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Comprehensive approach to restoring urban recreational reservoirs. Part 2 – Use of zooplankton as indicators for the ecological quality assessment

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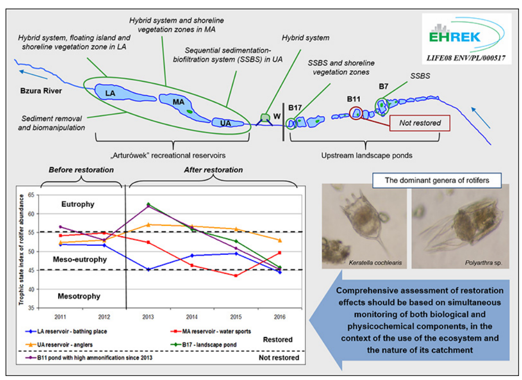
HIGHLIGHTS

- Zooplankton can be a useful indicator of ecological conditions in restored ecosystems.
- Rotifer trophic state index was positively correlated with trophic state index based on chlorophyll *a*.
- The biological indices should be used along with physicochemical monitoring when assessing water quality.
- Rotifer trophic state index was not reliable in ponds with high concentrations of ammonium.

GRAPHICAL ABSTRACT



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ABSTRACT

The presented research is part of the LIFE project ("EH-REK" LIFE08 ENV/PL/000517) on innovative restoration methods for small urban impoundments in the city of Łódź (Poland). The objective was to evaluate the usefulness of zooplankton as a biological quality element, when assessing the effectiveness of restoration efforts in three urban reservoirs and a one pond. The fifth unrestored pond was used as an example of the progressive eutrophication of an urban ecosystem. Studies were conducted during two periods: before (2010—2012) and after (2013–2016) restoration. A selection of zooplankton indices, including the rotifer trophic state index (TSI_{ROT}), was used. The influence of the supplying river resulted in the negligible responses of biological parameters to the restoration efforts in the Upper Arturówek (UA) reservoir, which is the first in cascade of reservoirs. However, clear symptoms of water quality improvements were observed in the other two reservoirs (the Middle Arturówek, MA; the Lower Arturówek, LA) and in the Bzura-17 (B17) pond. After restoration, the contribution of species indicative of high trophic levels decreased in these ecosystems. The TSI_{ROT} was strongly positively correlated with the trophic state index based on chlorophyll *a*, and both parameters significantly decreased in the MA, LA and B17. In the unrestored pond (B11), the successive increase in the concentrations of chemical parameters indicated progressing eutrophication. Interestingly, since 2013, the TSI_{ROT} values clearly decreased in B11, but the strong negative correlation between ammonium concentration and rotifer density indicated that the reduced TSI_{ROT} values didn't result from improvements in water quality; rather, they resulted from the increases in pollution and the associated harmful impacts on Rotifera. In conclusion, the TSI_{ROT} can be a useful tool for

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assessing the ecological quality of small urban ecosystems; however, the use of biological indices must be supported by also monitoring physicochemical parameters.

1. Introduction

In all ecosystems, the transfer of energy and nutrients among the trophic levels of food webs represents a key process. In aquatic systems, disruptions in the transfer of energy from primary producers to consumers may result in reductions in animal growth rates, increases in nuisance phytoplankton blooms and dissipation of energy from the system (McCauley and Kalff, 1981; Hairston and Hairston, 1993; Persson et al., 2007). Due to faster carbon recycling rates through the basal and first-order levels of the food chain, aquatic ecosystems appear to have a lower capacity to retain carbon during periods of increasing photosynthetic fixation than do terrestrial ecosystems (Cebrian, 2004). As a result, herbivorous and detritivorous zooplankton play a critical role in relocating carbon, energy and nutrients to the top of the trophic pyramid in water bodies (Cebrian and Lartigue, 2004). One example of this process is the transfer of essential fatty acids by herbivorous zooplankton through the food chain from phytoplankton and bacteria to organisms in higher trophic levels that cannot synthesise these fatty acids from precursors (Arts et al., 2009; Lampert, 2011).

Furthermore, the species structure of zooplankton as well as their biomass and density dynamics provide information about the relative importance of top-down and bottom-up controls and their impacts on water quality (Wojtal et al., 2008; Jeppesen et al., 2011). This is because the zooplankton community structure is primarily determined by the physical and chemical environment and is modified by biological interactions (Ejsmont-Karabin and Karabin, 2013). As zooplankton species composition, abundance and biomass change along a eutrophication gradient, the use of zooplankton as an indicator of the changes in trophic states and the ecological conditions of ecosystems caused by nutrient dynamics cannot be overstated (e.g., Gulati, 1984; Jeppesen et al., 2005). Small-sized rotifer species are particularly useful and important indicators of “bottom-up” processes because they are only slightly affected by fish predation, and their abundance is largely related with food type and quantity, which in turn, vary with changes in nutrient levels along the trophic gradient (Duggan et al., 2001; Jack and Thorp, 2002). However, this does not mean that rotifers are not dependent on top-down control, e.g., they are often exposed to the impact of competing Cladocera (Gilbert, 1988) and invertebrate predators (Lair, 1990). However, the sensitivity of rotifers to changes in the trophic states of lakes has been well documented by many researchers (e.g., Yoshida et al., 2003; Haberman and Haldna, 2014; Ochocka and Pasztaleniec, 2016; Yin et al., 2017).

Among the indices of the trophic states of lakes, only one index uses rotifer abundance and species composition to assess the degree of eutrophication of ecosystems. The rotifer trophic state index (TSI_{ROT}) was developed and tested in 74 Polish lakes (41 dimictic and 33 polymictic) of different trophy to assess the practical application of rotifers as potential indicators of the trophic state of a lake (Ejsmont-Karabin, 2012). During TSI_{ROT} development, the data on rotifer density and taxonomy were used to estimate the relationship between the rotifer community structure and the indices of the trophic state of the lakes (TSI); specifically, the concentration of chlorophyll *a* (Chla) and the Secchi disc visibility (as the estimator of suspended particulate material in the water) were calculated according to Carlson (1977). All regression equations were converted into formulas, which enable the calculation of the trophic state index from the parameters characterising the rotifer communities. These extensive and long-term studies (1976–2005) confirmed that the formulas can be used to help researchers assess the trophic status of any lake (both deep, stratified lakes and shallow, polymictic lakes) in central and northern Europe using the relevant rotifer data. These formulas may also be useful for preparing similar indices for lakes in other parts of the world, but when there are differences in the taxonomic structure of rotifer communities, formulating the relationships separately for particular climatic zones should be considered (Ejsmont-Karabin, 2012).

Nevertheless, to date, the usefulness of the TSI_{ROT} formulas, which were originally proposed for natural lakes, have not been verified in urban reservoirs or ponds. These small and shallow ecosystems are often inhabited by large and diverse communities of Rotifera. Because urban impoundments, due to their location and size, are more exposed to anthropogenic pressure than are natural lakes (Céréghino et al., 2008), monitoring through the use of bioindicators that are sensitive to deteriorated trophic status, such as rotifers, is highly recommended (Lodi et al., 2011). The degradation of these ecosystems often results from their improper use and a lack of protective measures. Small urban ecosystems that are frequently used for recreational purposes are exposed to the rapid and fluctuating delivery of high concentrations of nutrients, industrial pollution and pesticides (algicides) (Faulkner, 2004; Waajen et al., 2014; Suski et al., 2018) as well as mismanagement, which adversely affects the biodiversity of aquatic organisms (Declerck et al., 2006). As the extent and intensity of urbanization increases, more comprehensive protection and approaches for the restoration of urban ecosystems are necessary; moreover, ideal management should result in the reduction of nutrient inflow from the catchment and improve water quality (Hassall, 2014).

Our research focused on five impoundments (i.e., three small reservoirs and two ponds) in the city of Łódź (in Central Poland), four of which were restored within the LIFE project “Ecohydrologic rehabilitation of recreational reservoirs “Arturówek” (Łódź) as a model approach to rehabilitation of urban reservoirs” (EH-REK) (LIFE08 ENV/PL/000517, 2008; Jurczak et al., 2012); additionally, the fifth site was used as an example of the progressive eutrophication of urban ecosystems, where no restoration efforts have occurred. The effects of restoration on the biological elements of water were tested in the four reconditioned ecosystems. Considering the high indicative ability of zooplankton, the objectives of this studies were to: 1) investigate the suitability of zooplankton as a biological quality element in assessing the effectiveness of restoration efforts in urban reservoirs and ponds; we hypothesised that changes in the abundance and species composition of the main zooplankton groups (i.e., Rotifera, Cladocera, Copepoda) will reflect changes in the amount of nutrients and the water quality resulting from restoration efforts; and 2) test the usefulness of the rotifer trophic state index, which was originally proposed for natural lakes, as a biological indicator of water quality in small urban impoundments.

2. Materials and methods

2.1. Sampling site

The study was conducted in five urban impoundments located along the Bzura River in the north-eastern part of the city of Łódź, Poland (Fig. 1); these impoundments included the Lower Arturówek reservoir (LA), the Middle Arturówek reservoir (MA), the Upper Arturówek reservoir (UA), the Bzura-17 pond (B17) and the Bzura-11 pond (B11) (Table 1). The first three reservoirs (LA, MA and UA) form a cascade and are placed in area called “Arturówek”, which is the largest recreational area in the northern part of Łódź. Each reservoir, however, serves specific major functions, e.g., the LA is used as a bathing place, the MA is used for water sports, and the UA is restricted to angling use. The Bzura-11 and the Bzura-17 are typical water bodies of the landscape; they are

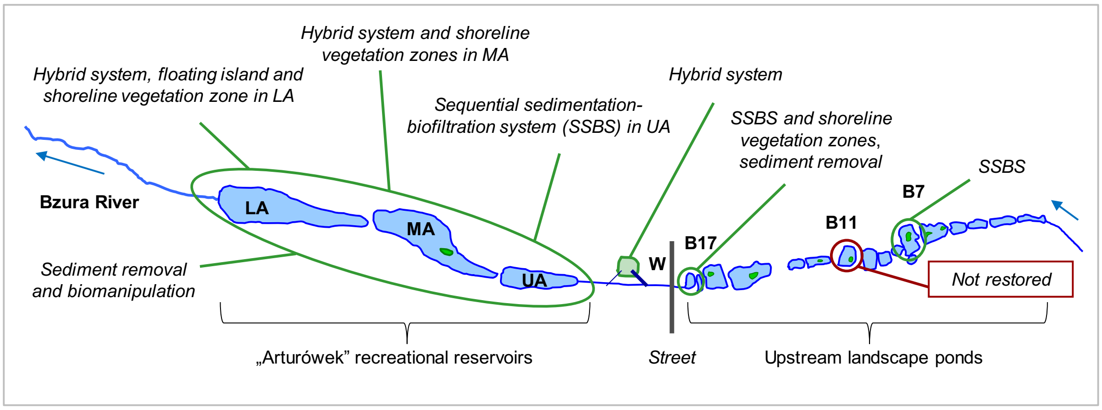

Fig. 1. Location of the studied ecosystems and range of the restoration actions implemented in 2013 in the Bzura ponds and Arturówek reservoirs.

located upstream of the Arturówek reservoirs and are diverted by other impoundments or river channels not covered in our current research. We used the term pond to refer to the impoundments that were built mainly by excavation as cut-and-fill ponds (Renfro, 1969; Coche et al., 1995); additionally, the length to width ratio of a pond is lower than 2. In contrast, the term reservoir was used to refer to impoundments that were created by constructing a barrier perpendicular to a flowing river and minor embankments, as a typical valley reservoir or barrage pond (Renfro, 1969; Coche et al., 1995), and had a length to width ratio >2. All five impoundments are shallow, have long water retention times, and are subjected to high concentrations of nutrients. These ecosystems were included in the restoration LIFE project due to the increase in eutrophication, which deteriorated the water quality.

Table 1
Average annual concentrations (±standard error) of chemical parameters and chlorophyll *a* (Chla) in the five studied impoundments (according to Jurczak et al., 2018a).

