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Tailwater habitat stability after dam removal and return to a natural hydrological regime

by

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Abstract

The study was conducted from 2000 to 2003 in the tailwater of the Drzewieckie Lake, an artificial reservoir in Central Poland. Short-term peaks in water flow were generated for the purpose of the operation of a whitewater slalom canoeing track built just downstream of the dam. In 2002, the reservoir was drawn down. The patterns in habitat samples were recognized with a Kohonen's unsupervised artificial neural network (SOM). The SOM spatial gradient was stronger than the SOM temporal gradient, which shows that the removal of the studied dam did not have a destructive impact on habitats' features, as shown in other studies, and that the patchy nature of the riverbed has been maintained. The complete emptying of the Drzewieckie Lake took place at the beginning of the vegetation season, which allowed plants to cover the exposed bottom of the reservoir and, consequently, reduce the downstream flow of organic matter accumulated there. Patterns in the displacement of aquatic macrophytes, inorganic substratum and different fractions of particulate organic matter are discussed. The amount of dissolved oxygen decreased because of the lack of intensive water discharge from the reservoir into the river, which would result in high water turbulence. Results of this study are important for planning the ecologically sound dam removals.

Key words: dam reservoir, impoundment, river restoration, temporary renaturisation, environmental parameters, hydropeaking, Kohonen artificial neural network, self-organizing map (SOM)

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Introduction

In order to prevent floods or inundations, many rivers have been regulated and dammed (Dynesius & Nilsson 1994). Over the past century, about 45 000 reservoirs with dams over 15 m high (Oud & Muir 1997; Robinson et al. 2003a,b) and countless smaller impoundments (Poff & Hart 2002) have been constructed worldwide. Regardless of the type and size of the dam and the system of its operation, each reservoir becomes shallow within a longer time. This is especially true for lowland regions. An average period of reservoir operation ranges from 60 to 120 years (Dendy & Champion 1978) and largely depends on reservoir morphometry and the nature of the river catchment. Both of these factors determine the sedimentation rate of organic matter transported over the basin bed (Morris & Fan 1998) and the subsequent reservoir shallowing and loss of its retention capacity.

Dam removal becomes a more and more frequent solution to the problems associated with reservoir shallowing, results of improper reservoir exploitation or decaying hydro-technical constructions. In the United States alone, 80% of approximately 2.5 million existing dams (National Research Council 1992) are expected to be decommissioned (River Alliance of Wisconsin and Trout Unlimited 2000). In addition, there have already been over 700 documented dam decommissions and/or removals in the USA in the past century (Gleick et al. 2009), including over 350 in the last decade (Pohl 2003, American Rivers 2009). Few dams are removed for ecological reasons (Poff & Hart 2002), although it is known that impoundment disturbs river functions by modifying the natural water and nutrient flow (Petts 1984, Shuman 1995). A decisive role in determining the decisions to remove dams is mostly played by economic and social causes. Usually, relevant decisions are made by local authorities based on the "profits and losses" analysis (Smith et al. 2000, Graber et al. 2001, Bowman 2002, Johnson & Graber 2002). It is more beneficial to bear lower costs resulting from dam removal than the costs of dam repair and/or protection of aquatic environments or human settlements against possible disasters (Kanehl et al. 1997, Born et al. 1998, Johnson & Graber 2002). According to Pejchar and Warner (2001), dam removal should be more disposed to ecological aspects in the future. This should be accompanied by effective renaturisation procedures reducing the risk of ecological catastrophes (Hart et al. 2002). While the impact of impoundment on the river biocenosis is relatively well known, drawing down a reservoir and dam removal still require more attention (Hart et al. 2002, Pollard & Reed 2004), as they can have negative consequences, such as river pollution with toxic compounds that have accumulated for years in bottom sediments of the reservoir (Shuman 1995) or colmatation, i.e. filling up the river bed with fine-grained mineral particles (Brunke 1999, Doyle et al. 2003). So far, only a few studies concerning the impact of dam removal on the river morphology and biocenosis downstream of (mostly large) impoundments have been reported in the literature (Shuman 1995, Hart et al. 2002, Burroughs et al. 2009).

The problem of shallowing also occurred in the Drzewieckie Lake, an artificial reservoir constructed in the fourth order stretch of the lowland Drzewiczka River in Central Poland. This over 70-year-old impoundment was created for the purpose of water retention for industry and recreation. Additionally, in 1980, a mountain-like whitewater slalom canoeing track (WSCT) was built just below the dam reservoir. In order to make it work, a specific hydrological regime, resulting in hydropeaking, was kept in the tailwater (Szczerkowska-Majchrzak et al. 2015). Each day, three to five times, for 2-3 hours, the water discharge was increased, as compared to natural conditions (Tszydel et al. 2009). Over time, however, the reservoir has lost its retention capacity. Therefore, the Drzewieckie Lake was drawn down, and its dam was reconstructed. As a result, the Drzewiczka River returned to its natural discharge (1.7-4.5 $m^3\ s^{-1}$) for several years, which was a specific kind of temporary tailwater renaturisation (Żmudziński et al. 2002). The latter was an unintended by-product. Nevertheless, this allowed the monitoring of environmental parameters of the Drzewiczka River stretch just after the river was freed from hydraulic stress, and for comparing them between the impoundment and post-impoundment periods.

The aim of this study was to assess changes in environmental factors during the return of the Drzewiczka River tailwater to its natural hydrological regime by answering the following questions:

(1) How the basic environmental parameters

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have changed in several selected habitats after the retreat of artificial high-frequency flow fluctuations?

(2) Which habitats are most stable after freeing the river from hydropeaking?

Materials and methods

Study area

The lowland Drzewiczka River is the biggest right-side tributary of the Pilica River (the Vistula River system), with a length of 81.3 km and a river basin area of 1082.9 km^3. The impoundment, known as "the Drzewieckie Lake", which has an area of 84 hectares and a retention volume of about 1.2 million m^3, was constructed in 1936 (Mackiewicz et al. 1986, Szudek et al. 1989). In 2001, a resolution to dredge the reservoir was adopted by the Drzewica City Council. The reservoir had been completely emptied before the cleaning.

