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Comprehensive approach to restoring urban recreational reservoirs. Part 1 – Reduction of nutrient loading through low-cost and highly effective ecohydrological measures

Tomasz Jurczak[a,*], Iwona Wagner[a], Adrianna Wojtal-Frankiewicz[a], Piotr Frankiewicz[a], Agnieszka Bednarek[a], Małgorzata Łapińska[a], Zbigniew Kaczkowski[a], Maciej Zalewski[b]

[a] Department of Applied Ecology, Faculty of Biology and Environmental Protection, University of Łódź, 90-237 Łódź, 12/16 Banacha str., Poland
[b] European Regional Centre for Ecohydrology PAS, 90-364 Łódź, 3 Tylna str., Poland

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ABSTRACT

The study presented in this paper aimed to improve the water quality of a small, stormwater-fed urban river and a cascade of ponds and reservoirs (Łódź, Poland) that are intensively used for recreation. We tested a combination of conventional restoration methods (bottom sediment removal and biomanipulation) and comprehensive ecohydrological restoration methods (hybrid systems, sequential sedimentation-biofiltration systems (SSBS), floating islands, landform-adjusted shoreline vegetation and plant harvesting). As a result of sediment removal (9795 m^3, cost: €80761), almost 12.5 tons of nitrogen and 197 kg of phosphorous were eliminated from the reservoirs. The hybrid systems and SSBSs reduced the nutrient transport and suspended solid levels by 49%–98.5% and 89.6%–98.6%, respectively, from the upstream catchments to the reservoirs (investment cost: €63929). Generally, restoration improved most of the water quality indicators, although an increase in the phosphate concentration was observed from 0.06 to 0.17 $mg\,L^{-1}$ before restoration to 0.13–0.28 $mg\,L^{-1}$ immediately after restoration as a result of disturbances resulting from bottom sediment removal. Four years after restoration, the concentrations of the total forms of nutrients and ammonium were still considerably lower than before restoration. The number of summer days with microcystins and with chlorophyll *a* concentrations above the WHO limits for recreational waters decreased. However, there was a gradual decline in the physicochemical parameters of the water due to the local impacts of human use (swimming and duck and fish feeding), rather than from other external or internal loads, which were effectively controlled.

1. Introduction

Small and shallow impoundments are very important elements in the landscapes and functions of cities. They create recreational areas, support biodiversity, regulate microclimates and retain stormwater. However, the quality of the retained water often does not meet quality standards (Szklarek et al., 2015; Urbaniak et al., 2015), the expectations of water users (Wagner and Zalewski, 2011) and the requirements of healthy ecosystems (Downing, 2010; Waajen et al., 2014; Hassall and Anderson, 2015; McAndrew et al., 2016). One of the primary threats to and, consequently, key driver of the ecological quality of small urban water ecosystems is stormwater runoff, which transports pollutants from impermeable urban areas (Yu and Stone, 2010; Schwartz et al., 2017). In the case of small urban reservoirs, high catchment to reservoir surface area ratios, high pollution loads from urban catchments or rainwater drainage systems, and long water retention times make them especially susceptible to the accumulation of contaminants. This leads to the accumulation of bottom sediments in the reservoirs, which may cause secondary pollution in the summer due to the release of the internal pollutant load into the open water (Pokorny and Hauser, 2002). As a result, a decrease in dissolved oxygen and an increase in the concentrations of nitrogen and phosphorus compounds are observed. In addition, anthropogenic pressures related to the local urban infrastructure (rainwater drainage systems) and different recreational activities, such as waterfowl and fish feeding, cause further water quality degradation.

High external and internal loads of nutrients, high water temperatures in the summer, and the stagnation of water intensify eutrophication symptoms in urban reservoirs (Oberholster et al., 2006; Peretyatko et al., 2010; Waajen et al., 2014). The hydromorphological

* Corresponding author.
E-mail address: tomasz.jurczak@biol.uni.lodz.pl (T. Jurczak).

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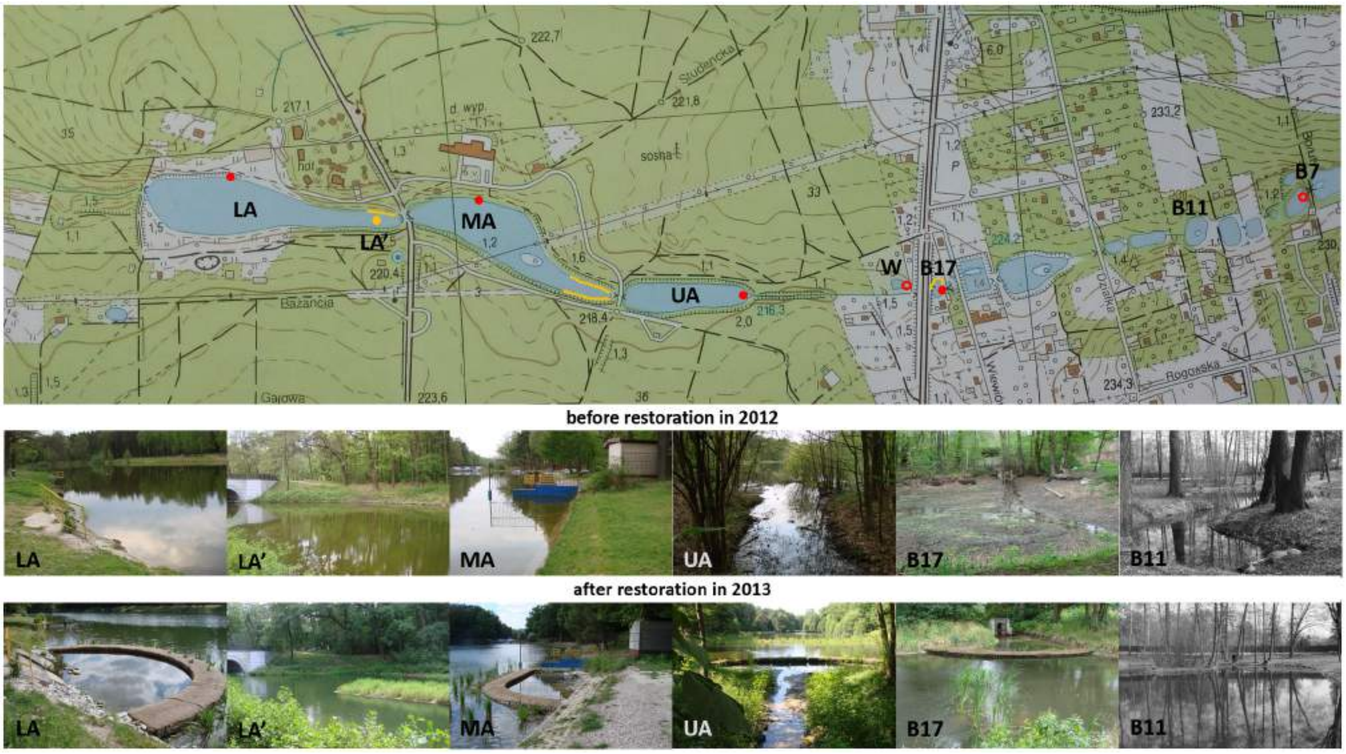

Fig. 1. Location of the project sites in the Bzura River and the Arturówek reservoirs. Red dots – SSBSs or hybrid systems; yellow dot – the floating island; yellow lines – shoreline vegetation zones; W and B7 – locations of the work described by Jurczak et al. (2018a, 2018b). Photos depict the situation at the study sites before restoration (upper photos) and after restoration (lower photos) (photos by T. Jurczak). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

and biological structures of urban ecosystems are often degraded, which diminishes their resilience. Toxic cyanobacterial blooms not only degrade biotic structure in waters but are also highly hazardous to the health, and even lives, of the people utilising these ecosystems and to wild and domestic animals (Chorus and Mur, 1999; Falconer, 2001). This phenomenon discourages city inhabitants from using reservoirs for recreational purposes, especially during the summer holiday season (Schagerl et al., 2010).

Common restoration measures that are often applied to reduce internal loading in small urban reservoirs include the mechanical removal of bottom sediments or the inactivation of phosphorus by the introduction of chemicals or bacteria into the water (e.g., Peretyatko et al., 2012; Park et al., 2016, Rosińska et al., 2017). Additionally, the aeration of the water layers above the bottom sediments is often used as a method to support good water quality (Peretyatko et al., 2009; Podsiadłowski et al., 2017). However, these methods are costly, and their effects are limited and temporary because they do not reduce the pollution sources (loads of external pollutants), they only address the impacts of the current pollution, diminishing internal loading. Management plans can be more complete and efficient by addressing the rehabilitation issue in a more comprehensive manner, simultaneously reducing the external supply of pollutants and the internal loading and regulating the biological structure in water ecosystems (Zalewski et al., 2012; Zalewski, 2014).

Such a complex restoration strategy was tested within the EU LIFE project "Ecohydrologic rehabilitation of recreational reservoirs "Arturówek" (Łódź) as a model approach to rehabilitation of urban reservoirs" (EH-REK) (LIFE08 ENV/PL/000517). The project was developed in a small urban river and a cascade of reservoirs and ponds, some of which are intensively used for recreation. The eutrophication of the reservoirs had been impacting water quality for years, in part due to the regular appearance of toxic algae blooms (Jurczak et al., 2012), restricting the reservoir use by Łódź citizens due to aesthetic and safety concerns. Therefore, the project implemented a number of simultaneous rehabilitation actions, including the removal of accumulated bottom sediments and rapidly developing vegetation and the construction of sequential sedimentation-biofiltration systems (SSBS) and hybrid systems to improve the quality of the inflowing water (Jurczak et al., 2018a; Szklarek et al., 2018). Simultaneously, multiple biotechnologies were employed, 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. The biological response of the system is presented in Jurczak et al. (2018b).

The aim of this paper is to present the extent to which the measures used for the restoration of small urban reservoirs affected the physicochemical quality of the water and the appearance of eutrophication symptoms (algae and toxic cyanobacteria blooms). We attempt also to compare the effectiveness and costs of the conventional restoration method such as bottom sediment removal, with comprehensive ecohydrological measures, such as hybrid system and SSBS. Comparison of these measures was designed to answer a question, about which approach is more effective in urban water ecosystem management: bottom sediment removal (performed once in approximately ten years) or protection of the reservoirs by the systems reducing inflow of contaminants to the reservoirs through the whole year and to the future.

2. Materials and methods

2.1. Study site

The study was conducted on the upstream stretch of the Bzura River, the western tributary to the Vistula River, with a catchment area of 7788 km^2 and a length of 166 km, of which 6.5 km of the river

Table 1
Characteristic of the Arturówek reservoirs, restoration actions implemented in 2013 during EH-REK project (LIFE08 ENV/PL/000517) and the investment and maintenance costs.

Parameter/Restoration	Lower Arturówek	Middle Arturówek	Upper Arturówek	Bzura-17	Bzura-11[1]	TOTAL
Bzura river location (km)	164.242	164.706	165.142	165.700	166.125	–
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	–
Conventional restoration effort:						
Bottom sediment removal						
Volume of sediments (m^3)	3040	2840	3548	367	–	9795
Efficiency of nutrient removal with sediment	6407 kg N 61.5 kg P	2634 kg N 70.6 kg P	2991 kg N 57.4 kg P	474.4 kg N 7.50 kg P	–	12506.4 kg N 197.0 kg P
Investment cost (€)[2]	24,883	23,238	29,032	3608	–	80,761
Annual operating cost (€)	0	0	0	0	–	0
10-year investigation and exploitation [€]	24,883	23,238	29,032	3608		80,761
20-year investigation and exploitation [€]	49,766	46,476	58,064	7216		161,522
Engineering biotechnological actions:						
Construction of SSBS and hybrid systems						
type of the system and area (m^2)	one underground separator +SSBS 150	two underground separators +SSBS100	SSBS 1200	SSBS 300	–	–
Efficiency of the SSBS or hybrid system in nutrient and TSS removal in 2014 (%)	TN (84.6) TP (80.1) NO_3^- (86.8) PO_4^{3-} (83.3) NH_4^+ (97.2) TSS (92.2)	TN (89.6) TP (83.6) NO_3^- (98.5) PO_4^{3-} (89.2) NH_4^+ (87.6) TSS (98.6)	TN (56.9) TP (57.1) NO_3^- (91.3) PO_4^{3-} (49.0) NH_4^+ (59.8) TSS (89.6)	n.a.	–	–
Annual volume of sediments in separators (m^3)	0.407	0.519	–	–		0.926
Efficiency of nutrients removal with sediments from separators per single cleaning operation	2.49 kg N 0.15 kg P	3.17 kg N 0.19 kg P	–	–		5.66 kg N 0.34 kg P
Annual volume of sediments in sedimentation zones (m^3)	5	3	8	5		21
Efficiency of nutrients removal with sediments from sedimentation zones	30.3 kg N 2.19 kg P	18.2 kg N 1.32 kg P	8.09 kg N 0.187 kg P	5.06 kg N 0.117 kg P		61.65 kg N 3.81 kg P
Area of plants in SSBS (m^2)	70	60	650	80		860
Number of plants in SSBS in 2015 (ind)	1895	2930	14,572	4007		23,404
Efficiency of nutrient removal by harvesting	0.27 kg N 0.045 kg P	0.23 kg N 0.038 kg P	2.46 kg N 0.415 kg P	0.30 kg N 0.051 kg P		3.26 kg N 0.549 kg P
Investment cost (€)[2]	6275	15,647	33,955	8052	–	63,929
Annual operating cost (€)	~400	~800	~1000	~300	–	~2500
10-year construction and exploitation [€]	10,275	23,647	43,955	11,052		88,929
20-year construction and exploitation [€]	14,275	31,647	53,955	14,052		113,929
Additional actions:						
Construction of the floating island						
Area (m^2)	100	–	–	–	–	100
Efficiency of nutrient removal by harvesting	0.38 kg N 0.064 kg P	–	–	–	–	0.38 kg N 0.064 kg P
Investment costs (€)[2]	45,923	–	–	–	–	45,923
Annual operating cost (€)	~200	–	–	–	–	~200
10-year construction and exploitation [€]	47,923					47,923
20-year construction and exploitation [€]	49,923					49,923
Construction of the shoreline vegetation zones						
Area (m^2)	200	350	–	150	–	700
Number of plants in 2015 (ind)	1164	1540		1010		3714
Efficiency of nutrient removal by harvesting	0,76 kg N 0.128 kg P	1.33 kg N 0.223 kg P	–	0.57 kg N 0,096 kg P	–	2.66 kg N 0,447 kg P
Investment costs (€)[2]	871	4777	–	1602	–	7250
Annual operating cost (€)	~100	~200	–	~100	–	~400
10-year construction and exploitation [€]	1871	6777		2602		11,250
20-year construction and exploitation [€]	2871	8777		3602		15,250
Hydrodictyon reticulatum removal						
Area (m^2) × density (m)	450 × 0.03	1,530 × 0.03	–	–	–	1,980 × 0.03
Efficiency of nutrient removal by harvesting	2.63 kg N 0.168 kg P	8.95 kg N 0.572 kg P	–	–	–	11.58 kg N 0.74 kg P
Investment costs (€)[2]	–	–	–	–	–	–
Annual operating cost (€)	174	590				764

n.a. – not analyzed, TN – total nitrogen, TP – total phosphorous, NO_3^- – nitrate, PO_4^{3-} – phosphate, NH_4^{3-} – ammonium, TSS – total suspended solids, [1] – not restored site, [2] – spent in PLN and calculated on EURO by average EURO exchange rate for the first half year of 2013 (1 Euro = 4.1783 PLN); it was assumed that 1 m^3 of sediments corresponds to 843 kg of dry weight, 1 m^2 of vegetation zone corresponds to 411 g of dry weight and 1 m^3 of H. reticulatum corresponds to 10.536 kg of dry weight.