Parameter (±SE)	TN [mg L^{-1}]	TP [mg L^{-1}]	Nitrite [mg L^{-1}]	Nitrate [mg L^{-1}]	Ammonia [mg L^{-1}]	Phosphate [mg L^{-1}]	Total Chla [μg L^{-1}]
LA							
2010	1.15 (0.28)	0.26 (0.09)	0.006 (0.002)	0.27 (0.07)	0.28 (0.19)	0.03 (0.01)	8.78 (2.04)
2011	1.35 (0.20)	0.08 (0.01)	0.002 (0.001)	0.48 (0.09)	0.05 (0.01)	0.11 (0.03)	12.74 (3.23)
2012	5.07 (1.59)	0.14 (0.04)	0.130 (0.04)	0.47 (0.18)	0.10 (0.04)	0.06 (0.03)	32.95 (6.32)
2013	0.50 (0.21)	0.05 (0.03)	0.008 (0.004)	1.14 (0.27)	0.02 (0.01)	0.11 (0.08)	2.65 (0.69)
2014	0.67 (0.15)	0.28 (0.11)	0.024 (0.01)	0.15 (0.07)	0.01 (0.004)	0.17 (0.04)	4.42 (0.95)
2015	1.38 (0.47)	0.11 (0.01)	0.063 (0.03)	0.41 (0.25)	0.01 (0.005)	0.08 (0.04)	10.82 (4.41)
2016	1.39 (0.55)	0.04 (0.01)	0.023 (0.01)	0.36 (0.21)	0.02 (0.01)	0.27 (0.05)	11.03 (2.35)
MA							
2010	0.57 (0.12)	0.12 (0.03)	0.004 (0.003)	0.27 (0.20)	0.08 (0.03)	0.02 (0.004)	6.63 (1.14)
2011	0.92 (0.11)	0.10 (0.01)	0.001 (0.000)	0.05 (0.02)	0.07 (0.02)	0.10 (0.03)	11.16 (0.98)
2012	4.55 (0.99)	0.12 (0.03)	0.113 (0.03)	0.07 (0.06)	0.04 (0.02)	0.07 (0.03)	24.46 (2.64)
2013	0.37 (0.15)	0.07 (0.02)	0.001 (0.000)	0.05 (0.02)	0.01 (0.004)	0.07 (0.05)	3.67 (1.17)
2014	0.42 (0.07)	0.26 (0.10)	0.025 (0.01)	0.03 (0.01)	0.01 (0.004)	0.21 (0.05)	4.03 (2.18)
2015	1.56 (0.58)	0.08 (0.02)	0.058 (0.03)	0.09 (0.07)	0.01 (0.004)	0.09 (0.05)	1.83 (0.19)
2016	1.33 (0.38)	0.06 (0.04)	0.067 (0.04)	0.14 (0.06)	0.02 (0.01)	0.16 (0.03)	5.06 (1.23)
UA							
2010	0.70 (0.17)	0.17 (0.06)	0.005 (0.003)	0.19 (0.16)	0.04 (0.01)	0.01 (0.003)	6.60 (1.42)
2011	1.31 (0.12)	0.14 (0.02)	0.002 (0.001)	0.07 (0.03)	0.04 (0.01)	0.11 (0.04)	27.11 (5.89)
2012	5.07 (1.32)	0.54 (0.17)	0.102 (0.03)	0.12 (0.11)	0.23 (0.10)	0.07 (0.03)	91.21 (36.95)
2013	0.47 (0.19)	0.08 (0.03)	0.002 (0.001)	0.04 (0.03)	0.02 (0.01)	0.06 (0.04)	6.68 (1.97)
2014	0.50 (0.11)	0.29 (0.11)	0.025 (0.01)	0.03 (0.01)	0.01 (0.001)	0.12 (0.02)	6.77 (1.64)
2015	1.33 (0.40)	0.10 (0.02)	0.062 (0.04)	0.02 (0.01)	0.01 (0.004)	0.09 (0.04)	5.25 (1.49)
2016	1.34 (0.56)	0.02 (0.01)	0.048 (0.03)	0.13 (0.09)	0.02 (0.01)	0.21 (0.05)	5.38 (1.02)
B17							
2010	1.15 (0.20)	0.20 (0.02)	0.006 (0.001)	0.18 (0.07)	0.79 (0.16)	0.13 (0.03)	n.a.
2011	1.44 (0.22)	0.16 (0.01)	0.016 (0.003)	0.24 (0.04)	0.31 (0.08)	0.19 (0.05)	n.a.
2012	4.85 (1.31)	0.41 (0.07)	0.093 (0.03)	0.20 (0.04)	0.62 (0.06)	0.22 (0.08)	n.a.
2013	1.13 (0.58)	0.08 (0.03)	0.004 (0.003)	0.12 (0.11)	0.64 (0.59)	0.03 (0.02)	32.51 (4.24)
2014	1.64 (0.21)	0.62 (0.10)	0.022 (0.005)	0.23 (0.10)	0.43 (0.15)	0.31 (0.06)	29.55 (3.95)
2015	0.79 (0.25)	0.28 (0.07)	0.036 (0.02)	0.02 (0.01)	0.07 (0.06)	0.09 (0.03)	14.50 (5.46)
2016	3.74 (1.75)	0.22 (0.17)	0.016 (0.01)	0.34 (0.26)	0.05 (0.02)	0.42 (0.11)	6.15 (1.69)
B11							
2010	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
2011	1.48 (0.18)	0.13 (0.01)	0.001 (0.000)	0.03 (0.01)	0.13 (0.06)	0.16 (0.04)	25.18 (3.90)
2012	2.67 (1.19)	0.21 (0.04)	0.071 (0.02)	0.01 (0.01)	0.10 (0.05)	0.12 (0.04)	32.70 (7.55)
2013	2.83 (1.14)	0.23 (0.06)	0.001 (0.001)	0.02 (0.005)	0.35 (0.32)	0.21 (0.11)	111.7 (51.59)
2014	2.15 (0.32)	0.85 (0.11)	0.029 (0.01)	0.06 (0.02)	0.95 (0.32)	0.42 (0.05)	52.93 (8.60)
2015	2.45 (0.54)	0.43 (0.09)	0.036 (0.02)	0.03 (0.01)	1.46 (0.73)	0.53 (0.23)	24.75 (5.53)
2016	4.86 (1.15)	0.38 (0.12)	0.017 (0.01)	0.09 (0.04)	2.89 (0.94)	0.69 (0.14)	19.71 (11.44)

n.a. – not analysed.

Restoration activities were conducted from January to June 2013 (Fig. 1) and included 1) sediment removal (in all the restored reservoirs/ponds); 2) construction of sequential sedimentation-biofiltration systems (SSBS) in the UA reservoir and B17 pond. The SSBS consist of three zones: sedimentation, geochemical and biological. The prototype was tested and described by Szklarek et al. (2018) and was further optimised and developed in this project; 3) construction of hybrid systems in the LA and MA reservoirs. The hybrid system combines engineering and biotechnological measures, including an underground separation system and SSBS, and was described by Jurczak et al. (2018b); 4) construction of a floating island in the upper part of the LA reservoir; 5) establishing shoreline vegetation zones in the LA and MA reservoirs and the B17 pond; and 6) biomanipulation that was applied by introducing predatory fish in 2013–2014 to enhance zooplankton development in the reservoirs LA, MA and UA and to enhance top-down control of phytoplankton development.

Additionally, one hybrid system that purified storm water collected from Wycieczkowa Street (W in the Graphical Abstract) and one SSBS in Bzura-7 (B7 in the Graphical Abstract) were constructed within the framework of the project; however, these results are not presented in this paper.

The B11 non-restored pond, was used as an example of the progressive eutrophication of urban ecosystems. Furthermore, B11 was a reference site for the B17 pond because of its similar size, morphology, type of catchment and usage. It could not serve as a control pond for the Arturówek reservoirs due to their large size and different characteristics.

Detailed descriptions of the applied restoration activities and the physicochemical aspects of the waters of these impoundments are presented by Jurczak et al. (2018a).

Between April and October of 2010–2016, water samples from each ecosystem were collected monthly in 2010, 2011, and 2014 and every two weeks in 2012, 2015 and 2016; these water samples were used for Chla concentration and physicochemical analyses. Zooplankton samples were collected monthly (except in 2010, when they were collected every two weeks) from April to October. All samples were collected in the central part of the respective ecosystems, which was the section considered to be the most representative; additionally, samples were collected approximately 8–10 m from the shore in the reservoirs and 5 m from the shore in the ponds (Fig. 1). The specific timing of sample collection for each water body was as follows:

a) in the two periods before restoration (i.e., 2010–2012) and after restoration (i.e., 2013–2016) at the Arturówek reservoirs (i.e., LA, MA, UA);
b) only after restoration (i.e., 2013–2016) at the B17 pond;
c) in the period from 2011 to 2016 at the B11 pond.

In 2013, the restoration activities were completed in June; thus, 2013 had only three series of samples that were collected from July to October.

Fish sampling was limited to the three reservoirs in Arturówek; specifically, only the Arturówek impoundments were deep and large enough to set gillnets and were subjected to fishery impacts, i.e., fish stocking and angling. Between 2010 and 2016, fish assemblages were studied once per year in October, with the exception of 2013, when fish sampling occurred in July because the reservoirs were drained before autumn to begin the process of sediment removal.

2.2. Physicochemical analyses

Physical parameters, including water temperature, oxygen concentration, pH and conductivity, were determined in situ during water sampling using the WTW Multi 340i (WTW, Weilheim, Germany). Water samples were filtered by GF/C membranes and analysed by ion chromatography (Dionex ICS-1000, Sunnyvale, California, U.S.A); additionally, the quality and quantity of cations was analysed with an Ion Pac CS15 column (NH_4^+), and the analysis of anions was conducted with an Ion Pac AS14A column (NO_2^-, NO_3^-, and PO_4^{3-}). The analysis of the total nitrogen (TN) concentration was performed in unfiltered water using the persulfate digestion method (method no. 10071; HACH, 1997). The samples used for the total phosphorus (TP) analysis were digested with the addition of Oxisolve® Merck reagent (Merck, Darmstadt, Germany) with the MerckMV 500 Microwave Digestion System and were determined using the ascorbic acid method (Golterman et al., 1978).

2.3. Chlorophyll a analyses

The concentration of Chla was measured immediately after sampling using the bbe Algae Online Analyser (AOA, Version 1.5 E1, bbe-Moldaenke company Kiel, Germany). Details are presented in Wojtal-Frankiewicz et al. (2014).

2.4. Zooplankton analyses

Zooplankton were sampled from a 1-m column of water using a 10-litre bucket. The bucket is an acceptable method of catching zooplankton in shallow (i.e., <1 m depth) lakes, ponds and rivers (e.g., Sluss et al., 2011; Lopes et al., 2014). The depth at the sampling points of the studied ecosystems ranged from 0.8 to 1 m. Under such conditions, the tubular sampler could not be used. The bucket was gently submerged to the depth of the arm (i.e., approximately 50 cm) at an angle of approximately 45°, and samples were collected from the middle part of the water column to avoid the risk of moving the sediments. Sampling was always conducted by the same person in the same way; thus, possible sampling error could be omitted. The samples were collected during the daytime, from 9:00 to12:00 p.m.

To identify the zooplankton species and determine their density, 10 L of water was filtered using a 20-μm mesh net, and the samples were concentrated to 10 mL and preserved in 4% Lugol's solution. In the laboratory, 2 mL of 10% formaldehyde was added to each sample (Ciecierska and Dynowska, 2013). Zooplankton taxa were distinguished under a Nikon 115 microscope (magnification of 6100–200) using a Sedgewick-Rafter counting chamber. Morphological analyses of the collected individuals and their taxonomic identification were performed according to Amoros (1984), Ejsmont-Karabin et al. (2004) and Benzie (2005). Cladocera and Rotifera individuals were identified as far as possible to species level, but Copepoda individuals were assigned only to order. The biomass for a specific zooplankton species or genus (mg L^{-1}; wet weight (WW)) was estimated based on the species-specific length/weight regression curves and formulas elaborated by Bottrell et al. (1976). The biomass values of the individual species of rotifers, cladocerans and copepods were calculated separately.

The rotifer trophic state index (TSI_{ROT}) values were calculated using formulas for the relationship between the trophic state of lakes and the parameters of abundance and structure of rotifer communities estimated for both dimictic and polymictic lakes (Ejsmont-Karabin, 2012). The TSI_{ROT} was calculated from 2011 to 2016 due to the insufficient precision of the taxonomic analysis of Rotifera in 2010. The following formulas for the TSI_{ROT} index were used: (1) density of rotifers (N, ind L^{-1}): $TSI_{ROT1} = 5.38 \text{ Ln(N)} + 19.28$; (2) total biomass of rotifer community (B, mg L^{-1}): $TSI_{ROT2} = 5.63 \text{ Ln(B)} + 64.47$; (3) percentage of bacterivores in the total number of rotifers (BAC, %): $TSI_{ROT3} = 0.23 \text{ BAC} + 44.30$; (4) percentage of the *tecta* form in the population of *Keratella cochlearis* (TECTA, %): $TSI_{ROT4} = 0.187 \text{ TECTA} + 50.38$; (5) ratio of rotifer biomass to density (B: N, mg L^{-1}): $TSI_{ROT5} = 3.85 \text{ (B:N)}^{-0.318}$; and (6) contribution of species that indicate high trophic states in the indicatory group's numbers (IHT, %): $TSI_{ROT6} = 0.203 \text{ IHT} + 40.0$. Based on the recommendation of Ejsmont-Karabin (2012), the genus *Asplanchna* (*A. priodonta* in the case of our research) was excluded from these analyses because of its size and weight, which were much larger than those of the other species, and thus, could considerably change the value of the index and make it difficult to interpret. The rotifer trophic state index (TSI_{ROT}) was established as a mean of the above indices. It was assumed that the lakes with a TSI_{ROT} lower than 35 were oligotrophic, those with a value between 35 and 45 were mesotrophic, those with a TSI_{ROT} value of 45–55 were meso-eutrophic, those with a TSI_{ROT} value of 55–65 were eutrophic and those with a TSI_{ROT} value >65 were hypereutrophic (Ejsmont-Karabin, 2012).

In addition, the following relations were analysed: (1) the mean zooplankton weight (mg; WW), the mean cladoceran weight (mg; WW), the mean copepod weight (mg; WW) and the mean rotifer weight (mg; WW); (2) the ratio of crustacean density to rotifer density (N_{Crust}/N_{Rot}), which is an auxiliary indicator for the TSI_{ROT} that enables the assessment of the share of rotifers in the zooplankton community; and (3) the ratio of cyclopoid copepod biomass to the biomass of Cladocera (B_{Cyc}/B_{Clad}). The abovementioned parameters were selected based on the achievements of Ejsmont-Karabin and Karabin (2013) and Haberman and Haldna (2014), who studied the use of biological parameters as valuable tools for assessing the trophic state and water quality of aquatic ecosystems.

