ in trophic, hydrologic, and land-use properties (Table 1). Using very general criterion, we distinguish:

– Wetlands not disturbed hydrologically, which include inland fens and transitional bogs without surface outflow, closely related to "potholes" from US nomenclature [15] (nos. 1 and 2 in Table 1)
– Wetlands with mire vegetation and an active peat accumulation process, however cut by watercourse, including complexes with shallow water bodies and transitional moor (nos. 3 and 4 in Table 1)
– Wetlands meliorated and cultivated, including directly and formerly drained peatlands in various landscape situations (nos. 5, 6, and 7 in Table 1)

The Masurian wetlands are much more diverse and complex. These areas listed represent the broad range of habitats joined by its more or less transitional position in the catchment areas of lakes and rivers. The bog peatland on one side, hydrologically isolated to a high degree, and meliorated systems on the other side, directly connected with lakes by drains or ditches, were studied (Table 1).

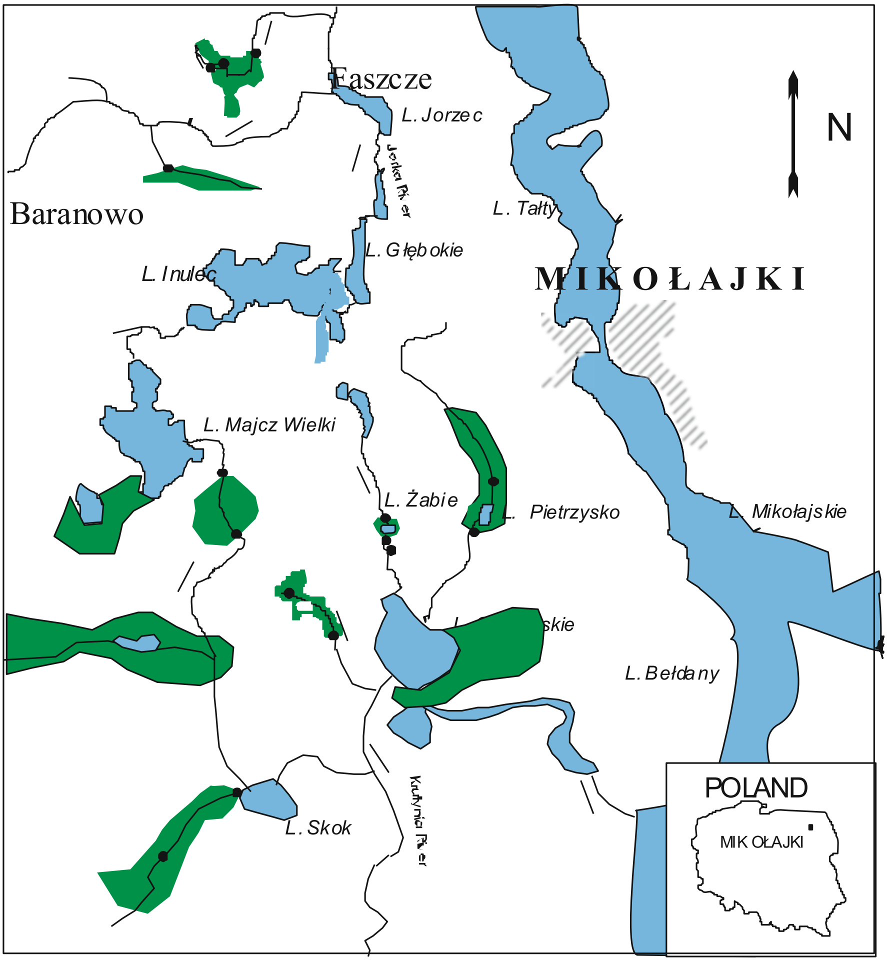

Fig. 1 Localization of study area near the town of Mikolajki in Masurian Lakeland, N-E Poland, with lakes (light gray patches) and larger wetlands (strong gray patches)

2.3 Methods

The methodic basis of the research was the ecosystem approach with a well-defined area of wetland with measurable inflows and outflows [6]. We simplified the general budget formula by omitting inflow from the atmosphere as a component not considered in the watershed-wetland-freshwater relations. Retention (+) or losses (−) of particular nutrient x (RL_x) were calculated as differences between inflow and outflow total loads using a general budget formula:

Table 1 Presentation of wetland types dominated in the study area (Masurian Lakeland, N-E Poland), after Kruk [15, 16]

No	Wetland type	Years of budget studies	Dominating vegetation	Position in lake/river catchment area
1	Minerotrophic fen without surface outflow	2	Cattail rushes	Infield, low catch-ment position
2	Minerotrophic wetland without surface outflow	1	Willow shrubberies	Infield, low catch-ment position
3	Transition bog without sur-face outflow	2	*Sphagnum* bog, sedge rushes, birch trees	Isolated, upper posi-tion, as patches
4	Peatland with a shallow lake with surface throughflow	2	Alder swamp, sedge rushes	Connected with lake by a stream
5	Peatland with a shallow lake with surface throughflow	2	Alder swamp, reed rushes	Connected with lake by a stream
6	Peatland with transition bog with surface throughflow	4	*Sphagnum* bog, sedge, and reed rushes	Connected with lake by a stream
7	Minerotrophic peatland directly drained	1	Willow shrubberies, nettle	Connected with lake by drain
8	Formerly drained peatland with meadow (midforest)	2	Wet, rich meadow	Connected with lake by a ditch
9	Formerly drained peatland with meadow (midfield)	4	Cultivated meadow	Connected with lake by drains and ditch

$$\mathrm{RL}_x = \mathrm{Iw}_x - \mathrm{Ow}_x,$$

where Iw is inflow from the catchment area and Ow streamflow or subsurface output.

The nutrient retention effectiveness, identical to nutrient removal effectiveness used by Johnston [6], was calculated as the relation between yearly retention and input amounts, expressed in percent. The volume of subsurface throughflow of water and nutrients was measured using hydrogeological examinations of hydraulic gradients of the water table and ground permeability (see details in Kruk) [14]. Discharge of streams, ditches, and drains supplying, as well as draining, the studied peatlands was measured using culvert pipes and immersible floats.

The water samples were collected once a month in each peatland during the years of the study. Total phosphorus (TP) concentration was determined using digestion in perchloric acid method and total nitrogen (TN) concentration by Kjeldahl method [16]. Dissolved inorganic phosphorus (PO_4-P) was examined by stannous chloride method and nitrates (NO_3-N) by potentiometric or reduction into nitrite methods. An error of estimation of the obtained N and P budget results was considered on the level ±25 to 30% [17].

3 Results and Discussion

3.1 Nutrient Retention in Wetland Types

Using the input-output ecosystem budget model [14], we demonstrated that wetlands listed above vary in N and P retention, in total dissolved and particulate forms, as well as in NO_3-N and PO_4-P, the most responsible forms for eutrophication effect.

From seven wetland types taken into account, five of them exhibit positive input-output budget in case of NO_3-N. The mean retention of this nutrient in 3–4 years of study in minerotrophic mires without a runoff and in wetlands with water bodies with stream throughflow reaches even 100 kg/ha/year. In contrast, transition peatlands remove only a few kg/ha/year of nitrites. Two of the three cultivated systems lose NO_3-N but not inconsiderable amounts (Table 2). The retention of total nitrogen (TN) shows a similar picture, but in this case, all examined drained peatlands lose N, even in a level greater than 10 (Table 2).

As drained peatlands tend to lose nitrates, similarly, minerotrophic and transition mires without surface outflow tend to lose phosphates but in very limited amounts in case of bogs (Table 3). The wetlands as the systems with surface throughflow intercept PO_4-P in quite a considerable level. A wetland with a boggy shallow lake can accumulate even 0.33 kg/ha/year of phosphate. Retention of this nutrient inflowing from watershed by drained peatlands was very low (Table 3). Interesting differences appear when we compare the described retention of phosphate with the studied wetlands' ability to retain total phosphorus. Tendencies to lose this nutrient

Table 2 Retention of NO_3-N and total nitrogen (TN) by wetland types in Masurian Region (N-E Poland)

Wetland types	*n*	Retention of NO_3-N		Retention of N	
		kg/ha/year	% of watershed inflow	kg/ha/year	% of watershed inflow
Minerotrophic peatland without surface outflow	3	65.5	*95*	66.7[a]	*69*
Transition bog without surface outflow	2	3.8	*73*	4.2[a]	*60*
Wetland with lake with surface throughflow	4	100.5	*19*	113.1	*11*
Transition bog with surface throughflow	4	5.5	*9*	9.3	*8*
Minerotrophic peatland directly drained	1	−6.7	*−231*	−10.3	*−123*
Formerly drained peatland with meadow (midforest)	2	−5.1	*−68*	−15.3	*−36*
Formerly drained peatland with meadow (pasture)	4	8.4	*13*	−0.8	*−1*

Retention is expressed in absolute values (kg/ha/year) and relative values as retention effectiveness (% of watershed outflow in italics). Mean from *n* years
[a]*DTN* dissolved total nitrogen

Table 3 Retention of PO_4-P and total phosphorus (TP) by wetland types in Masurian Region (N-E Poland)

Wetland types	*n*	Retention of PO_4-P kg/ha/year	Retention of PO_4-P % of watershed inflow	Retention of TP kg/ha/year	Retention of TP % of watershed inflow
Minerotrophic peatland without surface outflow	3	−0.08	*−53*	n.d.	n.d.
Transition bog without surface outflow	2	−0.005	*−69*	n.d.	n.d.
Wetland with shallow lake with surface throughflow	4	0.33	*9*	−1.99	*−20*
Transition bog with surface throughflow	4	0.12	*22*	0.18	*11*
Minerotrophic peatland directly drained	1	0.002	*1*	n.d.	n.d.
Formerly drained peatland with meadow (midforest)	2	0.04	*5*	−1.28	*−97*
Formerly drained peatland with meadow (pasture)	4	0.01	*6*	−0.35	*−106*

Retention is expressed in absolute values (kg/ha/year) and relative values as retention effectiveness (% of watershed outflow in italics). Mean from *n* years
n.d. no data

prevailed, and wetlands with a water body lost almost 2 kg/ha/year of total P, and formerly drained peatlands also lost a considerable amount (Table 3). Transition bog with throughflow stream was the only system examined able to intercept phosphorus but in a limited level (Table 3).