(8.85 km^2 of the catchment area) is located within the administrative borders of the city of Łódź (Bald et al., 1999). The mean discharge of the Bzura River in the study site ranges from 0.005 to 0.028 $m^3\ s^{-1}$ (Kujawa and Kujawa, 2011). The length of the investigated river section was 3.54 km downstream from the river source, and 56.6% of this length is now covered by man-made reservoirs (with a length to width ratio greater than 2) and ponds (with a length to width ratio < 2). These reservoirs and ponds include a cascade of 17 small ponds located upstream of the recreational area and a cascade of three reservoirs used for recreation: the Lower Arturówek (LA), Middle Arturówek (MA) and Upper Arturówek (UA) (Fig. 1). The research presented in this paper includes results from the LA, MA and UA reservoirs and two of the smaller upstream ponds, Bzura-11 (B11) and Bzura-17 (B17). The characteristics of all the investigated impoundments are presented in Table 1 and locations in Fig. 1.

The recreational reservoirs and ponds have different uses and characteristics. The LA reservoir has always endured the most human pressure, as it is intensively used for recreation with a beach and swimming area. The stormwater inflow and duck feeding are additional activities that decrease the water quality. The MA reservoir is used for water sports (primarily canoeing and paddle boats) and is impacted by intensive duck feeding. This reservoir is also under pressure resulting from stormwater inflow from impermeable areas (street, parking lots and hotel roof). The UA reservoir is mainly exploited by anglers and used for fishing.

The B11 and B17 ponds are typical small, shallow, decorative landscape water bodies. These ponds are under considerable human pressure resulting from the transfer of contaminants from other upstream ponds and from the unsewered catchment. In 2011, the B11 pond was included to the monitoring programme as an unrestored site. Furthermore, B11 served as a reference site for the B17 pond (only) due to a similar size, morphology, type of catchment and usage.

Restoration activities were done in LA, MA, UA and B17 from January to June 2013 and included conventional restoration effort and engineering biotechnological actions described in Tables 1 and 2.

The response of biological indices to this activity is described in Jurczak et al. (2018b).

2.2. Sampling sites and physicochemical analyses

Between 2010 and 2016, water samples from all the impoundments were analysed for physicochemical parameters, chlorophyll *a* and microcystin concentrations. The samples were collected from April to October every two weeks in 2010, 2011 and 2014 and monthly in 2012, 2013, 2015 and 2016. The data were divided into two periods: before restoration (2010–2012) and after restoration (2014–2016). Because there were only three monitoring events in 2013, the results for this year are presented in the figures, but they were not included in the statistical analysis.

Physical parameters, including water temperature, dissolved oxygen concentration, pH and conductivity, were determined *in situ* during water sampling by using a WTW Multi 340i instrument (WTW, Weilheim, Germany). Water samples filtered with GF/C membranes were analysed with ion chromatography (Dionex ICS-1000, Sunnyvale, California, USA); cations were analyzed with an Ion Pac CS15 column (ammonium, NH_4^+), while the anion analysis was conducted with an Ion Pac AS14A column (nitrite, NO_2^-; nitrate, NO_3^-; phosphate, PO_4^{3-}). Analysis of total nitrogen (TN) was performed on unfiltered water using the persulfate digestion method (method no. 10071; HACH, 1997). The samples for the total phosphorous (TP) analysis were digested with the addition of Oxisolve® Merck reagent (Merck, Darmstadt, Germany) with the Merck MV500 Microwave Digestion System and determined using the ascorbic acid method (Golterman et al., 1978). Nutrient levels were compared with the national water quality standards (Dz U., 2016, pos. 1187), which are adopted to the EU Water Framework Directive requirements.

2.2.1. Efficiency of nutrient removal with SSBSs and hybrid systems and additional actions

The efficiency of the individual SSBSs and hybrid systems in trapping nutrients and total suspended solids (TSS) from water was tested in 2014 with 5 and 7 individual sampling series for the hybrid systems constructed respectively in LA and MA (only during rainwater events), and 23 individual sampling series for the SSBS constructed in UA (during the whole season). The first sample from each series was always

Table 2
The characteristic of restoration actions implemented in 2013 during the EH-REK project.

Treatment system	Location	Description
Conventional restoration effort:		
Bottom sediment removal	B7, B17, W, UA, MA, LA	Conventional treatment restoration measure.
Engineering biotechnological actions:		
Construction of the SSBS	B7, B17, UA,	The SSBSs consists of three zones: sedimentation zone which reduce the speed of river flow and enhance sedimentation, geochemical zone made of limestones and dolomite with subsurface flow for water filtration and phosphate adsorption and biological zone covered with vegetation with surface flow for nitrogen compound reduction (Szklarek et al., 2018)
Construction of the hybrid system	W, MA, LA	The hybrid system combines engineering, i.e., an underground separator, and biotechnological measures (SSBS; see graphical abstract), where underground separators reduce oil substances and suspended solids and the biotechnological part – suspended matter and dissolved nutrients (Jurczak et al., 2018a)
Additional actions		
Construction of the floating island	LA	Floating island/floating treatment wetland (FTW) with an area of 100 m^2 was constructed in the upper part of reservoir. It assimilated dissolved nutrients in plant tissues flowing to the reservoir.
Construction of the shoreline vegetation zones	MA, LA, B17	Shoreline vegetation with a length of 100 m to 350 m and an initial width of 1 m was established along the banks of impoundments. Mix of selected native species was planted, depending on the site depth, water fluctuation and light availability. Planting sites were chosen based on the landforms in the surrounding area (steep slopes of the reservoir margin impacted by surface erosion) or based on the hydromorphological characteristics of the reservoir (the shallowest areas with high sediment accumulation and limited fishing access). The following aquatic vegetation species were planted: *Typha angustifolia* (L.), *Carex riparia* William Curtis, *Glyceria maxima* (Hartm.) Holmb., *Iris* sp. and *Ceratophyllum demersum* (L.).
Removal of *Hydrodictyon reticulatum*	MA, LA	Removal of *Hydrodictyon reticulatum* (L.) Bory de Saint-Vincent (also called water net), was conducted after the restoration activities had been implemented. It was additional ad hoc measure, due to the intensive growth of this species in the reservoirs.
Biomanipulation	B7, B17, UA, MA, LA	Biomanipulation was applied by introducing piscivorous fish (pike *Esox lucius* (L.) and pikeperch *Sander lucioperca* (L.)) from 2013 to 2014 to enhance zooplankton development in reservoirs and enhance top-down control of phytoplankton development. Details of this actions are described in Jurczak et al. (2018b).

collected at the inlet to the SSBS or hybrid system, and the final sample from downstream of the system, after the water had flowed through the biological section. The nutrient trapping efficiency was calculated as a ratio of the inflow and outflow concentrations, and the reduction was expressed as a percentage of the inflow concentration. Additionally, during annual maintenance work amount of removed nutrients in sediment removal from underground separators and sedimentation zones of the SSBSs and hybrid systems was assessed.

The sediment subsamples were analysed for TN, TP and heavy metal content. The analysis of TN, TP and heavy metals was performed on the dry mass of the sediment, according to PN-R-04023:1996 (1996) for phosphorus and PB 49 ed. 2 from 01.02.2007 for nitrogen, and PB 21 ed. 1 from 01.05.2004 for heavy metals (Krysiak et al., 2016). The amount of nutrients removed was calculated as the volume of sediment multiplied by the nutrient content in the analysed samples.

The biomass of plants harvested from hybrid systems and SSBSs was estimated based on the inventory (number of specimens per sub-sample area) and the average weight of the aboveground parts (stems and leaves) of single plants. Similarly, the efficiency of *Hydrodictyon reticulatum* (*L.*) *Bory de Saint-Vincent* and the shoreline vegetation zones in nutrient removal was calculated. Only, due to safety considerations, vegetation from the floating island was harvested in winter, when the reservoir was frozen, and the nutrient removal efficiency was extrapolated based on the data from the shoreline vegetation zones. The subsamples were analysed for nutrient content. The analyses of the plant content of TN and TP were performed according to PB 60 ed. 2 from 15.06.2012 for nitrogen and PB 13 ed. 1 from 11.06.2004 for phosphorous (Krysiak et al., 2016) by an accredited external laboratory of the Chemical and Agricultural Station in Łódź.

2.3. Chlorophyll a and microcystin analyses

The concentration of chlorophyll *a* was measured immediately after sampling using a bbe Algae Online Analyser (AOA, Version 1.5 E1, bbe-Moldaenke Company, Kiel, Germany). The bbe AOA measurement is based on the determination of the fluorescence spectrum and the fluorescence kinetics of the algae. By analysing the interaction between chlorophyll *a* and other pigments, AOA discriminates among the four main groups of algae (green algae, cyanobacteria, diatoms and cryptophytes). This method is recognised as a reliable digital analysis of chlorophyll *a* measurements (Cagnard et al., 2006) and as a useful tool for monitoring phytoplankton community composition, particularly as an early warning system for the detection of harmful algal blooms (Richardson et al., 2010).

The microcystins (MCs) were analysed in suspended matter (intracellular). One litre water samples were filtered through Whatman GF/C filter paper immediately after sampling. MCs were extracted in 75% aqueous methanol with the sonication process in a Misonix ultrasonicator (Farmingdale, NY, USA). The extracts were then centrifuged at $11000 \times g$ for 10 min in an Eppendorf 5804 centrifuge (Hamburg, Germany). The supernatants were collected and evaporated in an SC110A Speedvac Plus (Thermo Savant, Holbrook, NY, USA). MCs were analysed with high-performance liquid chromatography (HPLC) using the method described by Jurczak et al. (2005). MCs in the cyanobacterial extracts were identified using MC-LR, MC-RR and MC-YR standard stock solutions, with their characteristic absorption spectra and retention times.

2.4. Cost analysis of the restoration measures

We compared the cost of restoration measures for bottom sediment removal, hybrid systems, SSBSs, *H. reticulatum* removal, and vegetation harvesting from the shoreline vegetation zones and the floating island. The costs were calculated per 10-year period, which is the estimated interval between the necessary sediment removal for conventional protection measures in the Arturówek reservoirs, and per 20-year period, which shows the costs from a long-term perspective. Removal of *H. reticulatum* scum was calculated as a one-time, ad hoc action, with no maintenance costs. Bottom sediment removal (dragging, transport, and deposition) was calculated as a one-time action per 10 years, with no maintenance costs within a single 10-year period. For other measures, the one-time investment cost was added to the annual maintenance costs for 10-year and 20-year periods. The investment costs included both material and construction for all zones in the SSBSs and hybrid systems, shoreline vegetation zone and floating island. The maintenance costs included the cleaning and removal of sediment from the underground separators, sedimentation zones, annual vegetation removal, as well as necessary infrastructure maintenance.

2.5. Statistical analyses

To identify significant differences between the physicochemical and biological parameters before (2010–2012) and after (2014–2016) restoration, the general linear models (GLMs) was used separately for each ecosystem. To control inter-annual variation in the study parameters, the effect of year was included as fixed factor and nested within each period. Analyses were performed with JMP v.13 (SAS Institute Inc., Cary, NC, USA). All statistical analyses were conducted using log (x + 1) transformed data.

3. Results

3.1. Comparison of the water quality before and after restoration in three recreational reservoirs

3.1.1. Changes in physical parameters

In the LA, no differences were observed in the physical parameters of the water before and after restoration (Fig. 2). In MA, only a reduction in conductivity (Fig. 3) from 340 $\mu S\ cm^{-1}$ to 309 $\mu S\ cm^{-1}$ was significant ($p < 0.001$) (Table 3). Similarly, in the UA, a significant reduction in conductivity was observed ($p < 0.001$) (Table 3, Fig. 4). The concentration of dissolved oxygen as well as conductivity were below 4 $mg\ L^{-1}$ and 800 $\mu S\ cm^{-1}$ respectively, the thresholds for good water quality (Dz U., 2016, pos. 1187).

3.1.2. Changes in nutrient concentration

The average TN concentration in 2010–2012 was relatively high in all three recreational reservoirs. TN ranged from 1.97 $mg\ L^{-1}$ in the LA reservoir to 1.46 $mg\ L^{-1}$ in the MA reservoir (Table 3). After restoration, TN concentration decreased significantly in all three reservoirs (Figs. 2–4) and ranged from 0.92 $mg\ L^{-1}$ in UA to 1.03 $mg\ L^{-1}$ in LA. The threshold for good water quality was 2 $mg\ L^{-1}$ according to Polish regulations (Dz U., 2016, pos. 1187). Only in 2012 the concentration of TN was above 2 $mg\ L^{-1}$ for all three reservoirs. The maximum concentration of TN ranged from 8.8 $mg\ L^{-1}$ in MA to 9.8 $mg\ L^{-1}$ in LA. After restoration they were reduced and they did not exceed 4.10–4.50 $mg\ L^{-1}$ in all three reservoirs.

The average concentration of ammonium before restoration ranged from 0.07 to 0.15 $mg\ L^{-1}$. After restoration, they were from 4 to 10 times lower than pre-restoration values, and these decreases were statistically significant (Table 3). Similarly, nitrate values decreased two times after restoration, but these changes were not statistically significant. In contrast, nitrite concentrations increased from 0.02 to 0.03 $mg\ L^{-1}$ before restoration to 0.03–0.04 $mg\ L^{-1}$ after restoration in all reservoirs, but this trend was also not significant.

