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Impact of the restoration of an incised mountain stream on habitats, aquatic fauna and ecological stream quality

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ABSTRACT

Mitigation of adverse effects of channel incision and the loss of alluvial bed substrate on the ecosystems of mountain watercourses downstream from check dams requires effective sediment entrapment in the incised channels. We examined changes in the ecosystem of mountainous Krzczonówka Stream, Polish Carpathians, resulting from lowering of a high check dam and installation of several block ramps in the downstream reach. Physical habitats, benthic macroinvertebrates and fish were examined in 10 cross-sections downstream from the dam over 5 years of the restoration project. Halfway into the restoration project, sediments flushed out by a flood from the reservoir of the lowered check dam were entrapped by block ramps, re-establishing alluvial bed substrate in the incised channel. Increases in bed elevation, bankfull channel width and near-bed flow velocity and a reduction in bankfull channel depth significantly changed physical habitat conditions. A significant increase in taxonomic richness of benthic macroinvertebrates was inversely related to a change in bankfull channel depth in given cross-sections. Fish species richness and the abundance of subadult and adult fish individuals did not increase, but the structure of fish community changed towards more natural one. Assessments of restoration effects on ecological stream quality performed with the invertebrate-based BMWP-PL index and the European Fish Index yielded different results, with the former indicating a significant improvement and the latter a lack of quality change. A combination of the applied measures appears useful in restoring the ecosystem of incised mountain streams, though effects on different groups of aquatic fauna may vary.

1. Introduction

Rapid channel incision during the twentieth century has been documented for numerous watercourses worldwide (Darby and Simon, 1999). Incision was especially common in European mountain streams and rivers (Rinaldi et al., 2013) as they have relatively high energy facilitating channel adjustments, usually were subjected to engineering modifications and at the same time experienced a decrease in sediment supply caused by land use changes in the catchments. Channel incision was also a principal process modifying the morphology of watercourses in the Polish Carpathians over the last century (Wyżga, 2008; Wyżga et al., 2016b; Hajdukiewicz et al., 2019). The incision had a negative influence on hydromorphological quality of the watercourses (Wyżga et al., 2009; Hajdukiewicz et al., 2019) and was reflected in the impoverishment of benthic macroinvertebrate and fish communities (Wyżga et al., 2009, 2013).

Engineering structures fragmenting the course of streams and rivers—such as dams, check dams and weirs—were another critical impact on river ecosystems (Belletti et al., 2020). These structures acted as barriers disrupting longitudinal connectivity of the ecosystems and prevented fish migration along the channels (Dynesius and Nilsson, 1994). In European mountain streams, this impact was predominantly related to check dams that were constructed to reduce sediment flux in the channels, which prior to the widespread 20th-century land use changes were supplied with substantial amounts of material from extensively deforested hillslopes (Rinaldi et al., 2013; Piton et al., 2016); here, this impact was especially detrimental because most species living in these streams migrate over large distances, particularly for spawning

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(Schlosser, 1991; Gido et al., 2015). In the Polish Carpathians, numerous closed check dams were constructed in the Raba River catchment (Fig. 1). Construction of check dams in the Polish Carpathians and other mountain areas in Europe coincided with a decline in agricultural use and reforestation of hillslopes (e.g., Boix-Fayos et al., 2008), which aggravated the sediment deficit downstream from the transversal structures. As the deficit intensified channel incision and induced transformation of alluvial beds into bedrock beds, negative impacts of check dams on stream biotic communities became apparent not only in the upstream reaches but also in downstream ones.

Currently many old dams and check dams are in poor technical condition and their reservoirs are filled with sediment (O'Connor et al., 2015)—as is the case with check dams in the Polish Carpathians (Ratomski, 1991). Many of such structures are being decommissioned worldwide (O'Connor et al., 2015; Foley et al., 2017), with their removals being a significant tool of river restoration (Magilligan et al., 2016a; Sneddon et al., 2017). The removal of transversal structures or their reconstruction to enable fish passage is the first step towards the restitution of riverine ichthyofauna (Doyle et al., 2005; O'Hanley et al., 2013; Foley et al., 2017). The release of coarse sediments previously stored in dam reservoirs may improve habitat conditions in deeply incised downstream reaches and have a positive effect on stream biotic communities in the long term (Tullos et al., 2014), but dam removals themselves and the resultant mobilization of impounded sediments constitute a disturbance to stream communities in the downstream reaches (e.g., Chiu et al., 2013; Chang et al., 2017; Foley et al., 2017). However, with only a small proportion of dam removals monitored, understanding of changes of stream ecosystem occurring during and after these restoration activities is incomplete (Hart et al., 2002; Foley et al., 2017).