The calculation of retention effectiveness for the studied wetland ecosystems surprisingly show that it is difficult to find the most beneficial management situation or category of wetland, from a water quality point of view. Firstly, undrained minerotrophic peatland and transition bog retained about 70–85% of total N, including 95% of nitrates (Table 2), but at the same time, they lost more than 50% of incoming phosphates (Table 3). Wetlands with a shallow lake or transition bog crossed by watercourse were considered as typical through-flowing systems with almost equal amounts of inflowing and outflowing nutrients. Drained peatland was the most dangerous in exporting N and P forms into recipient lakes and rivers (Tables 2 and 3). Directly drained minerotrophic fen lost even 70% of nitrates in the first year after melioration. However, this leaching effect did not cause phosphates. Formerly drained peatlands can also lose nitrates, but originally, they lost even 80–100% of total phosphorus input, mostly in organic and particulate forms. All drained peatlands examined did not retain or lose effectively total nitrogen and phosphates (Tables 2 and 3).

3.2 The Variability of Nutrient Retention by Wetland Types

We have analyzed the variability of above two indicators of nutrient retention properties of wetlands, namely, retention in absolute values and relative effectiveness of retention. They have been described separately, but we are aware that only a few combinations of both should be useful in estimating wetlands' role in protecting freshwater bodies. The presentation of every wetland and study year on the plot was conducted, in which *x*-axis represents absolute retention of nutrient in kg/ha/year and y-axis its retention effectiveness in % of watershed inflow (Fig. 2). We also used for comparison more general categories of wetlands described in Sect. 2.2, namely,

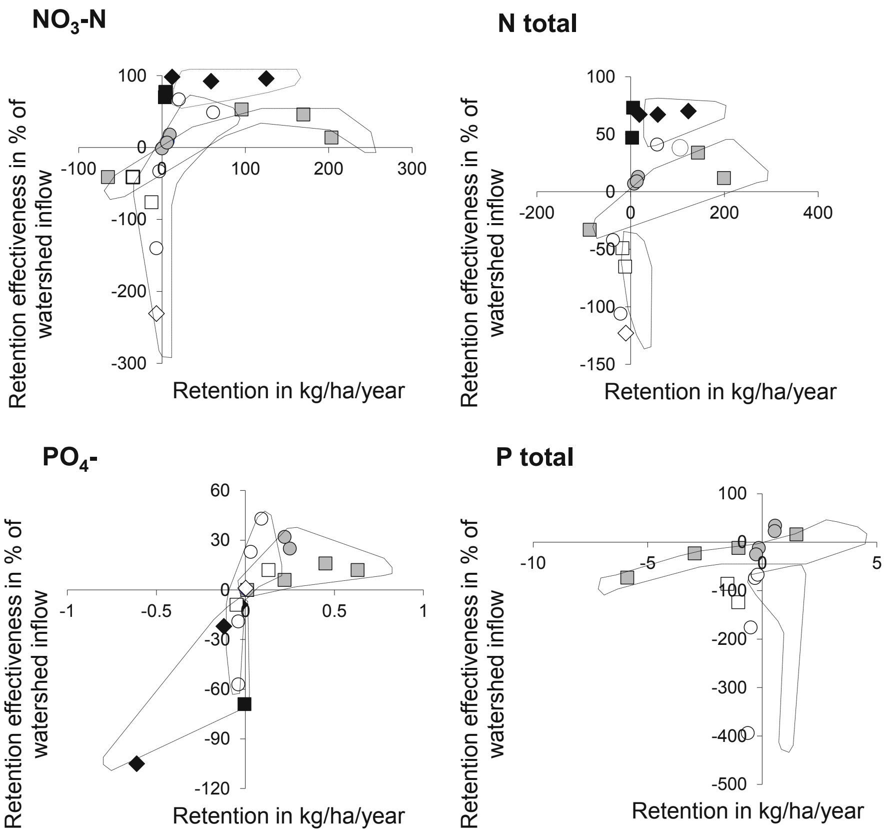

Fig. 2 Wetland nutrient retention plots are demonstrating relations between retention of nutrient in kg/ha/year and its retention effectiveness expressed in % of watershed inflow. Wetlands without surface outflow – black: black diamonds – minerotrophic fens, black square – ombrotrophic bog. Wetlands along streams – gray: gray square – wetlands with shallow lakes and alder carr, gray circle – wetlands with transition bog. Drained peatlands – white: white diamond – directly drained minerotrophic fen, white square – formerly drained peatland in a forested landscape, white circle – formerly drained peatland in agricultural use

undisturbed mires without surface mires, wetlands with through-flowing streams, and drained, cultivated peatlands. Such an analysis aims to classify nutrient retention properties of wetlands examined and to evaluate its benefits or threats for the water quality in recipient lakes and rivers.

The analysis of the retention of NO_3-N and N total demonstrates a similar picture (Fig. 2). The category of mires without surface outflow, located in the highest position in the plots, indicated the great potential to retain N forms. Another wetlands, with streams, were considerably stretched horizontally. Their abilities to retain nitrogen were limited, but increased amounts of N retention and losses had happened. Losses of N forms prevail noticeably in the majority of drained peatlands, creating the most dangerous category of wetlands, in case of N flowing into recipient waters (Fig. 2).

Analyzing plot distribution of wetland retention indicators for phosphate and total phosphorus, we also find the most consistent and consequently PO_4-P retention association of wetlands which creates ecosystems with streams. Drained peatlands were almost symmetrically divided between a tendency to retain and lose phosphates. In contrast to nitrogen retention, mires without surface outflow proved the most favorable for phosphate transport into freshwaters (Fig. 2). In case of wetland distribution due to P total retention, very visible differences arise between wetlands with through-flowing waters and drained peatlands. The first mentioned systems exhibit a dominant tendency to lose phosphorus, but not very effectively, and some examples of P retention occur. Based on this, drained peatlands represent a considerable source, especially in relative values, of total phosphorus into lakes and rivers (Fig. 2).

3.3 Wetland Nutrient Retention and Water Quality

In Lakeland areas, as in the Masurian Region, lakes and wetlands connected by groundwater and a net of watercourses constitute one functional self-dependent system with relations between inner watershed without surface outflow wetlands and water-recipient bodies, mostly lakes. A considerable amount of nitrogen can be intercepted in through-flowing groundwater even in the higher elevated watershed, which is caused probably by very intensive denitrification in submerged organic soils of these relic wetlands [18]. The sediment processes are responsible for phosphate retention by wetlands with shallow lakes and through-flowing streams [19]. Here, especially transition peatlands with retention of about 1/3 of an incoming load of phosphates should be positively evaluated (Fig. 2).

Greater or less potential of examined wetlands to remove nitrogen and phosphorus (Fig. 2) could be economically evaluated by analysis of the difference in nutrient loading into lakes with and without wetlands or with undrained and drained wetlands. The counter-eutrophication effect of maintaining undrained wetlands in lake watershed can then be visibly demonstrated. This positive effect could be economically evaluated from the benefits of fisheries more noble species and the usefulness of water for recreation and aquatic sports (Fig. 3).