The sampling site was located in the tailwater, at the end of the 2 km long WSCT and, thus, about 2 km downstream of the dam. Along a 160-m-long river stretch, the following five sampling habitats $H_{(h)}$ reflecting the diversity of environmental parameters were selected (Fig. 1): **H$_P$** – pool, with a neighboring island and a fallen tree that increased the speed of the river; **H$_S$** – stagnant habitat, with very low flow and a large amount of particulate organic matter (POM), silt and emergent macrophytes; **H$_M$** – midstream habitat with a bed covered by sand and submerged macrophytes; **H$_B$** – habitat along the riverbank, with a clayey bed covered with gravel and pebbles; **H$_R$** – riffle habitat, with high flow and depth, and a bed covered with gravel, pebbles and cobbles.

Habitat sampling

The sampling was conducted in the location of 20°29'14"E and 51°27'08"N (Fig. 1). A total of 120 samples were collected from each sampling habitat once a month, in two annual cycles; the first cycle (S1)

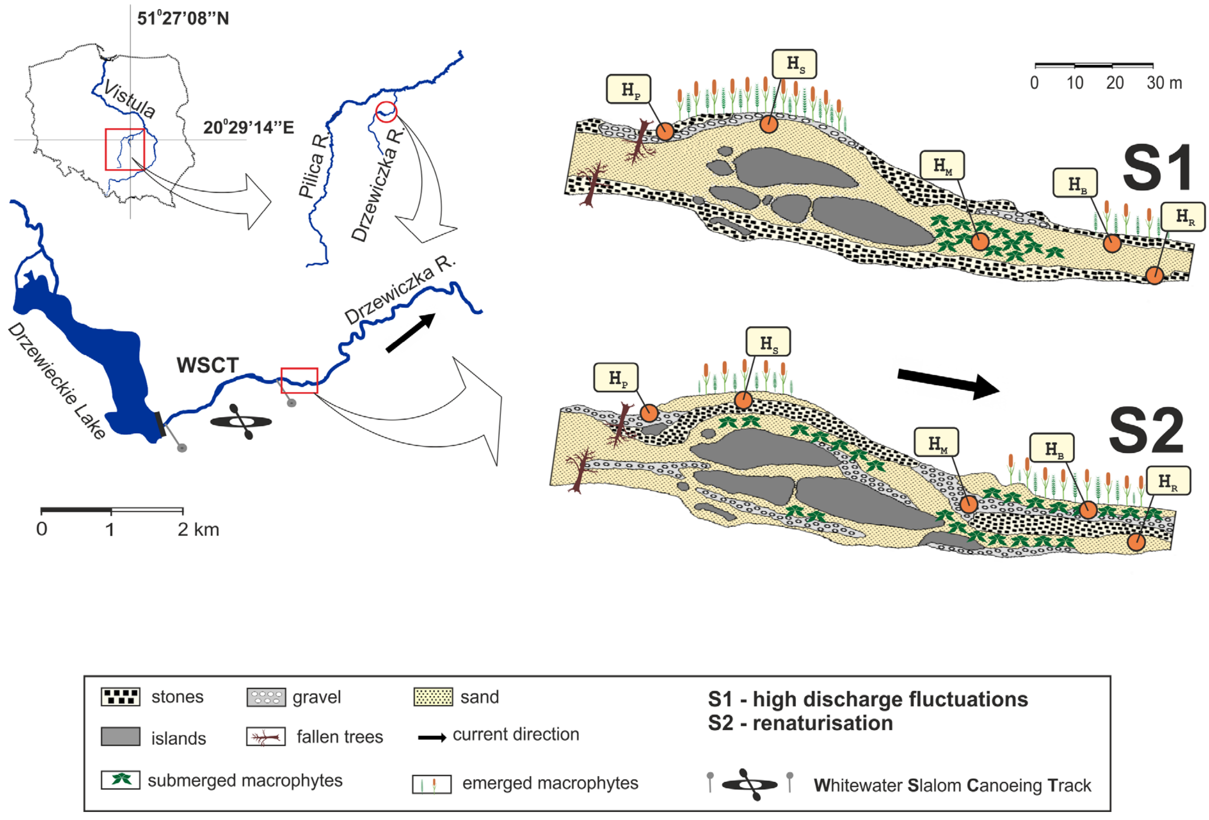

Figure 1
The study area

lasted from November 2000 to October 2001 when hydropeaking occurred, and the second cycle (S2) lasted from November 2002 to October 2003 during the river's return to the natural hydrological regime after the dam had been pulled down. Emptying the water reservoir began in January 2002, i.e. between both sampling cycles, and lasted four months.

At each of the sampling habitats, the following parameters were measured: water temperature, concentration of dissolved oxygen, pH, water depth and current velocity. On each sampling occasion, 250 cm^2 of riverbed was collected by means of a tubular sampler. The composition of particulate inorganic substratum (granulation) was analyzed according to Cummins (1962), which allowed for the estimation of the substratum index SI (Quinn & Hickey 1990) for each sampling habitat. The same bed samples were used to determine 1) the POM content in the bottom sediment, divided into fine POM (BFPOM) with particles <1 mm and coarse POM (BCPOM) with particles >1 mm (Petersen et al. 1989), and 2) periphyton quantity, which was expressed by the content of chlorophyll (Golterman et al. 1978). In order to determine the amount of fine POM transported in suspension (TFPOM), triplicate water samples were collected in 10 dm^3 plastic containers.

The above data were already used by Tszydel et al. (2009) while presenting the influence of dam decommissioning on trichopteran assemblages. However, the abiotic data were not compared in detail between the sampling habitats and the sampling cycles, although such comparisons seem to be crucial for the assessment of the dam removal effects on the river's functioning.