As deeply incised mountain watercourses are typified by high values of bed shear stress at given flood discharges (Czech et al., 2016; Wyżga et al., 2021b), sediments released from dam reservoir might be transferred downstream without being retained in the incised channel below a decommissioned check dam. Sediment retention in such channels may be promoted by fish-passable, transversal structures constructed from rocky material that mimic riffles existing in equilibrium channels with alluvial beds. In the literature, such structures are described as rock riffles (Newbury et al., 2011; Newbury, 2013) or block ramps (Tamagni et al., 2010), but because of their artificial character, in this study we call them block ramps to avoid confusion with natural river features.

The monitoring of stream ecological conditions existing before and after implementation of restoration measures is fundamental to evaluation of the success of restoration projects (Wyżga et al., 2021a). It is known that particular groups of riverine biota differ in the level of sensitivity to a change in a given environmental factor such as hydromorphology (e.g., Hering et al., 2006; Yates and Bailey, 2011; Marzin et al., 2012). The Water Framework Directive of the European Union recommended four groups of biota for the bioassessment of watercourses (European Commission, 2000), of which benthic macroinvertebrates and fish are considered most useful to evaluate restoration-induced changes in the ecological quality of mountain

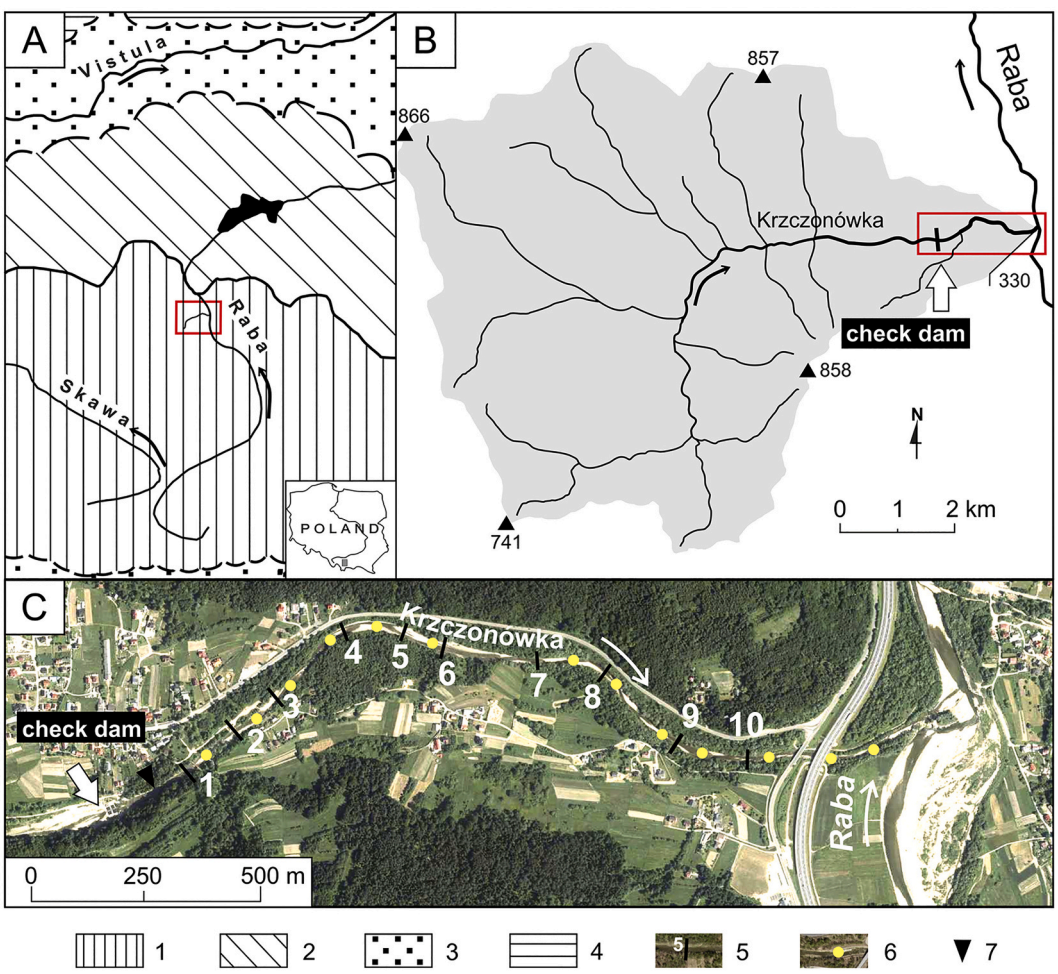

Fig. 1. (A) Location of Krzczonówka Stream in relation to physiographic regions of southern Poland. (B) Krzczonówka Stream catchment; (C) orthophoto from 2015 showing the studied stream reach and the location of surveyed cross-sections and block ramps constructed in 2013. 1 – mountains of intermediate and low height; 2 – foothills; 3 – intramontane and submontane basins; 4 – uplands; 5 – surveyed stream cross-sections; 6 – block ramps; 7 – water-gauge station.

streams. Establishing the response of stream biocoenosis to implemented restoration measures and the resultant change in ecological stream quality should thus be based on surveys of these two groups of biota (cf. Heino, 2010).