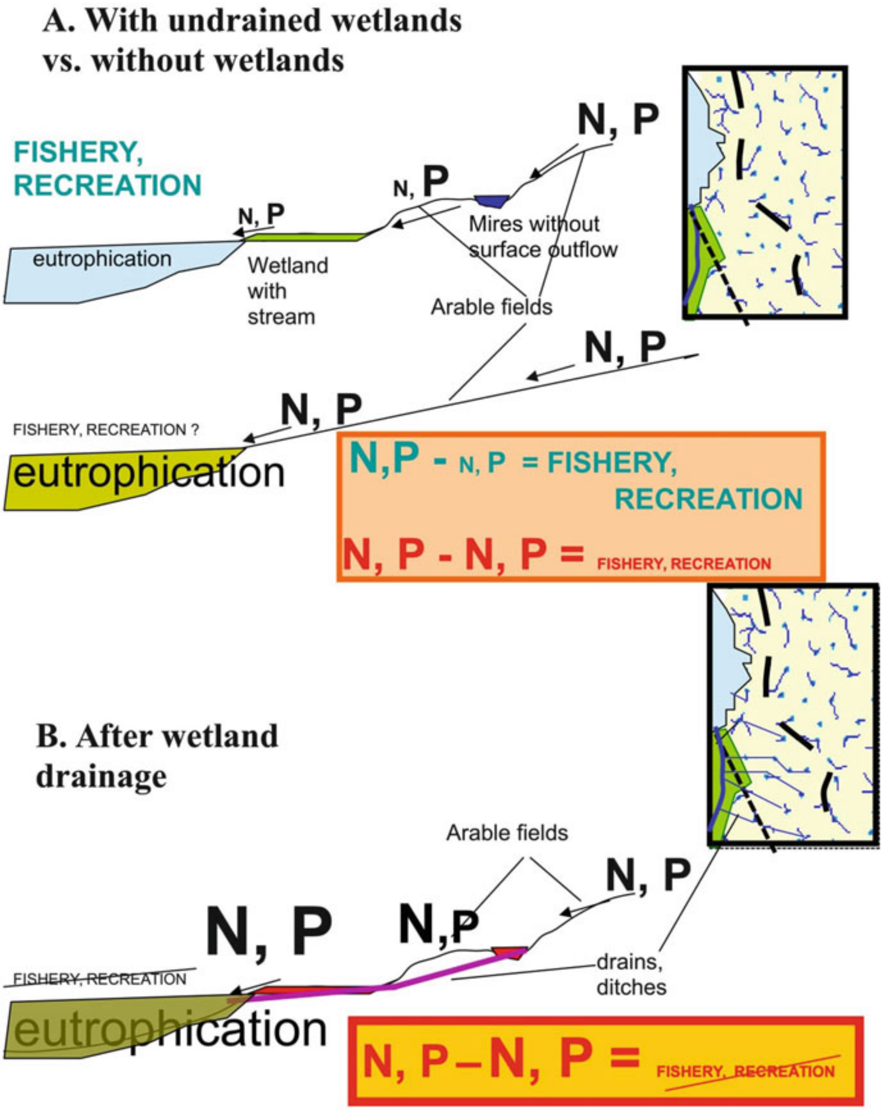

Fig. 3 Schematic presentation of wetlands role in transporting nutrients from arable fields into the lake in the watershed (maps in boxes) and in socioeconomic benefits concern usefulness of recipient waters by fishery and recreation activities. Changes in nitrogen and phosphorus loads by Table 2 and Fig. 2. (**a**) Illustration of undrained wetland values. The occurrence of wetlands in watershed influences positively on the fishery and recreation (symbolized by large letters) vs. situation without wetlands, enhanced eutrophication, and inhibits usefulness of lake waters (symbolized by small letters). (**b**) Drainage of wetlands in lake watershed (see box) increases loads of N and P into the lake, accelerate eutrophication, and can minimize or eliminate benefits from fishery and recreation. The dotted line on maps – course of the transect

The analysis conducted above using retention plots shows a kind of biogeochemical diversity or specialization among analyzed types of wetlands. Namely, mires without surface outflow intercept nitrates in a high degree but lose phosphates, which can be retained in lower located wetlands along streams. The conclusion is that a topographic configuration of different types of wetlands in a watershed can be more effective in nutrient interception than a particular wetland type alone.

We propose a comparative evaluation of wetlands in a lake or river watershed as a system for improving water quality in recipient freshwater bodies based on the calculation of two environmental situations: watershed with and without undrained wetlands or drained ones. The basic and initial calculations of this evaluation come from the ecosystem budget formula, which can be transformed into economic benefits in sequence, schematically presented in Fig. 3.

It is possible to calculate and demonstrate the real economic value of wetlands as buffers of nonpoint agricultural pollution for freshwater ecosystems, using the presented scheme. It should, however, be noticed that the situation without wetlands rather rarely happened and a comparison between situations with undrained and drained wetlands is realistic and widely happens in watersheds of lakes or rivers in N-E Poland and many regions of Europe. It is important to differentiate due to ecosystem budget results, indicating that drainage of wetland moves an additional load of N and P (Fig. 2). The threat to water quality is even greater in this case than in a situation when watershed only consists of agricultural fields. As a consequence, the eutrophication effect and cost of losses in fishery and recreation would be considerable (Fig. 3). There is also an arising need to integrate research and policy of wetland management with agriculture and sustainable water environmental policies to see paramount importance of wetland ecosystems.

4 Conclusions

(a) Wetland types characteristic for Lakeland landscape of N-E Poland vary in nutrient retention to a considerable degree. It depends on anthropogenic hydrological transformations and position in the watershed.
(b) The best potential for N forms retention, as the resultant of absolute and relative values of retention, was presented by inland without surface outflow mires and – the highest threat potential for waters – by drained peatlands.
(c) Moderate potential for retention of phosphate is characteristic for wetlands with through-flowing streams. This nutrient is, however, lost by inland mires. P total is consequently eroded from drained peatlands – constituting a high-risk potential for freshwater bodies.
(d) Due to biogeochemical differences among wetlands, a specific landscape spatial system of various wetland types in the watershed can be more effective in nutrient retention than a particular wetland type alone.
(e) Based on ecosystem budget calculations, it seems to be possible to assess for recipient waters users: socioeconomic benefits caused by wetland conservation, as well as losses resulted from wetland drainage.

References

1. Waring RH, Running SW (1998) Forest ecosystems. Analysis at multiple scales. Academic Press, San Diego, p 370
2. Vadeboncoeur MA, Hamburg SP, Yanai RD, Blum JD (2014) Rates of sustainable forest harvest depend on rotation length and weathering of soil minerals. For Ecol Manag 318:194–205
3. Cherry KA, Shepherd M, Withers PJA, Mooney SJ (2008) Assessing the effectiveness of actions to mitigate nutrient loss from agriculture: a review of methods. Sci Total Environ 406:1–23
4. Yanai RD, See CR, Campbell JL (2018) Current practices in reporting uncertainty in ecosystem ecology. Ecosystems 21:971–981
5. Junk WJ, An S, Finlayson CM, Gopal B, Květ J, Mitsch WJ, Robarts RD (2013) Current state of knowledge regarding the world's wetlands and their future under global climate change: a synthesis. Aquat Sci 75:151–167
6. Johnston CA (1991) Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. Crit Rev Environ Control 21:491–565
7. Fellman JB, D'Amore DV (2007) Nitrogen and phosphorus mineralization in three wetland types in Southeast Alaska, USA. Wetlands 27:44–53
8. Gorham E (1991) Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecol Appl 1:182–195
9. Kruk M (1997) Effect of draining on nitrogen flow through mires in agricultural landscape. Ekol Pol 45:441–460
10. Marttila H, Karjalainen S-M, Kuoppala M, Nieminen ML, Kløve B, Hellsten S (2018) Elevated nutrient concentrations in headwaters affected by drained peatland. Sci Total Environ 643:1304–1313
11. Kruk M (2005) Trace metal fluxes in a Sphagnum peatland – humic lake system as a consequence of drainage. Water Soil Air Pollut 168:213–233
12. Kruk M (1999) Peatlands – barrier or source of phosphorus inflow to lakes? Acta Hydrobiol 41 (Suppl):73–81
13. Powers SM, Tank JL, Robertson DM (2015) Control of nitrogen and phosphorus transport by reservoirs in agricultural landscapes. Biogeochemistry 124:417–439
14. Kruk M (1990) The processing of elements by mires in agricultural landscape: mass balances based on sub-surface hydrology. Ekol Pol 38:73–117
15. Mitsch WJ, Gosselink JG (2015) Wetlands. Wiley, New York, p 456
16. Golterman HL (1969) Methods for chemical analysis of fresh waters. IBP handbook no. 8, Blackwell Scientific, Oxford, p 172
17. Kruk M (2000) Biogeochemical functioning of hydrologically modified peatlands and its effect in eutrophication of freshwaters. Pol J Ecol 48:103–161
18. Koops JG, van Beusichem ML, Oenema O (1997) Nitrogen loss from grassland on peat soils through nitrous oxide production. Plant Soil 188:119–130
19. Jonsson A (1997) Fe and Al sedimentation and their importance as carriers for P, N and C in a large humic lake in northern Sweden. Water Soil Air Pollut 99:283–295

Nature of Polish Tatra Lakes

Joanna Galas and Grzegorz Tończyk

Contents

Abstract The Tatra Mountains are small range of mountains of alpine character which belong to the Carpathian chain. All components of Tatra nature have a unique character. Tatra lakes are valuable and special environments for hydrobionts. The geological and hydrological attributes as well as water chemistry determining the specificity of those lakes have been described. The uniqueness of life forms present there has been described with regard to epilithic algal, zooplankton, macro-zoobenthos and fish communities. The main features determining the structure of Tatra lakes hydrobionts communities are lake altitude, catchment character, water temperature and ice cover period. The influence of anthropogenic activities such as water acidification is also important. Fish introduction to the originally fishless lakes or their natural presence also has significant impact. Vertical distribution of lake organisms is distinctly marked in particular lakes. Based on many parameters, the Tatra lakes can be divided into two groups: of alpine and subalpine character. The water bodies situated on the Polish side of the Tatras are not acidified when compared to those in the Republic of Slovakia. The Tatras and their lakes are the

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farthest located in Northern Europe centre of endemism and the largest site of high mountain fauna and flora occurrence in Central Europe.

Keywords Abiotic and biotic factors · Hydrobiont assemblages · Poland · Subalpine and alpine lakes · Tatra Mts.

1 Introduction

The Tatra Mountains (the Tatras) constitute a small but distinctly separated massif of the Carpathians that evolved during the Alpide orogeny, at the same time as the Alps, Pyrenees, Apennines and Himalayas. The Tatra Mts. occupy an area of 785 km^2, of which a bigger part (610 km^2) lies in the Republic of Slovakia, while the Polish Tatras occupy the smaller part (175 km^2). The length of the Tatras is 56.3 km, and they are divided into Western Tatra and Eastern Tatra with Tatra Bielskie and High Tatra. Unique parts of these mountains are protected by Tatra National Park (TPN) in Poland and Tateranský Národný Park (TANAP) in Slovakia. As a whole the Tatras are also on UNESCO biosphere reserves list.

Despite the small area, the Tatra massif has evident and well-developed postglacial relief, which resulted in Tatras being the farthest northern centre of endemism in Central Europe with the highest number of high mountain flora and fauna species in Central Europe. All components of Tatra nature are unique, and this concerns particularly subalpine and alpine lakes.