The average TP concentrations increased slightly in LA and MA after restoration from 0.16 $mg\ L^{-1}$ and 0.11 $mg\ L^{-1}$ to 0.18 $mg\ L^{-1}$ and 0.17 $mg\ L^{-1}$, respectively. While the UA experienced an decrease 0.05 $mg\ L^{-1}$ to 0.18 $mg\ L^{-1}$ after restoration, but in all cases, the observed changes were not statistically significant. In 2015 and 2016 concentrations of TP in all reservoirs were below 0.12 $mg\ L^{-1}$, the threshold for a good water quality (Dz U., 2016, pos. 1187). The

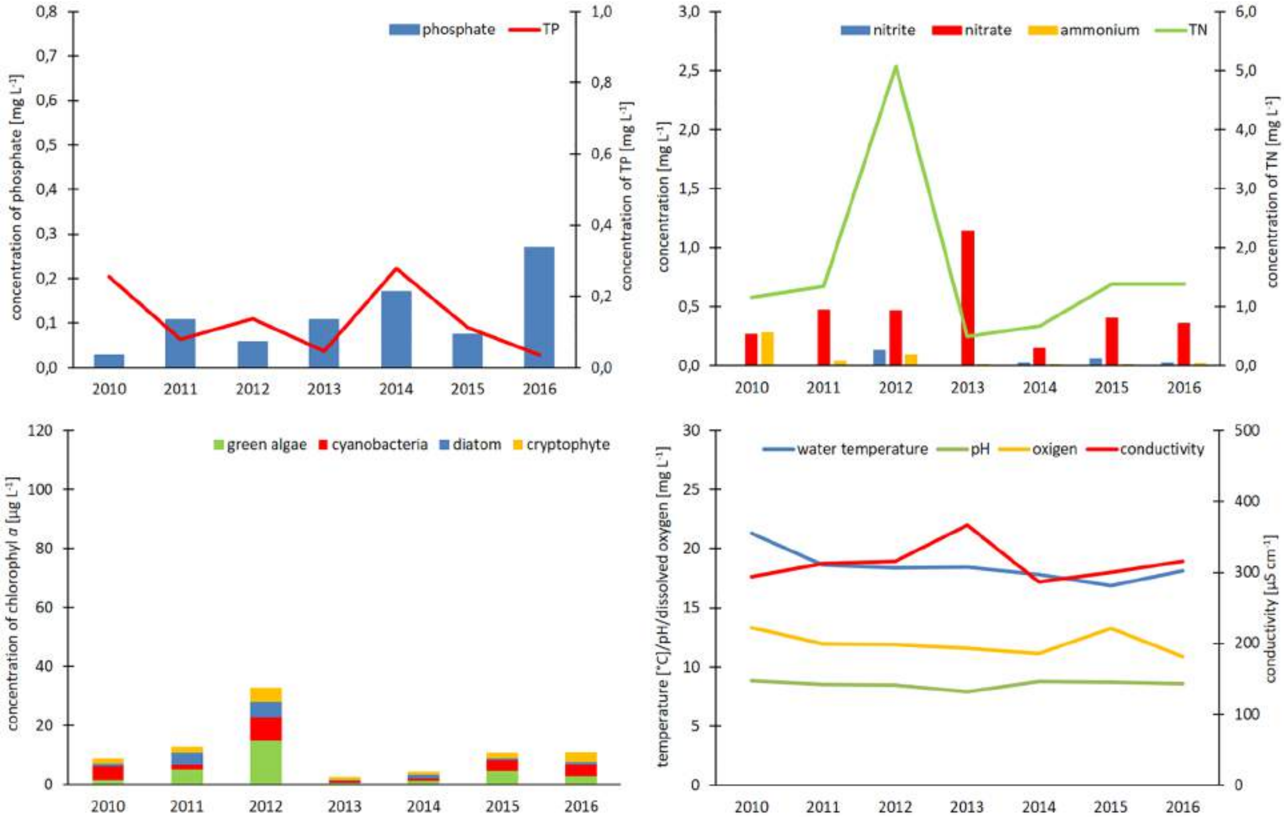

Fig. 2. The annual average changes in chlorophyll *a* concentrations and physicochemical parameters in the LA reservoir in 2010–2012 (before restoration) and 2013–2016 (after restoration).

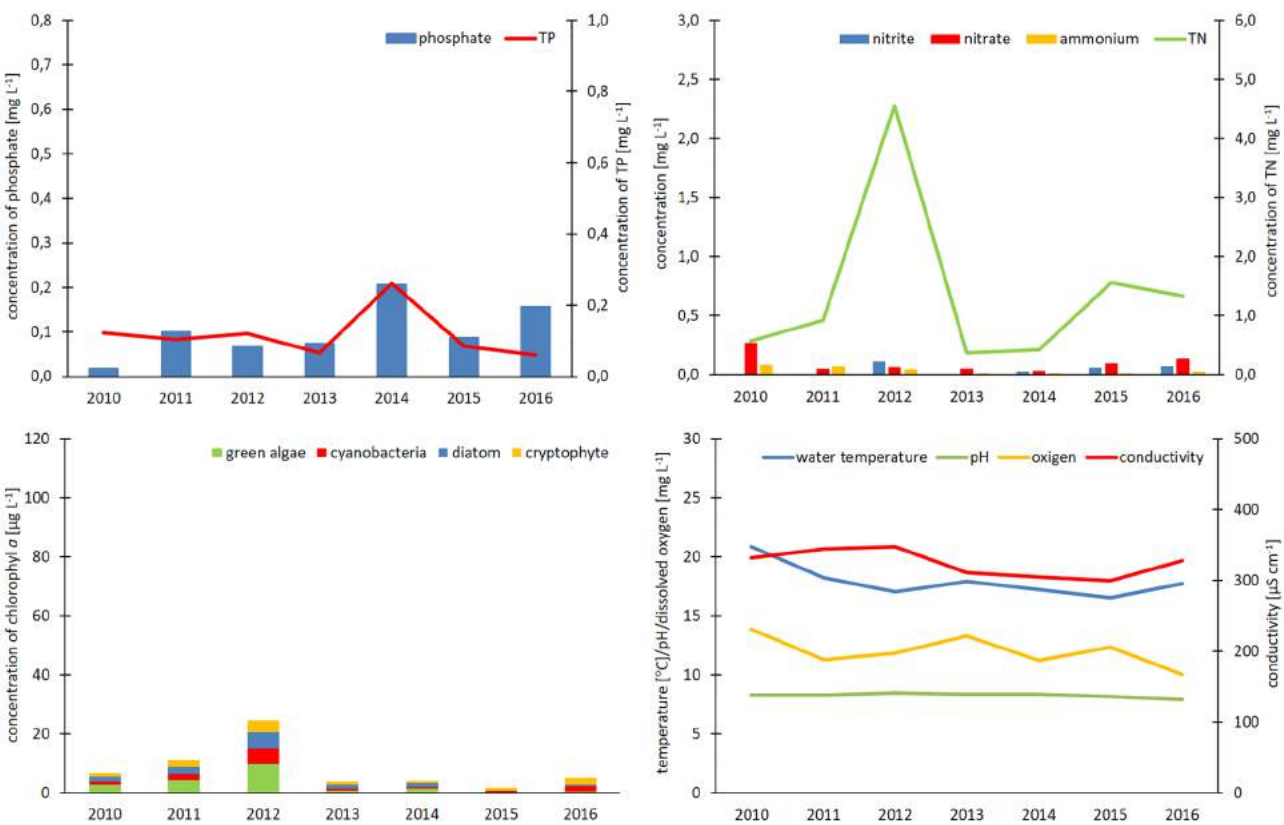

Fig. 3. The annual average changes in chlorophyll *a* concentrations and physicochemical parameters in the MA reservoir in 2010–2012 (before restoration) and 2013–2016 (after restoration).

Table 3
Comparison of the water quality before (2010–2012) and after (2014–2016) restoration in three recreational reservoirs: LA, MA, UA (the differences in all study parameters between the two periods were tested with the general linear models (GLMs) separately for each pond).

parameter	LA						MA		
	3y before 2010	2y before 2011	1y before 2012	1y after2014	2y after2015	3y after2016	3y before 2010	2y before 2011	1y before 2012
Physical parameters									
Temperature [°C]	21.3 ± 1.8 19.7 ± 1.0 n.s.	18.7 ± 1.2	18.4 ± 2.7	17.8 ± 1.2 17.7 ± 1.0	16.9 ± 2.9	18.1 ± 2.5	20.8 ± 1.7 19.1 ± 1.0 n.s.	18.3 ± 1.2	17.0 ± 2.3
Conductivity [µS cm^{-1}]	293.5 ± 4.3 305.4 ± 3.4 n.s.	312.9 ± 2.9	315.2 ± 11.2	286.6 ± 7.2 297.3 ± 5.6	300.1 ± 11.8	315.9 ± 13.1	332.3 ± 4.0 340.0 ± 3.0 $F_{(1,54)}$ = 18.20p < 0.001	344.5 ± 4.8	347.0 ± 6.1
pH	8.85 ± 0.11 8.64 ± 0.08 n.s.	8.52 ± 0.12	8.45 ± 0.15	8.78 ± 0.14 8.72 ± 0.12	8.72 ± 0.41	8.61 ± 0.19	8.31 ± 0.05 8.34 ± 0.06 n.s.	8.31 ± 0.14	8.49 0.07
dissolved oxygen [mg L^{-1}]	13.35 ± 0.69 12.52 ± 0.48 n.s.	11.98 ± 0.76	11.89 ± 1.14	11.15 ± 0.49 11.61 ± 0.36	13.24 ± 0.65	10.89 ± 0.70	13.87 ± 0.84 12.45 ± 0.54 n.s.	11.31 ± 0.79	11.84 ± 0.94
Chemical parameters									
TN [mg L^{-1}]	1.15 ± 0.28 1.97 ± 0.41 $F_{(1,54)}$ = 8.17p = 0.006	1.35 ± 0.19	5.07 ± 1.59	0.67 ± 0.15 1.03 ± 0.19	1.38 ± 0.47	1.39 ± 0.55	0.57 ± 0.12 1.46 ± 0.32 $F_{(1,54)}$ = 8.63p = 0.005	0.92 ± 0.11	4.55 ± 0.99
Ammonium [mg L^{-1}]	0.28 ± 0.19 0.151 ± 0.080 $F_{(1,54)}$ = 5.12p = 0.029	0.05 ± 0.01	0.10 ± 0.04	0.01 ± 0.00 0.014 ± 0.003	0.01 ± 0.01	0.02 ± 0.01	0.08 ± 0.03 0.070 ± 0.016 $F_{(1,54)}$ = 7.63p = 0.008	0.07 ± 0.02	0.04 ± 0.02
Nitrate [mg L^{-1}]	0.27 ± 0.07 0.39 ± 0.06 n.s.	0.48 ± 0.09	0.47 ± 0.18	0.15 ± 0.07 0.27 ± 0.08	0.41 ± 0.25	0.36 ± 0.21	0.27 ± 0.20 0.14 ± 0.08 n.s.	0.05 ± 0.02	0.07 ± 0.06
Nitrite [mg L^{-1}]	0.006 ± 0.002 0.028 ± 0.011 n.s.	0.002 ± 0.001	0.130 ± 0.040	0.024 ± 0.006 0.034 ± 0.009	0.063 ± 0.034	0.023 ± 0.008	0.004 ± 0.003 0.023 ± 0.009 n.s.	0.001 ± 0.000	0.113 ± 0.027
TP [mg L^{-1}]	0.26 ± 0.09 0.16 ± 0.04 n.s.	0.08 ± 0.01	0.14 ± 0.04	0.28 ± 0.11 0.18 ± 0.06	0.11 ± 0.02	0.04 ± 0.01	0.12 ± 0.03 0.11 ± 0.01 n.s.	0.10 ± 0.01	0.12 ± 0.03
Phosphate [mg L^{-1}]	0.03 ± 0.01 0.07 ± 0.01 $F_{(1,54)}$ = 15.37p < 0.001	0.11 ± 0.03	0.06 ± 0.03	0.17 ± 0.04 0.17 ± 0.03	0.08 ± 0.04	0.27 ± 0.05	0.02 ± 0.00 0.06 ± 0.01 $F_{(1,54)}$ = 7.53p = 0.008	0.10 ± 0.03	0.07 ± 0.03
± ± N:P	10:1 ± 3:1 201:1 ± 184:1 n.a.	19:1 ± 2:1	1010:1 ± 979:1	6:1 ± 1:1 17:1 ± 4:1	18:1 ± 10:1	38:1 ± 9:1	7:1 ± 2:1 34:1 ± 21:1 n.a.	10:1 ± 1:1	145:1 ± 106:1
Biological parameters									
total chlorophyll*a* [µg L^{-1}]	8.78 ± 2.04 14.92 ± 2.46 $F_{(1,54)}$ = 7.47p = 0.008	12.74 ± 3.12	32.95 ± 6.32	4.42 ± 0.95 7.67 ± 1.34	10.82 ± 4.41	11.03 ± 2.35	6.63 ± 1.14 11.82 ± 1.38 $F_{(1,54)}$ = 80.62p < 0.001	11.16 ± 0.94	24.46 ± 2.64