A combination of the decommissioning of old check dam and installation of block ramps in the downstream reach is a novel approach that may considerably increase beneficial effects of ecosystem restoration in deeply incised mountain streams as a result of enhanced effectiveness of the entrapment of sediments released from the dam reservoir. A restoration project realised in Krzczonówka Stream, Polish Carpathians, between 2012 and 2016 consisted in reduction of the height of a check dam and construction of several block ramps in the deeply incised downstream reach (Wyżga et al., 2021a, 2021b). This study aims to assess the influence of these restoration measures on the stream ecosystem downstream from the lowered check dam, including changes in physical habitat parameters, fish and benthic macroinvertebrate communities and ecological stream quality.

2. Material and methods

2.1. Geographical setting

Krzczonówka is a 17-km-long stream in the Outer Western Carpathians in southern Poland (Fig. 1). It drains a catchment 92.9 km^2 in area with low-mountain relief and altitudes ranging from 330 to 867 m (Fig. 1B). Local geology is typified by flysch complexes composed of alternating sandstones and shales with a secondary occurrence of marls and conglomerates.

Mean annual precipitation in the Krzczonówka catchment amounts to 800 mm (Niedźwiedź and Obrębska-Starklowa, 1991). Low water storativity of the flysch bedrock is reflected in highly variable stream discharge; at the Krzczonów water-gauge station situated 1.9 km upstream from the stream confluence with the Raba River (Fig. 1C), mean annual discharge is 1.52 m^3 s^{-1} and averages for the lowest and the highest annual discharges amount to 0.23 m^3 s^{-1} and 12 m^3 s^{-1}, respectively. Larger floods occur during late spring and summer; they are caused by a few days-long, advective rainfall of moderate (8–10 mm h^{-1}) intensity and the total sum of precipitation of 200–300 mm (cf. Wyżga et al., 2016a).

The study was carried out in the lowest, 2-km-long stream reach that experienced considerable human impacts during the twentieth century and was subjected to restoration activities in recent years. In the years 1935–1951 a closed check dam with a height of 3.7 m was built at a distance of 2 km from the stream mouth (Fig. 1C). This resulted in a persistent depletion of sediment in the downstream reach. Channelization works encompassing training of the stream with groynes and construction of gabions or rip-rap along concave banks were conducted since the late 1950s, resulting in up to a threefold narrowing of the active channel until the mid-2000s (Lenar-Matyas et al., 2015). Sediment starvation combined with increased transport capacity of the stream as a result of its channelization induced rapid bed degradation. Until the early 2010s up to 2 m of channel incision occurred and the alluvial channel bed was transformed into a bedrock–alluvial or bedrock bed.