Moreover, Carlson's trophic state index (Carlson, 1977) was applied using two parameters:

a) Chla concentration (μg dm^{-3}), according the equation: $TSI_{CHL} = 9.81 \ln(\text{Chla}) + 30.6$;
b) total phosphorus concentration (μg dm^{-3}), according the equation: $TSI_{TP} = 14.42 \ln(\text{TP}) + 4.15$

The values of Carlson's index range from 0 to 100 and can be used to classify the trophic state of the lakes. The lakes with TSI < 30–40 were oligotrophic, the lakes with TSI = 40–50 were mesotrophic, the lakes with TSI = 50–70 were eutrophic, and the lakes with TSI = 70–100+ were hypereutrophic (Carlson and Simpson, 1996). The relationship between the TSI of Carlson's index and the TSI_{ROT} was calculated to estimate the suitability of the rotifer community structure for assessing the trophic state of urban ponds/reservoirs.

2.5. Gillnetting and stocking of fish

Gillnetting was performed for 12 h over night, starting 1 h before sunset, with gillnet panels set perpendicular to the shoreline. Nets were placed in the middle section of the reservoirs, with one set of nets located closer to the dam and the second set of nets located more upstream. The 30-m long and 1.5-m deep standardised multi-mesh gillnets were used (prEN 14757, 2005). Each gillnet was composed of 12 different mesh-sizes, ranging from 5 mm to 55 mm from knot to knot (the order of mesh panel sizes were as follows: 43, 19.5, 6.25, 10, 55, 8, 12.5, 24, 15.5, 5, 3, 5 and 29 mm). After sampling, data on the catch within each gillnet were recorded, including the number of individuals and the total length and weight of each specimen. Fish assemblages were characterised by grouping individual specimens into family (i.e., cyprinids and percids without pikeperch, *Sander lucioperca*) or genus (*Esox* sp. and *Sander* sp.). Cyprinids and percids were categorised into three size classes: small fish (<9.9 cm TL), medium-sized fish (10.0–19.9 cm TL) and large fish (>20.0 cm). For each fish group, the percentage ratio and catch per unit effort (CPUE) was calculated for both numerical abundance and biomass.

The capture of wild animals for the purpose of taking biometric measurements and determining their taxonomic classification is exempt from the requirements of the Ethics Committee approval in Poland (Art. 1 pt. 2.4 of the Act on the Protection of Animals Used for Scientific or Educational Purposes of 15 January 2015, which is the Polish implementation of Directive 2010/63/EU of the European Parliament and of the Council on the Protection of Animals Used for Scientific Purposes from 22 September 2010 (O.J.EUL276 of 20.10.2010, p.33)). Sampling via gillnetting was performed in accordance with the Act on Inland Fishery with the permission of the Lodz Voivodeship Marshal. After measurements, fish were donated to the Zoological Garden in Lodz. None of the activities in this study involved endangered or protected species.

Biomanipulation by stocking piscivorous fish began in the autumn of 2013. Pike (*Esox lucius*) and pikeperch (*Stizostedion lucioperca*) were stocked individually in UA and LA, respectively, while both species were stocked in MA. In the autumn of 2013 and 2014, 50 adult pike specimens (mean TL of 37.9 and 44.5 cm and mean weight of 328 and 552 g in 2013 and 2014, respectively) were stocked in UA. Beginning in 2014, 0+ pike and pikeperch were stocked every year in spring, with pike in UA (3000 specimens) and MA (1500 specimens) and pikeperch in LA (3000 specimens) and MA (1500 specimens).

Beginning in 2013, >150 specimens of tench (*Tinca tinca*) (mean TL: 21 cm, 127 g) and 350 specimens of crucian carp (*Carassius carassius*) (mean TL: 18.0 cm, 120 g) were stocked annually into all three reservoirs to maintain the attractiveness of the reservoirs to anglers. In addition to those late-spawning limnophillic cyprinids, rheophilic ide (*Leuciscus idus*) was stocked twice in 2013 and 2014 (mean TL: 23.1 cm, 135 g) in UA and once in 2014 in MA. The cyprinids used for stocking were of age 1+ and were released in autumn.

2.6. Statistical analyses

To determine whether there were significant differences between the biological parameters before (i.e., 2010–2012) and after (i.e., 2013–2016) the restoration efforts, a simultaneous adjustment for the imbalance in the number of collected samples was applied using the Monte Carlo permutation *t*-test. In cases where the assumptions of homogeneity of variances in compared groups were violated, the *t*-test designed to compare means with unequal variances was used. To determine the relationship between selected parameters, the Pearson correlation coefficient (r) was calculated.

Principal component analysis (PCA) was used to analyse the relationships between the physicochemical and biological parameters. PCA was conducted separately for each ecosystem. The ordination diagrams are presented as a projection of the variables (A) and the sampling terms (B) on the plane defined by the PCA axes. Only the first two components with the highest eigenvalues were plotted to analyse the relationships between variables in all ecosystems, as the inclusion of other factors did not provide any additional significant information necessary to interpret the ecological relevance of the data. To assess the significance of differences in the contribution of sampling terms to the PC axes, one-way ANOVA followed by Tukey HSD pairwise test for unequal N was applied; moreover, the factor coordinates of the sampling terms were used as the dependent variables, and the years were used as the categorical predictors.

Prior to analyses, all data were log $(x + 1)$-transformed for variance stabilisation.

The *t*-test and calculations of correlations were conducted using PAST v3.15 software (Hammer et al., 2001). PCA and ANOVA were performed in Statistica 12 (StatSoft).

3. Results

3.1. Comparison of chemical and biological parameters before and after restoration in three recreational reservoirs

3.1.1. Changes in the concentrations of chemical parameters

The concentrations of total phosphorus increased after restoration in both LA and MA. Before restoration, the average concentration of TP in LA was 0.16 mg L^{-1}, and this value increased to 0.18 mg L^{-1} after the restoration work. The average concentration of TP increased from 0.11 to 0.17 mg L^{-1} in MA. However, restoration efforts have contributed to the reduction in the total nitrogen concentrations in both ecosystems; for example, the total nitrogen concentration decreased from 1.97 to 1.03 mg L^{-1} in LA and from 1.46 to 0.93 mg L^{-1} in MA. In UA, the average concentrations of the total forms of nitrogen and phosphorus were reduced from 1.77 to 0.92 mg L^{-1} for TN and from 0.23 to 0.18 mg L^{-1} for TP as an effect of restoration.

After restoration, the average concentration of ammonium decreased in all reservoirs. The concentration changed from 0.15 mg L^{-1} to 0.01 mg L^{-1} in LA. The average ammonium concentration also declined from 0.070 mg L^{-1} to 0.016 mg L^{-1} after restoration in MA and declined from 0.076 mg L^{-1} in 2010–2012 to 0.01 mg L^{-1} after restoration in UA. In contrast, the average phosphate concentration changed from 0.06 mg L^{-1} in the period of 2010–2012 to 0.16 mg L^{-1} after restoration in LA, and the average phosphate concentration increased from 0.07 mg L^{-1} to 0.17 mg L^{-1} after restoration in MA. Similarly, in UA, the average concentration of phosphate before restoration was 0.06 mg L^{-1}, and this concentration increased to 0.13 mg L^{-1} in 2014–2016. The details on the physicochemical data and changes in Chla concentrations are presented in Table 1 and are described by Jurczak et al. (2018a).

3.1.2. Changes in zooplankton community structure

In all three reservoirs, the zooplankton species composition changed slightly after restoration. However, the share of individual species and groups differed with years (Fig. 2A, B, C). In all studied years, Copepoda was dominated by Cyclopoida and their nauplii. Before restoration, the dominant genera of Cladocera were *Diaphanosoma*, *Ceriodaphnia* and *Bosmina*. Furthermore, *Daphnia* and *Scapholeberis* were found in UA. After restoration, Cladocera were present in low numbers and were mainly composed of *Bosmina* sp. in LA, as well as *Ceriodaphnia* sp. and *Daphnia* sp. in both MA and UA. In 2016, cladoceran species were not found in the samples from UA. Before restoration, the most numerous rotifer species in all reservoirs were *Keratella* sp., *Polyarthra* sp., *Trichocerca* sp., *Brachionus* sp., and *Asplanchna* sp. Additionally, *Filinia* sp., *Lecane* sp. and *Lepadella* sp. were identified in LA. After restoration,

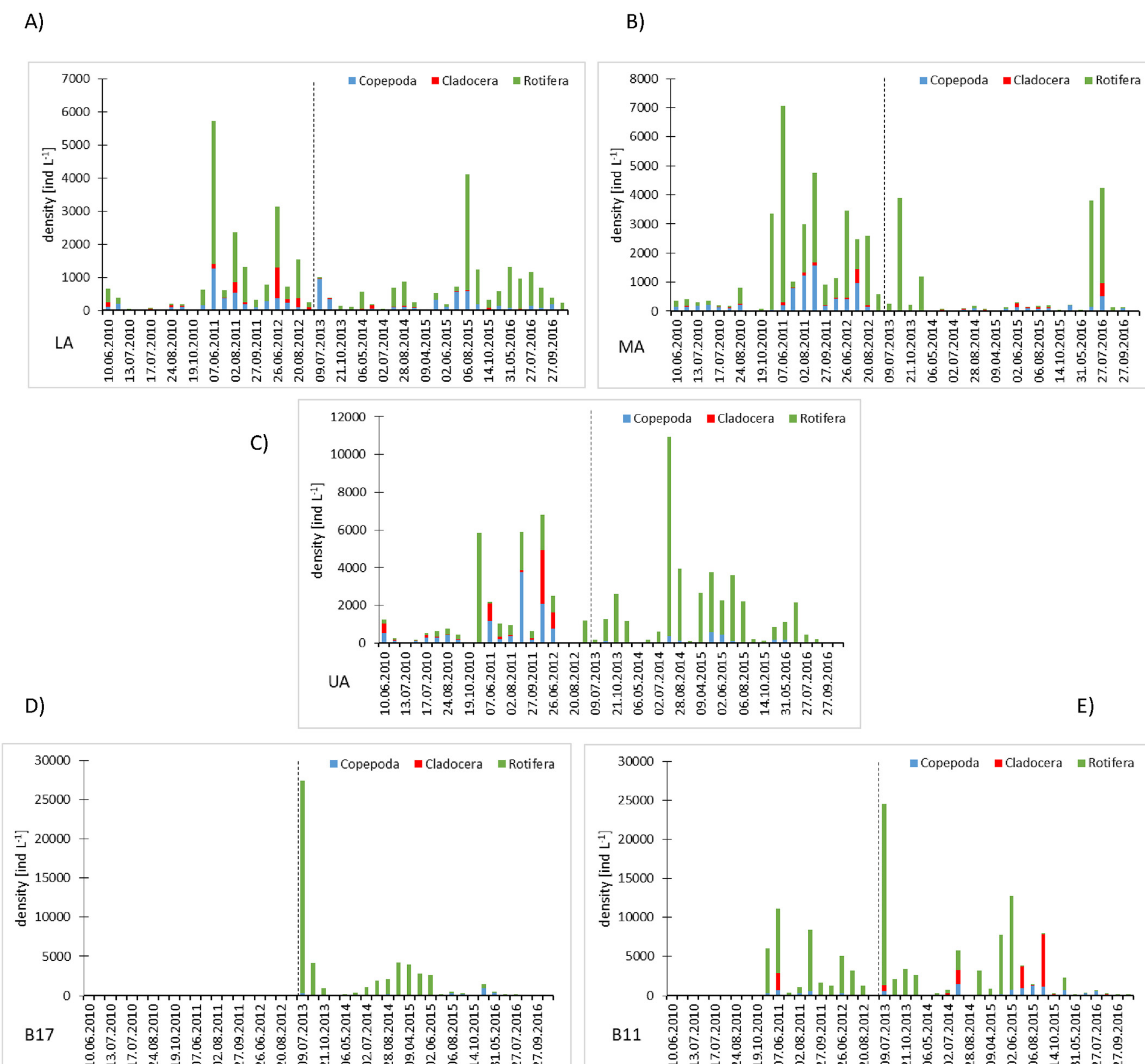

Fig. 2. The dynamics of zooplankton density groups (Copepoda, Cladocera, Rotifera) in: A) LA reservoir, B) MA reservoir, C) UA reservoir, D) B17 pond, and E) B11 pond in 2010–2016. The scales of y-axis for reservoirs and ponds are different. The dashed vertical line separates the periods before and after restoration.

Table 2
Characteristic of the five studied impoundments, with the list of the identified zooplankton species.