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Table 3 (*continued*)

parameter	LA						MA		
	3y before 2010	2y before 2011	1y before 2012	1y after2014	2y after2015	3y after2016	3y before 2010	2y before 2011	1y before 2012
cyanobacteria chlorophyll *a* [μg L^{-1}]	4.79 ± 1.77 4.12 ± 0.99 n.s.	1.62 ± 0.42	8.09 ± 3.06	0.73 ± 0.16 2.28 ± 0.66	3.78 ± 2.32	3.90 ± 1.23	0.90 ± 0.13 2.22 ± 0.42 $F_{(1,54)}$ = 13.60p < 0.001	2.18 ± 0.50	5.16 ± 1.40
MCs [μg L^{-1}]	4.31 ± 1.71 2.34 ± 0.76 n.a.	3.90 ± 0.24	2.89 ± 1.11	0.65 ± 0.02 1.52 ± 0.77	0.67 ± 3.01	1.52 ± 0.70	0.00 ± 0.00 0.08 ± 0.04 n.a.	0.01 ± 0.01	0.39 ± 0.20

parameter	MA			UA					
	1y after2014	2y after2015	3y after2016	3y before 2010	2y before 2011	1y before 2012	1y after2014	2y after2015	3y after2016
Physical parameters									
Temperature [°C]	17.2 ± 1.2 17.2 ± 1.0 n.s.	16.6 ± 2.9	17.7 ± 2.5	21.0 ± 1.9 19.3 ± 1.0 n.s.	18.4 ± 1.3	17.3 ± 2.5	17.7 ± 1.2 17.4 ± 1.1	16.4 ± 3.1	17.8 ± 2.7
Conductivity [μS cm^{-1}]	304.3 ± 9.8 308.9 ± 6.4 $F_{(1,54)}$ = 18.20p < 0.001	299.4 ± 12.7	327.4 ± 12.3	337.5 ± 4.2 345.4 ± 4.5 $F_{(1,54)}$ = 75.75p < 0.001	346.1 ± 5.4	360.8 ± 18.4	247.5 ± 9.2 270.3 ± 6.9	285.7 ± 9.9	300.3 ± 11.3
pH	8.35 ± 0.08 8.20 ± 0.10 n.s.	8.16 ± 0.33	7.95 ± 0.16	8.10 ± 0.07 8.24 ± 0.11 n.s	8.06 ± 0.17	8.92 ± 0.30	8.59 ± 0.15 8.51 ± 0.11	8.47 ± 0.36	8.37 ± 0.15
dissolved oxygen [mg L^{-1}]	11.22 ± 0.39 11.22 ± 0.33 n.s.	12.36 ± 0.71	10.06 ± 0.74	10.56 ± 0.60 10.11 ± 0.55 n.s.	8.51 ± 0.68	12.62 ± 1.77	9.93 ± 0.58 10.17 ± 0.38	11.63 ± 0.70	9.21 ± 0.66
Chemical parameters									
TN [mg L^{-1}]	0.42 ± 0.07 0.93 ± 0.18 $F_{(1,54)}$ = 8.63p = 0.005	1.56 ± 0.58	1.33 ± 0.38	0.70 ± 0.17 1.77 ± 0.38 $F_{(1,54)}$ = 11.70p = 0.001	1.31 ± 0.11	5.07 ± 1.32	0.50 ± 0.11 0.92 ± 0.18	1.33 ± 0.40	1.34 ± 0.56
Ammonium [mg L^{-1}]	0.01 ± 0.00 0.016 ± 0.003 $F_{(1,54)}$ = 7.63p = 0.008	0.01 ± 0.00	0.02 ± 0.01	0.04 ± 0.01 0.076 ± 0.022 $F_{(1,54)}$ = 24.97p < 0.001	0.04 ± 0.01	0.23 ± 0.10	0.01 ± 0.00 0.010 ± 0.002	0.01 ± 0.00	0.02 ± 0.01
Nitrate [mg L^{-1}]	0.03 ± 0.00 0.07 ± 0.02 n.s.	0.09 ± 0.07	0.14 ± 0.06	0.19 ± 0.16 0.13 ± 0.07 n.s.	0.07 ± 0.03	0.12 ± 0.11	0.03 ± 0.01 0.05 ± 0.02	0.02 ± 0.01	0.13 ± 0.09
Nitrite [mg L^{-1}]	0.025 ± 0.005 0.044 ± 0.012 n.s.	0.058 ± 0.031	0.067 ± 0.037	0.005 ± 0.003 0.022 ± 0.008 n.s.	0.002 ± 0.001	0.102 ± 0.027	0.025 ± 0.007 0.040 ± 0.012	0.062 ± 0.039	0.048 ± 0.034
TP [mg L^{-1}]	0.26 ± 0.10 0.17 ± 0.05 n.s.	0.08 ± 0.02	0.06 ± 0.04	0.17 ± 0.06 0.23 ± 0.05 n.s.	0.14 ± 0.02	0.54 ± 0.17	0.29 ± 0.11 0.18 ± 0.06	0.10 ± 0.02	0.02 ± 0.01
Phosphate [mg L^{-1}]	0.21 ± 0.05 0.16 ± 0.03 $F_{(1,54)}$ = 7.53p = 0.008	0.09 ± 0.05	0.16 ± 0.03	0.01 ± 0.00 0.06 ± 0.02 $F_{(1,54)}$ = 7.47p = 0.008	0.11 ± 0.04	0.07 ± 0.03	0.12 ± 0.02 0.13 ± 0.02	0.09 ± 0.04	0.21 ± 0.05
± ± N:P	2:1 ± 0:1 22:1 ± 6:1 n.a.	30:1 ± 12:1	55:1 ± 17:1	6:1 ± 2:1 112:1 ± 103:1 n.a.	11:1 ± 2:1	559:1 ± 550:1	3:1 ± 1:1 22:1 ± 7:1	17:1 ± 7:1	64:1 ± 23:1
Biological parameters									
total chlorophyll*a* [μg L^{-1}]	4.03 ± 2.18 3.74 ± 1.07 $F_{(1,54)}$ = 80.62p < 0.001	1.83 ± 0.19	5.06 ± 1.23	6.60 ± 1.42 30.80 ± 8.77 $F_{(1,54)}$ = 51.13p < 0.001	27.11 ± 5.67	91.21 ± 36.95	6.77 ± 1.64 6.04 ± 0.88	5.25 ± 1.49	5.38 ± 1.02

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Table 3 (*continued*)

parameter	MA			UA					
	1y after2014	2y after2015	3y after2016	3y before 2010	2y before 2011	1y before 2012	1y after2014	2y after2015	3y after2016
cyanobacteria chlorophyll *a* [µg L^{-1}]	0.62 ± 0.11 0.87 ± 0.14 $F_{(1,54)} = 13.60 p < 0.001$	0.54 ± 0.12	1.68 ± 0.45	0.48 ± 0.12 12.95 ± 8.13 $F_{(1,54)} = 8.67 p = 0.005$	3.32 ± 0.90	60.85 ± 40.20	2.51 ± 1.19 1.75 ± 0.58	0.67 ± 0.23	1.32 ± 0.33
MCs [µg L^{-1}]	0.00 ± 0.00 0.01 ± 0.01 n.a.	0.01 ± 0.01	0.04 ± 0.02	0.00 ± 0.00 2.13 ± 1.81 n.a.	0.02 ± 0.01	11.33 ± 9.34	0.00 ± 0.00 0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00

± n.s. – not significant, n.a. – not analyzed, F – values of used test, threshold for good status of water: ± TN ≤ 2 mg L^{-1}, TP ≤ 0.12 mg L^{-1}, conductivity ≤ 800 µS cm^{-1}, dissolved oxygen ≤ 4 mg L^{-1}. (Dz. U. 2016, pos. 1187).

maximum concentration of TP increased in all reservoirs, from 1.03 mg L^{-1}, 0.36 mg L^{-1} and 1.19 mg L^{-1} before restoration to 1.37 mg L^{-1}, 1.51 mg L^{-1} and 1.57 mg L^{-1} after restoration (all sampled in 2014) for LA, MA and UA, respectively. The annual average concentration of phosphates changed significantly ($p < 0.01$) in all three reservoirs, increasing by 0.07–0.1 mg L^{-1} after restoration (Table 3). Similarly, the maximum phosphate concentrations after restoration were from 2 to 2.5 times higher for LA and MA, respectively.

As a result of the above changes, the N:P ratio in all three reservoirs considerably decreased initially after restoration to an exceptionally low level and then steadily increased during the subsequent years. In 2016, the N:P ratios reached 38:1, 55:1 and 64:1 in LA, MA and UA, respectively (Table 3).

3.1.3. Changes in chlorophyll *a* concentration

During the post-restoration period, the total chlorophyll *a* concentrations were significantly reduced ($p < 0.01$) in all three reservoirs from 14.92 µg L^{-1}, 11.82 µg L^{-1} and 30.80 µg L^{-1} to 7.67 µg L^{-1}, 3.74 µg L^{-1} and 6.04 µg L^{-1} after restoration for LA, MA and UA, respectively. The maximum concentration of chlorophyll *a* before restoration was observed in 2012 in UA and amounted to 271.1 µg L^{-1}. In LA and MA it was 45.85 µg L^{-1} and 31.8 µg L^{-1}, respectively. After restoration we observed reduction of maximum chlorophyll *a* concentration to 27.5 µg L^{-1} (2015) and 24.2 µg L^{-1} (2014), but only in LA and UA, respectively.

The concentration of cyanobacterial chlorophyll *a* decreased in all three reservoirs after the restoration efforts, but the changes were statistically significant only in MA and UA ($p < 0.005$). In the MA, the concentration of cyanobacterial chlorophyll *a* decreased after restoration from 2.22 µg L^{-1} to 0.87 µg L^{-1}. In turn, the average concentration of cyanobacterial chlorophyll *a* in UA was more than seven times lower (1.75 µg L^{-1}) after restoration, with maximum 16.1 µg L^{-1} detected in 2014.

3.1.4. Changes in microcystin concentrations

MC-RR, MC-YR and MC-LR were detected each year in the LA before restoration and in 2011 and 2012 in MA and UA. In the LA MCs were detected in 23 of the 32 sampling series, with concentrations ranging from 0.02 µg L^{-1} to 21.5 µg L^{-1}. After restoration, toxins were detected less frequently (in only 13 of the 31 sampling series), with maximum concentrations ranging from 0.05 µg L^{-1} to 4.54 µg L^{-1}. Incidentally, one high concentration was detected in September 2015 (21.9 µg L^{-1}). There were no MCs detected in 2013, and they only appeared during one sampling at the end of August in 2014, with a low concentration of 0.24 µg L^{-1}.

In the MA reservoir, MCs were detected in 5 of the 32 samples before restoration, with maximum concentrations of 0.06 µg L^{-1} in 2011 and 1.07 µg L^{-1} in 2012. After restoration, MC-YR was detected only once, in May 2015, with a maximum concentration of 0.04 µg L^{-1}. MC-YR and MC-LR were detected only 3 times, in the summer of 2016, with a maximum concentration of 0.13 µg L^{-1}.

Before restoration in the UA reservoir, MC-RR, MC-YR and MC-LR were detected only four times: twice in 2011 (with a maximum concentration of 0.178 µg L^{-1}) and twice in 2012 (with a maximum concentration of 57.2 µg L^{-1}). After restoration, only one of the 31 sampling series analysed over 4 years indicated the presence of MC-YR, at a low concentration of 0,07 µg L^{-1} in May 2015.

3.2. Comparison of the water quality before and after restoration of Bzura-17 and in the unrestored Bzura-11 pond

3.2.1. Changes in physical parameters

Significant differences were detected for temperature and dissolved oxygen in the B17 pond between pre-restoration and post-restoration periods and only for conductivity in the B11 pond (Table 4, Fig. 6). The oxygen concentration significantly increased after restoration efforts

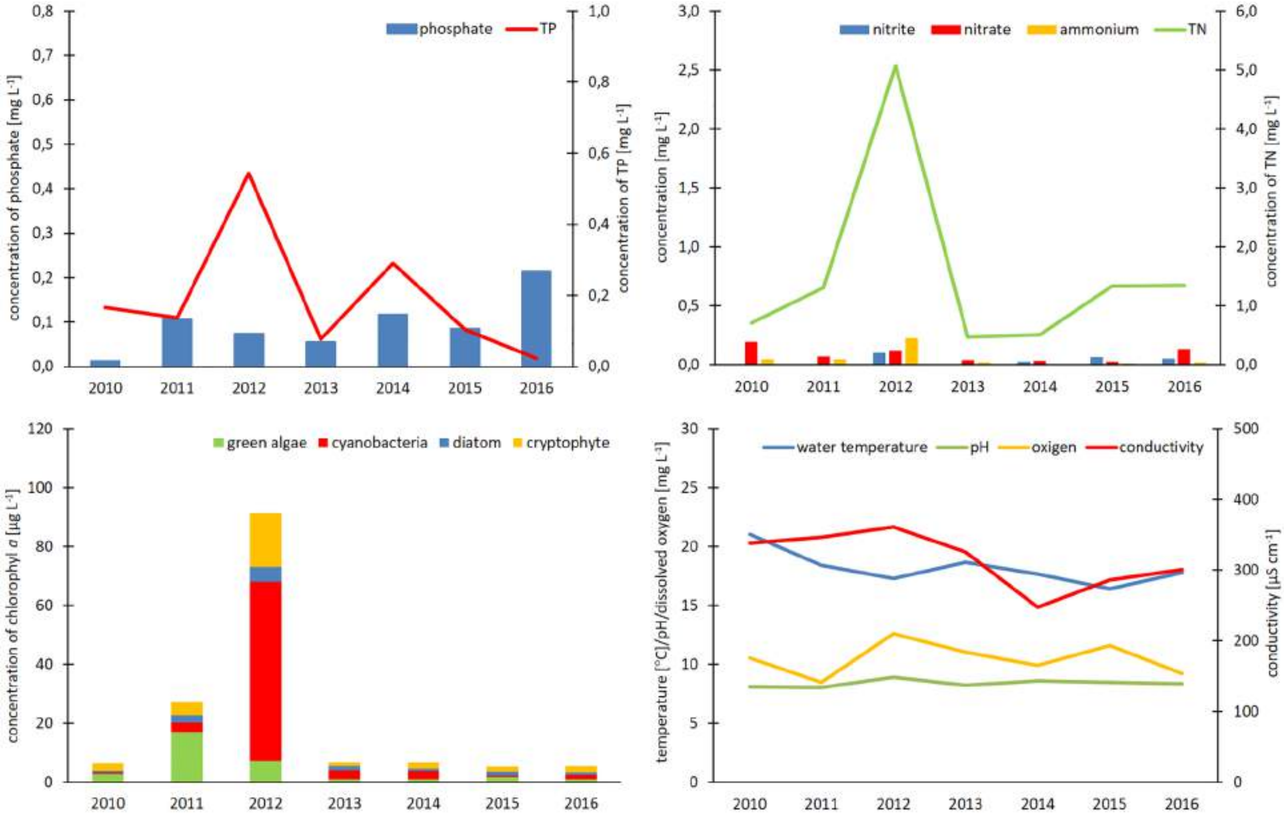

Fig. 4. The annual average changes in chlorophyll *a* concentrations and physicochemical parameters in the UA reservoir in 2010–2012 (before restoration) and 2013–2016 (after restoration).

from 4.5 mg L^{-1} to 6.8 mg L^{-1} in B17 (Table 4). Simultaneously, the dissolved oxygen concentration decreased in B11 from 7.5 mg L^{-1} in 2011–2012 to 5.6 mg L^{-1} in 2014–2016. In both ponds, conductivity increased, which differentiates these impoundments from the reservoirs described in the section 3.1 (Fig. 6).