2.2. River restoration activities in the stream

In 2012 a decision was made to lower a check dam on Krzczonówka Stream to enable migration of fish through the structure. The dam was not completely removed because of concerns that this would have increased lateral channel migration upstream from the dam potentially threatening terrace stability and the existing settlements and road on the banks. To promote entrapment of the sediment released from the dam reservoir in the deeply incised downstream reach, construction of block ramps in the incised channel was arranged before the check-dam lowering. As the ramps would represent protruding channel features mimicking riffles, they were expected to reduce flow capacity of the channel, while remaining relatively stable. To determine the grain size of material for the ramps and their geometric parameters (average spacing along the channel, minimum bankfull width and average and maximum bankfull depth at the ramp crest) that would ensure persistence of the ramps at variable rates of bedload supply from the upstream, simulations with the Hey and Thorne regime equations (Thorne et al., 1997) were performed. The results showed that the material ca. 3–4 times coarser than that stored upstream from the check dam should be used for construction of the ramps (Jeleński et al., 2016).

Block ramps were installed in the stream in March 2013. Material used to form the ramps was sandstone rubble (sourced from a local quarry) that was covered with a layer of coarse gravel and mechanically compacted. The ramps were placed at thalweg inflection points, similar to natural riffles, at an average distance of 120 m along the channel. The exception was the middle part of the reach, where installation of ramps was inhibited by the lack of access for heavy machinery; here, the spacing between succeeding ramps was 325 m (Fig. 1C). The ramps were concave in cross-section to concentrate flow in the central part of the channel (Fig. 2A). The slope of their downstream side was a few times steeper than the average channel slope and the ramps impounded low to medium flows on their upstream side (Fig. 2A).

The check dam was rebuilt between April and October 2014; the crest of the structure and its total height were lowered by 1.7 m. The rebuilt check dam, with a total height of 2 m, is composed of three weirs with a trapezoidal notch ca. 0.4 m deep dissecting the central part of each weir to facilitate fish migration through the structure. During check dam reconstruction a moderate flood occurred (May 2014), flushing out a substantial amount of gravelly material from the dam reservoir and depositing it in the downstream reach of Krzczonówka Stream.

Fig. 2. View of Krzczonówka Stream in the vicinity of cross-section K4 shortly after the installation of a block ramp in 2013 (A) and after the passage of the flood of May 2014 (B). Note the entrapment of a considerable amount of gravel that buried the block ramp.

2.3. Study methods

2.3.1. Surveys of physical habitat characteristics

In 2012, at the beginning of the restoration project, 10 stream cross-sections located between 180 and 1810 m downstream from the check dam were delimited for the study (Fig. 1C). The cross-sections run across pools at half-distance between the planned location of block ramps, and their positions were kept unchanged throughout the restoration project, even though the actual location of some ramps was adjusted to the potential access of heavy machinery to the stream. Channel morphology, hydraulic conditions at base flow and bed-material grain size in the cross-sections were surveyed each summer between 2012 and 2016. Elevation profiles of the cross-sections were surveyed with a Topcon AT-B4 optical level, with measurement points within low-flow channels spaced 0.5 m apart. Water depth, depth-averaged and near-bed flow velocity and grain size of surface bed material were subsequently established at these measurement points and low-flow channel width was measured. Flow velocity was measured 1 cm above bed surface (near-bed velocity) and at 0.6 of the depth (depth-averaged velocity) using a Valeport model 801 electromagnetic current meter. Transect sampling of surface bed material (Wolman, 1954) that produces results equivalent to bulk sieve analysis (Diplas and Sutherland, 1988) was used to establish grain size of gravelly substrate, with 15 particles measured at each measurement point within the area characterized by the velocity and depth measurements. Samples of sand and mud deposits were transported to a laboratory to establish their grain-size distribution through either sieving (sands) or hydrometer (muds) analyses. Median (D_{50}) grain size of gravelly and fine-grained samples was determined from their grain-size distribution. Means and coefficients of variation of flow velocity, water depth and bed-material grain size were next calculated for each study cross-section. In each cross-section, a first break in the cross-section profile below permanent vegetation was used to identify bank edge (Radecki-Pawlik, 2002), and bankfull channel width and depth were determined.