Parameters		LA	MA	UA	B17	B11
Area (ha)		3.05	2.58	1.08	0.1	0.21
Average depth (m)		1.33	1.35	0.93	~1.15	~0.98
Capacity (m^3)		40,600	34,900	10,000	1200	2100
Bzura river location (km)		164.242	164.706	165.142	165.700	166.125
GPS location of the sampling points	N	51°49′24.25″	51° 49′22.34″	51°49′16.98″	51°49′15.94″	51°49′18.60″
	E	019°28′12.53″	019°28′34.59″	019°28′50.90″	019°28′15.22″	019°28′38.66″
Zooplankton species before restoration (2010–2012)						
Rotifera						
Anuraeopsis fissa (Gosse, 1851)		+++	++	++	n.a.	+
Asplanchna priodonta (Gosse, 1850)		++	++	++	n.a.	+
Brachionus angularis (Gosse, 1851)		++	++	+	n.a.	+
Filinia longiseta (Ehrenberg, 1834)		+++	++	+	n.a.	++
Gastropus stylifer (Imhof, 1888)					n.a.	++
Keratella cochlearis (Gosse, 1851)		+++	+++	+++	n.a.	++
Keratella quadrata (Müller, 1786)		++	+++	++	n.a.	++
Polyarthra remata (Skorikov, 1896)		++	++	++	n.a.	+
Polyarthra vulgaris (Carlin, 1943)		+++	+++	+++	n.a.	+++
Lepadella ovalis (O.F. Muller, 1786)		+	+	+	n.a.	+
Trichocerca pusilla (Jennings, 1903)		+++	+++	++	n.a.	+
Trichocerca rousseleti (Voigt, 1902)				+	n.a.	
Colurella sp.					n.a.	+++
Lecane sp.		+	+	+	n.a.	+
Testudinella sp.					n.a.	++
Cladocera						
Bosmina longirostris (O.F. Müller, 1785)		++	+	++	n.a.	++
Chydorus sphaericus (O.F. Müller, 1785)		+		+	n.a.	+
Daphnia longispina (O.F. Müller, 1785)		+	+	+	n.a.	+
Daphnia galeata (Sars, 1863)		+	+	+	n.a.	
Diaphanosoma branchyurum (Lievin, 1848)		+	+	+	n.a.	+
Alona sp.		+	+	+	n.a.	+
Ceriodaphnia sp.		+++	++	++	n.a.	
Copepoda						
nauplius		+++	+++	+++	n.a.	+++
Cyclopoida		++	++	++	n.a.	++
Zooplankton species after restoration (2013–2016)						
Rotifera						
Anuraeopsis fissa (Gosse, 1851)		+	++	+++	++	+++
Asplanchna priodonta (Gosse, 1850)		+	+	+	+	++
Brachionus angularis (Gosse, 1851)		+	+	+	+	++
Filinia longiseta (Ehrenberg, 1834)		+	+	+	+	++
Keratella cochlearis (Gosse, 1851)		++	+	++	+++	++
Keratella quadrata (Müller, 1786)		++	+	+	+	++
Polyarthra longiremis (Carlin 1943)		+	+++	+		
Polyarthra remata (Skorikov, 1896)		+	++	+	+++	
Polyarthra vulgaris (Carlin, 1943)		+++	+++	+++	+++	+++
Trichocerca pusilla (Jennings, 1903)		+	+	+	+	+++
Trichocerca similis (Wierzejski, 1893)		++				
Lepadella ovalis (O.F. Muller, 1786)		+	+	+	+	+
Lecane sp.		+	+	+	+	+
Synchaeta sp.		++	+	++	++	+
Cladocera						
Bosmina longirostris (O.F. Müller, 1785)		+	++	+	+	+++
Chydorus sphaericus (O.F. Müller, 1785)			+		+	+
Daphnia longispina (O.F. Müller, 1785)		+	+	+	+	+
Polyphemus pediculus (Linnaeus, 1761)			+	+		
Alona sp.					+	
Ceriodaphnia sp.		+	+	+	+	
Copepoda						
Nauplius		+++	++	++	++	+++
Cyclopoida		+	+	+	+	++

\+ density from 1 to 20 [ind L^{-1}].
++ density from 21 to 200 [ind L^{-1}].
+++ density > 201 [ind L^{-1}].
n.a. – data not analysed.

the species composition of rotifers did not change much in the studied reservoirs; in addition to the abovementioned genera, *Synchaeta* sp. appeared (Table 2).

3.1.3. Changes in the TSI indices

In LA, the TSI_{ROT} value was just over 51 in both 2011 and 2012, and this value indicated meso-eutrophy. After restoration efforts, the index decreased to 45.20 and then began to increase in the following years, again indicating meso-eutrophic waters. Changes in the TSI_{ROT} values in MA were similar to those in LA. The index was just over 54 in 2011–2012 (i.e., meso-eutrophy). In the years 2013 and 2014, the value of the index decreased to approximately 46, and in 2016, it increased to 49.64 (i.e., meso-eutrophy). In 2011–2012, the TSI_{ROT} values for UA corresponded to meso-eutrophy. However, the TSI_{ROT} increased to 55–57 (i.e., eutrophy) in the next three years. In 2016, the TSI_{ROT} reached a value of 53, which again corresponded to meso-eutrophy (Table 3).

Table 3
Average annual concentrations (±standard error) of trophic state indexes: based on rotifer abundance (TSI_{ROT}), chlorophyll *a* concentration (TSI_{CHL}) and total phosphorus concentration (TSI_{TP}), in five studied impoundments.

Parameter (±SE)	TSI_{ROT}	Trophic state acc. TSI_{ROT}	TSI_{CHL}	Trophic state acc. TSI_{CHL}	TSI_{TP}	Trophic state acc. TSI_{TP}
LA						
2010	n.a.	–	48.22 (2.66)	Mesotrophic	76.86 (3.87)	Hypereutrophic
2011	51.82 (1.90)	Meso-eutrophic	51.21 (2.85)	Eutrophic	65.71 (1.89)	Eutrophic
2012	51.59 (1.61)	Meso-eutrophic	62.99 (3.31)	Eutrophic	64.81 (12.34)	Eutrophic
2013	45.20 (1.93)	Meso-eutrophic	39.27 (3.14)	Oligo/mesotrophic	54.27 (9.16)	Eutrophic
2014	48.86 (1.62)	Meso-eutrophic	42.37 (2.06)	Mesotrophic	76.61 (3.76)	Hypereutrophic
2015	49.43 (1.48)	Meso-eutrophic	47.68 (4.73)	Mesotrophic	71.34 (2.29)	Hypereutrophic
2016	44.47 (2.64)	Mesotrophic	52.23 (2.75)	Eutrophic	51.85 (5.09)	Eutrophic
MA						
2010	n.a.	–	47.73 (1.48)	Mesotrophic	70.19 (2.69)	Eu/hypereutrophic
2011	54.17 (2.12)	Meso-eutrophic	53.73 (0.99)	Eutrophic	69.79 (1.71)	Eutrophic
2012	54.89 (1.44)	Meso-eutrophic	61.60 (1.27)	Eutrophic	66.04 (9.33)	Eutrophic
2013	52.34 (2.20)	Meso-eutrophic	42.31 (3.22)	Mesotrophic	63.06 (4.92)	Eutrophic
2014	46.29 (1.92)	Meso-eutrophic	38.22 (2.21)	Oligotrophic	78.26 (3.11)	Hypereutrophic
2015	43.51 (1.30)	Mesotrophic	36.25 (0.98)	Oligotrophic	64.76 (4.22)	Eutrophic
2016	49.64 (2.41)	Meso-eutrophic	43.48 (3.64)	Mesotrophic	55.08 (5.22)	Eutrophic
UA						
2010	n.a.	–	46.43 (2.12)	Mesotrophic	73.14 (2.70)	Hypereutrophic
2011	52.33 (2.59)	Meso-eutrophic	60.27 (2.16)	Eutrophic	73.65 (2.03)	Hypereutrophic
2012	53.04 (1.16)	Meso-eutrophic	71.89 (3.20)	Hypertrophic	81.72 (14.20)	Hypereutrophic
2013	57.11 (0.56)	Eutrophic	48.08 (3.60)	Mesotrophic	64.59 (5.50)	Eutrophic
2014	56.68 (1.64)	Eutrophic	46.49 (2.02)	Mesotrophic	77.71 (3.94)	Hypereutrophic
2015	55.91 (1.70)	Eutrophic	44.95 (2.38)	Mesotrophic	68.28 (4.01)	Eutrophic
2016	53.03 (2.27)	Meso-eutrophic	46.05(1.86)	Mesotrophic	47.01 (3.79)	Mesotrophic
B17						
2010	n.a.	–	n.a.	–	79.70 (1.49)	Hypereutrophic
2011	n.a.	–	n.a.	–	76.74 (0.97)	Hypereutrophic
2012	n.a.	–	n.a.	–	90.03 (2.03)	Hypereutrophic
2013	62.44 (2.96)	Eutrophic	64.58 (1.34)	Eutrophic	63.41 (8.41)	Eutrophic
2014	55.86 (1.22)	Eutrophic	62.25 (1.77)	Eutrophic	93.62 (2.97)	Hypereutrophic
2015	52.73 (2.68)	Meso-eutrophic	52.43 (3.87)	Eutrophic	82.48 (4.01)	Hypereutrophic
2016	45.71 (1.35)	Meso-eutrophic	46.71 (2.24)	Mesotrophic	64.23 (8.08)	Eutrophic
B11						
2010	n.a.	–	n.a.	–	n.a.	–
2011	56.5 (2.08)	Eutrophic	60.63 (1.75)	Eutrophic	74.53 (0.74)	Hypereutrophic
2012	53.00 (1.17)	Meso-eutrophic	63.63 (2.09)	Eutrophic	80.11 (2.98)	Hypereutrophic
2013	61.91 (0.43)	Eutrophic	73.55 (6.41)	Hypertrophic	81.21 (4.44)	Hypereutrophic
2014	56.29 (0.92)	Eutrophic	67.11 (2.21)	Eutrophic	99.96 (1.85)	Hypereutrophic
2015	50.81 (2.77)	Meso-eutrophic	60.57 (2.26)	Eutrophic	88.05 (4.86)	Hypereutrophic
2016	44.47 (2.64)	Mesotrophic	53.28 (4.26)	Eutrophic	83.56 (6.13)	Hypereutrophic

n.a. – not analysed.

After restoration, a decline in the Chla concentration was observed in all three reservoirs. In LA, the Chla values decreased from 14.92 $\mu g\ L^{-1}$ to 7.67 $\mu g\ L^{-1}$. However, Chla concentrations have shown an upward trend in recent years. The average concentration of Chla clearly decreased from 11.82 to 3.74 $\mu g\ L^{-1}$ in MA, and it decreased from 41.64 to 6.02 $\mu g\ L^{-1}$ in UA, both as a result of restoration efforts. The correlation between the Chla concentration and the density of *Asplanchna priodonta* was positive and significant in LA ($r = 0.51$; $p = 0.002$) and MA ($r = 0.53$; $p = 0.004$).

In LA, the trophic state index based on Chla (TSI_{CHL}) indicated eutrophy before restoration, an improvement in trophic status (i.e., mesotrophy) from 2013 to 2015, and then a return to eutrophy in 2016 (Table 3). The correlation between the TSI_{CHL} and the TSI_{ROT} was positive and significant ($r = 0.50$; $p = 0.002$) in LA. The TSI_{CHL} values for MA indicated eutrophic conditions before restoration; however, after restoration efforts, water quality improved to mesotrophic conditions and even oligotrophic conditions in 2014–2015 (Table 3). The TSI_{CHL} and the TSI_{ROT} were positively and strongly correlated ($r = 0.74$; $p < 0.001$) in MA. In UA, the TSI_{CHL} showed an increase in the eutrophication of this reservoir relative to the values before restoration (i.e., from mesotrophic in 2010 to hypereutrophic in 2012); additionally, the trophic conditions improved to become mesotrophic again in 2013–2016 (Table 3). The UA was the only reservoir in which the correlation between TSI_{CHL} and TSI_{ROT} was very weak and not significant ($r = 0.03$; $p = 0.875$).

The trophic state index based on total phosphorus (TSI_{TP}) showed eutrophy/hypereutrophy in LA during the study periods (Table 3). The correlation between the TSI_{TP} and the TSI_{ROT} was insignificant ($r = 0.01$; $p = 0.957$) in this ecosystem. In MA, the trophic state index based on total phosphorus (TSI_{TP}) was largely disproportionate relative to TSI_{CHL}; according to this index, the water in MA was eutrophic (in 2011–2013 and 2015–2016) and hypereutrophic (in 2010 and 2014) (Table 3). The correlation between the TSI_{TP} and the TSI_{ROT} was insignificant ($r = 0.15$; $p = 0.399$). In UA, the TSI_{TP} showed hypereutrophic conditions from 2010 to 2012, but there was a gradual improvement of trophic conditions to mesotrophy in 2016 (Table 3). The correlation between the TSI_{TP} and the TSI_{ROT} was insignificant ($r = 0.07$; $p = 0.634$).

3.1.4. Changes in other biological parameters

3.1.4.1. PCA. Because the results of PCA obtained for each of the Arturówek reservoirs were similar, the overall PCA results for the combined data from all three reservoirs are presented. The results showed that all the biological parameters (i.e., the densities of Cladocera, Copepoda and Rotifera, the concentration of Chla and the TSI_{ROT}) and the concentration of ammonium were negatively correlated with PC1, while the physicochemical parameters contributed to PC2; specifically, nitrate concentration and conductivity had negative contributions, and temperature had a positive contribution (Fig. 3A). Only parameters that had a correlation with a given PC axis that was higher than 0.5 or lower than −0.5 were indicated here and elsewhere. The correlation matrix is available in the supplementary material (Table A.1). Based on the distribution of cases along PC1 (Fig. 3B), statistically significant differences were found between years (ANOVA: $F_{5,132} = 37.43$, $p < 0.001$), with a distinct separation of years before and after restoration (Tukey test: 2011 and 2012 < 2013, 2014, 2015); specifically, before restoration, sampling terms were related to higher values of biological parameters and ammonium concentrations.