3.2.2. Changes in nutrient concentration

In the restored pond, the average TN concentrations before and after restoration were comparable (slightly < 2 mg L^{-1}), and the differences were not statistically significant (Table 4). In the unrestored B11 pond, a systematic increase in the TN concentration has been observed every year since 2011. The concentration increased from 1.48 mg L^{-1} in 2011 to 4.86 mg L^{-1} in 2016.

The ammonium concentration in the B17 pond significantly ($p < 0.001$) decreased after restoration, when the average concentration declined to 50% of the values recorded from 2010 to 2012 (0.56 mg L^{-1}). In the B11 pond, ammonium values significantly ($p < 0.001$) increased and were finally more than 20 times higher in 2016 (2.89 mg L^{-1}) than in 2011 and 2012 (Table 4). Mean nitrate values were similar in the pre- and post-restoration periods (0.21 mg L^{-1}) in the B17 pond. In the B11 pond, the nitrate concentrations recorded from 2014 to 2016 doubled in relation to the previous values but remained low (below 0.1 mg L^{-1}), and the difference had no statistical significance. In 2014–2016 nitrite concentrations were slightly lower in the B17 pond and slightly higher in the B11 pond then in previously years and amounted to 0.024 mg L^{-1} and 0.028 mg L^{-1}, respectively. However, these changes were statistically not significant.

The average TP concentration in B17 doubled from 0.22 mg L^{-1} in 2010–2012 to 0.44 mg L^{-1} after restoration, but the increase was not statistically significant (Table 4). The annual average concentration of TP after restoration decreased each year, from 0.62 mg L^{-1} in 2014 to 0.22 mg L^{-1} in 2016 (Table 4). A similar trend was observed in not-restored B11 pond, where average concentration of TP decreased each year, from 0.85 mg L^{-1} in 2014 to 0.38 mg L^{-1} in 2016. But the average TP concentration in this period was 4 times higher than in 2011–2012 and amounted to 0.63 mg L^{-1} (Table 4). We observed increase of phosphate concertation in B17 from 0.17 mg L^{-1} in pre-restoration period to 0.28 mg L^{-1} after restoration. In not-restored B11 pond concentration of this parameter in 2014–2016 was as high as 0.51 mg L^{-1} and was more than 3 times higher than in 2011–2012.

In 2010–2012 the N:P ratio was 9:1 in B17, with a maximum value of 14:1 in 2012 (Table 4). After restoration, the average N:P ratio decreased to 4:1 and 6:1 in 2014 and 2015, respectively. But in 2016 we observed increased of N:P ratio to 81:1. In B11 pond the N:P ratio decreased from 11:1 in 2011 and 2012 to 3:1 in 2014, than increased to 9:1 and 27:1 in 2015 and 2016, respectively (Table 4).

3.2.3. Changes in chlorophyll a and microcystin concentrations

The average concentration of total chlorophyll *a* in 2014–2016 was two-fold less in B17 than in B11. But in B17 as well as in B11 approximately the 50% reduction of this parameter was observed each year from 2014 (Table 4). The cyanobacterial chlorophyll *a* was presented in the minor concentration in B17 and B11 (Fig. 5 and Fig. 6), but increased significantly in B11 from 1.68 µg L^{-1} in 2011–2012 to 4.06 µg L^{-1} in 2014–2016. In B17 it was only 2.59 µg L^{-1} in 2014–2016. No MCs were detected in either pond.

3.3. Efficiency and cost of the restoration measures

3.3.1. Nutrient removal with SSBSs and hybrid systems

The SSBS constructed at the inflow of the river to the UA reservoir reduced the loads of TN and TP from the river by approximately 57% and the supply of dissolved nutrients and TSS from the river by 49% and 91.3%, respectively (Table 1). The hybrid systems implemented in the LA and MA reservoirs reduced the nutrients and TSS inflowing into

Table 4
Comparison of the water quality before (2010–2012) and after (2014–2016) restoration in the Bzura-17 and unrestored Bzura-11 ponds (the differences in all study parameters between the two periods were tested with the general linear models (GLMs) separately for each pond).

parameter	B17						B11					
	3y before 2010	2y before 2011	1y before 2012	1y after2014	2y after2015	3y after2016	3y before 2010	2y before 2011	1y before 2012	1y after2014	2y after2015	3y after2016
Physical parameters:												
Temperature [°C]	16.4 ± 1.2 16.6 ± 0.7 $F_{(1,54)} = 5{,}32 p = 0.025$	16.6 ± 0.9	16.9 ± 1.8	14.5 ± 0.9 14.1 ± 0.8	13.5 ± 1.9	13.8 ± 1.8	n.d. 16.1 ± 0.8 n.s.	16.6 ± 1.1	15.1 ± 2.0	16.2 ± 1.1 15.8 ± 0.9	15.0 ± 2.5	15.8 ± 2.2
Conductivity [μS cm^{-1}]	339.7 ± 3.20 371.5 ± 15.9 n.s.	362.7 ± 5.50	459.3 ± 78.3	385.6 ± 2.3 378.7 ± 3.5	365.9 ± 6.8	377.9 ± 11.4	n.d. 323.7 ± 5.6 $F_{(1,42)} = 36.32 p < 0.001$	339.6 ± 6.2	289.3 7.3	339.8 3.1 346.0 ± 3.0	350.9 ± 7.6	353.6 ± 7.1
± pH	7.23 ± 0.02 7.41 ± 0.07 n.s.	7.30 ± 0.08	8.06 ± 0.17	7.43 ± 0.05 7.38 ± 0.06	7.40 ± 0.21	7.25 ± 0.06	n.d. 7.60 ± 0.10 n.s.	7.53 ± 0.16	7.75 ± 0.19	7.49 ± 0.09 7.40 ± 0.06	7.32 ± 0.20	7.32 ± 0.05
dissolved oxygen [mg L^{-1}]	2.02 ± 0.21 4.50 ± 0.54 $F_{(1,54)} = 5.54 p = 0.022$	5.08 ± 0.65	8.62 ± 0.89	6.63 ± 0.79 6.84 ± 0.61	9.07 ± 1.33	5.01 ± 1.20	n.d. 7.50 ± 0.82 n.s.	7.17 ± 1.27	8.22 ± 1.93	5.97 ± 1.11 5.62 ± 0.65	6.13 ± 1.29	4.43 ± 1.03
Chemical parameters:												
TN ± [mg L^{-1}]	1.15 ± 0.20 1.96 ± 0.36 n.s.	1.44 ± 0.22	4.85 ± 1.31	1.64 ± 0.21 1.95 ± 0.45	0.79 ± 0.25	3.74 ± 1.75	n.d. 1.86 ± 0.30 $F_{(1,42)} = 4.28 p = 0.045$	1.48 ± 0.17	2.67 ± 0.19	2.15 ± 0.32 2.90 ± 0.38	2.45 ± 0.54	4.86 ± 1.15
ammonium[mg L^{-1}]	0.79 ± 0.16 0.561 ± 0.081 $F_{(1,54)} = 19.67 p < 0.001$	0.31 ± 0.08	0.62 ± 0.06	0.43 ± 0.15 0.237 ± 0.076	0.07 ± 0.06	0.05 ± 0.02	n.d. 0.120 ± 0.032 $F_{(1,42)} = 18.45 p < 0.001$	0.13 ± 0.05	0.10 ± 0.05	0.95 ± 0.32 1.560 ± 0.339	1.46 ± 0.73	2.89 ± 0.94
nitrate[mg L^{-1}]	0.18 ± 0.07 0.21 ± 0.03 n.s.	0.24 ± 0.04	0.20 ± 0.04	0.23 ± 0.10 0.21 ± 0.08	0.02 ± 0.01	0.34 ± 0.26	n.d. 0.03 ± 0.00 n.s.	0.03 ± 0.01	0.01 ± 0.01	0.06 ± 0.02 0.06 ± 0.01	0.03 ± 0.01	0.09 ± 0.04
nitrite[mg L^{-1}]	0.006 ± 0.001 0.026 ± 0.008 n.s.	0.016 ± 0.003	0.093 ± 0.027	0.022 ± 0.005 0.024 ± 0.005	0.036 ± 0.016	0.016 ± 0.008	n.d. 0.023 ± 0.007 n.s.	0.001 ± 0.000	0.071 ± 0.018	0.029 ± 0.006 0.028 ± 0.006	0.036 ± 0.020	0.017 ± 0.015
TP[mg L^{-1}]	0.20 ± 0.02 0.22 ± 0.02 n.s.	0.16 ± 0.01	0.41 ± 0.07	0.62 ± 0.10 0.44 ± 0.07	0.28 ± 0.07	0.22 ± 0.17	n.d. 0.16 ± 0.01 $F_{(1,42)} = 24.64 p < 0.001$	0.13 ± 0.01	0.21 ± 0.04	0.85 ± 0.11 0.63 ± 0.07	0.43 ± 0.09	0.38 ± 0.12
phosphate[mg L^{-1}]	0.13 ± 0.03 0.17 ± 0.03 n.s.	0.19 ± 0.04	0.22 ± 0.08	0.31 ± 0.06 0.28 ± 0.05	0.09 ± 0.03	0.42 ± 0.11	n.d. 0.15 ± 0.02 $F_{(1,42)} = 18.63 p < 0.001$	0.16 ± 0.04	0.12 ± 0.04	0.42 ± 0.05 0.51 ± 0.07	0.53 ± 0.23	0.69 ± 0.14
N:P	6:1 ± 1:1 9:1 ± 1:1 n.a.	9:1 ± 1:1	14:1 ± 4:1	4:1 ± 1:1 23:1 ± 13:1	6:1 ± 4:1	81:1 ± 53:1	n.d. 11:1 ± 1:1 n.a.	11:1 ± 1:1	11:1 ± 3:1	3:1 ± 0:1 10:1 ± 4:1	9:1 ± 5:1	27:1 ± 15:1
Biological parameters:												
total chlorophyll *a*[μg L^{-1}]	n.d. n.d. n.a.	n.d.	n.d.	29.55 ± 3.95 19.94 ± 2.90	14.50 ± 5.46	6.15 ± 1.69	n.d. 27.55 ± 2.74 n.s.	25.18 ± 3.76	32.70 ± 7.55	52.93 ± 8.60 37.58 ± 5.67	24.75 ± 5.53	19.71 ± 11.44
cyanobacteria chlorophyll *a*[μg L^{-1}]	n.d. n.d. n.a.	n.d.	n.d.	2.68 ± 0.49 2.59 ± 0.60	3.89 ± 2.32	1.13 ± 0.29	n.d. 1.68 ± 0.18 $F_{(1,42)} = 10.58 p < 0.001$	1.81 ± 0.25	1.40 ± 0.47	3.84 ± 0.70 4.06 ± 0.55	5.57 ± 1.48	2.98 ± 1.09
MCs [μg l^{-1}]	n.d. n.d. n.a.	n.d.	n.d.	0.00 ± 0.00 0.00 ± 0.00	0.01 ± 0.01	0.00 ± 0.00	n.d. n.d. n.a.	n.d.	n.d.	0.00 ± 0.00 0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00

n.s. – not significant, n.d. – no data, n.a. – not analyzed, F – values of used test, threshold for good status of water: TN ≤ 2 mg L^{-1}, TP ≤ 0.12 mg L^{-1}, conductivity ≤ 800 μS cm^{-1}, dissolved oxygen ≤ 4 mg L^{-1}. (Dz U., 2016, pos. 1187).

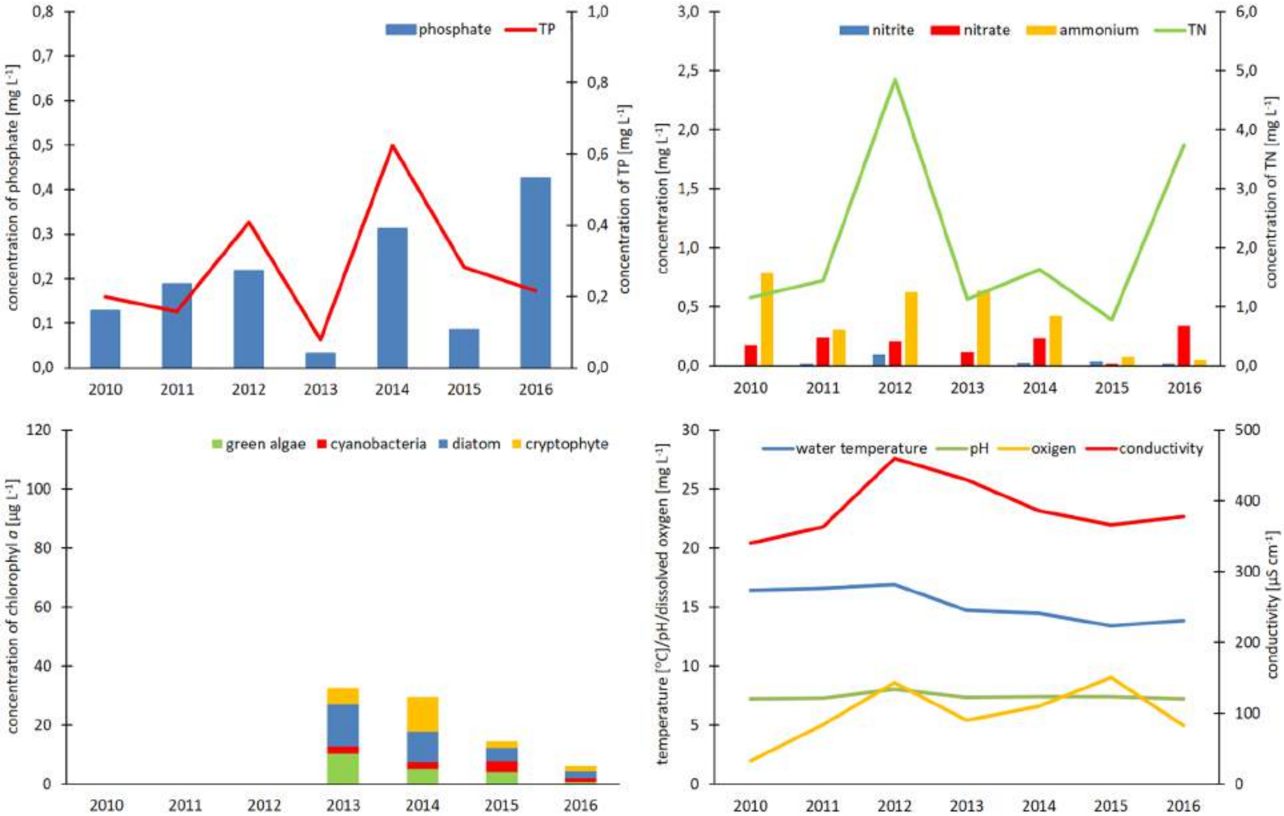

Fig. 5. The annual average changes in chlorophyll *a* concentrations and physicochemical parameters in the B17 reservoir in 2010–2012 (before restoration) and 2013–2016 (after restoration).