2.3.2. Benthic macroinvertebrate sampling

Each year between 2012 and 2016 sampling of benthic macroinvertebrates was carried out in the study cross-sections on a seasonal basis, i.e. in spring (late April–May), summer (July) and autumn (September), because many events in the life history of aquatic invertebrates, particularly insects, are triggered by seasonal changes in environmental conditions including water temperature (Ward and Stanford, 1982; Jackson et al., 2007). In each channel cross-section, samples were collected at 3 sites representing prime, visually determined habitat conditions (combinations of flow velocity, water depth and substrate type). At each sampling site, we collected invertebrates from approximately 0.25 m^2 of the channel bed, and sampling time was kept similar for all collected samples (cf. Fiałkowski et al., 2005). Invertebrates were collected with a bottom scraper, a triangular dip net with 500 μm mesh size, a mosquito dipper and tweezers (to gather the sprawlers from cobbles), with particular techniques chosen depending on substrate type (cf. McCafferty, 1998). Using a range of techniques of macroinvertebrate collection from different substrate types (from cobbles to mud) provided reliable data to determine taxonomic richness of invertebrate community but did not allow for analysis of invertebrate abundance.

During the survey and transit from the field, samples were kept at about 3 °C in order to minimize animal predation. Invertebrates were identified mostly from unpreserved material within 2–3 days after sampling, and partly from samples preserved with 70% ethanol. All the specimens were identified with the key of Kołodziejczyk and Koperski (2000) to the lowest unambiguously identifiable level.

2.3.3. Electrofishing

In the period 2012–2016 species composition and abundance of fish community in the study cross-sections were investigated twice a year: in the season of the best thermal conditions (July) and the season with advanced development of juvenile fish (September). A backpack pulsed-DC electroshocker (IUP-1,2; Radet, Poland) was used to conduct fish sampling. Electrofishing was conducted wading upstream within a 10-m-wide strip across each cross-section. Captured fishes were identified and classified into two age categories: young-of-the-year (YOY) and subadult/adult individuals (1-year old and older) on the basis of their total length. Based on available data on the age and growth of fishes occurring in the study area (Brylińska, 2000), the total length threshold between YOY and older individuals was assumed to be 10–12 cm for brown trout and 5–7 cm for other species. The number of fish recorded in a cross-section was considered to represent the standard CPUE (catch per unit effort).

2.3.4. Establishing ecological stream quality

For each study cross-section, scores of the BMWP-PL index (Dumnicka et al., 2006)—a Polish variant of the Biological Monitoring Working Party Score System (Hawkes, 1998)—were calculated on the basis of individual samplings of benthic macroinvertebrates. This index was originally considered to represent river water quality, but subsequent studies demonstrated that it should rather be regarded as an indicator of the ecological status of watercourses, dependent on their water-quality and hydromorphological characteristics (Wyżga et al., 2013). Scores of the index derived from data from spring, summer and autumn surveys of each year were averaged, providing a final score indicating a particular class of ecological quality represented by a study cross-section in a given year.

Data from individual electrofishings were used to calculate scores of the European Fish Index (EFI+) for each study cross-section. This index was calculated with an online software which compares the composition and the size structure of recorded fish assemblages with those expected to occur at a given site based on geographical location and local environmental conditions (EFI, 2009). Averaging of the scores based on summer and autumn electrofishings yielded a final score for a given year, which was next used to allocate given cross-sections to relevant classes of ecological quality.

2.3.5. Data analysis

Statistical significance of the differences in physical habitat parameters, taxonomic richness of benthic macroinvertebrates, abundance and species richness of fish, and scores of the BMWP-PL and EFI+ indices among different years of the restoration project was verified with a nonparametric Friedman test (Friedman, 1937), while significance of the differences between pairs of the years was determined with a Fischer's least significant difference *post hoc* test (Carmer and Walker, 1985).

The analysis of differences in mean elevation of the channel bed in the cross-sections between different years of the project required a special approach. If the analysis focused on differences in the relative bed elevation between different years, all values in the initial sample would be zero and the lack of variance in the sample would prevent statistical testing. Therefore, we estimated a linear trend of bed elevation in the study reach during the first year of the project on the basis of all cross-sections, and mean elevation of the bed in each cross-section in relation to the elevation shown by this trend was subsequently computed for different years of the project and compared between these years. On long distances, longitudinal elevation profiles of rivers typically have a concave shape (Wheeler, 1979), but in the 1.8-km-long study reach of Krzczonówka Stream a downstream decrease in bed elevation was very well described by the linear trend ($R^2 = 0.99$, $p = 0.000001$). Differences and simple linear regression models examined in the study were regarded statistically significant if p value was <0.05.