The first data on Tatra lakes biology was published by the Enlightenment multidisciplinary activist priest Stanisław Staszic [1]. The detailed history of the Tatra lakes studies was summarized by Kownacki and Ślusarczyk [2, 3]. More information on biological investigations carried there are available in a few monographs of the Tatra National Park, e.g. Mirek ed. [4]. The aim of this work is to present data concerning various aspects of Tatra lakes function against a background of current hydrobiological knowledge. This synthetic paper in English should allow the reader to introduce to the unique nature of Tatra lakes located in Poland in a broader way than general papers of local character concerning this issue published mainly in Polish.

2 Geology, Hydrology and Water Chemistry

In the Tatras there are over 260 permanent and seasonal lakes (big and small water bodies). Only 76 of them are located in Poland in the area of the Tatra National Park (Fig. 1, Table 1). Most of them are lying in the High Tatra Mts.; they are of alpine character resulting from the crystalline core composed largely of magmatic rock (granitoids) highly resistant to erosion. In the Western Tatras, which are built of crystalline metamorphic rock and located at a lower altitude, only eight small ponds

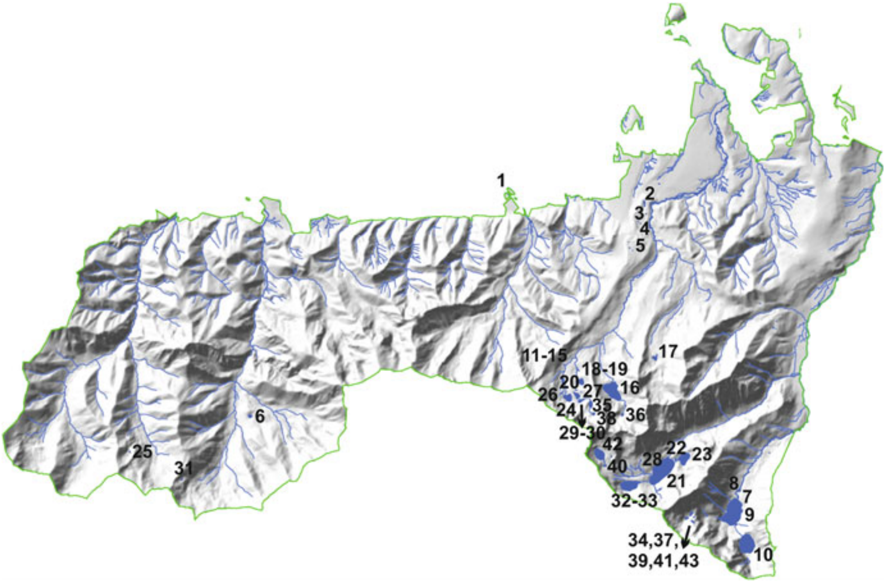

Fig. 1 Lakes of the Polish Tatra Mts. The order by altitude position. (1) Stawki pod Capkami Lake, (2) Toporowy Niżni Staw Lake, (3) Toporowy Wyżni Staw Lake, (4) Toporowy Przedni Staw Lake, (5) Kopanica Marsh (6) Smreczyński Staw Lake, (7) Żabie Lake, (8) pond near Morskie Oko Lake, (9) Morskie Oko Lake, (10) Czarny Staw Lake, (11) Jedyniak Lake, (12) Dwoiśniak Lake, (13) Troiśniak Lake, (14) Samotniak Lake, (15) Litworowy Staw Lake, (16) Czarny Gąsienicowy Staw Lake, (17) Czerwony Staw Lake in Pańszczyca Valley, (18) Dwoisty Wschodni Staw Lake, (19) Dwoisty Zachodni Staw Lake, (20) Dwoiśniaczek Lake, (21) Wielki Staw Lake in the Five Polish Lake Valley, (22) Mały Staw Lake in the Five Polish Lake Valley, (24) Przedni Staw Lake in the Five Polish Lake Valley, (25) Dudowe Stawki Lake, (26) Kotlinowy Stawek Lake, (27) Kurtkowiec Lake, (28) ponds I–IV in the Five Polish Lake Valley, (29) Czerwony Wschodni Staw Lake, (30) Czerwony Zachodni Staw Lake, (31) Siwe Stawki Lake, (32) Szpiglasowe ponds I–III, (33) Czarny Staw Lake in the Five Polish Lake Valley, (34) Staw Staszica Lake, (35) Długi Staw Lake, (36) Zmarzły Lake, (37) Zadni Staw Lake, (38) Wyżni Mnichowy Stawek I Lake, (39) Wyżnie Mnichowe Stawki II–VIII Lakes, (40) Wyżni Mnichowy Stawek IX Lake, (41) Wole Oko Lake, (42) Zadni Staw Lake, (43) Zadni Mnichowy Stawek Lake

Table 1 Classification of Polish Tatra lakes (ac. Szaflarski [5], modified)

Type of lake	Big, forest zone	Small, shallow	Deep	Shallow, high located
Altitude (m a.s.l.)	900–1,400	1,450–1,750	1,490–1,800	1,800–2,050
Months without ice	6	3–5	4–7	3
Max. water temperature (°C)	16–20	4–18	8–14	8
Example of lake	Morskie Oko Lake	Siwy Stawek Pond	Czarny pod Rysami Lake	Zadni Staw Lake in the Five Polish Lake Valley

are present. The dominant vegetation in the area is coniferous forests below 1,550 m a.s.l., dwarf pine between 1,550 and 1,800 m a.s.l. and alpine meadows or bare rocks in the alpine zone above 1,800 m a.s.l. The number of the High Tatra lakes lying in the forest zone is small – the biggest one being the Morskie Oko Lake (Table 1, Fig. 2). The High Tatra lakes are of glacial origin and are situated mostly above the tree line in the alpine environment and have small, steep and sparsely vegetated catchments (Figs. 3 and 4) and undeveloped (lithosol, ranker) soils. The deep lakes situated at the altitude between 1,490 and 1,800 m a.s.l. have maximum water temperature between 8 and 14°C, and they are ice-covered for 4–7 months, while the highest located ones have no ice cover for only 3 months [5] (Table 1).

Most of the lakes are filled with water throughout the whole year, fed mainly by precipitation and melting snow, but they are also supplied by numerous springs and water draining through the rocky catchment [6]. Lakes water level fluctuates from the highest during snow melting season and heavy rains to the lowest one in fall and winter.

Water chemistry of Tatra lakes is largely determined by the atmospheric loads, low rock weathering processes and limited soil development, which results in highly diluted waters, both in terms of major salts and nutrients. The waters of the granitic High Tatra lakes are oligotrophic, with very low conductivity and ionic content, while lakes and ponds in the karstic Western Tatras have higher concentration of base cations and mineral content [7, 8]. Bedrock geology, amount of soil and kind of

Fig. 2 Morskie Oko Lake

vegetation in the catchments strongly influence these differences. Concentration of phosphorus, organic carbon, nitrogen and chlorophyll-*a* is the lowest in lakes with sparse vegetation and thin soil cover in the catchment areas, while concentration of nitrate shows an opposite trend [9]. Due to the poor buffering capacities of soil and rocks in the watersheds, the High Tatra lakes are sensitive to acidification. Extremely high emissions of sulphur and nitrogen compounds in Central Europe with the maximum of acid deposition in the middle of the 1980s of the twentieth century [7] determined the presence of high nitrate levels in the water of many remote lakes including the Polish Tatra ones [10]. During that time, water chemistry of mostly smaller and shallow, poorly buffered High Tatra lakes showed various stages of acidification [11, 12]. The Tatra lakes have exhibited significant recovery from acidification due to the rapid decline in S and N emission in Poland and Central Europe since the year 1990 [13]. After that time the decrease in nitrate concentration and increased acid neutralising capacity (ANC) values were noted [7]. In 2000 based on several environmental variables, most Polish Tatra lakes year were classified as subalpine or alpine non-acidified lakes (Fig. 3) [14]. It can be supposed that such a classification is still valid.

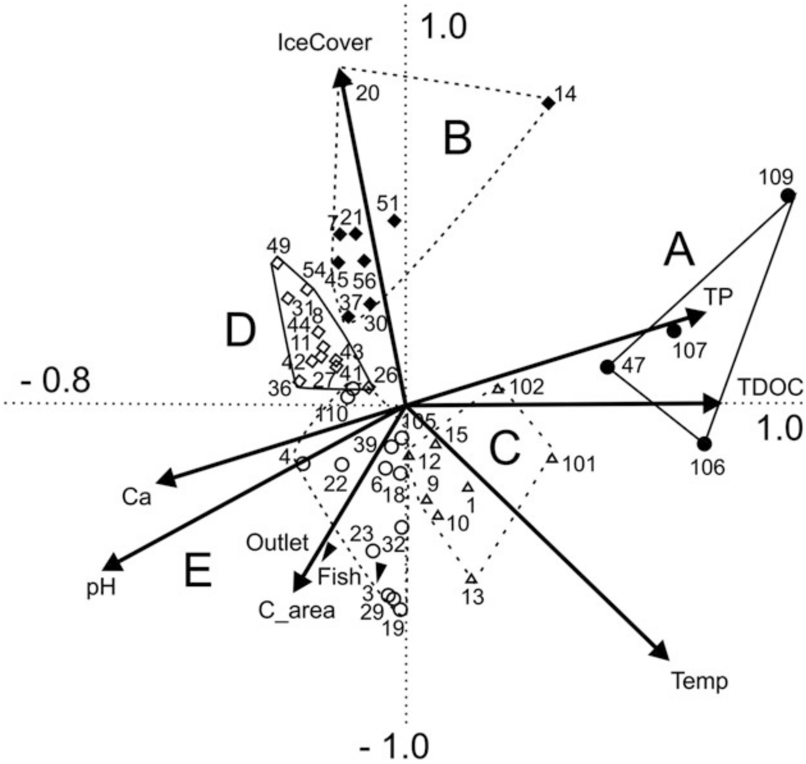

Fig. 3 CCA analyses based on environmental parameters, which classified the Tatra lakes (**a**) strongly acidified lakes, (**b**) alpine acidified lakes, (**c**) subalpine acidified lakes, (**d**) subalpine non-acidified lakes, (**e**) alpine non-acidified lakes (modified from Krno et al. [14])

Fig. 4 Długi Staw Lake

The alpine lakes, usually oligotrophic ones have no littoral, phytoplankton growth is limited and thus water transparency is high: e.g. Secchi depth 17 m is in the Czarny pod Rysami Lake (Galas, unpublished data). The water transparency in lakes located at lower altitude decreases up to 2 m, e.g. in Morskie Oko Lake, which has higher species variability.