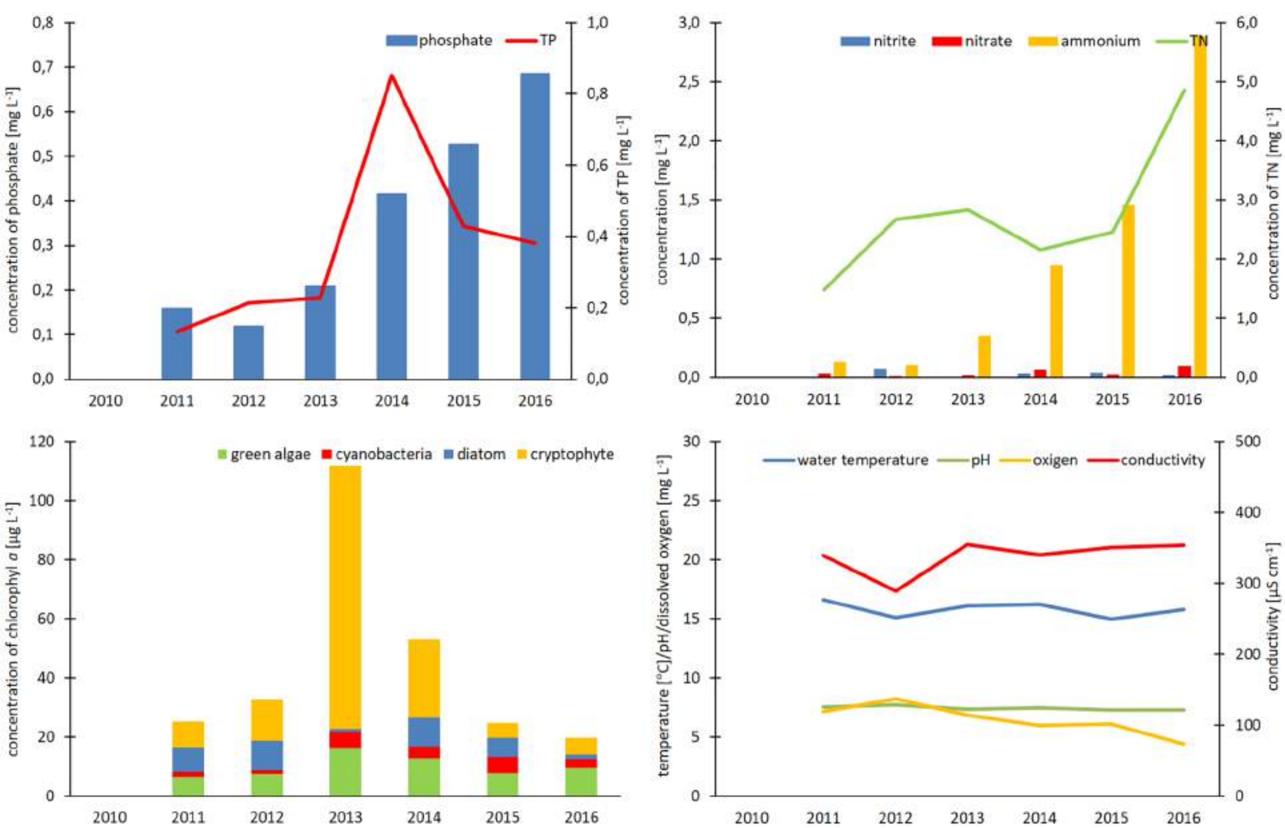

Fig. 6. The annual average changes in chlorophyll *a* concentrations and physicochemical parameters in 2010–2016 in the B11 reservoir as a reference site.

Table 5
Effectiveness of nutrient reduction by sediment removal from the reservoirs, in comparison to the other restoration actions.

Parameter	TN	TP	Pb	Cd	Cu	Cr	Ni	Zn	Fe	Mn
	mg kg^{-1} dry weight									
Bottom sediment in LA[1]	2500	24,0	11,3	0,2	8,5	4,6	5,6	36,5	5131	70,2
Bottom sediment in MA[1]	1100	29,5	19,4	0,2	15,9	8,2	9,4	73,6	6741	62,7
Bottom sediment in UA[1]	1000	19,2	7,4	0,2	4,4	4,0	4,3	21,2	3504	49,2
Underground separators in hybrid systems[2]	7233	435	93,4	0,89	107	37,8	32,8	805	19,901	321
Sedimentation zones in hybrid systems[3]	7200	520	79,5	1,52	105,8	35,2	32,4	454	23,375	341
Sedimentation zones in SSBS[3]	1200	27,7	17,4	0,2	10,8	4,3	5,6	50,5	3245	74,6
Vegetation in SSBS, hybrid systems and shorline zones[4]	9218	1553	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Hydrodictyon reticulatum[5]	18,517	1183	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.

n.a. – not analyzed, [1] – samples collected in February 2013 during investigation works, [2] – samples collected in April and June 2014 during operational works, [3] – samples collected in December 2013 during operational works, [4] – samples collected in September 2014 during operational works, [5] – Hydrodictyon reticulatum samples collected in May and August 2014 from surface water of MA and LA.

the reservoirs from the stormwater drainage systems by more than 80% and 90%, respectively (Table 1).

The underground separators constructed in the hybrid systems retained 0.4–0.5 m^3 of sediment transported from impermeable urban areas, corresponding to 343 kg (LA) and 438 kg (MA) of sediment dry weight. In our project, the operation of separators twice per year allowed for the removal of at least 5.0 kg of nitrogen and over 0.30 kg of phosphorous with the sediment removed from the underground separators (Table 1). In addition, the separators eliminated heavy metals several times more effectively (Table 5) than removal of bottom sediment from the reservoir.

The vegetation zones in the SSBSs and hybrid systems assimilated nutrients in plant tissue during the growing season. Approximately 3350 plants were planted in 2013 to construct these systems. In 2015, in the two hybrid systems in LA and MA and the two SSBSs in UA and B17, we identified 7 times more plants for a total number of 23,404 plants. Removal of the plants from these four systems in the autumn of 2015 allowed for the removal of 3.26 kg of nitrogen and 0.55 kg of phosphorous (Table 1). The vegetation zone eliminated 3.8 g N m^{-2} and 0.64 g P m^{-2} per year.

The cost of construction and maintenance of two hybrid systems in LA and MA (underground separators, sedimentation zone and vegetation zone) was €33922 for a 10-year period. We estimate that in the perspective of a 20-year period, this cost will increase to €45922. The cost of construction and maintenance for two SSBSs in UA and B17 (sedimentation zone and vegetation zone) was €55007 over a 10-year period and is estimating to €68007 for a 20-year period (Table 1).

3.3.2. Nutrient removal with bottom sediment

Bottom sediment was removed from the LA, MA, and UA reservoirs and the B17 pond in volumes of 3040 m^3, 2840 m^3, 3548 m^3 and 367 m^3, respectively. One cubic meter of fresh sediment corresponded to approximately 843 kg of sediment dry weight. The concentrations of nitrogen and phosphorous removed with these sediments ranged from 1.0 to 2.5 g for N and from 19.2 to 29.5 mg for P per kilogram of dry weight (Table 5). The highest amounts, 6407 kg N and 61.5 kg P, were removed with the sediment from LA. Although the sediment volume removed from UA was 15% higher than that the sediment removed from LA, the amount of the removed nutrients was lower – only 2991 kg N and 57.4 kg P were eliminated. The highest amount of phosphorous, 70.6 kg P, was removed with the sediment from MA (Table 1).

The removal cost for bottom sediments exceeded €80000. No further maintenance costs were associated with this action. Usually, for a 10-year period, the sediments should be removed again, which at least doubles the costs for a long-term perspective (Table 1).

3.3.3. Vegetation harvesting from the shoreline zones and floating island

In 2015, we identified 3714 plants in the three vegetated shorelines planted at LA and B17 in 2013, and at MA in 2014. The autumn harvesting of vegetation from the shoreline vegetation zones removed over 2.5 kg of nitrogen and almost 0.5 kg of phosphorous (Table 1). Approximately 1000 plants were planted on the floating island, an area of 100 m^2, which was constructed in LA in 2013. In all, 0.38 kg of N and 0.06 kg of P was estimated to have been removed from the floating island. The cost of construction and maintenance of the shoreline vegetation was €11250 from a 10-year perspective and is estimating to €15250 from a 20-year perspective (Table 1).

3.3.4. Hydrodictyon reticulatum removal

In 2014, *H. reticulatum* was removed from 450 m^2 and 1530 m^2 areas in LA and MA, respectively. The removal of *H. reticulatum* from LA and MA eliminated 18.5 g and 1.18 g of nitrogen and phosphorous, respectively, for each kilogram of dry biomass of these green algae (Table 5). In total, by removing 59.4 m^3 of algae, 11.6 kg of nitrogen and 0.74 kg of phosphorous were extracted from the reservoirs (Table 1). The cost of the one-time removal of *H. reticulatum* was €764.

4. Discussion

4.1. Effect of restoration measures on nutrient availability

As a result of the restoration efforts undertaken in this study, the water quality in the reservoirs improved, particularly when considering the most hazardous aspect, phytoplankton growth (Fig. 7). Although the concentrations of phosphate and nitrite increased after restoration in all inpoundments, the TN, nitrate and ammonium concentrations were significantly lower, especially in three recreational reservoirs. Before the restoration measures, high concentrations of ammonium were detected in all the impoundments. After restoration, these concentrations were reduced 7-fold in LA, MA and UA. Interestingly, in the same period, concentrations of ammonium were only reduced two-fold in B17, while these levels increased ten-fold in the unrestored pond B11. High concentrations of ammonium in these small and shady ponds might have been caused by mineralisation and ammonification occurring in the bottom sediment (Vymazal, 2007; Tomaszek and Gruca-Rokosz, 2007; Jurczak et al., 2018c).

The increased concentrations of phosphate could have resulted from bottom sediment removal. Exposing the remaining sediment to oxygen during drying of the impoundment increases the mineralisation rate. Consequently, this exposure increases the internal load and the reservoir productivity after re-flooding. This process is well known and commonly used in pond aquaculture (Knoesche et al., 2000). Therefore, Dunalska et al. (2015) suggested that phosphorous inactivation should be considered a second stage of the restoration process, in addition to supporting actions, such as biomanipulation. For this purpose, the researchers recommended dosing water with chemicals, e.g., iron sulfate and magnesium chloride (Huser et al., 2016). However in the case of small, shallow and highly polymictic recreational reservoirs, aluminium sulfate may damage *Microcystis* cells and induce microcystin release (Han et al., 2016), posing a health risk to swimmers and wildlife (Chen et al., 2009, Amado and Monserrat, 2010).

Ecological lake management often aims to restore a clear water state with submerged or benthic macrophyte vegetation. After sediment removal in the northern basin of Lake Kraenepoel (Belgium), Van Wichelen and co-authors (2007) observed the intensive growth of submerged macrophytes, which covered almost 40% of the lake bottom and 35% of the water volume. In the Arturówek reservoirs, the sediment removal and re-flooding was associated with high phosphate concentrations and contributed to the intensive development of primary producers, unfortunately not only macrophytes but also *H. reticulatum* scum (Fig. 8). This result seems to be a natural ecosystem reaction, as *H. reticulatum* prefers conditions of long water retention times and high phosphate concentrations (e.g., Lelková and Poulíčková, 2004; Wojtal-Frankiewicz and Frankiewicz, 2011). Still, the appearance of this green alga in a large biomass initially seemed like a challenge. However, in the end, *H. reticulatum* was relatively easily to mechanically remove from the reservoir, and with its biomass, a fraction of available phosphate and nitrate was efficiently extracted from the ecosystem. The removal of *H. reticulatum* (Fig. 8) not only reduced the phosphate concentration in the water in the third quarter of 2014 but also, in the end, increased the N:P ratio (Table 3). We suppose that a low N:P ratio in 2014 was one of the reasons for the weaker cyanobacteria and could have additionally promoted the development of *H. reticulatum*, as a low N:P ratio is optimal for its growth (Hawes and Smith, 1993). The results confirm the thesis by Jeppesen and co-author (2007) that nitrogen plays a more significant role in freshwater ecosystem restoration than is usually anticipated.

4.2. Effect of restoration measures on the chlorophyll a and microcystin concentrations

Urban impoundments may enrich municipal recreational offers and provide unique entertainment opportunities. However, low water quality and degraded hydromorphological characteristics may also limit these functions. For years, due to the significant pollution inflow from the suburbs, the reservoirs in Arturówek have faced problems of poor water quality and summer cyanobacterial blooms.

The problem of cyanobacterial blooms in urban reservoirs is not uncommon and has been reported by several authors (Waajen et al., 2014, Genuário et al., 2016). Before restoration in LA, MA and UA concentration for chlorophyll *a*, exceeding the WHO (2003) threshold ($10\,\mu g\,L^{-1}$) for a low human risk, was detected in 14, 17 and 19 sampling series, out of a series of 32 samples in each reservoir. After restoration, this threshold concentration appeared only in 9, 1 and 4 samplings out of a series of 31 samples in each reservoir, respectively.

The maximum concentration of chlorophyll *a* before restoration, was observed in UA in 2012, where the total chlorophyll *a* concentration reached $271.1\,\mu g\,L^{-1}$, with concentrations of MCs as high as $57.2\,\mu g\,L^{-1}$. It was most likely caused by the discharge of water from the pond located above the UA reservoir during the spring. However, such a large cyanobacterial bloom did not occur in MA or LA in 2012. This shows that the reservoirs, although connected, function to some extend independently from one another, and the bloom does not necessary transfer downstream. After restoration, microcystins were observed in UA only once and at a safe concentration of $0.07\,\mu g\,L^{-1}$. In LA, the reservoir used mainly by swimmers, toxins were also detected less frequently with maximum concentrations ranging from $0.05\,\mu g\,L^{-1}$ to $4.54\,\mu g\,L^{-1}$. An exceptionally high chlorophyll *a* concentration with cyanobacteria presence appeared only once, in LA in 2015. This instance may have been the result of a portable toilet thrown into the reservoir by vandals on the 30th of July.