Models applied

The patterns in habitat samples were recognized with a self-organizing map (SOM), i.e. Kohonen's unsupervised artificial neural network (Kohonen 1982, 2001). The SOM was trained with the use of the SOM Toolbox (Vesanto et al. 2000), which was developed by the Laboratory of Information and Computer Science at the Helsinki University of Technology (http://www.cis.hut.fi/projects/somtoolbox).

Artificial neural networks are simple structural and functional models of a human brain, and they are built of neurons (data processing units). They easily deal even with variables that exhibit strongly skewed distributions and/or are related in a complex way. Moreover, they do not require any *a priori* specification of the model describing studied phenomena (Brosse et al. 2001).

The neurons in the Kohonen artificial neural network are arranged in two (input and output) layers. The input layer serves to receive data, while the results are formed in the output layer. The dataset (8 abiotic variables × 120 samples) was repeatedly presented to the input neurons. The number of the latter was the same as the number of variables in the dataset (eight) because each input neuron received data of one variable. The output neurons were arranged into a two-dimensional grid (9 × 6). To choose the size of the output layer, we adopted the rule saying that the number of output neurons should be close to $5\sqrt{n}$ (Vesanto & Alhoniemi 2000), where n is the number of habitat samples (result: 55 output neurons). During the SOM training, each input neuron transmitted information to all output neurons, and the weight of connections (that is, their intensity) for pairs of each input neuron and each output neuron were modified (strengthened or weakened). Based on such established links between the input (receiving data) layer and the output layer (in which results are formed), a virtual habitat sample (VHS) was created in each output neuron. A similarity of VHSs was related to their positions in the output layer: VHSs in distant neurons were considerably different, while those in neighboring neurons were similar. Clustering of VHSs (and respective output neurons) was performed with hierarchical cluster analysis, with the Ward linkage method and the Euclidean distance measure (Ward 1963, Vesanto & Alhoniemi 2000). Finally, each real habitat sample was assigned to the best matching VHS (and a respective output neuron). Consequently, almost identical real habitat samples were located in one neuron, similar samples were located in neighboring neurons of the same (sub) cluster, and considerably different samples were in distant neurons or in adjoining neurons belonging to different (sub)clusters (Chon 2011, Penczak 2011, Qu et al. 2013, Bae et al. 2014). In summation, the Kohonen artificial neural networks, with a moderate loss of information, reduced *n*-dimensional data to a two-dimensional map, on which similar studied

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objects (habitat samples) were located nearby, while different studied objects were assigned to neurons in distant regions of the output layer.

More details on the SOM used for ecological applications can be found, among others, in similar studies (Chon et al. 1996, Park et al. 2001, Lek et al. 2005, Conti et al. 2012, Li et al. 2013, Park et al. 2013).

Spatial comparisons (between the five sampling habitats) and temporal comparisons (between the sampling cycle S1 and S2) in all the eight abiotic variables, and additionally, in the amount of periphyton and macrophyte cover, were performed with the Friedman test for multiple dependent groups of samples, with an adopted significance level of $p<0.05$ in multiple comparisons (two-tailed test) in the XLSTAT package working in Microsoft Excel.

The Pearson's correlation coefficient r was used to determine the mutual relations between environmental parameters. Appropriate calculations were performed with the use of the STATISTICA data analysis software system v.10 (StatSoft Inc., 2011).

Results

The SOM quantization and topographic errors were 0.383 and 0.042, respectively. The output layer of SOM was partitioned into two main clusters: X and Y, each with respective subclusters: X_1 and X_2, and Y_1, Y_2 and Y_3 (Fig. 2). The spatial origin of the habitat samples was decisive for their assignation to clusters X and Y (and, thus, for the vertical gradient over SOM) (Fig. 2). Cluster X contained

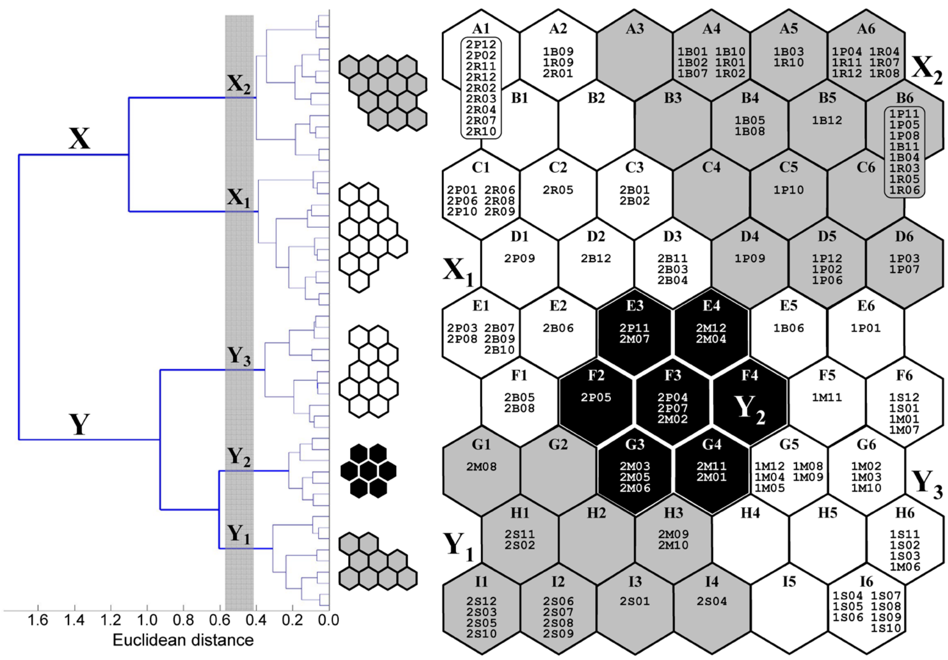

Figure 2

Habitat samples assigned to SOM output neurons A1-I6, grouped into clusters X and Y, and subclusters X_1, X_2, Y_1, Y_2 and Y_3 (the latter ones are marked with shading), which were distinguished on the basis of the hierarchical cluster analysis. Each sample code consists of: (1) indication of the sampling cycle (1 – S1, from November 2000 to October 2001, pre-dam-removal phase with hydropeaking; 2 – S2, from November 2002 to October 2003, post-dam-removal renaturisation phase); (2) habitat code (P – pool, S – stagnant habitat, M – midstream habitat, B – habitat along the riverbank, R – riffle habitat), and (3) the number of the month of sampling