The Tatra lakes catchments are relatively undisturbed because grazing, logging and any other kind of land use have been prohibited since the 1950s when the Tatras became the National Park. Tourism does not significantly influence lake water chemistry because swimming and fishing are forbidden there and tourist trails are usually situated not close to the lakes.

The extreme climatic conditions in terms of temperature and light of high mountain lakes strongly influence their ecology: lake waters are mainly oligotrophic with low productivity and simple ecosystem structure. It consists with short food webs, usually dominated by only a few species. Lake biota may respond to short-term seasonal changes of temperature and light regimes by changing the length of life cycles of organisms living there.

3 Epilithic Algal Communities

The algal communities from the Tatra standing waters are represented mainly by diatoms (Bacillariophyceae), whereas green algae (Chlorophyta), golden algae (Chrysophyceae) and euglenids (Euglenophyceae) are less abundant than epilithic diatoms. The blue-green algae (*Cyanobacteria*) now classified as prokaryote also occur in Tatra lakes.

In the dystrophic and rich in organic matter small lakes and ponds situated in the forested or meadow catchments as well as in alpine lakes with introduced fish, euglenophytes occur [15]. They are represented by widespread cosmopolitan taxa such as *Trachelomonas armata* var. *longispina* Playfair 1915 and *Phacus wettsteinii* Drezepolski 1925, both found for the first time in Tatra lakes more than half a century ago. The colourless species *Menoidium distractum* Wermel 1924 and *Petalomonas klebsii* Christen 1962 have recently been found in Toporowy Staw Wyżni Lake. Also green algae prefer shallow ponds and lakes situated at a lower elevation [16]. In Tatra water bodies, up to now, 90 taxa of chlorophytes were found. Among them three species, *Monoraphidium tatrae* Hindák 1977, *Pediastrum braunii* Komárek and Jankovská 2001 and *Thelesphaera olivacea* Fott 1976, occur mostly in the mountainous areas. *Pediastrum braunii* has a boreal-alpine distribution and is regarded as an arctic-alpine form that occurs very rarely in the world. Litworowy Staw Lake was the most algae-rich. In this lake, *Cosmarium hornavanense* Ruzicka 1949, a new species for science, was found.

Forty different taxa of golden algae have been reported from the Tatra National Park so far [17]. Higher abundance of chrysophytes was stated in smaller Zmarzły Staw Gąsienicowy Lake than in larger Czarny Gąsienicowy Lake [18], which could be attributed to the difference in the surface and catchment areas between these two lakes. It is worth nothing that other groups of algae are represented in these two lakes in similar quantities and abundance; Zmarzły Staw Gąsienicowy Lake has richer algal species composition than Czarny Staw Gąsienicowy Lake [19].

Diatoms are the largest group of algae of the Tatra waters, and their communities, especially in streams, are best described [20]. In the sub-mountain and high-altitude big and small lakes of the Polish Tatras, epilithic diatoms are the most species-rich primary producers. In alpine lakes they are also good bioindicators, since they may react to changes in environmental conditions [21]. In standing waters from the whole Tatra Mts., there are 447 diatom species and varieties that have been identified [22]. Littoral diatom communities in the High Tatra lakes are dominated by alpine and acidophilous taxa (mainly belonging to the genus *Achnanthes* Bory de Saint-Vincent 1822 *s.l.*), depending on various stages of acidification. In shallow, moderately acidified Polish Tatra lakes or ponds such as Mnichowy, Długi and Zmarzły Gąsienicowy lakes, acidophilous diatoms prevail, with *Psammothidium marginulatum* (Grunow) Bukhtiyarova and Round 1996 (=*Achnanthes marginulata*) being the most numerous [8, 19]. In mostly very deep, non-acidified lakes such as Czarny pod Rysami, Wielki and Czarny Polski, where circumneutral organisms prevail and among them *Achnanthidium minutissimum* (Kützing) Czarnecki 1994 (= *Achnanthes minutissima*) is dominant in most cases. Littoral diatom composition structure mostly confirms the status of alpine lakes based on their water chemistry. In blue-green algae association, *Pleurocapsa aurantiaca* Geitler 1931 (= *Scopulonema polonicum*) and *Chamaesiphon polonicus* (Rostafinski) Hansgirg 1893 dominate and grow at the lake shores [23]. The black colour ("czarny" in Polish) of their thalli gives the name to several lakes, e.g. Czarny Staw Gąsienicowy or Czarny Staw pod Rysami. As the lake depth increases, stratification in blue-green algae community distribution and chromatic adaptation to changing light condition are observed. In the Wielki Staw Lake in Five Polish Lake Valley, *Chamaesiphon subglobosus* (Rostafinski) Lemmermann 1907 and *Pseudophormidium tatricum* (Starmach) Anagnostidis 2001 (= *Plectronema tatrica*) grow at the depth of 5–40 m.

4 Zooplankton

The zooplankton of the oligotrophic Tatra lakes is characterized by low diversity, low numbers of animals and small biomass [24, 25]. The important feature, connected with the Tatra lakes hydrogeomorphology, is lack of differentiation onto pelagial and littoral zooplankton [26]. The impact of lake altitude and orientation (situation on the northern or southern slope of the range) on the occurrence of particular species is also an important factor [25, 27]. Anthropogenic parameters also influenced the character of the zooplankton communities. Water acidification, observed in the middle of the 1980s of the twentieth century, caused dramatic changes in the quality and quantity of zooplankton [7, 10, 11, 27, 28]. Such a phenomenon has now lower impact on the structure of planktonic species community than a few decades ago, but effects of environmental changes caused by atmospheric pollution can be still observed [7]. The other anthropogenic feature strongly influencing zooplankton species composition was the introduction of salmonids into initially fishless lakes [29, 30].

Vertical distribution is a characteristic feature of planktonic organisms. In the uppermost water layer of Tatra lakes, there is no zooplankton, and its higher abundance is found at the depth of 10–30 m and at the bottom since they have good oxygen conditions. Vertical distributions have daily character – zooplankton fauna concentrates during higher insolation close to the bottom, while it appears close to the surface layer when light is dispersed on clouded days or at nights. The habitat selection by different groups of zooplankton has been stated, e.g. rotifers at the depth of approximately 15 m and at the bottom layer. Crustaceans prefer the depth of 30–40 m [25, 31].

Three groups of zooplankton, cladocerans (Cladocera), copepodes (Copepoda) and rotifers (Rotifera), can be found in the mountain lakes. In the Polish Tatra lakes, the presence of 36 species of cladocerans has been stated. The most common ones are *Chydorus sphaericus* (Müller 1776) – also in water bodies situated above 2000 m a.s.l.; *Eurycercus lamellatus* (Müller 1776); *Alona quadrangularis* (Müller 1776); *A. affinis* Leydig, 1860; *Holopedium gibberum* Zaddach 1855; *Polyphemus pediculus* (L. 1758); *Simocephalus vetulus* (Müller 1776); *Daphnia pulicaria* Forbes 1893; and *D. longispina* (Müller 1776). Individual species show clear distribution depending on the altitude (Fig. 5) [25, 31–33].

Live cycles of the mentioned species are closely correlated with harsh environmental conditions typical for mountain waters. Usually young organisms hatch after ice melting in May (except for lakes situated at the highest altitude, with long period of ice cover, where the development of young organisms is possible only in July). Reproduction usually occurs in the second part of July, August and September. Most species have a monocycle type of the live cycle, while zooplankton from lowland waters has clearly a polycyclic type [25].

The second group of crustacean zooplankton copepods (Copepoda) is represented by 36 species in Tatra lakes. The most common one is *Cyclops abyssorum* Sars 1893, which can also be found in fish ponds as the only one representative of this group. The rare species occurring only in fishless water bodies are *Heterocope saliens* (Lilljeborg 1962) and *Acanthodiaptomus denticornis* (Wierzejski 1887). In lakes situated at lower altitude in the forests or in peat bogs, *Acanthocyclops vernalis* (Fischer 1853) is the typical species. In small ponds, rich in detritus on the bottom, *Mixodiaptomus tatricus* (Wierzejski 1883) is mostly found. Some species noted in the Tatra waters belong to eurythermic forms, which are also known from lowland waters, e.g. *Acanthocyclops vernalis*, while others are stenothermic, e.g. *Bryocamptus zschokkei* (Schmeil 1883) or *B. alpestris* (Vogt 1845). For most of them, production of one generation per year is typical. Reproduction starts even before ice melting, during the warm season when organisms undergo particular developmental stages (nauplius, copepodit) and reach maturity during winter. The distribution of copepods as well as cladocerans depends on lake location: it is wider in the range of altitude 900–2,000 m a.s.l. or limited to the zone 1,500–2,000 m a.s.l [25, 34].