In the post-restoration period, a minor increase in the chlorophyll *a* concentration was observed only in 2015 and 2016 in LA and in 2016 in MA. This may suggest a slow return of these ecosystems to the state recorded before restoration. Interestingly, such an increase was not observed in the most upstream reservoir (UA), which is directly supplied by the river. This suggests that the decrease in the water quality of the most downstream reservoir (LA) was not caused by the inflow of pollutants transported by the river or rainwater collectors, which were protected by SSBSs and hybrid systems, but resulted from other reasons. Hypothetically, this might be the result of extensive recreational use of the reservoirs and/or extended retention time due to drought reduced flow, during extremely dry and warm summers, which occurred from 2014 to 2016 in Poland. However analysis of the effect of climatic and hydrological conditions on the effectiveness of restoration has not been the subject of this study.

Fig. 7. Long-term effects of the ecohydrological restoration of the Arturówek reservoirs on water quality improvements: toxic cyanobacterial blooms before restoration (upper row) and water quality improvements after completed restoration measures (lower row).

Fig. 8. Occurrence of *Hydrodictyon reticulatum* and macrophytes in Arturówek reservoirs during the first year following restoration (lower row), as a result of the sediment removal in 2013 (upper row) (photos by T. Jurczak).

4.3. Cost and efficiency of nitrogen and phosphorous removal

4.3.1. Removal of nutrients with sediment

Before restoration efforts, the bottom sediments from the Arturówek reservoirs were removed in 2000 by the Łódź City Government. The sediment thickness in UA in 2013, when it was dredged again, reached up to 40 cm and 3548 m^3. In total, 9795 m^3 of bottom sediment was dredged from three reservoirs and B17 pond. The recommended period before the next dredging would be 10 years for this case. But we estimate that limiting the inflow of pollutants through the construction of SSBS and hybrid systems and other activities will extend this time to 20 years.

A literature review shows that the amount of nitrogen and phosphorous in bottom sediment is highly variable. In the Arturówek reservoirs, the amounts ranged from 1.0 to 2.5 g for TN and from 19.2 to 29.5 mg for TP per each kilogram of dry weight (Table 5). Studies conducted by Waajen et al. (2014) on three reservoirs, Dongen Pond (2500 m^2), Eindhoven Pond (6500 m^2) and Heesch Pond (1600 m^2) in the Netherlands, indicated that sediment contained 0.26 mg, 0.43 mg and 0.11 mg of phosphorous in one gram of dry weight, respectively. These values were 10 times higher than those of the sediment removed from the Arturówek reservoirs (0.024 mg P g^{-1} dry weight) and almost equivalent to those of sediment removed from the separators (0.435 mg P g^{-1} dry weight). This finding indicates in our case that the separator-removed sediment was a rich source of phosphorous and proves the efficiency of the separators as a part of the purifying systems (Jurczak et al., 2018a).

Bottom sediment removal is a very efficient and widely accepted, but a costly and often short-term, method of water quality improvement in small reservoirs. The sediment removal cost in the project was relatively high (€80761), compared to the investment cost of all other measures implemented in the project (€117102). Still, in routine practice, the total cost of sediment removal can even be as much as twice the cost of the amount spent during the project. In this case, avoiding transportation and utilisation and local use of the sediments for other purposes (land levelling), allowed to reduce total costs of the operation. Furthermore, sediment removal cannot be considered a restoration or rehabilitation measure *per se*. This method instantly removes the effects of previous long-term sedimentation, but sediment removal does not protect a reservoir from continuous degradation and future nutrient loads. Additionally, this method often activates nutrients that were previously buried under the removed sediment, thus exposing them to mineralisation during the removal operation. The constant accumulation of nutrients transported to the reservoir by rivers and stormwater outflows results in gradually worsening water quality, so that the reservoirs degrade very quickly and recreation restrictions have to be applied. Therefore, in spite of investing considerable funds, reservoirs operators may face a situation in which they may encounter algal or even cyanobacterial blooms, leading to low aesthetic values, health risks, city beach closures in summer, and low user satisfaction, even a short time after investment (Steffensen, 2008).

The hybrid systems designed in our study aimed to protect the reservoirs against continuous loading. These systems effectively reduced nutrient inflow to LA and MA directly from the stormwater outflows (from 80.1% to 98.5%), similar corresponds to the results from Szklarek et al. (2018) and Jurczak et al. (2018a). The underground separators were the most effective elements for TSS and nutrient removal in the hybrid systems. However, further treatment stages (a sedimentation zone and biofiltration zones) also fulfilled an important role. Although the contaminants concentrations removed therein were smaller to those removed by the separators, there were able to assimilate dissolved and easily available nutrients. Plants, as natural nutrient competitors with algae and cyanobacteria assimilate nutrients. They also stabilise bottom sediments, minimise resuspension and internal loading, and provide habitats for zooplankton, fish, amphibains, insects and birds. Higher biodiversity increases ecosystem resistance to stress and enhances ecosystem services (Steudel et al., 2012). Normally, the effectiveness of nitrogen and phosphorus removal depends on the inflow loading to wetlands, and aboveground N and P standing stock values can vary from 0.6 to88 g N m^{-2} and from 0.1 to19 g P m^{-2} per year (Vymazal, 2007). We assume that the efficiency of the nutrient removal by vegetation in the hybrid systems, as well as in the SSBSs, shoreline zones and floating islands, will increase each year, following the establishment of a more stable vegetation zone (Rozema et al., 2016).

The two SSBSs also effectively reduced inflowing nutrients and TSS (from 49% to 91.3% for the SSBSs). Studies conducted by Dierberg (1989) indicated that a removal treatment, including the stormwater-detention-filtration of wetlands at Maginnis arm in Lake Jackson, cost 40,530 USD (c.a. €32424) per ha. The system, comprised of a detention basin (163000 m^3), filtration field (18 ha) and an artificial marsh (2.5 ha), removed 95% of the TSS and 76% and 90% of the TN and TP, respectively, under normal flow conditions. In our case, the cost of construction and 10 years of maintenance for the two hybrid systems and the two SSBSs was €88929, which when combined is approximately 10% higher than a one-time sediment removal (€80761). From a 20-year perspective, the cost is 30% lower. An additional sediment removal after the next 10 years of the reservoirs siltation, double the total costs to €161552, while in the case of hybrid systems and SSBSs, the total costs increase only by 27% to €113929. This calculation assumes at least a 20-year utilisation of these systems, which is yet to occur. However, the benefit of implementing protective measures over a conventional approach is that the reservoirs maintain good water quality, with high clarity and aesthetic value, are safe in terms of the appearance of toxic cyanobacterial blooms, and can be used for swimming and other forms of recreation.

4.3.2. Removal of nutrients with vegetation

Planting and subsequent harvesting of vegetation is often used for water quality protection (Verhofstad et al., 2017). In our work, the efficiency and costs differed for different vegetation types (Iamchaturapatr et al., 2007, Kumwimba et al., 2017).

The construction of a 100 m^2 floating island was a very expensive investment and, in our study, the cost was very high compared to the efficiency. Nevertheless, several studies show that in locations where other measures are impossible to install, a floating island seems to be a well-functioning tool. Lynch et al. (2015) showed reductions of 40% of the initial nitrogen and 48% of the initial phosphorus concentrations in a mesocosm experiment. McAndrew et al. (2016) showed that a 50 m^2 floating island, constructed in the stormwater Mason Pond (area of 7100 m^2) in Virginia, effectively removed 191 g of nitrogen from the pond. In an experiment in North Carolina on two retention ponds (areas of 0.36 ha and 0.05 ha), Winston et al. (2013) constructed 12 and 4 floating islands, respectively, with an area of 23 m^2 each, which reduced TN and TP concentrations by 36–59% and 36–57%, respectively.

The cost of construction and maintenance of shoreline vegetation was comparatively low from a 10-year perspective and only increased insignificantly over the next 10 years. However, considering the amount of N and P removed, the harvesting of the shoreline vegetation zone was not a very efficient measure. Still, we assume that the cost efficiency of the shoreline vegetation will improve over time with increasing biomass, yield and, consequently, N and P assimilation (Gachango et al., 2015, Kumwimba et al., 2017). The shoreline vegetation also had other ecological functions, such as providing habitat for zooplankton and reservoir fauna (Chesnes et al., 2011, Herrmann, 2012).

4.4. Integrated system from a design and engineering perspective

The conventional measures for reservoir rehabilitation require significant technical interventions, constrains the reservoirs functioning and significant requires expanse to restore good water quality. Unconventional approach to water management, especially those including regulation of ecological processes, may cost more in the initial stage (Chen et al., 2014), however are more beneficial in the long term perspective. In our study, the conventional restoration measures were only 10% less and 30% more costly than comprehensive engineering biotechnological actions and their 10-year and 20-year maintenance, respectively. However, the systems which are limiting contaminant inflow to reservoirs extend the time in between sediment removal from the reservoirs reducing these costs in a longer perspective, and help to maintain good water quality for a long time. The reservoirs can be constantly used for recreation, therefore the ecological effects of the restoration and ecosystem services provided are much higher.

The SSBS and hybrid systems integrate engineering measures with ecological biotechnologies, which increases the efficiency of stormwater purification. If maintained properly and systematically, they provide long-term prevention of eutrophication and siltation, instead of treatment of the already degraded system. The systems can be improved by adjusting the size of zones to the loads. They proved to be efficient in spite of their relatively small size and surface area, therefore they can be used successfully in cities, where land is expensive or availability is limited. Additionally, they provide habitats for biodiversity, making the space more friendly to nature and people and resistant to stressors. These kinds of systems are being now implemented as an example of nature-based solution for city adaptation to climate change, for example in Radom city (Poland) in the framework of the LIFE14 CCA/PL/000101 project.

5. Conclusions

Based on the results obtained in this study, we conclude the following:

1. A combination of conventional and innovative restoration measures considerably improved the chemical water quality in a cascade of three urban recreational reservoirs. Reductions in TN and ammonium and establishment of a favourable N:P ratio were observed in all reservoirs. A significant increase in the oxygen concentration and a decrease in the ammonium concentration were observed in the B17 pond.
2. Bottom sediment removal increased the phosphate level in the water column and resulted in the growth of *H. reticulatum* scum. However, harvesting the scum allowed for the quick removal of P and N that had accumulated in the algal tissue, resulting in rapid water quality improvement.
3. A decrease in the frequency of cyanobacterial bloom occurrences in recreational reservoirs and lower MC concentrations were observed.
4. The construction cost of the SSBSs and hybrid systems for reducing nutrient loads to the reservoirs was lower than that of sediment removal. Sediment removal is costly (€80761) and only has the temporary effect of removing already accumulated pollution. SSBSs and hybrid systems have long-term effects, protecting the reservoirs from inflowing pollution and reducing the accumulation rate at a much lower investment cost (€63929). In 10-year perspective, the total cost of construction and maintenance of SSBSs and hybrid systems is only 10% higher than that of sediment removal.
5. Four years after restoration was completed, a slow decline in water quality was observed in the most downstream (LA and MA) reservoirs, where the chlorophyll *a* concentration started to increase, most likely due to extensive recreational use of the reservoirs.
6. After restoration, the N:P ratio initially (2014) decreased to below 10:1 and then increased in 2016 from 38:1 and to 64:1, which can stimulate growth of cyanobacteria. However, the low concentration of phosphorus restricted cyanobacteria appearance compared to the pre-restoration period. Implementation of post-restoration measures focusing on efficient nitrogen removal should be considered to maintain high water quality for an extended period.

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References

Amado, L.L., Monserrat, J.M., 2010. Oxidative stress generation by microcystins in aquatic animals: Why and how. Environ. Int. 36, 226–235. https://doi.org/10.1016/j.envint.2009.10.010.

Bald, K., Krakowska, E., Karski, K., 1999. Program rewitalizacji doliny rzeki Bzury. Zarząd Miasta Zgierza. [Program for revitalization of the Bzura river valley. The City of Zgierz], Zgierz 1999 (in Polish).

Cagnard, O., Boudin, I., Lemoigne, I., Cartnick, K., 2006. Assessment of emerging optic sensors (fluoroprobes) for algae on-line monitoring. pp. 1–10. Water Quality Technology Conference (WQTC) Proceedings. American Water Works Association, Denverss.

Chen, C., Meurk, C., Chen, J., Lv, M., Wen, Z., Jiang, Y., Wu, S., 2014. Restoration design for Three Gorges Reservoir shore lands, combining Chinese traditional agro-ecological knowledge with landscape ecological analysis. Ecol. Eng. 71, 584–597. https://doi.org/10.1016/j.ecoleng.2014.07.008.

Chen, J., Xie, P., Li, L., Xu, J., 2009. First identification of the hepatotoxic microcystins in the serum of a chronically exposed human population together with indication of hepatocellular damage. Toxicol. Sci. 108, 81–89. https://doi.org/10.1093/toxsci/kfp009.

Chesnes, T., Duncan, S., Swick, K., Jabaly, C., 2011. Biodiversity of submerged aquatic vegetation in Lake Worth Cove, a protected region of Lake Worth Lagoon, Florida. Biodiversity 12 (2), 90–96. https://doi.org/10.1080/14888386.2011.599705.

Chorus, I., Mur, L., 1999. Preventive measures. E& FN Spon, London, pp. 235–273.

Dierberg, F.E., 1989. Lake management techniques in costs and water quality effects. Environ. Manage. 13 (6), 729–742.

Downing, J.A., 2010. Emerging global role of small lakes and ponds: little things mean a lot. Limnetica 29, 9–24.

Dunalska, J.A., Grochowska, J., Wiśniewski, G., Napiórkowska-Krzebietke, A., 2015. Can we restore badly degraded urban lakes? Ecol. Eng. 82, 432–441. https://doi.org/10.1016/j.ecoleng.2015.05.037.

Dz. U. 2016, pos. 1187. Rozporządzenie Ministra Środowiska z dnia 21 lipca 2016 r. w sprawie sposobu klasyfikacji stanu jednolitych części wód powierzchniowych oraz środowiskowych norm jakości dla substancji priorytetowych [Journal of Laws 2016, item 1187. Regulation of the Minister of the Environment of July 21, 2016 on the method of classification of the state of surface water bodies and environmental quality standards for priority substances] (in Polish).

Falconer, I.R., 2001. Toxic cyanobacteria bloom problems in Australian waters: Risks and impacts on human health. Phycologia 40, 228–233. https://doi.org/10.2216/i0031-8884-40-3-228.1.