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66 samples, only from H_P, H_B and H_R, while among 54 samples in cluster Y, there were all the samples from H_S and H_M, and additionally, six samples from H_P and H_B (five and one, respectively). The abiotic variables associated with the SOM vertical gradient were current velocity, SI (both higher in X) and amounts of BCPOM (higher in Y) (Fig. 3). A clear horizontal gradient was also visible at the lower typology (subcluster) level, and it was related to the temporal origin of samples (Fig. 2). Subcluster X_1 contained samples from the sampling cycle S2 (with two exceptions, in neuron A2), while all the samples in X_2 came from S1. Similarly, subclusters Y_1 and Y_2 contained only samples from S2, and Y_3 contained only samples from S1 (Fig. 2). In summary, the vertical spatial gradient (differences between clusters) was stronger than the horizontal temporal gradient (differences between subclusters).

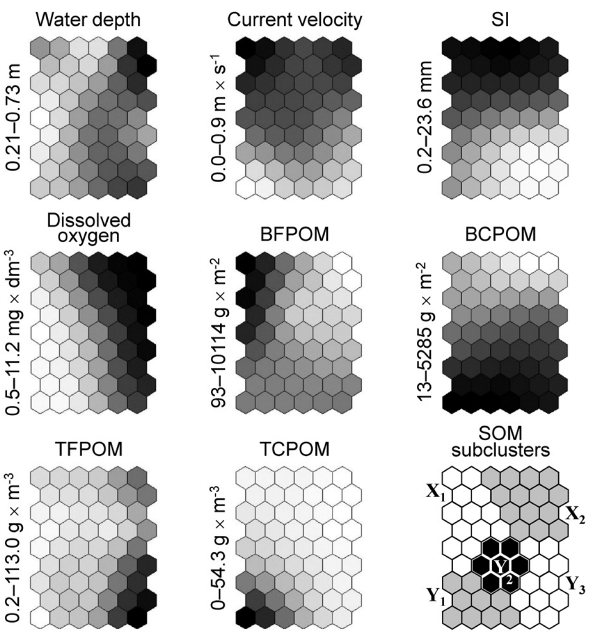

Figure 3

The intensity (stronger if darker shading; based on virtual habitat samples) of eight abiotic variables (used for SOM training) in particular SOM regions. The intensity of the grey color is scaled independently for each variable (the range of variability in the dataset is presented on the left side of each plane). The variables exhibiting a vertical gradient considerably contributed to identification of SOM clusters X and Y, while others influenced the identification of SOM subclusters X_1, X_2, Y_1, Y_2 and/or Y_3 (compare with Figure 2). In general, variables with similar greyness patterns are positively correlated, and variables with mirror greyness patterns are negatively correlated.

The above rule was also evident in the much higher number of spatial differences (between sampling habitats) compared to the number of temporal differences (between the sampling cycles S1 and S2) in the studied variables found with conventional statistical methods (Table 1). The spatial differences in most studied variables were big enough to be observed in both sampling seasons. Water depth in H_R was significantly (p<0.05) higher than in H_B. Current velocity was significantly lowest in H_S (Fig. 4, Table 1). Additionally it was significantly higher in H_R compared with H_M and H_B. Similarly, bed substratum was significantly finer, i.e. SI was significantly lower in H_S and H_M compared with H_B and H_R, and in H_M compared with H_P. The quantity of periphyton (chlorophyll *a*) was highest in H_S compared with all other habitats except H_M. Significant spatial differences were also found in the amounts of BCPOM (Fig. 5, Table 1). They were significantly lower in H_B and H_R. Some additional differences between habitats in the above variables were observed only in the S1 or S2 sampling cycles (Fig. 4, 5, Table 1).

In general, spatial differences in the remaining variables were recorded only in a particular sampling cycle. In S1, the concentrations of dissolved oxygen were significantly higher in H_R than H_S (Fig. 4, Table 1). Significant differences in BFPOM between H_P habitat and H_S and H_B habitats reversed, i.e. significantly lower values in H_P in S1 became significantly higher in S2; in S2, H_P and H_R were the richest in BFPOM (Fig. 5, Table 1). In S1, the highest amounts of TFPOM were recorded in H_S. In S1, little coarse POM transported in suspension (TCPOM) was recorded in all the habitats, although in significantly lower amounts in H_R than in H_M. In S2, H_R remained poorer in TCPOM than H_S. In S1, the macrophyte cover was larger in H_M than in H_P and H_S (Fig. 4, 5, Table 1).