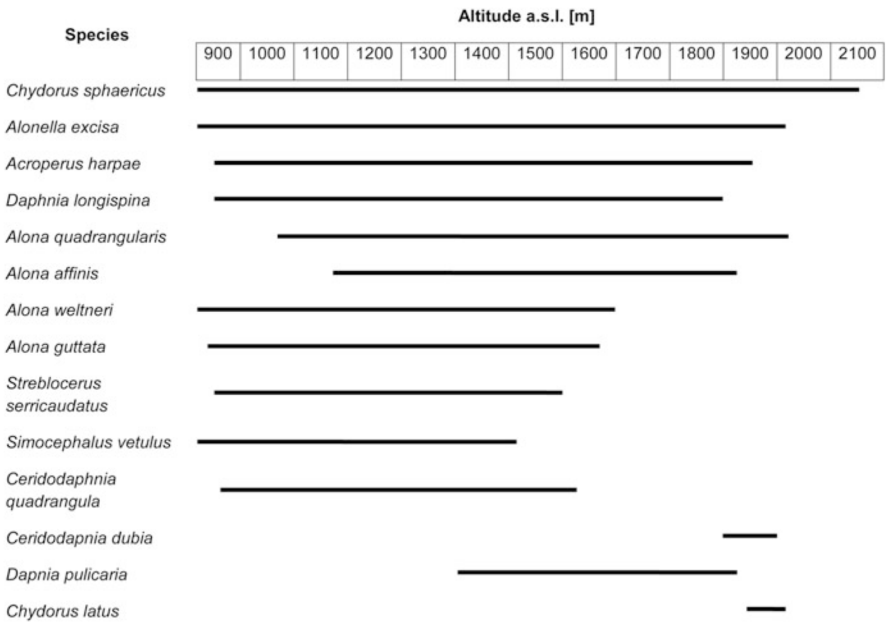

Fig. 5 Vertical distribution of Cladocera in the Tatra lakes (modified from Kownacki and Żurek [25])

In the Tatra waters, fauna of rotifers (Rotifera) is represented by 143 species, but only 20 of them are typical planktonic forms. Others inhabit mosses and psammon – coastal moist sand – or they are cosmopolites. In Tatras, 123 rotifer taxa have been found mostly in the biggest and very deep lakes [35]. In the lakes with the depth >10 m, 79 taxa have been stated while 76 in lakes with surface >10 ha. The typical rotifer species of the Tatra waters is *Polyarthra dolichoptera* Idelson 1925 – a stenotherm, living in cold, high-altitude lakes. The other typical species for Tatra water is *Keratella quadrata* (Müller 1786) – the organism which has wider tolerance to water temperature (eurytherm) and is found in various water bodies, very numerously in lakes lying in the forest zone [25]. The separate group is rotifers inhabiting psammon. In the Tatra lakes, this group forms 22–24 species, which also live in the pelagic zone, e.g. *Cephalodella megalocephala* (Glascott 1893) and *Colurella colurus* (Ehrenberg 1830) [25, 35, 36].

Hořická et al. [27] classified lakes on the basis of Cladocera and Copepoda communities studied in over 100 Tatra lakes (in Slovakia and Poland). *Acanthodiaptomus denticornis* and *Daphnia longispina* are typical and dominant species inhabiting lakes in the subalpine zone. *Arctodiaptomus alpinus* (Imhof 1885), *Cyclops abyssorum*, *Daphnia longispina*, *Daphnia pulicaria* and *Holopedium gibberum* are dominant species in the alpine zone. *Ceriodaphnia quadrangular* (Müller 1785), *Daphnia obtuse* Kurz 1875, *Daphnia pulex* Leydig 1860 and *Mixodiaptomus tatricus* are species dominating in acidified waters or with high amount of dissolved organic matter. The zooplankton of the Polish Tatra lakes is typical for non-acidified subalpine and alpine zone lakes.

5 Macroinvertebrates

The benthic macroinvertebrates of the Tatra lakes are represented by two main kinds. The high taxonomic diversity of the first one is connected with dystrophic lakes situated in the forest zone with the shores overgrown by higher plants (Fig. 6). The second one has low taxonomic diversity and is typical for lakes situated above the upper tree line, with no littoral vegetation (Fig. 7) [25, 26].

The taxonomic composition of macroinvertebrates in the Tatra water bodies is similar to that in the inland lakes – the same types of animal groups are represented there. Only the proportion of particular groups is different – in lowland lakes, species richness is much more higher, while in the Tatra lakes, they are represented by few species. In lakes lying in the forest zone such as Toporowy Niżni or Smreczyński Staw Lakes, biodiversity is higher than in alpine ones. Over 100 species of macroinvertebrates have been found in the Tatra water bodies up to now [14, 25, 37, 38]. More detailed information about particular groups of animals can be found in separate research papers, e.g. Oligochaeta [39], Odonata [40] or Chrionomidae [41].

The larvae of Diptera dominated among littoral benthic macroinvertebrates of 45 alpine and subalpine lakes in the Tatras and were recorded from all the sampling sites [14]. The second most frequent group was Oligochaeta (96% of sites),

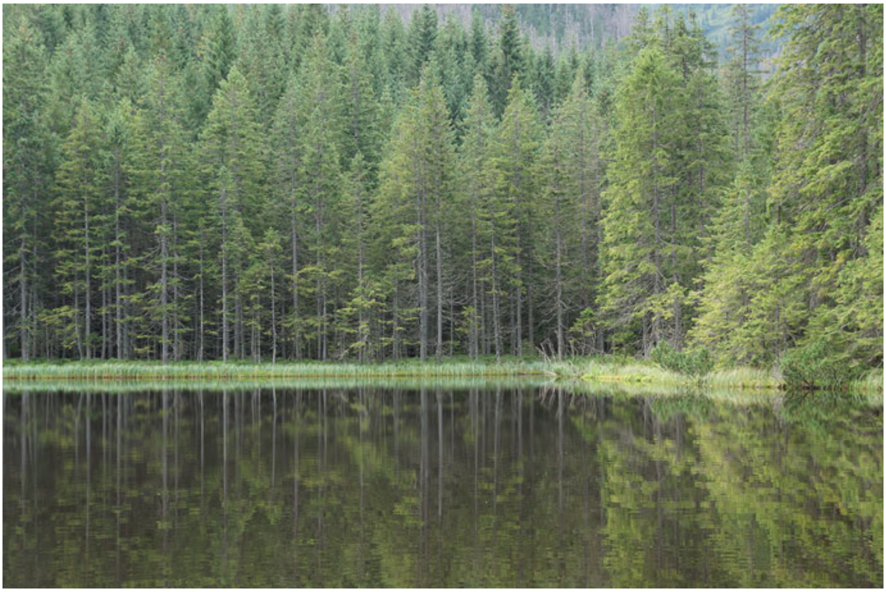

Fig. 6 Phytolittoral in the Smreczyński Staw Lake

Fig. 7 Stony littoral in Czarny Staw Gąsienicowy Lake

Plecoptera (73%), Trichoptera (69%) and Turbellaria (59%). Representatives of the remaining groups such as Hydrozoa, Mollusca and Heteroptera were found sporadically. The dominance of Chironomidae and Oligochaeta was also observed by other authors, e.g. Galas [42], Kownacka and Kownacki [43], Kownacki and Kownacka [44] and Kownacki et al. [45, 46].

The most common macroinvertebrate species recorded from the littoral of the Tatra lakes which partially undergo life cycles belong to Diptera from Chironomidae family: *Heterotrissocladius marcidus* (Walker 1856) (78% of the studied sites), *Micropsectra* spp. (73%) and *Corynoneura* spp. (62%) [14]. The Oligochaeta such as *Nais variabilis* Piguet 1906 (69%), *Cernosvitoviella atrata* Bretscher 1903 (56%), *Stylodrilus heringianus* Claparéde 1862 (53%) and flatworm *Crenobia alpina* (Dana 1766) (56%) were the most often found species undergoing the whole life cycles in the Tatra waters. In the Tatra lakes, among insects aside from Diptera, stoneflies *Capnia vidua* Klapálek 1904 (42%) and *Nemurella pictetii* Klapálek 1900 (33%), mayflies *Ameletus inopinatus* Eaton 1887 (31%), caddisflies from genus *Allogamus* Schmid 1955 (31%) and beetle *Agabus bipustulatus* (Linnaeus 1767) (29%) were noted.

In each lake the variability of the benthic fauna communities is related to the lake character and its location as well as to its depth. The most abundant groups of macroinvertebrates in the littoral of the lakes located in the forest zone and in peatbogs (Toporowy Niżni Staw Lake, Wielka Pańszczycka Młaka, Rówień Waksmundzka) are oligochaetes, leaches, snails, water mites, dragonflies, true bugs, true flies, caddisflies and beetles. In the muddy bottoms, covered by detritus, the benthic fauna is abundant but with less species, where nematodes, oligochaeta and a chironomid *Psectrocladius* Kieffer 1906 dominate. Small freshwater clams *Pisidium casertanum* (Poli 1791) are very often found there [25, 47]. There is no vegetation in the lakes situated above the forest line; their banks are steep and stony. The benthic macroinvertebrates near the shores are represented by *Hydra* Linnaeus 1758, flatworms, oligochaeta, *Ancylus fluviatilis* Müller 1774 (Gastropoda), *Pisidium* Pfeiffer 1821 (Bivalvia), alderfly from genus *Sialis* Latreille 1802, caddisflies and Chironomidae species such as *Chironomus* Meigen 1803, *Heterotrissocladius marcidus*, *Psectrocladius*, *Tanytarsus* van der Wulp 1874, *Paratanytarsus austriacus* (Kieffer, 1924), *Dicrotendipes* Kieffer 1913, *Anatopynia* Johannsen 1905 and *Micropsectra* Kieffer 1909. At the depth of 10–25 m, the benthos composition is similar to the one near shores; the only difference is the presence of a Chironomidae *Procladius* Skuse 1889. The pronounced changes in invertebrates' communities are observed at the depth 20–80 m, where the bottom is covered by mud. Only Nematoda, Oligochaeta and *Psectrocladius* (Diptera) can be found there. Fauna composition of the Polish Tatra lakes, Morskie Oko Lake and Wielki Staw Lake, in the Five Polish Lake Valley is best documented [25, 44] (Fig. 8).