Gachango, F.G., Pedersen, S.M., Kjaergaard, C., 2015. Cost-effectiveness analysis of surface flow constructed wetlands (SFCW) for nutrient reduction in drainage discharge from agricultural fields in denmark. Environ. Manag. 56, 1478–1486. https://doi.org/10.1007/s00267-015-0585-y.

Genuário, D.B., Lorenzi, A.S., Agujaro, L.F., de Lima Isaac, R., de Paiva Azevedo, M.T., Neto, R.C., Fiore, M.F., 2016. Cyanobacterial community and microcystin production in a recreational reservoir with constant *Microcystis* blooms. Hydrobiologia 779, 105–125. https://doi.org/10.1007/s10750-016-2802-y.

Golterman, H.L., Clymo, R.S., Ohstand, M.A., 1978. Methods for Physical and Chemical Analysis of Freshwater. Scientific Publication, Londres pp. 214.

HACH Water Analysis Handbook, HACH Company, 1997 pp. 1309.

Han, J., Jeon, B.-S., Park, H.-D., 2016. Microcystin release and *Microcystis* cell damage mechanism by alum treatment with long-term and large dose as in-lake treatment. J. Environ. Sci. Health A 51 (6), 455–462. https://doi.org/10.1080/10934529.2015.1128708.

Hassall, Ch., Anderson, S., 2015. Stormwater ponds can contain comparable biodiversity to unmanaged wetlands in urban areas. Hydrobiologia 745, 137–149. https://doi.org/10.1007/s10750-014-2100-5. DOI: 10.1007/s10750-014-2100-5.

Hawes, I., Smith, R., 1993. Influence of environmental factors on the growth in culture of a New Zealand strain of the fast-spreading alga *Hydrodictyon reticulatum* (water-net). J. Appl. Phycol. 5, 437–445. https://doi.org/10.1007/BF02182736.

Herrmann, J., 2012. Chemical and biological benefits in a stormwater wetland in Kalmar, SE Sweden. Limnologica 42, 299–309. https://doi.org/10.1016/j.limno.2012.07.003.

Huser, B.J., Futter, M., Lee, J.T., Perniel, M., 2016. In-lake measures for phosphorus control: The most feasible and cost effective solution for long-term management of water quality in urban lakes. Water Res. 97, 142–152. https://doi.org/10.1016/j.watres.2015.07.036.

Iamchaturapatr, J., Yi, S.W., Rhee, J.S., 2007. Nutrient removals by 21 aquatic plants for vertical free surface-flow (VFS) constructed wetland. Ecol. Eng. 29, 287–293. https://doi.org/10.1016/j.ecoleng.2006.09.010.

Jeppesen, E., Søndergaard, M., Meerhoff, M., Lauridsen, T.L., Jensen, J.P., 2007. Shallow lake restoration by nutrient loading reduction – some recent findings and challenges ahead. Hydrobiologia 584, 239–252. https://doi.org/10.1007/s10750-007-0596-7.

Jurczak, T., Wagner, I., Kaczkowski, Z., Szklarek, S., Zalewski, M., 2018a. 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.

Jurczak, T., Wojtal-Frankiewicz, A., Frankiewicz, P., Kaczkowski, Z., Oleksińska, Z., Bednarek, A., Zalewski, M., 2018b. Comprehensive approach to restoring urban recreational reservoirs. Part 2 – use of zooplankton as indicators for the ecological quality assessment. Sci. Total Environ. https://doi.org/10.1016/j.scitotenv.2018.08.006.

Jurczak, T., Wojtal-Frankiewicz, A., Kaczkowski, Z., Oleksińska, Z., Bednarek, A., Zalewski, M., 2018c. Restoration of a shady urban pond – the pros and cons. J. Environ. Manage. 217, 919–928. https://doi.org/10.1016/j.jenvman.2018.03.114.

Jurczak, T., Wagner, I., Zalewski, M., 2012. Urban aquatic ecosystems management. Public Serv. Rev.: Europe 24, 178.

Jurczak, T., Tarczynska, M., Izydorczyk, K., Mankiewicz, J., Zalewski, M., Meriluoto, J., 2005. Elimination of microcystins by water treatment processes – examples from Sulejow Reservoir, Poland. Water Res. 39, 2394–2406. https://doi.org/10.1016/j.watres.2005.04.031.

Knoesche, R., Schreckenbach, K., Pfeifer, M., Weissenbach, H., 2000. Balances of phosphorus and nitrogen in carp ponds. Fisheries Manage. Ecol. 7, 15–22. https://doi.org/10.1046/j.1365-2400.2000.00198.x.

Krysiak, S., Majchrowska, A., Papińska, E., Adamczyk, J., 2016. Methods. In: Krysiak, S., Adamczyk, J. (Eds.), The Ecological Role of Abandoned Agricultural Lands in Buffer Zones Around Landscape Parks in the Łódź Voivodeship. Wydawnictwo Uniwersytetu Łódzkiego, Łódź, pp. 11–14.

Kujawa, I., Kujawa, M., 2011. Ekohydrologiczna rekultywacja zbiorników rekreacyjnych "Arturówek" (Łódź) jako modelowe podejście do rekultywacji zbiorników miejskich. Koncepcja programowo-przestrzenna. Część hydrologiczna. [Ecohydrologic rehabilitation of recreational reservoirs "Arturówek" (Łódź) as a model approach to rehabilitation of urban reservoirs. Program and spatial concept. The hydrological part.] Łódź, 2011 (in Polish).

Kumwimba, M.N., Zhu, B., Muyembe, D.K., Dzakpasu, M., 2017. Growth characteristics and nutrient removal capability of eco-ditch plants in mesocosm sediment receiving primary domestic wastewater. Environ. Sci. Pollut. Res. 24, 23926–23938. https://doi.org/10.1007/s11356-017-9992-3.

Lelková, E., Poulíčková, A., 2004. The influence of *Hydrodictyon reticulatum* (L.) LAGERH. on diurnal changes in environmental variables in a shallow pool, Czech Phycology. Olomouc 4, 103–109.

Lynch, J., Fox, L.J., Owen Jr., J.S., Sample, D.J., 2015. Evaluation of commercial floating treatment wetland technologies for nutrient remediation of stormwater. Ecol. Eng. 75, 61–69. https://doi.org/10.1016/j.ecoleng.2014.11.001.

McAndrew, B., Ahn, C., Spooner, J., 2016. Nitrogen and sediment capture of a floating treatment wetland on an urban stormwater retention pond – the case of the rain project. Sustainability 8, 972. https://doi.org/10.3390/su8100972.

Oberholster, P.J., Botha, A.M., Cloete, T.E., 2006. Toxic cyanobacterial blooms in a shallow, artificially mixed urban lake in Colorado, USA. Lakes Reser. Res. Manage. 11, 111–123. https://doi.org/10.1111/j.1440-1770.2006.00297.x.

Park, G.S., Khan, A.R., Kwak, Y., Hong, S.-J., Jung, B.K., Ullah, I., Kim, J.-G., Shin, J.-H., 2016. An improved effective microorganism (EM) soil ball-making method for water quality restoration. Environ. Sci. Pollut. Res. 23, 1100–1107. https://doi.org/10.1007/s11356-015-5617-x.

Peretyatko, A., Teissier, S., De Backer, S., Triest, L., 2009. Restoration potential of biomanipulation for eutrophic peri-urban ponds: the role of zooplankton size and submerged macrophyte cover. Hydrobiologia 634, 125–135. https://doi.org/10.1007/s10750-009-9888-4.

Peretyatko, A., Teissier, S., De Backer, S., Triest, L., 2010. Assessment of the risk of cyanobacterial bloom occurrence in urban ponds: probabilistic approach. Ann. Limnol. – Int J. Lim. 46, 121–133. https://doi.org/10.1051/limn/2010009.

Peretyatko, A., Teissier, S., De Backer, S., Triest, L., 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.

PN-R-04023, 1996. Analiza chemiczno-rolnicza gleby. Oznaczanie zawartości przyswajalnego fosforu w glebach mineralnych [Chemical and agricultural analysis of soil – Determination of available phosphorus content in mineral soils] (in Polish).

Podsiadłowski, S., Osuch, E., Przybył, J., Osuch, A., Buchwald, T., 2017. Pulverizing aerator in the process of lake restotation. Ecol. Eng. (in press). https://doi.org/10.1016/j.ecoleng.2017.06.032.

Pokorny, J., Hauser, V., 2002. The restoration of fish ponds in agricultural landscapes. Ecol. Eng. 18, 555–574. https://doi.org/10.1016/S0925-8574(02)00020-4.

Richardson, T.L., Lawrenz, E., Pinckney, J.L., Guajardo, R.C., Walker, E.A., Paerl, H.W., MacIntyre, H.L., 2010. Spectral fluorometric characterization of phytoplankton community composition using the Algae Online Analyser. Water Res. 44, 2461–2472. https://doi.org/10.1016/j.watres.2010.01.012.

Rosińska, J., Kozak, A., Dondajewska, R., Gołdyn, R., 2017. Cyanobacteria blooms before and during the restoration process of a shallow urban lake. J. Environ. Manage. 198, 340–347. https://doi.org/10.1016/j.jenvman.2017.04.091.

Rozema, E.R., Gordon, R.J., Zheng, Y., 2016. Harvesting plants in constructed wetlands to increase biomass production and Na^{+} and Cl^{-} removal from recycled greenhouse nutrient solution. Water Air Soil Pollut. 227, 136. https://doi.org/10.1007/s11270-016-2831-1.

Schagerl, M., Bloch, I., Angeler, D.G., Fesl, C., 2010. The use of urban clay-pit ponds for human recreation: assessment of impacts on water quality and phytoplankton assemblages. Environ. Monit. Assess. 165, 283–293. https://doi.org/10.1007/s10661-009-0945-2.

Schwartz, D., Sample, D.J., Grizzard, T.J., 2017. Evaluating the performance of a retrofitted stormwater wet pond for treatment of urban runoff. Environ. Monit. Assess. 189, 256. https://doi.org/10.1007/s10661-017-5930-6.

Steffensen, D.A., 2008. Economic cost of cyanobacterial blooms. Adv. Exp. Med. Biol. 619, 855–865. https://doi.org/10.1007/978-0-387-75865-7_37.

Steudel, Bastian, Hector, Andy, Friedl, Thomas, Löfke, Christian, Lorenz, Maike, Wesche, Moritz, Kessler, Michael, Gessner, Mark, 2012. Biodiversity effects on ecosystem functioning change along environmental stress gradients. Ecol. Lett. 15 (12), 1397–1405. https://doi.org/10.1111/j.1461-0248.2012.01863.x.

Szklarek, S., Stolarska, M., Wagner, I., Mankiewicz-Boczek, J., 2015. The microbiotest battery as an important component in the assessment of snowmelt toxicity in urban watercourses – preliminary studies. Environ. Monit. Assess. 187, 16. https://doi.org/10.1007/s10661-014-4252-1.

Szklarek, S., Wagner, I., Jurczak, T., Zalewski, M., 2018. Sequential Sedimentation-Biofiltration System for the purification of a small urban river (the Sokolowka, Lodz) supplied by stormwater. J. Environ. Manage. 205, 201–208. https://doi.org/10.1016/j.jenvman.2017.09.066.

Tomaszek, J.A., Gruca-Rokosz, R., 2007. Rates of dissimilatory nitrate reduction to ammonium in two polish reservoirs: impacts of temperature, organic matter content and nitrate concentration. Environ. Technol. 28 (7), 771–778. https://doi.org/10.1080/09593332808618834.

Urbaniak, M., Zieliński, M., Wagner, I., 2015. Seasonal distribution of PCDDs/PCDFs in the small urban reservoirs. Int. J. Environ. Res. 9 (2), 745–752. https://doi.org/10.22059/IJER.2015.948.

Van Wichelen, J., Declerck, S., Muylaert, K., Hoste, I., Geenens, V., Vandekerkhove, J., Michels, E., De Pauw, N., Hoffmann, M., De Meester, L., Vyverman, W., 2007. The importance of drawdown and sediment removal for the restoration of the eutrophied shallow Lake Kraenepoel (Belgium). Hydrobiologia 584 (1), 291–303. https://doi.org/10.1007/978-1-4020-6399-2_26.

Verhofstad, M., Poelen, M., van Kempen, M., Bakker, E., Smolders, A., 2017. Finding the harvesting frequency to maximize nutrient removal in a constructed wetland dominated by submerged aquatic plants. Ecol. Eng. 106, 423–430. https://doi.org/10.

1016/j.ecoleng.2017.06.012.
Vymazal, J., 2007. Removal of nutrients in various types of constructed wetlands. Sci. Total Environ. 380 (1–3), 48–65. https://doi.org/10.1016/j.scitotenv.2006.09.014.
Waajen, G.W.A.M., Faassen, E.J., Lürling, M., 2014. Eutrophic urban ponds suffer from cyanobacterial blooms: Dutch examples. Environ. Sci. Pollut. Res. 21, 9983–9994. https://doi.org/10.1007/s11356-014-2948-y.
Wagner, I., Zalewski, M., 2011. System solutions in urban water management: the Lodz (Poland) perspective. In: Howe, C., Mitchell, C. (Eds.), Water Sensitive Cities. IWA Publishing, London pp. 231e245.
WHO Guidelines for safe recreational water environments Volume 1 2003 Coastal and fresh waters. World Health Organization Geneva.
Winston, R.J., Hunt, W.H., Kennedy, S.G., Merriman, L.S., 2013. Evaluation of floating treatment wetland as retrofits to existing stormwater retention ponds. Ecol. Eng. 54, 254–265. https://doi.org/10.1016/j.ecoleng.2013.01.023.
Wojtal-Frankiewicz, A., Frankiewicz, P., 2011. The impact of pelagic (*Daphnia longispina*) and benthic (*Dreissenapolymorpha*) filter feeders on chlorophyll and nutrient concentration. Limnologica 41 (3), 191–200. https://doi.org/10.1016/j.limno.2010.09.001.
Yu, C., Stone, M., 2010. Sediment and Nutrient Transport Dynamics in an Urban Stormwater Impoundment. IAHS-AISH Publication, pp. 99–105.
Zalewski, M., Wagner, I., Fratczak, W., Mankiewicz-, Boczek J., Parniewki, P., 2012. Blue-green city for compensating global climate change. Parlim. Magaz. 6.
Zalewski, M., 2014. Ecohydrology and hydrologic engineering: regulation of hydrology-biota interactions for sustainability. J. Hydrol. Eng. 20https://doi.org/10.1061/(ASCE)HE.1943-5584.0000999. A4014012-1-A4014012-14.