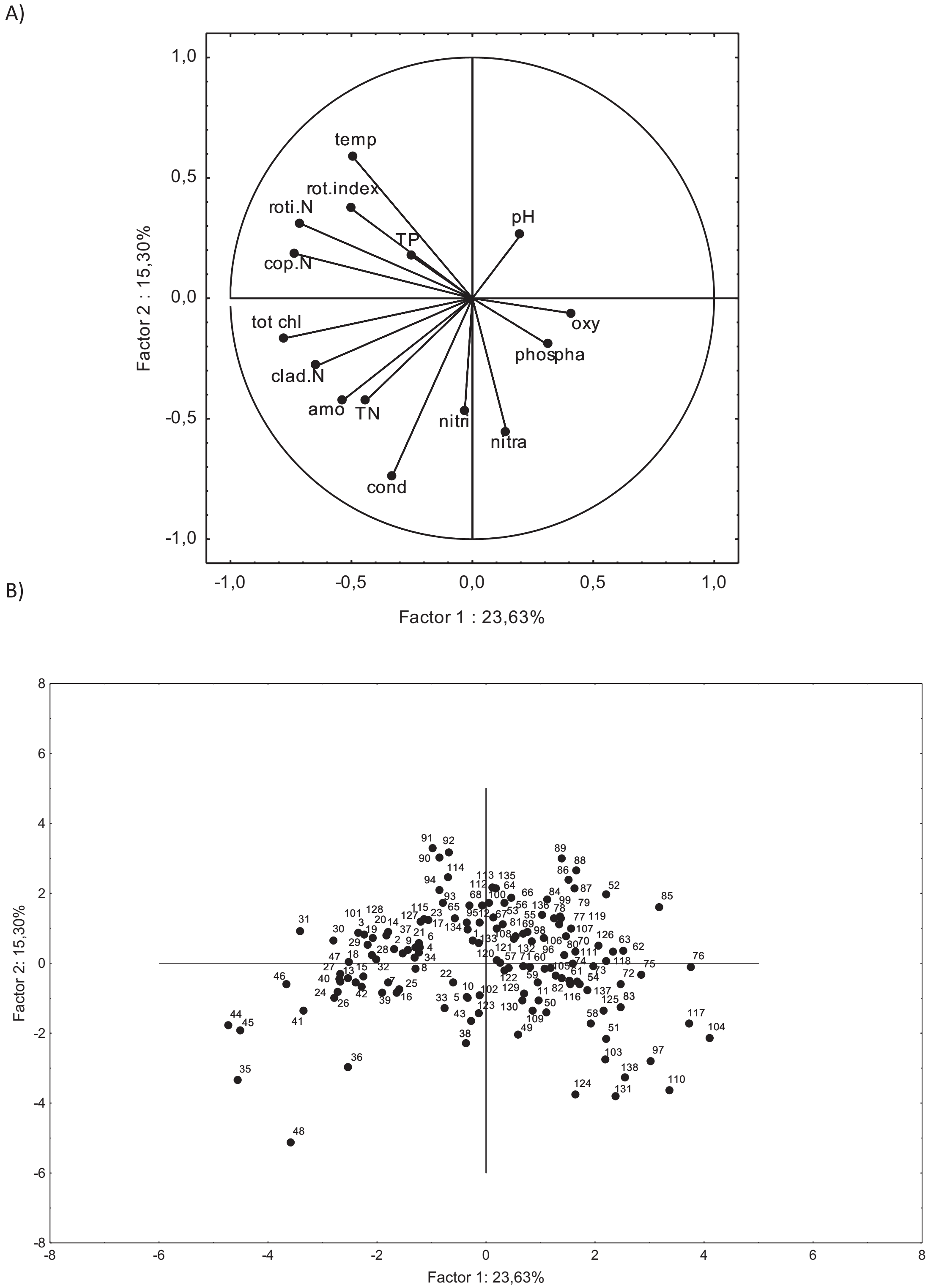

Fig. 3. Results of the principal component analysis (PCA) conducted on the combined data from the three Arturówek reservoirs: A) Projection of variables on the plane defined by the first two PCA factors. B) Projection of the sampling terms on the plane defined by the first two PCA factors. The numbers indicate the consecutive terms of sampling in 2011 (1−33), 2012 (34–48), 2013 (49–57), 2014 (58–96), 2015 (97–117) and 2016 (118–138). Legend: clad. N – density of Cladocera; cop. N – density of Copepoda; roti. N – density of Rotifera; roti index – the rotifer trophic state index; tot chl – concentration of chlorophyll *a*; cond – conductivity; oxy – concentration of oxygen; temp – temperature; amo – concentration of ammonium; nitra – concentration of nitrate; nitri – concentration of nitrite; phospha – concentration of phosphate; TN – total nitrogen; TP – total phosphorus.

3.1.4.2. Comparison of zooplankton parameters before and after restoration. In all three reservoirs, biological parameters that differed significantly before (i.e., 2010–2012) and after (i.e., 2013–2016) restoration were identified. All of them had higher values before restoration (Table 4). In LA, these parameters included the density and biomass of Cladocera (N_{Clad} and B_{Clad}), the biomass of Rotifera (B_{Rot}), the mean weight of zooplankton and rotifers (W_{Zoo} and W_{Rot}, respectively), the percentage of bacterivores in the total number of rotifers (BAC, %), and the TSI_{ROT} values.

In MA, these parameters included the densities of Copepoda, Rotifera, total zooplankton and total Crustacea (i.e., N_{Cop}, N_{Rot}, N_{Zoo}, and N_{crust}, respectively); the biomass values of Copepoda, Rotifera, Cyclopoida and total zooplankton (B_{Cop}, B_{Rot}, B_{Cyc}, and B_{Zoo}, respectively); the mean weight of rotifers (W_{Rot}); the density of Crustacea (N_{Crust}); the ratio of cyclopoid copepod biomass to the biomass of Cladocera (B_{Cyc}/B_{Clad}); the values of the percentage of bacterivores in the total number of rotifers (BAC, %); and the TSI_{ROT}.

In contrast, UA had higher values before restoration in terms of the densities of Copepoda, Cladocera and Crustacea (N_{Cop}, N_{Cla}, and N_{Crust}, respectively); the biomasses of Copepoda, Cyclopoida, Cladocera, Rotifera and total zooplankton (B_{Cop}, B_{Cyc}, B_{Cla}, B_{Rot}, and B_{Zoo}, respectively); the mean weights of Copepoda, Cladocera, Rotifera and total zooplankton (W_{Cop}, W_{Cla}, W_{Rot}, and W_{Zoo}, respectively); and the ratio of crustacean density to rotifer density (N_{Crust}/N_{Rot}) (Table 4).

3.1.4.3. Changes in fish community structure. Before restoration, the fish assemblage in LA was formed mainly by the perch (*Perca fluviatilis*) population (Table 5). Specimens of four cyprinid species, i.e., roach (*Rutilus rutilus*), crucian carp, tench and Prussian carp (*Carassius gibelio*) constituted over 21% of the numerical abundance, but constituted up to 67% of the biomass from the gillnet sample. Fishes smaller than 10 cm TL constituted <40% of the community, except for the results from 2012. Most of the small fishes were small perch specimens, which dominated the community in 2012 (i.e., 66.3% of the numerical abundance). After restoration, the participation of small fish clearly increased, usually constituting >70% of the fish specimens caught in the gillnets. The sunbleak (*Leucaspius delineatus*) and gudgeon constituted >98% of the fish specimens caught in 2013, and sunbleak were twice as abundant as gudgeon. In 2014, the situation changed dramatically. A total of 77.1% of the fish specimens were stocked pikeperch (85% of the biomass) and Prussian carp, and a much smaller percentage, i.e., <3%, of gudgeon, perch and stocked crucian carp were found. No sunbleak were found. In 2015 and 2016, the numerical abundance of pikeperch decreased, but pikeperch maintained high rank in terms of biomass contribution. Perch became more abundant, and its numerical abundance and biomass in the fish community increased. The stocked cyprinids, i.e., tench, crucian carp and ide, were present in small numbers, but these species never constituted >3% of the gillnet sample in terms of numerical abundance and biomass. Sunbleak specimens became very abundant again in 2015 (i.e., over 85%) but dropped below 2% in 2016. CPUE was higher for the period after restoration (Table 5).

Before restoration, the fish assemblage in MA was similar to the community found in LA; specifically, the fish assemblage was dominated mainly by perch. The specimens of four cyprinid species, i.e., crucian carp, tench, Prussian carp and roach, constituted over 13% of the numerical abundance, with crucian carp being the most abundant species (10.7%). Two species, i.e., the crucian carp and tench, constituted over 50% of the gillnet sample biomass. Fishes smaller than 10 cm TL constituted up to 20% of the community (Table 5), with the exception of 2012. Most of the small fishes were small perch specimens. After restoration, the contribution of small fish clearly increased, usually constituting over 50% of the fish specimens caught in the gillnets. Shortly after restoration, the reservoir was settled by two species, i.e., sunbleak and non-native Prussian carp, which constituted over 97% of the fish specimens caught in 2013. In 2014, the water level in the MA reservoir was dramatically lower than usual from the beginning of spring to the beginning of summer; this reduction occurred so the LA reservoir could be fully inundated, and, as a result, almost all fishes migrated downstream. Only one perch specimen and one roach specimen were found in the reservoir. In 2015 and 2016, only four fish species were found in the reservoir and low fish abundances were maintained in the reservoir (Table 5). In 2015, pike was the most abundant species in gillnets. In 2016, pike maintained a high share of the biomass, but perch became the dominant species, and its proportion of the fish community increased. The CPUE was much lower during the period after restoration (Table 5).

In contrast to the downstream reservoirs, the fish assemblage in UA was dominated by sunbleak before and after restoration. However, on average, three times as many fish were caught before restoration (see CPUE values, Table 5). Perch and ide were subdominant groups, with perch being more abundant and ide primarily contributing to the catch biomass (51%). After restoration, Prussian carp were second in terms of numerical abundance (25.9%), but they had a limited contribution to biomass (6.9%). Pike contributed to almost 10% of the catch and

Table 4
Comparison of the significance of the differences in biological parameters before (2010–2012) and after (2013–2016) restoration of three Arturówek reservoirs and the B11 pond, using the Monte Carlo permutation *t*-test.

Parameters	LA	MA	UA	B11
N_{Cop}	n.s.	t = 3.67, p < 0.001 before > after	t = 3.14, p = 0.003 before > after	n.s.
N_{Clad}	t = 4.05, p < 0.001 before > after	n.s.	t = 8.28, p < 0.001 before > after	n.s.
N_{Rot}	n.s.	t = 3.61, p < 0.001 before > after	n.s.	t = 3.17, p = 0.022 before > after
N_{Zoo}	n.s.	t = 3.43, p = 0.002 before > after	n.s.	n.s.
N_{Crust}	n.s.	t = 3.26, p = 0.003 before > after	t = 3.91, p < 0.001 before > after	n.s.
N_{Crust}/N_{Rot}	n.s.	n.s.	t = 4.55, p < 0.001 before > after	n.s.
B_{Cop}	n.s.	t = 4.00, p < 0.001 before > after	t = 4.42, p < 0.001 before > after	n.s.
B_{Cyc}	n.s.	t = 4.50, p < 0.001 before > after	t = 4.35, p < 0.001 before > after	n.s.
B_{Clad}	t = 3.72, p < 0.001 before > after	n.s.	t = 6.47, p < 0.001 before > after	n.s.
B_{Rot}	t = 2.61, p = 0.003 before > after	t = 6.70, p < 0.001 before > after	t = 4.18, p < 0.001 before > after	t = 2.82, p < 0.001 before > after
B_{Zoo}	n.s.	t = 3.36, p = 0.002 before > after	t = 6.02, p < 0.001 before > after	n.s.
B_{Cyc}/B_{Clad}	n.s.	t = 5.32, p < 0.001 before > after	n.s.	n.s.
W_{Cop}	n.s.	n.s.	t = 4.01, p < 0.001 before > after	n.s.
W_{Clad}	n.s.	n.s.	t = 4.67, p < 0.001 before > after	n.s.
W_{Rot}	t = 3.70, p < 0.001 before > after	t = 3.98, p < 0.001 before > after	t = 4.03, p < 0.001 before > after	n.s.
W_{Zoo}	t = 4.34, p < 0.001 before > after	n.s.	t = 7.83, p < 0.001 before > after	n.s.
BAC [%]	t = 3.03, p = 0.005 before > after	t = 5.75, p < 0.001 before > after	n.s.	n.s.
TSI_{ROT}	t = 2.26, p = 0.029 before > after	t = 3.83, p < 0.001 before > after	n.s.	n.s.

Legend: density of Copepoda (N_{Cop}), Cladocera (N_{Clad}), Rotifera (N_{Rot}), Zooplankton (N_{Zoo}) and Crustacea (N_{Crust}); the ratio of crustacean density to rotifers density (N_{Crust}/N_{Rot}); biomass of Copepoda (B_{Cop}), Cyclopoida (B_{Cyc}), Cladocera (B_{Clad}), Rotifera (B_{Rot}) and Zooplankton (B_{Zoo}); the ratio of cyclopoid copepod biomass to the biomass of Cladocera (B_{Cyc}/B_{Clad}); mean weight of Copepoda (W_{Cop}), Cladocera (W_{Clad}), Rotifera (W_{Rot}) and Zooplankton (W_{Zoo}); percentage contribution of bacterivores (BAC, %); value of the rotifer trophic state index (TSI_{ROT}); n.s. – not significant.