There were fewer significant temporal differences than the spatial differences presented in the two previous chapters. Among the temporal differences (that is, between the sampling cycles

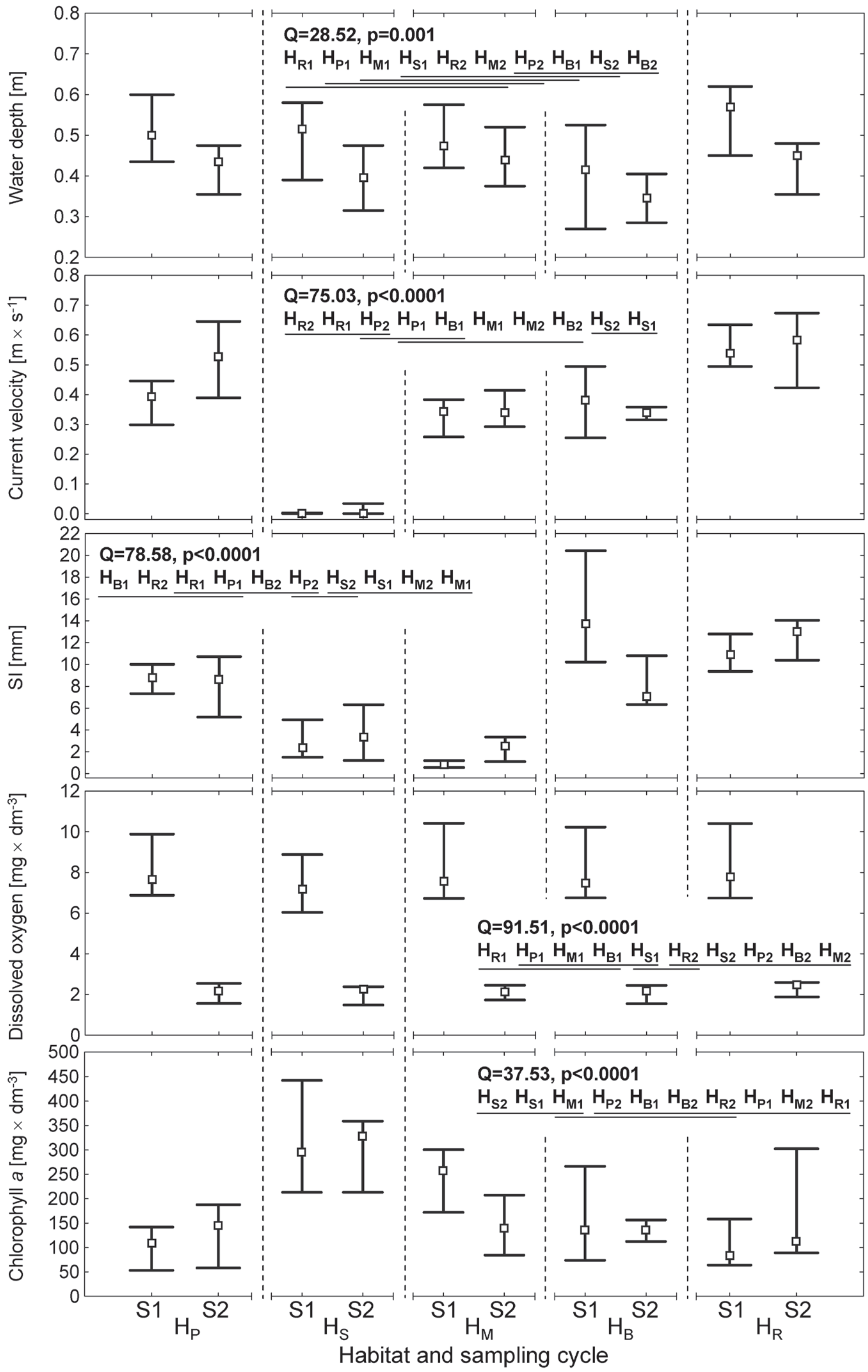

Figure 4

Spatial (between sampling habitats: H_P – pool, H_S – stagnant habitat, H_M – midstream habitat, H_B – habitat along the riverbank, H_R – riffle habitat) and temporal (between the sampling cycles S1 and S2, pre- and post-dam removal, respectively) comparisons of: water depth, water current velocity, the substratum index (SI), concentration of oxygen dissolved in water, and concentration of chlorophyll *a*. Point – median, whiskers – interquartile range, Q – statistics of the Friedman test (df = 9, n = 12). For sampling occasions whose symbols are underlined with the same line, no significant difference ($p \leq 0.05$) in *post-hoc* comparisons was recorded (a summary of *post-hoc* comparisons available in Table 1).