Studies on Tatra lakes fauna allowed to determine not only its taxonomic composition or vertical distribution but also to establish which environmental variables have an important influence on macroinvertebrate communities. Studies carried out by Krno [14] show that across the whole Tatras, the specificity of zoobenthos to the highest degree depends on factors connected with lake altitude, i.e. water temperature and ice cover duration. The other equally important

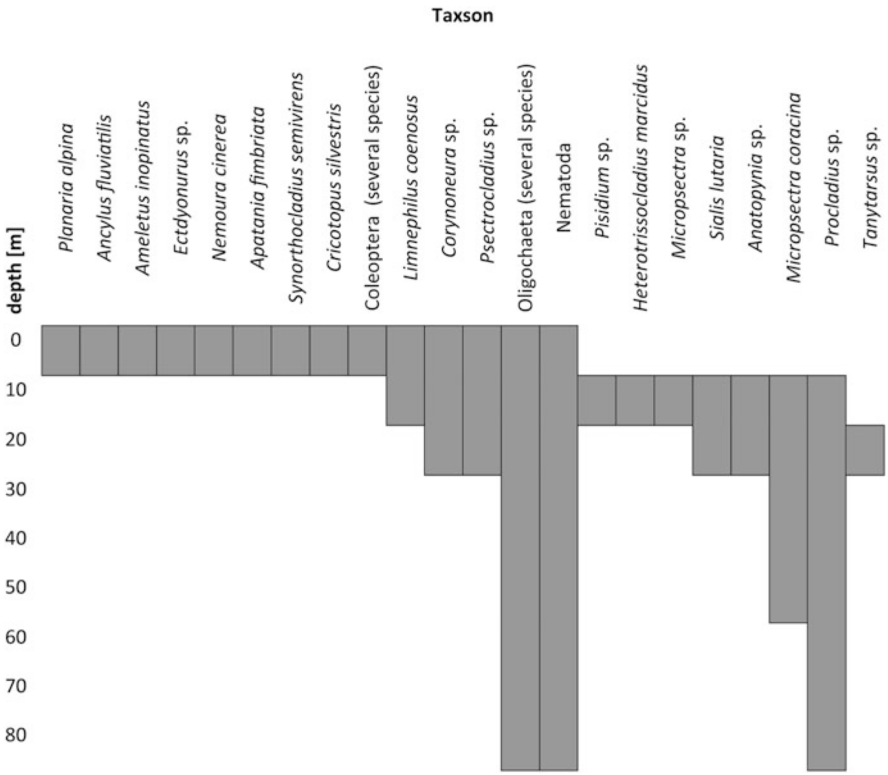

Fig. 8 Vertical distribution of zoobenthos in the Wielki Staw Lake (modified from Kownacki and Żurek [25])

parameters are connected with water chemistry and its trophy: pH, Ca, TP and TDOC contents. The size of a lake and the presence of an outlet and fish are less important factors influencing zoobenthos. Such a setup of factors deciding on the composition of zoobenthos has been presented also in previous papers [48].

The lakes acidification strongly influenced both the zooplankton and macroinvertebrate communities [7, 14, 49]. The influence of this process on zoobenthos structure has not yet been fully understood. The reaction was slightly different in the case of phyto- and zooplankton. It is hard to distinguish anthropogenic changes from natural ones ongoing in lakes. Hydrobiological studies made by Kownacki et al. [8] on a few small Tatra ponds and lakes situated at the altitude of 1,600–1,850 m a.s.l. confirmed that suggestion. The results of the above mentioned studies did not reveal if changes observed in zoobenthic community structure were the effect of air pollution or if they were connected with natural catchment character.

The Tatra lakes classification including objects located both in Slovakia and Poland presented by Krno et al. [14] seems the best describing the character of mountain Tatra lakes. Five types of lakes (A–E) were distinguished (Table 2).

Table 2 Types of the Tatra lakes [14]

Type	Characteristic	Indicative taxa
Type A. Strongly acidified lakes	Small lake basin, low pH, high concentration of total phosphorus and highest % of organic matter of littoral substrate; high proportion of Diptera in macroinvertebrate assemblages	*Sialis lutaria*, *Limnephilus coenosus*, *Zalutschia tatrica*
Type B. Alpine acidified lakes	The smallest proportion of particulate organic matter in the littoral and low values of some biotic metrics: number of genera, diversity index and % EPT taxa	*Leuctra rosinae*, *Allogamus* spp., *Pseudodiamesa nivosa*
Type C. Subalpine acidified lakes	Low diversity index and a dominating share of Diptera in the assemblages	*Cernosvitoviella tatrica*, *Tubifex ignotus*, *Nemurella pictetii*
Type D. Alpine non-acidified lakes	High values of some biotic metrics: number of families, proportion of crenal and metarhithral taxa/littoral taxa	*Haplotaxis gordioides*, *Arcynopteryx compacta*, *Leuctra rosinae*, *Allogamus* spp., *Pseudodiamesa nivosa*
Type E. Subalpine non-acidified lakes	High values of some biotic metrics: families, number of genera, BMWP, EPT taxa and % ETP taxa	*Diura bicaudata*, *Pedicia rivosa*, *Eukiefferiella* spp., *Micropsectra* spp., *Zavrelimyia* spp.

6 Fish

The lakes in the Tatra Mountains are mostly fishless, except for a few lakes in High Tatra Mts. artificially stocked with the non-native brook trout (*Salvelinus fontinalis* (Mitchell 1814)) (Fig. 9) and brown trout (*Salmo trutta* m. *fario* L. 1758). The only lake with natural population of the brown trout is Morskie Oko Lake. Some lakes remained fishless until 1950–1960 and some are fishless even now. The introduction of salmonids to the Tatra lakes caused a drastic change in their crustacean zooplankton communities, which has been well documented by comparing plankton samples collected since 1881 [30].

The fry stockings of Tatra waters which started in 1881 were without success; most species did not acclimatize and became extinct [50]. The next several attempts of fish stocking were undertaken in the middle of the twentieth century, this time only with fry of brook trout which survived and created stable but isolated populations [51]. They are usually able to reproduce and many of them have survived till now. The alien species *Salvelinus fontinalis* originating from the North America now occurs in the Lakes Zielony, Czarny and Litworowy, all situated in the Gąsienicowa Valley, as well as in Lakes Przedni and Czarny in the Five Polish Lake Valley, where that fish find excellent conditions [52]. Those lakes are the only one habitat in Poland with stable populations of *Salvelinus fontinalis*. The result of the introduction of a brook trout into Tatra lakes is the expansion of brown trout range, which originally and naturally occurred in Morskie Oko Lake but now is noted also in the lakes Zielony Gąsienicowy and Przedni in the Five Polish Lake

Fig. 9 *Salvelinus fontinalis* from the Zielony Staw Lake

Valley. Fish in the Tatra National Park are currently under strict protection; stocking is not allowed, and no fishing grounds are available.

7 Concluding Remarks

Tatra lakes are of glacial origin. Those located at lower elevations have catchment areas in the forest, whereas those located above the tree line have an alpine character. The Tatra alpine and subalpine lakes are characterized by the following features:

– Waters of the deepest ones located at 1,500–1,800 a.s.l. have maximal temperature 8–14°C, and they have an ice cover duration for 4–7 months.
– The chemical parameters of the Tatra lakes waters are determined mainly by the atmospheric loads, the majority of them is of oligotrophic character.
– From the ecological point of view, Tatra lakes are characterized by low productivity and simple ecological structure, and the main factors determining these features are extreme climatic features and temperature.
– Biological components (epilithic algae, zooplankton and benthos) show clear differences depending on lake location, catchment area character and temperature. Differences connected with lake depth are also evident.
– According to the classification proposed by Krno et al. [14], Tatra lakes located in Poland can be ascribed to two types: non-acidified subalpine lakes and non-acidified alpine lakes.