Table 5
Characteristic of the fish assemblage in the three reservoirs of Arturówek before (2010–2012) and after restoration (2013–2016) in respect to the fish family (Cyprinids, Percids[a]) or genus (*Esox* sp. and *Sander* sp.), size class (cm TL), the share in abundancy (n, % of specimens) and biomass (B, % of sample weight), total number of caught specimens (N_{tot}, specimens) and gillnet sample biomass (B_{tot}, kg), average catch per unit of effort for abundancy ($CPUE_{Ntot}$, specimens $(m^2\ night)^{-1}$) and biomass ($CPUE_{Btot}$, g $(m^2\ night)^{-1}$) and biomanipulation results presented as the variability of specimens and biomass contribution of small fishes (<9.9 cm TL), *Esox* sp. and *Sander* sp. between 2010 and 2016.

Fish family	Size class	LA		MA		UA	
		n	B	n	B	n	B
Contribution of cyprinids, percids, Esox sp. and Sander sp. in fish assemblage before restoration							
Cyprinids	< 9.9	1.48	0.30	0.26	0.02	42.17	1.59
	10.0–19.9	12.17	19.51	2.81	5.94	7.23	8.05
	> 20.0	8.01	54.45	12.02	72.46	21.08	68.94
Percids[a]	< 9.9	56.68	7.79	70.59	9.39	5.42	0.90
	10.0–19.9	21.36	9.10	12.53	6.29	22.29	10.10
	> 20.0	0.30	8.85	1.28	2.64	1.20	7.47
Esox sp., *Sander* sp.		0.00	0.00	0.51	3.27	0.60	2.95
N_{tot}/B_{tot}		337	13.47	391	24.01	166	9.44
$CPUE_{Ntot}/CPUE_{Btot}$		1.2	49.9	1.4	88.9	0.6	35.0
Contribution of cyprinids, percids, Esox sp. and Sander sp.in fish assemblage after restoration							
Cyprinids	< 9.9	71.43	4.71	68.42	3.97	64.20	2.18
	10.0–19.9	8.89	6.22	6.58	5.17	8.64	6.45
	> 20.0	1.22	9.73	4.61	26.78	17.28	65.53
Percids[a]	< 9.9	5.75	0.81	10.53	0.61	0.00	0.00
	10.0–19.9	0.70	0.55	0.66	0.79	0.00	0.00
	> 20.0	2.09	18.92	5.92	31.54	0.00	0.00
Esox sp., *Sander* sp.		9.93	59.07	3.29	31.14	9.88	25.84
N_{tot}/B_{tot}		574	23.80	152	11.51	81	9.66
$CPUE_{Ntot}/CPUE_{Btot}$		1.6	66.1	0.4	32.0	0.2	26.8

Fish group	Year	LA		MA		UA	
		n	B	n	B	n	B
Contribution of fishes < 9.9 cm TL, Esox sp. and Sander sp. in fish assemblage between 2010 and 2016 in relation to the whole gillnet sample							
Fishes < 9.9 cm	2010	9.1	0.7	10.3	0.2	5.9	0.2
	2011	31.6	2.5	20.8	0.7	67.7	2.6
	2012	68.2	14.0	81.7	18.4	48.3	2.9
	2013	80.1	45.7	91.2	43.4	87.0	26.1
	2014			50.0	0.5	40.0	0.3
	2015	85.4	3.5			57.1	0.2
	2016	72.3	3.1	55.2	1.0		
Esox sp., *Sander* sp.	2010			5.1	13.1		
	2011						
	2012					0.8	4.3
	2013						
	2014	77.1	84.8			15.0	6.3
	2015	6.8	42.0	50.0	37.0	42.9	99.8
	2016	23.4	91.8	3.4	33.3	25.0	21.9

[a] Percids without pikeperch *Sander lucioperca*.

over 25% of the sample biomass. Ide and crucian carp were responsible for >66% of the catch biomass and >23% of the numerical abundance. No perch were found in the reservoir after restoration. There was no clear difference in the presence of small fish before and after restoration.

3.2. Comparison of chemical and biological parameters before and after restoration in Bzura-17 and the unrestored Bzura-11 pond

3.2.1. Changes in the concentrations of chemical parameters

In the B17 pond, only the phosphorus parameters increased after restoration. The average concentrations of TN and TP before restoration were 2.48 and 0.22 mg L^{-1}, respectively. After restoration, the average concentration of TN decreased to 1.82 mg L^{-1}, while the average concentration of TP increased to 0.44 mg L^{-1}. The average concentration of ammonium declined from 0.56 mg L^{-1} (in 2010–2012) to 0.24 mg L^{-1} (in 2013–2016). The average concentration of phosphate increased from 0.17 to 0.26 mg L^{-1} after restoration efforts.

In the B11 pond, all chemical parameters showed an upward trend during the analysed years. The average annual concentrations of TN and TP changed, with nitrogen ranging from 1.48 mg L^{-1} (2011) to 4.86 mg L^{-1} (2016) and phosphorus ranging from 0.13 mg L^{-1} (2011) to 0.38 mg L^{-1} (2016). Moreover, intensive increases in the phosphate and ammonium concentrations were observed from 2011 to 2016. The average annual concentration of phosphate increased from 0.16 to 0.69 mg L^{-1} and that of ammonium increased from 0.13 to 2.88 mg L^{-1}. However, substantial and regular increases in ammonium were observed since 2013, particularly during the second half of the year, until the maximum value of 7.23 mg L^{-1} was reached in September 2016. The increasing ammonium concentration corresponded to the decrease in the density of rotifers, and a significant negative correlation was found between these parameters ($r = -0.59$; $p < 0.005$). For comparison, in other impoundments, such high ammonium values were not observed, and the analogous correlation was not statistically significant, i.e., in B17, $r = 0.30$, (n.s.); in UA, $r = -0.005$, (n.s.); in MA, $r = -0.18$, (n.s.); and in LA, $r = -0.26$, (n.s.). Details are in Table 1 and in Jurczak et al. (2018a).

3.2.2. Changes in zooplankton community structure

Zooplankton was analysed in 2012–2016 in B11 and in only 2013–2016 in B17. In both ponds, the zooplankton community was very similar and was dominated by Cyclopoida and their nauplii as well as by Rotifera. Cladocera were generally present in small densities (with the exception of the high density of *Bosmina* sp. in B11), and these genera were mainly composed of *Bosmina*, *Daphnia*, *Ceriodaphnia* and *Chydorus*. In both impoundments, the dominant genera of rotifers were *Keratella*, *Polyarthra*, *Trichocerca*, *Filinia*, and the species

Anuraeopsis fissa was also found (Table 2). Since 2014, the density of Rotifera has decreased in the B11 pond, but the species composition did not change until 2016, when the only genera found in the samples were *Keratella* and *Polyarthra*.

3.2.3. Changes in the TSI indices

In the B17 pond, the rotifer trophic state index was analysed for only the 2013–2016 period. In 2013, the high index value (62.44) corresponded to eutrophic conditions. Over the following years, the TSI_{ROT} gradually decreased, reaching its lowest value (i.e., 45.71) in 2016 (Table 3). The values of the TSI_{ROT} in the B11 pond mainly corresponded to eutrophic conditions and ranged from 53 to almost 70 in 2011–2014. Then, the values of the index decreased to 50.81 in 2015 and decreased to 44.47 (mesotrophy) in 2016 (Table 3). However, in contrast to the B17 pond, the low index values calculated for the years 2015 and 2016 in B11 did not result from the improvements in water quality; rather, they were a result of the significant decrease in Rotifera density, as the components of the index ($TSI_{ROT1-ROT3}$) consider both the density and the biomass of these organisms. An analogous situation did not occur in the other impoundments.

Chla was not studied in 2010–2012 in the B17 pond. In the period from 2013 to 2016, the average concentration of Chla was 20.68 μg L^{-1}; however, the average annual values showed a downward trend in two recent years (i.e., 14.50 μg L^{-1} in 2015 and 6.15 μg L^{-1} in 2016) in this pond. In the B11, the average concentration of Chla in 2011–2016 was 44.49 μg L^{-1}, with the highest value of 111.7 μg L^{-1} recorded in 2013. The correlation between the Chla concentration and the density of *Asplanchna priodonta* was positive and significant ($r = 0.379$; $p = 0.031$).

The TSI_{CHL} showed changes in the trophic state of the B17 pond, i.e., from eutrophic conditions in 2013–2015 to mesotrophic conditions in 2016 (Table 3), while in the B11, the TSI_{CHL} value indicated eutrophic conditions in all years except 2013, when the conditions were hypereutrophic according to this index. A strong positive correlation between the TSI_{CHL} and the TSI_{ROT} was found ($r = 0.81$; $p < 0.001$) in the B17, and the correlation was positive and significant ($r = 0.57$; $p < 0.001$) in the B11 pond. The trophic state index based on total phosphorus (TSI_{TP}) indicated hypereutrophic conditions for the B17 pond in 2010–2012 and 2014–2015 and eutrophic conditions in 2013 and 2016 (Table 3). The correlation between the TSI_{TP} and the TSI_{ROT} was not significant ($r = 0.17$; $p = 0.420$) in this pond. In the case of the B11 pond, the trophic state index based on total phosphorus (TSI_{TP}) showed hypereutrophic conditions during all studied periods (Table 3). The correlation between the TSI_{TP} and the TSI_{ROT} was also not significant ($r = 0.03$; $p = 0.887$).

3.2.4. Changes in other biological parameters

In the B17 pond, the Chla concentration, rotifer density, TSI_{ROT}, conductivity and oxygen concentration were identified by PCA as the main negative contributors to the PC1 axis; in contrast, the total nitrogen and phosphate concentrations were identified as the main positive contributors. In turn, oxygen was positively correlated with PC2, and temperature, total phosphorus and ammonium concentrations were negatively correlated with PC2 (Fig. 4A). The correlation matrix is available in the supplementary material (Table A.2). In the B17, based on the distribution of cases along PC1, statistically significant differences were found between years (ANOVA: $F_{3,25} = 9.63$; $p < 0.001$), mostly due to the separation of the year 2016 (Tukey test: 2016 > 2013, 2014, 2015) (Fig. 4B, cases 25–30), which was when high concentrations of total nitrogen, nitrate and phosphate were observed.

In the pond B11, PCA revealed that the Chla, rotifer density, TSI_{ROT}, pH and oxygen concentration negatively contributed to PC1, while the total nitrogen, nitrate, ammonium and phosphate concentrations were the primary positive contributors. Copepoda and Cladocera densities and temperature were negatively correlated with PC2 (Fig. 5A). In B11, the distribution of sample terms on the plane defined by the PCA axes were different than that in the Arturowek reservoirs. The correlation matrix is available in the supplementary material (Table A.3). Although there were significant differences among study years (ANOVA: $F_{5,37} = 16.99$; $p < 0.001$), these differences mostly resulted from the presence of points from the year 2016 on the positive side of the PC1 axis (Tukey test: 2016 > 2011) (Fig. 5B, cases 39–44), which were characterised by high concentrations of ammonium and phosphate as well as low Rotifera density and TSI_{ROT}.

Among the biological parameters, only density and rotifer biomass had higher values in 2010–2012 than in 2013–2016 in pond B11 (Table 4). During sampling, the presence of Prussian carp and sunbleak was observed.

4. Discussion

4.1. Relation of zooplankton parameters with trophic condition of the ecosystems

In LA, MA, UA, B17 and B11, Rotifera was the most represented group in terms of both the number of species and their densities. In 2010–2012, the rotifer communities of the B11 pond and the Arturówek reservoirs were dominated by taxa that are typical for high trophic conditions, including *Anuraeopsis fissa*, *Brachionus angularis*, *Filinia longiseta*, *Keratella quadrata*, *Keratella cochlearis* f. *tecta* and species belonging to the genus *Trichocerca* (Pejler, 1983; Karabin, 1985). During this period, other symptoms of eutrophic conditions associated with zooplankton (according to Ejsmont-Karabin and Karabin, 2013) were observed, such as the occurrence of cladoceran species characteristic of eutrophic waters (i.e., *Bosmina longirostris*, *Diaphanosoma brachyurum*, and *Chydorus sphaericus*), the high abundance of Cyclopoida and their nauplii (in all ecosystems), and the high ratio of cyclopoid biomass to Cladocera biomass (in MA).

After restoration of LA, MA and UA, chemical and biological conditions changed, as evidenced by PCA results; specifically, in all three ecosystems, a distinct separation of years was found before and after restoration, and sampling terms before restoration were related to higher values of the biological parameters. The decreased values of many biological parameters in the period from 2013 to 2016 were a symptom of the positive influence of restoration on some parameters, but these influences were unfavourable or neutral for other indices. After restoration, the contribution of species that were indicative of high trophic conditions decreased in the LA and MA reservoirs. This phenomenon could have been related to the reduction in detritus after the removal of sediments or/and the decrease in un-grazed algal biomass entering the detritus-based food web (Ejsmont-Karabin, 2012). Only in the UA did the percentage share of bacterivores and TSI_{ROT} values not change, despite the removal of sediments and the observed decrease in the concentration of Chla. However, the UA was the only impoundment among all study sites that has a short stretch of river neighbouring it upstream rather than another pond (Fig. 1); therefore, we assumed that its rotifer community mainly originated from the river. This would explain the negligible responses of the biological parameters to restoration in this ecosystem.