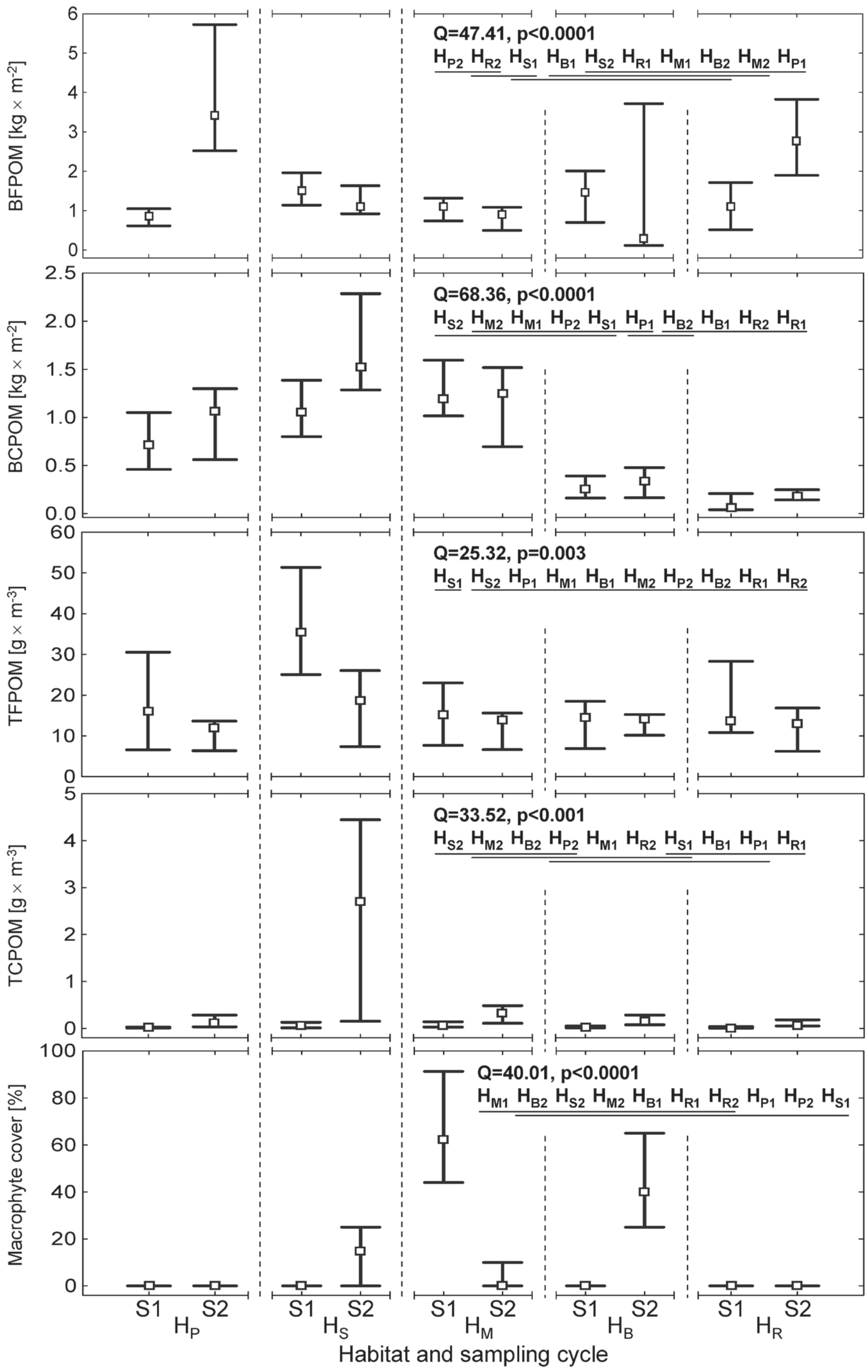

Figure 5

Spatial (between sampling habitats: H_P – pool, H_S – stagnant habitat, H_M – midstream habitat, H_B – habitat along the riverbank, H_R – riffle habitat) and temporal (between the sampling cycles S1 and S2, pre- and post-dam-removal, respectively) comparison of the amounts of benthic fine particulate organic matter (POM), benthic coarse POM, transported fine POM, transported coarse POM, and macrophytes covering the river bottom. Point – median, whiskers – interquartile range, Q – statistics of the Friedman test (df = 9, n = 12). For sampling occasions whose symbols are underlined with the same line, no significant difference ($p \leq 0.05$) in *post-hoc* comparisons was recorded (a summary of *post-hoc* comparisons available in Table 1).

Table 1

Summary of *post-hoc* comparisons presented in Figures 4 and 5. The number and character of significant 1) spatial differences between sampling habitats and 2) temporal differences between pre- (S1) and post-dam removal (S2) sampling cycles are presented. Explanations: H_P – pool, H_S – stagnant habitat, H_M – midstream habitat, H_B – habitat along the riverbank, H_R – riffle habitat

Variable	Unit	Spatial differences observed (between sampling habitats)			Temporal differences (between S1 and S2)
		in both S1 and S2	in S1 only	in S2 only	
Water depth	m	1: H_B–H_R	1: H_P–H_B	1: H_M–H_B	
Current velocity	m s^{-1}	6: H_S–(H_P,H_M,H_B,H_R) H_R–(H_M,H_B)	1: H_P–H_R	2: H_P–(H_M,H_B)	
SI	mm	5: H_P–H_M, (H_S,H_M)–(H_B,H_R)	1: H_P–H_S	2: H_R–(H_P,H_B)	1: H_B
Dissolved oxygen	mg dm^{-3}		1: H_S–H_R		5: H_P, H_S, H_M, H_B, H_R
Chlorophyll *a*		3: H_S–(H_P,H_B,H_R)	2: H_M–(H_P,H_R)	1: H_S–H_M	1: H_M
BFPOM	kg m^{-2}		2: H_P–(H_S,H_B)	6: (H_P,H_R)–(H_S,H_M,H_B)	2: H_P, H_R
BCPOM		6: (H_B,H_R)–(H_S,H_M,H_P)			
TFPOM	g m^{-3}		4: H_S–(H_P,H_M,H_B,H_R)		1: H_S
TCPOM			1: H_M–H_R	1: H_S–H_R	3: H_S, H_B, H_R
Macrophyte cover	%		2: H_M–(H_P,H_S)		
Total number of differences		21	15	13	13

S1 and S2), the most distinct ones were recorded in the concentrations of oxygen dissolved in water (Fig. 4, Table 1). In each habitat, significantly lower values were recorded in the sampling cycle S2. Although a certain reduction in water depth was visible, no significant differences between S1 and S2 were recorded in any habitat. Periphyton amounts significantly declined in H_M. No significant temporal changes were recorded in BCPOM, while the amounts of BFPOM significantly increased in H_P and H_R (Fig. 5, Table 1). The quantities of TFPOM and TCPOM significantly increased in H_S; an increase in TCPOM was also recorded in H_B and H_R. Within the period of permanent hydropeaking (S1), submerged plants (mainly *Potamogeton* sp.) occurred only in the middle zone of the riverbed (H_M), whereas in the period of renaturisation, the macrophytes also appeared in habitats H_S and H_B (Fig. 1). Nevertheless, the temporal differences were insignificant. In both sampling cycles, habitat H_S was covered with emerged plants, but in the renaturisation period, the meander bank retreated together with amphiphytes (Fig. 1).

In both S1 and S2, it was the current velocity that was negatively correlated with the amount of various organic matter fractions (Table 2). Amounts of fine POM and macrophyte cover were negatively correlated with concentrations of oxygen dissolved in water (Table 2). These results are consistent with the SOM results (Fig. 3).