References

1. Staszic S (1805) O Ziemiorództwie Karpatów i innych gór i równin Polski, vol 1955. Wydawnictwa Geologiczne, Warszawa. (reprint)
2. Kownacki A, Ślusarczyk JM (2009) Badania faunistyczne jezior tatrzańskich do roku 1914. Kwart Hist Nauki Tech 54(1):95–113
3. Kownacki A, Ślusarczyk JM (2015) Badania faunistyczne polskiej części Tatr w latach 1919–1939. Kwart Hist Nauki Tech 60(3):67–77
4. Mirek Z (ed) (1996) Przyroda Tatrzańskiego Parku Narodowego. TPN, Kraków-Zakopane, 786 pp
5. Szaflarski J (1932) Z badań nad termiką jezior tatrzańskich. Przegląd Geograficzny 12:181–194
6. Barczyk G (2008) Vauclusian springs of the Tatra Mountains. Karst systems of vauclusian springs in the Polish Tatra Mountains. TPN, Zakopane
7. Kopáček J, Stuchlik E, Hardekopf D (2006) Chemical composition of the Tatra Mountain Lakes: recovery from acidification. Biologia Bratislava 61(Suppl 18):21–33
8. Kownacki A, Dumnicka E, Kwandrans J, Galas J, Ollik M (2006) Benthic communities in relation to environmental factors in small high mountain ponds threatened by air pollutants. Bor Environ Res 11:481–492
9. Stuchlik E, Kopáček J, Fott J, Hořicka Z (2006) Chemical composition of the Tatra lakes: response to acidification. Biologia Bratislava 61(Suppl 18):11–20
10. Kopáček J, Prochazkova L, Stuchlik E, Blazka P (1995) The nitrogen-phosphorus relationship in mountain lakes: influence of atmospheric input, watershed, and pH. Limnol Oceanogr 40:930–937
11. Wojtan K, Galas J (1994) Acidification state of small mountain lakes in the High Tatra mountains, Poland. Hydrobiologia 174:179–182
12. Rzychoń D, Worsztynowicz A (1995) The role of nitrogen in acidification of Tatra Mountains lakes. Water Air Soil Pollut 85:481–486
13. Kopáček J, Vesely J, Stuchlik E (2001) Estimation on sulphur and nitrogen fluxes in the Bohemian Forest and Tatra Mountains during industrial revolution (1980 to 2000). Hydrol Earth Syst Sci 5:391–405
14. Krno I, Šporka F, Galas J, Hamerlík L, Zatovičová Z, Bitušik P (2006) Littoral benthic macroinvertebrates of mountain lakes in the Tatra Mountains (Slovakia, Poland). Biologia Bratislava 61(Suppl 18):147–166
15. Poniewozik M, Wołowski K (2017) Euglenoids (Euglenophyta) in water-peatbog habitats of the Tatra Mts. Frag Flor Geobot et Pol 24:283–294
16. Lenarczyk J (2017) The history and perspectives of green algae studies (Chlorophyta) in mountains lakes of the Tatra National Park (Western Carpathians). Frag Flor et Geob Pol 24:339–351
17. Piątek J (2017) Studies of golden algae (Chrysophyceae) and their spores (stomatocysts) in Tatra National Park. Frag Flor et Geob Pol 24:295–302
18. Piątek J (2006) Stomatocysts of the Dolina Gąsienicowa Valley in the Tatra Mts (Poland). 1. Czarny Staw Gąsienicowy and Zmarzły Staw Gąsienicowy lakes. Pol Bot J 51:61–77
19. Kawecka B, Galas J (2003) Diversity of epilithic diatoms in high mountain lakes under the stress of acidification (Tatra Mts, Poland). Ann Limnol Int J Limnol 39:239–253
20. Kawecka B (2012) Diatom diversity in streams of the Tatra National Park (Poland) as indicator of environmental conditions. Institute of Botany Polish Academy of Sciences, Kraków
21. Battarbee RW, Charles DF, Dixit SS, Renberg I (1999) Diatoms as indictors of surface water acidity. In: Stoermer EF, Smol JP (eds) The diatoms: applications for the environmental and earth sciences. Cambridge University Press, Cambridge, pp 85–127
22. Kawecka B (2017) Diatom (Bacillariophyceae) communities of springs, standing and flowing waters in the Polish part of the Tatra Mts. Frag Flor et Geobot Pol 24:303–328
23. Kawecka B (1996) Algae. In: Mirek Z (ed) Nature of the Tatra National Park. TPN, Kraków, pp 347–362

24. Hrbáček J, Brtek J, Vranovský M, Štěrba O (1974) Zooplanktón a význační zástupcovia niektórych skupin drobného vodného živočíšstva tatranských plies. Zborník TANAP 16:105–109
25. Kownacki A, Żurek R (1996) Fauna jezior. In: Mirek Z (ed) Nature of the Tatra National Park. TPN, Kraków, pp 535–553
26. Sowa R (1965) Invertebrate water animals of the Polish Tatra. In: Starmach K (ed) Tatra Mountains. Polish Academy of Sciences, Kraków, pp 41–49
27. Hořická Z, Stuchlík E, Hudec I, Černy M, Fott J (2006) Acidification and the structure of crustacean zooplankton in mountain lakes: the Tatra Mountains (Slovakia, Poland). Biologia Bratislava 61(Suppl 18):121–134
28. Fott J, Pražakova M, Stuchlik E, Stuchlikowá Z (1994) Acidification of lakes in Šumava (Bohemia) and in the High Tatra Moutains (Slovakia). Hydrobiologia 274:37–47
29. Gliwicz ZM (1963) Wpływ zarybienia na biocenozy jezior tatrzańskich. Chrońmy Przyrodę Ojczystą 5:27–35
30. Gliwicz ZM (1967) Zooplankton and temperature-oxygen conditions of two alpine lakes of the Tatra Mountains. Pol Arch Hydrobiol 14:53372
31. Woźniczka K (1965) The zooplankton of the valley of the five Polish Lakes in the Tatra Mountains. Kom Zagosp Ziem Górskich PAN 11:19–31
32. Woźniczka-Starzykowa K (1966a) Zooplankton strefy litoralnej Przedniego Stawu w Dolinie Pięciu Stawów Polskich w Tatrach. Acta Hydrobiol 6:329–334
33. Woźniczka-Starzykowa K (1966b) Zooplankton drobnych stawków w Dolinie Pięciu Stawów Polskich w Tatrach. Acta Hydrobiol 8:335–339
34. Prószyńska M (1963) Cladocera and Copepoda of small water bodies of Tatra Mountains and Podhale region. Some remarks on typology pools. Pol Arch Hydrobiol 11:157–167
35. Bielańska-Grajner I (2010) Dotychczasowy stan badan wrotków (Rotifera) w wodach Tatrzańskiego Parku Narodowego. In: Mirek Z (ed) Nauka a zarządzanie obszarem Tatr i ich otoczeniem. Tom II. Nauki biologiczne. TPN Polskie Towarzystwo Przyjaciół Nauk o Ziemi – Oddział Krakowski, Zakopane, pp 87–94
36. Wiszniewski J (1936) Notatki o psammonie. IV–V. Rotiferes psammiques de quelques lacs de Tatras. Arch Hydrobiol Ryb 10:235–243
37. Minkiewicz S (1914) Przegląd fauny jezior tatrzańskich. Spraw Kom Fizyogr AU 48:114–137
38. Novikmec M, Veselská M, Bitušík P, Hamerlík L, Matúšova Z, Reduciendo Klementová B, Svitok M (2015) Checklist of benthic macroinvertebrates of high altitude ponds of the Tatra Mountains (Central Europe) with new records of two species for Slovakia. Check List 11:1–12. https://doi.org/10.15560/11.1.1522
39. Dumnicka E, Galas J (2010) Stan wiedzy o skąposzczetach wodnych (Oligochaeta) Tatrzańskiego Parku Narodowego. In: Mirek Z (ed) Nauka a zarządzanie obszarem Tatr i ich otoczeniem. Tom II. Nauki biologiczne. TPN Polskie Towarzystwo Przyjaciół Nauk o Ziemi – Oddział Krakowski, Zakopane, pp 93–100
40. Tończyk G (2010) Ważki (Odonata) Tatr – historia i teraźniejszość. In: Mirek Z (ed) Nauka a zarządzanie obszarem Tatr i ich otoczeniem. Tom II Nauki biologiczne. TPN Polskie Towarzystwo Przyjaciół Nauk o Ziemi – Oddział, Krakowski, pp 101–105
41. Kownacki A (2010) Chironomidae (Diptera, Insecta) Tatrzańskiego Parku Narodowego rozmieszczenie, ekologia, zoogeografia. In: Mirek Z (ed) Nauka a zarządzanie obszarem Tatr i ich otoczeniem. Tom II. Nauki biologiczne. TPN Polskie Towarzystwo Przyjaciół Nauk o Ziemi – Oddział, Krakowski, pp 113–118
42. Galas J (2004) Invertebrate communities of high mountain lakes (Polish Tatra Mts.). Teka Kom Ochr Środ Przyr 1:57–63
43. Kownacka M, Kownacki A (1965) Fresh water invertebrates of Stawki Michowe pools in the Tatra Mountains. Kom Zagosp Ziem Górskich PAN 11:81–90
44. Kownacki A, Kownacka M (1965) The bottom fauna of the lakes Morskie Oko and Wielki Staw in the Polish Tatra Mountains. Kom Zagosp Ziem Górskich PAN 11:33–38

45. Kownacki A, Dumnicka E, Galas J, Kawecka B, Wojtan K (1997) Ecological characteristics of a high mountain lake-outlet stream (Tatra Mts., Poland). Arch Hydrobiol 139:113–128
46. Kownacki A, Galas J, Dumnicka E, Mielewczyk S (2000) Invertebrate communities in permanent and temporary high mountain lakes (Tatra Mts.). Ann Limnol 36:181–188
47. Kasprzak K, Šporka F (1991) *Pisidium casertanum* (POLI) (Sphaeriidae, Bivalvia) v tatranských jazerách. Zborník TANAP 31:183–190
48. Krno I (1991) Macrozoobenthos of the Tatra lakes littoral (the High Tatras) and its affection by acidification. Biologia Bratislava 45:495–506
49. Galas J (2010) Wpływ czynników środowiskowych w wysokogórskich jeziorach tatrzańskich na różnorodność ich fauny dennej. In: Joniak T (ed) Funkcjonowanie i ochrona ekosystemów wodnych T3. Uniwersytet A Mickiewicza, Poznań, pp 30–36
50. Witkowski A (1996) Fishes. In: Mirek Z (ed) Nature of the Tatra National Park. TPN, Kraków, pp 485–492
51. Dawidowicz P, Gliwicz ZM (1983) Food of brook charr in extreme oligotrophic conditions of an alpine lake. Environ Biol Fish 8:55–60
52. Kot M, Witkowski A (2017) Fishing in the Tatra Mountains and adjacent regions. Chrońmy Przyrodę Ojczystą 73:464–473

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