Moreover, in the MA and UA reservoirs the abundance and biomass of cyclopoid copepods decreased, as well as the ratio of cyclopoid copepod biomass to the biomass of Cladocera (B_{Cyc}/B_{Clad}) in MA. The ratio of B_{Cyc}/B_{Clad} is related to the trophic status of the water, and its high value is characteristic of eutrophic conditions (Ejsmont-Karabin and Karabin, 2013). This is because eutrophic lakes provide favourable food conditions for many omnivorous Cyclopoida species, including the availability of detritus and bacteria for the juvenile forms (Maier, 1998). Thus, reductions in the dominance of Cyclopoida and decreases in the B_{Cyc}/B_{Clad} ratio indicated the water quality was improving. These results were similar to the values of TSI_{ROT}, which significantly decreased after restoration in the LA and MA reservoirs. However, the reductions in some of the zooplankton parameters, such as the abundance of

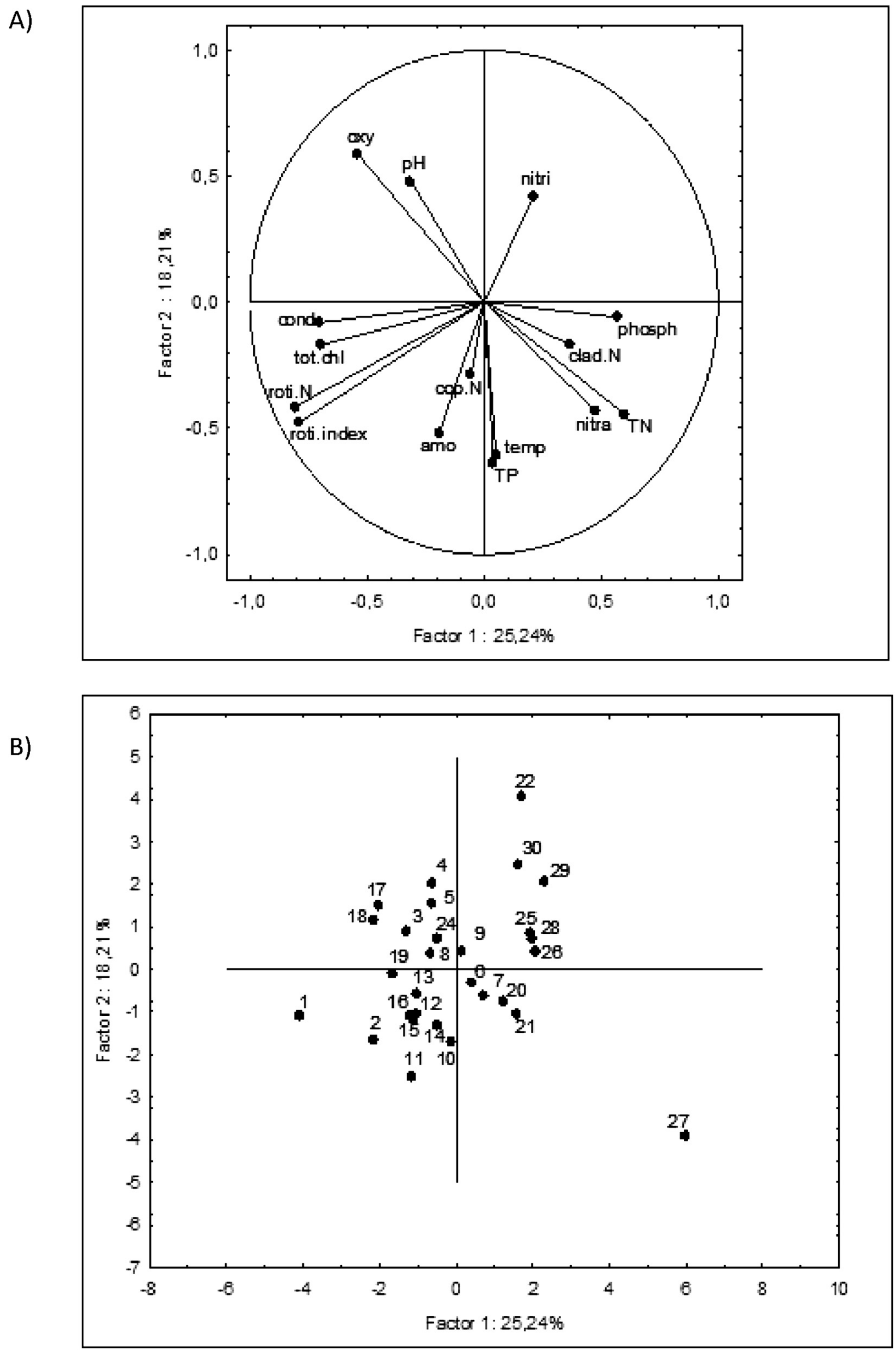

Fig. 4. Results of the principal component analysis (PCA) conducted on data from the Bzura-17 pond: A) Projection of variables on the plane defined by the first two PCA factors. B) Projection of the sampling terms on the plane defined by the first two PCA factors. The numbers indicate the consecutive terms of sampling in 2013 (1–3), 2014 (4–16), 2015 (17–23) and 2016 (24–30). The legend is the same as in Fig. 3.

Cladocera and Rotifera, their biomass and the average weight of the individual in the period of 2013–2016, represented less favourable changes that were observed in the Arturówek reservoirs (Table 4). These changes were related to the fact that cladocerans such a *Daphnia* are recognised the keystone genus responsible for good water quality in freshwater temperate ecosystems due to their high filtering potential. Furthermore, low biomass and zooplankton individuals with low weights are characteristic of eutrophic waters (e.g., Jeppesen et al.,

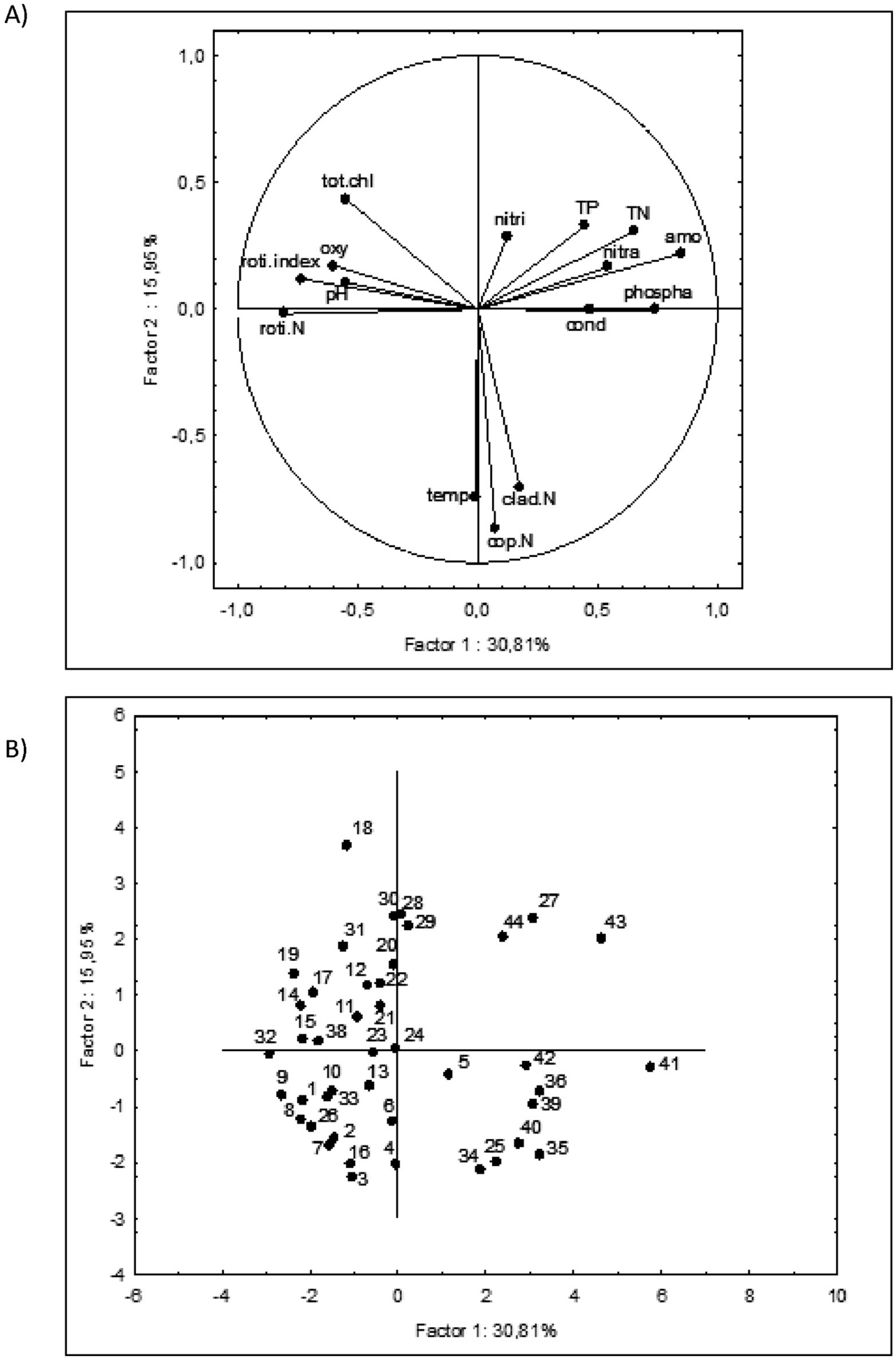

Fig. 5. Results of the principal component analysis (PCA) conducted on data from the Bzura-11 pond: A) Projection of variables on the plane defined by the first two PCA factors. B) Projection of the sampling terms on the plane defined by the first two PCA factors. The numbers indicate the consecutive terms of sampling in 2011 (1−11), 2012 (12–15), 2013 (16–18), 2014 (19–31), 2015 (32–37) and 2016 (38–44). The legend is the same as in Fig. 3.

2000; Haberman and Laugaste, 2003). This is because, in aquatic ecosystems with increased nutrient loading and elevated fish predation, smaller zooplankton species achieve competitive dominance because they are less inhibited by toxic cyanobacteria or their large filament size, and they are less exposed to fish pressure (Vakkilainen et al., 2004). In all three Arturówek reservoirs, reductions in rotifer biomass and average weight were observed. This phenomenon is often a result of increasing dominance of small detritophagous species (Ejsmont-Karabin, 2012). However, in LA and MA, the contribution of species such as *Anuraeopsis fissa*, *Keratella cochlearis*, *Brachionus angularis* and *Filinia longiseta* in the total number of rotifers was significantly lower after restoration than before restoration, and this pattern was also reflected in the values of the TSI_{ROT} (Table 4). Therefore, we assumed that the reduction in the biomass and the average weight of rotifers was caused by the decrease in the proportion of *Asplanchna priodonta* in the Rotifera community, which was observed in 2013–2016 (Table 2). One probable reason for the decline in *A. priodonta* density could be changes in the amount of food base available after restoration. Many studies have demonstrated that *A. priodonta* is an opportunistic feeder whose trophic position entirely depends on the structure of the plankton community; this is because its diet may consist of algae, including diatoms, dinoflagellates, cyanobacteria, Volvocales, Chlorococcales, and Desmidiales as well as small rotifers (e.g., *Keratella cochlearis* and *Polyarthra* spp.) and protozoans, such as *Codonella*, *Difflugia* and *Tintinnida* (Chang et al., 2010; Oganjan et al., 2013). The improvements in water quality in the Arturówek reservoirs after restoration were associated with a significant reduction in phytoplankton abundance, which was expressed by the reduced Chla concentrations and the decrease in the density of some rotifer species, which are potential food for *A. priodonta*. In some ecosystems, *A. priodonta* is a grazer rather than a predator (Kappes et al., 2000; Pociecha and Wilk-Woźniak, 2008); thus, we assumed that the amount of phytoplankton may have been of key importance to the dynamics of the *A. priodonta* population in the studied ecosystems. This hypothesis may be supported by the positive correlation between the density of *A. priodonta* and the concentration of Chla in all of the studied ecosystems (however, it was significant only in LA, MA and B11). However, we do not exclude that the decrease in *Asplanchna* density could also be related to fish predation.

In the smallest restored impoundment, i.e., the B17 pond, the values of the chemical parameters clearly decreased after restoration, and this state remained until 2015. Unfortunately, in 2016, the concentrations of total nitrogen, nitrate and phosphate significantly increased. Nonetheless, changes in the biological parameters pointed to the progressive improvement in water quality in the years after restoration. According to the PCA results, the Chla concentration, rotifer density, TSI_{ROT} and conductivity were positively correlated. The significant decrease in the Chla concentration was associated with a decline in the TSI_{ROT} values. In addition, positive changes were evidenced by the fact that the mean zooplankton weight, the mean rotifer weight and the ratio of crustacean density to rotifer density (N_{Crust}/N_{Rot}) increased and were significantly higher in 2016 than in 2013. Moreover, the percentage contribution of bacterivores in the total number of rotifers declined, which was analogous to LA and MA.