Table 2

Significant Pearson correlations between environmental parameters of the investigated habitats of the Drzewiczka River in both sampling cycles (S1 – during high short-term discharge fluctuations; S2 – after a return to the natural hydrological regime). Explanations: chl. *a* – chlorophyll *a*, subm. macr. – submerged macrophytes; significance level: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; (–) – negative correlation

Parameter	Unit	Sampling cycle	
		S1	S2
Water depth	m		chl. *a**, –subm. macr.***
Current velocity	m s^{-1}	–chl. *a****, –BFPOM*, –BCPOM***, –TFPOM*, –TCPOM*	–chl. *a***, –BFPOM***, –BCPOM***, –TFPOM***
Inorganic substratum index (SI)	mm	–chl. *a**, –BCPOM***, –subm. macr.***	–BCPOM***, –TCPOM*, –subm. macr.***
Dissolved oxygen	mg dm^{-3}	–BFPOM*, –TFPOM*, –subm. macr.*	–TFPOM*, –subm. macr.***

Discussion

In any habitat, a change in environmental parameters due to disturbance can be different and depends on disturbance intensity (Palmer et al. 1995, 1996a, 1996b, 1997). A return to the natural hydrological regime in the tailwater, resulting from emptying a dam reservoir, allows for the monitoring of the stability of river habitats (Death & Winterbourn 1995, Gore 1996). The most stable sections of riverbeds with respect to environmental parameters are called refugia. They fulfil a very significant function, first of all, in river recolonization by zoobenthos groups after any disasters occurring with a higher or lower frequency (Matthaei et al. 1998, 2000, Matthaei & Townsend 2000).

When the Drzewieckie Lake functioned, the hydrological regime in the tailwater (Tszydel et al. 2009, Szczerkowska-Majchrzak et al. 2015) resembled the one observed downstream of hydropower plants, which operate according to diurnal fluctuations in energy demand (Troelstrup & Hergenrader 1990, Moog 1993). Despite short-term releases of large water volumes (much greater than the natural discharge) from the Drzewieckie Lake to the WSCT (canoeing track), very heterogeneous environmental conditions prevailed in the tailwater (Szczerkowska-Majchrzak et al. 2010). The patchy character of the studied riverbed was preserved even when the dam was removed, which was clearly expressed in the SOM spatial gradient, and was visibly stronger than the temporal gradient. The dam removal and the emptying of the Drzewieckie Lake did not have as a destructive impact on environmental parameters as described by Perry and Perry (1986) or Pizzuto (2002) for other river ecosystems. A comparison of values of abiotic parameters between the permanent disturbance and renaturisation periods (S1 and S2, respectively) displayed that, in all the habitats studied, it was the amount of oxygen dissolved in water that changed most. The reason for this lies in the lack of intensive water discharges from the reservoir to the river in S2, which would result in high water turbulence and induce increased saturation of water with oxygen. In S2, the amount of oxygen dissolved in water decreased also due to the increased quantity of decomposing organic matter from exposed sediments washed downstream to the river (Table 2).

The maintained mosaicity of habitats in the Drzewiczka River in S2 compared to S1 is very important for planning the ecologically sound dam removals (Downes 1990, Lancaster & Hildrew 1993, Palmer et al. 1995). A runoff of considerable amounts of benthic POM accumulated over many years in the reservoir may be a threat (Randle et al. 1996, Lisle et al. 2001, Doyle et al. 2003). The transfer of POM to the river can be extremely high, as it may potentially reach up to 50% of the fine POM deposits accumulated on the bed (Randle et al. 1996). The presence of ecotone plants (amphiphytes) overgrowing the exposed reservoir basin can considerably reduce the POM transfer (Bednarek 2001, Shafroth et al. 2002). The complete emptying of the Drzewieckie Lake took place at the beginning of the vegetation season, which allowed plants to totally cover the exposed bed within the first four months of the vegetation season (Fig. 6). This reduced the downstream flow of organic matter accumulated in large amounts on the reservoir's bed. Therefore, in H_P and H_R habitats in the Drzewiczka River, although an increase in BFPOM after dam removal was significant, it was much lower than catastrophic levels observed by Stanley et al. (2002) or Doyle et al. (2003) in the tailwaters of large dam reservoirs where fine-particle sediments were from 35 to 160 cm in thickness. Moreover, no significant changes in the BCPOM amounts were observed in the tailwater in S2.

The increased transport of inorganic substratum and organic matter to the tailwater during dam removal usually leads to the redevelopment of a riverbed (Kanehl et al. 1997, Egan 2001). Rapid emptying of a dam reservoir and the river returning to its natural flow can also cause the destabilization of sediment organic matter fractions (Rae 1987, Scrimgeour et al. 1988, Asaeda & Rashid 2012, Sawaske & Freyberg 2012). In the Drzewiczka River during the emptying of the reservoir, mostly BFPOM was transported downstream, and then was observed in significantly greater amounts on the tailwater bed in H_P and H_R. The rapid displacement of large amounts of BFPOM from the reservoir between S1 and S2 resulted in recording similar or even significantly lower amounts of TFPOM in the tailwater during the renaturisation period (S2). An opposite pattern was observed for the coarse POM fractions. The displacement of BCPOM was not so

Figure 6

Successive development of vegetation covering the exposed bottom of the emptied Drzewieckie Lake between March and June 2002, i.e. a few months before the second sampling cycle (S2) (by M. Tszydel).

effective, and as a result: 1) no significant increase in BCPOM accumulated on the tailwater bed was recorded after dam removal (S2), and 2) in S2, the coarse POM was still present on the exposed reservoir's bed and, thus, it was transported (as TCPOM) in significantly higher amounts to the river. Moreover, the proportion of TFPOM to TCPOM in both sampling cycles was similar, i.e. a clear quantitative predominance of TFPOM over TCPOM was maintained. Destabilization and transport of sole fine sediment POM shows that the emptying of the Drzewieckie reservoir was not sudden. The non-catastrophic nature of the dam removal is confirmed by: 1) the fact that this process lasted several months, and 2) the lack of significant differences between the sampling cycles in the current velocity in any habitat. It should also be stressed that the amounts of transported organic matter in the study area in any sampling cycle were not high in comparison to studies conducted by Shafroth et al. (2002) and Doyle et al. (2003). However, retention of both POM fractions on the river bottom was not only dependent on the downstream transport of organic matter, but also on the macrophyte cover.