Interesting changes in the dynamics of both chemical and biological parameters were observed in the non-restored B11 pond from 2011 to 2016. The successive increase in the values of the chemical parameters (mainly TP, TN, phosphates and ammonium concentration; Table 1) indicated progressing eutrophication and water pollution. The significant differences between years, which were found by PCA, showed that only one year (2016) was distinctly separated from other years due to abnormally high concentrations of ammonium and phosphate and low Rotifera density and TSI_{ROT} values. The significant strong negative correlation between biological and chemical parameters and specifically between ammonium concentration and rotifer density indicated that the decline in the TSI_{ROT} values did not result from improvements in water quality (which was the original assumption of this index); in contrast, changes were caused by the increasing pollution and its harmful impact on Rotifera. Ammonia and nitrite are the most common toxic nitrogenous compounds in aquatic ecosystems (Cheng et al., 2013). Although many authors have documented the toxic effects of ammonia on Rotifera, most of these studies consisted of laboratory experiments using the genus *Brachionus*. For example, Schlüter and Groeneweg (1985) found that the reproduction rate of *B. rubens* significantly decreased when the ammonia concentration ranged from 3 to 5 mg L^{-1}; however, above 5 mg L^{-1}, rotifers died within 2 days. Few field studies have indicated a trend of declining populations for most rotifer species under concentrations of NH_4-N > 2 mg L^{-1} (Ji et al., 2013) and of unionised ammonia levels over 2.5 mg L^{-1} (Arauzo, 2003). The increasing NH_3-N concentration in water is highly associated with bacterial denitrification and nitrate ammonification, i.e., with mineralisation of organic matter that typically occurs in the sediments and in anaerobic environments (Ward, 1996). Pond B11 is small and surrounded by trees; thus, in terms of hydrological conditions, it is stagnant due to the fencing effect of the trees, and as a consequence, lacks water mixing caused by wind and has high inputs of allochthonous organic matter in the form of leaf litter, which results in intensive mineralisation. In addition, this pond was generally characterised by high concentrations of Chla. These facts are important because high levels of ammonia (NH_3-N), which is present as ammonium (NH_4^+) and unionised ammonia (NH_3), may also be produced during bloom degradation (Codd et al., 2005). Aquatic ecosystems must balance unionised ammonia (NH_3) and nontoxic ionised ammonia (ammonium; NH_4^+). This equilibrium is controlled by pH and temperature. For example, as pH and temperature increase, the ionic form decreases, and the balance shifts towards the unionised species. In eutrophic and hypereutrophic ecosystems, this type of chemical dynamics plays a key role in ammonia toxicity (Arauzo and Valladolid, 2003). The strong negative correlations between the ammonium concentrations and rotifer density observed in B11 may be an indirect indication of a high ammonia concentration and its harmful effect on the rotifer community.

4.2. Impact of fish on zooplankton in Arturówek reservoirs

The relatively low density of Cladocera (particularly, species of *Daphnia*) and domination of Rotifera may have resulted from periodic fish pressure, which concerns rotifers to a small extent. Rotifers are known to be a valuable primary food for larval fish and are currently widely used in aquaculture (Awais et al., 1992; Yúfera, 2001); however, most fish species quickly shift their interest to larger prey (Hillbricht-Ilkowska, 1964; Lazzaro, 1987; Solovyev et al., 2014). However, some bias in the literature might exist with respect to estimations of rotifer frequency in fish diets because the soft-bodied structures of many rotifers make them easier to destroy in the digestive tract (Stenson, 1982). This may be one of the reasons why most of the feeding studies that cover the early life stages do not mention rotifer presence in the diet or present them as a highly limited food item (Penttinen and Holopainen, 1992; Rezsu and Specziár, 2006; Specziár and Rezsu, 2009; Dukowska and Grzybkowska, 2014; Dukowska et al., 2014; Özdilek and Jones, 2014; Solovyev et al., 2014; Yazicioğlu et al., 2016). Even the research covering small fish species such as sunbleak do not indicate the importance of rotifers as a food source (Gozlan et al., 2003), even though it was shown that freshwater fish can forage on larger rotifer species (Stenson, 1982). Nevertheless, studies that have focused on the fish-zooplankton community relationship provide evidence for direct or indirect fish-Rotifera interactions (Benndorf et al., 2000; Shao et al., 2001). There is a common assumption, based on empirical evidence (Hillbricht-Ilkowska, 1964; Stenson, 1982; Telesh, 1993; Benndorf et al., 2000; Shao et al., 2001; Shao and Xie, 2003), that the presence of fish results in increases in the density, biomass, production, fecundity and size of small filtering herbivores, with rotifers as the dominant zooplankton, especially in waters that are strongly impacted by fishes (Hillbricht-Ilkowska, 1964; Lazzaro, 1987). Our results presented a decrease in the density of some rotifer species, e.g., *A. priodonta*, which was recognised as a species that was subjected to direct predation from fish (Stenson, 1982; Shao et al., 2001), and these results seem to confirm these observations. However, we observed that all groups of rotifers decreased in density, even small *Keratella* sp. (Table 2), which was not consistent with the results of other researchers (Stenson, 1982; Shao et al., 2001). This supports the assumption presented above, i.e., that habitat transformations played a primary role in shaping the structure of the zooplankton community at Arturówek. In our case, the size of the ecosystems could also be significant because, as Karatayev et al. (2005) found, lakes have higher local richness of cladoceran communities than do small reservoirs, ponds and pools.

Before restoration, the fish assemblage in the Arturowek reservoirs was dominated by a "cycling" population of one particular year class of perch (e.g., the strong year class in 2012), such as ichtyocenose found in approximately 9% of European reservoirs (Kubečka, 1993). Our biomanipulation efforts combined with additional engineering restoration actions did not provide results that supported our expectations. Even in LA in 2014, where an increased number of piscivorous pikeperch and a sharp decline in small fishes was observed, no visible improvement was registered in the zooplankton community. This result confirms that successful biomanipulation is a complicated process (DeMelo et al., 1992; Mehner et al., 2004), and increasing the number of piscivores does not always transfer directly to better water quality estimators (Table 3). For example, Prejs et al. (1997) concluded that fish removal was needed to achieve improvements in water quality and higher zooplankton densities. Our observations in MA confirm these assumptions. The very low fish densities in 2014 and 2015 were correlated with much higher trophic state indexes (Table 3). In the LA and UA reservoirs, where the stocking of pike and pikeperch was initially successful, did not transfer into long-term effects. Our presumption was that cannibalism hindered the long-term efficiency of stocking pike and pikeperch, and this was especially visible in the case of the LA reservoir. Such intraspecific impacts are well documented in the literature (Frankiewicz et al., 1996, 1999; Yazicioğlu et al., 2016). To enhance the temporal efficiency of biomanipulation, stocking should be combined with the intensive removal of larger piscivores, e.g., through commercial or recreational fisheries, to decrease the cannibalistic predation level, as suggested by Scharf (2008). However, the sudden appearance of large perch in the LA and MA reservoirs due to illegal stocking might undermine the cost-efficiency of biomanipulation efforts, making stocking of 0+ pike and pikeperch pointless.

4.3. Influence of ecosystem characteristics on biological indices studied

Our research indicates that the differential responses of the studied urban impoundments could result from their location, size and human impact (see also Jurczak et al., 2018a). An example is MA, which is a reservoir that has lower human pressure than that of LA and UA, has the least recreational use, and, due to its location, has a limited inflow of pollutants directly from the Bzura River. The biological indices of this water body improved the most after restoration. Biological parameters also indicated that the changes after restoration were generally less explicit in the larger Arturówek reservoirs than in the small B17 pond. In addition, there were symptoms that the ecological conditions of the Arturówek reservoirs were gradually returning to the state before restoration, which was probably due to intense human pressure. However, despite the fact that the examined ecosystems responded in different ways to the restoration efforts, we noticed repetitive dependencies. One of them was the strong positive correlation between TSI_{ROT} and TSI_{CHL}, which indicated that phytoplankton abundance was a key parameter related to rotifer populations dynamics. Thus, our research confirmed the thesis that the composition and abundance of rotifers was mainly regulated by bottom-up forces (Ejsmont-Karabin, 2012), although the TSI_{ROT} was not correlated with the TSI_{TP} in any of the studied ecosystems. It should be noted that Rotifera depend only indirectly on the amount of phosphorus present. The close correlations between Chla and phosphorus concentrations occur in ecosystems where phosphorus is the major limiting factor for algal growth, and the concentrations of all forms of phosphorus present are a function of algal biomass (Kalff, 2002). In eutrophic ecosystems, this relationship is less visible, and biological parameters often change significantly during the season based on mutual interactions and environmental factors other than phosphorus availability (e.g., temperature, light, oxygen, and micronutrients) (Carlson, 1977; Filstrup and Downing, 2017).

Our results indicate that the TSI_{ROT} is useful as a biological indicator of water quality in small urban impoundments, although we plan to test it during the upcoming years. This index has been used to assess the state of eutrophication of waters in 84 lakes situated in northeastern Poland, within four geomorphological units, i.e., Masurian, Iława and Suwałki Lake Districts and Lubawa Upland (Ejsmont-Karabin, 2012; Ochocka and Pasztaleniec, 2016), as well as in Lake Võrtsjärv, which is situated in Central Estonia (Haberman and Haldna, 2014). On the basis of these studies, the TSI_{ROT} seems to be a good indicator wherever there is a Rotifera community characteristic of freshwater waters of the temperate zone, with trophic statuses that range from oligo- to eutrophy. However, despite the promising results, this index has a limited application for evaluating the water quality in dystrophic lakes (Ejsmont-Karabin, 2012) and in brackish water bodies (Gutkowska et al., 2013). Therefore, studies on the TSI_{ROT} in different types of ecosystems and various conditions are still needed.

4.4. Assessment of the impact of investment actions on biological indexes in restored ecosystems

In the implemented EU LIFE project "Ecohydrologic rehabilitation of recreational reservoirs "Arturówek" (Łódź) as a model approach to rehabilitation of urban reservoirs" (EH-REK), a complex restoration strategy was tested (details in Part 1: Jurczak et al., 2018b). Most of these strategies concerned environmentally friendly solutions that naturally regenerate the ecosystem and increase its resistance to anthropogenic stress (Zalewski, 2000, 2015). A few examples of these strategies include the following: developing vegetation and the construction of sequential sedimentation-biofiltration systems (SSBS) and hybrid systems to improve the quality of the inflowing water as well as multiple biotechnologies, such as the development of landform-adjusted shoreline vegetation, to reduce the nutrient concentrations in surface runoff, the construction of floating islands to remove dissolved nutrients, and the regulation of the biological structure of the ecosystem to enhance the above described "technical" activities used for water quality maintenance (Jurczak et al., 2018a, 2018b). However, bottom sediment removal has also been used as a widely accepted conventional method of water quality improvement in small reservoirs. The necessity to use this method resulted from two reasons: 1) the reduction of internal loads by removing nutrients deposited in sediments as the first stage of restoration; and 2) to compare the effectiveness and costs of a traditional restoration method, such as the removal of sediments, with innovative ecohydrological measures, such as the hybrid system and SSBS.

However, the removal of sediments is a very invasive method that can seriously disturb the dynamics of the zooplankton population in the ecosystem. This is because bottom sediments of the lake accumulate the egg-bank of cladocerans and rotifers as well as pools of diapausing copepodites, and these processes have several implications for zooplankton ecology, genetics, and evolution (Gyllström and Hansson, 2004). Most resting stage eggs are located within a few centimetres of the sediment surface layer (Brendonck and De Meester, 2003); thus, they are removed during dredging. Although we do not have direct evidence for this, we suppose that the removal of sediments in the studied ecosystems has reduced the bank of resting eggs, especially those of cladocerans. Perhaps this phenomenon is confirmed by the significant reduction of Cladocera species in all ecosystems where the removal of sediments was applied. Similar changes were not observed in B11, which was not restored. It also seems that the removal of sediments had a smaller impact on the Rotifera community because the changes in the dominance structure mainly concerned the reduction of the share of bacterivore species that respond to improvements in water quality. However, the monitoring of biological indicators in subsequent years in terms of the long-term response of zooplankton species to investment activities is necessary.

5. Conclusions

Understanding "water-biota interactions" is necessary for regulating ecosystem structure and processes to increase their carrying capacity

(Zalewski, 2000, 2015). The importance of zooplankton in these biotic interactions, namely, its ability to regulate phytoplankton biomass and energy transfer to higher trophic levels, can be useful in developing the concept of restoration of a given ecosystem, and then to evaluate the effectiveness of the efforts made (Jeppesen et al., 2011). Our research confirmed the high potential of biological indicators in assessing the ecological quality of water in small urban aquatic ecosystems, despite the varied responses of the studied water bodies to restoration. These differences in responses to restoration efforts resulted not only from the ecological characteristics of the ecosystem but also from the size and location, as well as the character of the catchment and its manner of use. We have also demonstrated some restrictions of the rotifer trophic state index, indicating that the TSI_{ROT} should not be used in ecosystems with abnormally high nutrient peaks that limit the abundance of rotifers. Thus, it might be argued that a comprehensive assessment of restoration effects should be based on the simultaneous monitoring of both biological and physicochemical components in terms of the use of the ecosystem and the nature of the catchment. Using biomanipulation in small impoundments should be combined with active fishery management that focuses on the annual removal of larger piscivorous fish to decrease the cannibalism ratio.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.08.006.

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