The rate and size of the displaced inorganic grains depend on the flow intensity, season and reservoir's capacity (Lisle et al. 2001, Doyle et al. 2002, 2003). Fine-grained deposits are usually accumulated in bank habitats and meanders, whereas they are washed out from the middle zone of a riverbed (Leopold 1992). In the Drzewiczka, a significant change in the particle size of the inorganic substrate concerned only H_B, which was covered with smaller grains caught by submerged vegetation and emergent plants in the ecotone zone. The stability of coarse inorganic grains (similar to the case of coarse organic matter), resulted from the stable current velocity in both sampling cycles.

In general, submerged macrophytes growing on the river bottom and emergent plants of the ecotone zone considerably determine most of the changes in the substrate particle size and accumulation of benthic POM on the bottom of medium and large rivers (Naiman et al. 1988, Hachmöller et al. 1991). The growth of submerged macrophytes in a river usually belongs to mitigating factors that reduce the current velocity and bed erosion (Kanehl et al. 1997, Pardo & Armitage 1997, Matthei et al. 1998), and contribute to an increased sedimentation of fine fractions of inorganic substratum and organic matter (Pardo & Armitage 1997, Asaeda et al. 2004, Collier 2004). During both permanent hydrological disturbances and renaturisation, a considerable part of the Drzewiczka River's bed was overgrown with submerged plants. During the Drzewieckie Lake functioning, H_M was the most stable and could be recognized as a refuge area for many invertebrates (Tszydel et al. 2004). After the dam removal and the return to the natural hydrological regime, the macrophyte patches moved partly from H_M to H_B. Although submerged macrophytes usually constitute a trap for both organic matter and fine fractions of inorganic substratum on the riverbed (Kanehl et al. 1997, Asaeda & Rashid 2012), no significant differences in benthic organic matter were noted in habitats with submerged plants during renaturisation of the Drzewiczka. This may partly result from the fact that the research was conducted throughout the year, while the plants were present only in the growing season, which considerably reduced the significance of differences they caused. The displacement of some submerged plants from H_M into H_B influenced only granulation of the inorganic substratum in the latter habitat (Asaeda et al. 2011). Thus, formation of new or reconstruction of existing patches of emergent and submerged plants can determine the extent and nature of changes caused by dam removal in particular habitats (Walker et al. 1986, Gurnell et al. 2004, Steiger et al. 2005).

Generally, periphyton amounts on the Drzewiczka River bed were comparable with those of many rivers in the temperate climatic zone (Blinn & Cole 1991). The growth of algae (periphyton and epiphyton) in rivers is promoted by high water transparency, temperature stability, high accessibility of biogens, absence of severe floods and droughts, and the presence of submerged macrophytes and coarser inorganic substrate particles (Blinn et al. 1998, Dodds et al. 1998). The bottom stability can also be of primary importance (Gresens & Lowe 1994, Cattaneo et al. 1997). The most suitable elements of substratum for the growth of periphyton are pebbles and cobbles, while silt and sand are less suitable for algae attachment (Cattaneo et al. 1997). On the one hand, emptying of the Drzewieckie Lake altered the cover of the Drzewiczka riverbed with submerged plants in H_M and resulted in a decrease in periphyton mass in this habitat in S2. On the other hand, the increased accumulation of fine POM in coarse-grained habitats (H_P and H_R) did not result in significant changes in the periphyton.

Previous studies suggest that the river administrators and the ecologists should seriously consider the risks and possible detrimental effects on tailwater/downstream areas during dam removal and river restoration. The ability to predict the effects of dam removal is increasingly important, as the number of such dam removal cases continues to grow. The potential impacts of dam removal are specific and can vary substantially depending on the size of impoundment and local environmental conditions. This paper complements other studies on the effects and efficiency of liquidation of small reservoirs used for the purposes of local recreation, mills, hydroelectric power plants and/or fish farms.

Conclusions

1. The permanent moderate hydropeaking caused by a small impoundment on the Drzewiczka River and the functioning of the whitewater slalom canoeing were conducive to considerable habitat differentiation in the tailwater.
2. After the dam removal, the mosaicity of habitats was generally preserved with the exception of the mid-channel macrophyte habitat, which lost its stability after emptying the reservoir when the amount of transferred organic matter increased. The riffle habitat seemed the most resistant, with high water flow and depth, and the bed covered with coarse bed substratum.
3. Among environmental parameters, the amount of oxygen dissolved in water changed most. The reason for this lies in the lack of intensive water discharges from the reservoir to the river after dam removal, which would result in high water turbulence and would induce an increased saturation with oxygen. The amount of oxygen dissolved in water also decreased due to the increased quantity of decomposing organic matter from exposed sediments, which was washed downstream into the river.
4. The dam removal and emptying of the Drzewieckie Lake did not have such a destructive impact on environmental parameters as was described for other removed dams. The time of complete emptying of a reservoir is crucial for ecologically sound dam removals. The Drzewieckie Lake was emptied at the beginning of the vegetation season, which allowed plants to totally cover the exposed bed within the first four months of the vegetation season and considerably reduce the downstream transfer of large amounts of organic matter accumulated on the reservoir's bed.
5. The non-catastrophic character of the dam removal also resulted from the fact that the emptying of the Drzewieckie Lake was not sudden (it lasted several months) and, thus, destabilization and increased transport of sole fine sediment POM was observed.
6. In conclusion, the dam removal process does not necessarily constitute an ecological disaster. During the planning process, the time of water release and the season should be taken into account. The best solution seems to be a slow

water release, and exposing the dam reservoir bed convergent with the beginning of the growing season so that the terrestrial vegetation can begin to develop on the reservoir's bed and reduce the downstream flow of organic matter accumulated there.

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