Relationships between changes in either the taxonomic richness of macroinvertebrates or the macroinvertebrate-based index of ecological quality and changes in physical habitat parameters recorded between the first and the last year of the restoration project were first examined using a stepwise multiple regression analysis with forward selection of

variables. Subsequently, a redundancy analysis (RDA) with forward selection of variables (ter Braak and Prentice, 1988) was performed to explain the combined pattern of variability in the temporal changes of taxa richness and BMWP-PL scores among the study cross-sections by temporal changes in physical habitat variables. As multiple regression and redundancy analyses examine relationships with multiple independent variables, a significance level for the inclusion of these variables in the final models was set at 0.1.

3. Results

3.1. Changes in physical habitat characteristics

During the flood of 2014, ca. 15,650 m^3 of the gravelly sediments released from the dam reservoir were deposited in the downstream reach of the stream. In turn, the amount of bed material deposited downstream of the check dam between 2012 and 2016 was estimated at ca. 16,600 m^3. Not only did the sediment deposition re-establish the occurrence of an alluvial bed in the whole downstream reach, burying bedrock exposures on the channel bed, but it also buried block ramps (Fig. 2B) on a distance of 1.2 km from the check dam (Wyżga et al., 2021b).

Because the flood of 2014 and the associated entrapment by block ramps of the sediment released from the lowered check dam caused the most important habitat changes over the project duration, changes in physical habitat parameters in the study cross-sections were compared between 2012 (the beginning of the restoration project), 2014 (after the sediment release from the check dam) and 2016 (the end of the project). The comparison indicated that only a proportion of the analysed parameters differed significantly among the years (Fig. 3, Table 1). The most significant were differences in mean elevation of the channel bed (Friedman test, $p = 0.0005$), which reflected an increase in bed elevation by 0.57 m on average between 2012 and 2014 (Fischer's LSD test, $p = 0.001$) followed by a slight reduction of the elevation by 0.06 m between 2014 and 2016 (Fig. 3, Table 1). Bed aggradation in the years 2012–2016 was larger close to the check dam (Fig. 4), with the average change in bed elevation in cross-sections 1–5 amounting to 0.68 m and that in cross-sections 6–10 to 0.34 m (Table S1 in the Supplementary data). However, the smallest increase in bed elevation was recorded in cross-section K6 (Table S1) located at a considerably larger distance from the downstream block ramp than the remaining cross-sections.

Table 1
Average values of analysed physical habitat parameters in the study cross-sections of Krzczonówka Stream in 2012, 2014 and 2016, and results of a Friedman test for significance of the differences of the parameters between the years. p values < 0.05 are indicated in bold.

Physical habitat parameter	Mean value			Significance of Friedman test
	2012	2014	2016	
Bed elevation in relation to downstream trend line (m)	0.00	0.57	0.51	**$p = 0.0005$**
Bankfull channel depth (m)	1.42	1.01	1.03	**$p = 0.0006$**
Bankfull channel width (m)	22.5	23.3	23.8	**$p = 0.003$**
Low-flow channel width (m)	12.1	11.2	10.9	$p = 0.72$
Flow depth: mean (m)	0.13	0.12	0.12	$p = 0.30$
Flow depth: coefficient of variation	0.492	0.638	0.561	$p = 0.90$
Near-bed velocity: mean (m s^{-1})	0.15	0.25	0.23	**$p = 0.02$**
Near-bed velocity: coefficient of variation	0.684	0.533	0.478	$p = 0.15$
Depth-averaged velocity: mean (m s^{-1})	0.26	0.32	0.31	$p = 0.27$
Depth-averaged velocity: coefficient of variation	0.557	0.482	0.418	$p = 0.20$
Bed-material grain size: mean (mm)	51.8	44.1	51.1	**$p = 0.025$**
Bed-material grain size: coefficient of variation	0.245	0.236	0.254	$p = 1.00$

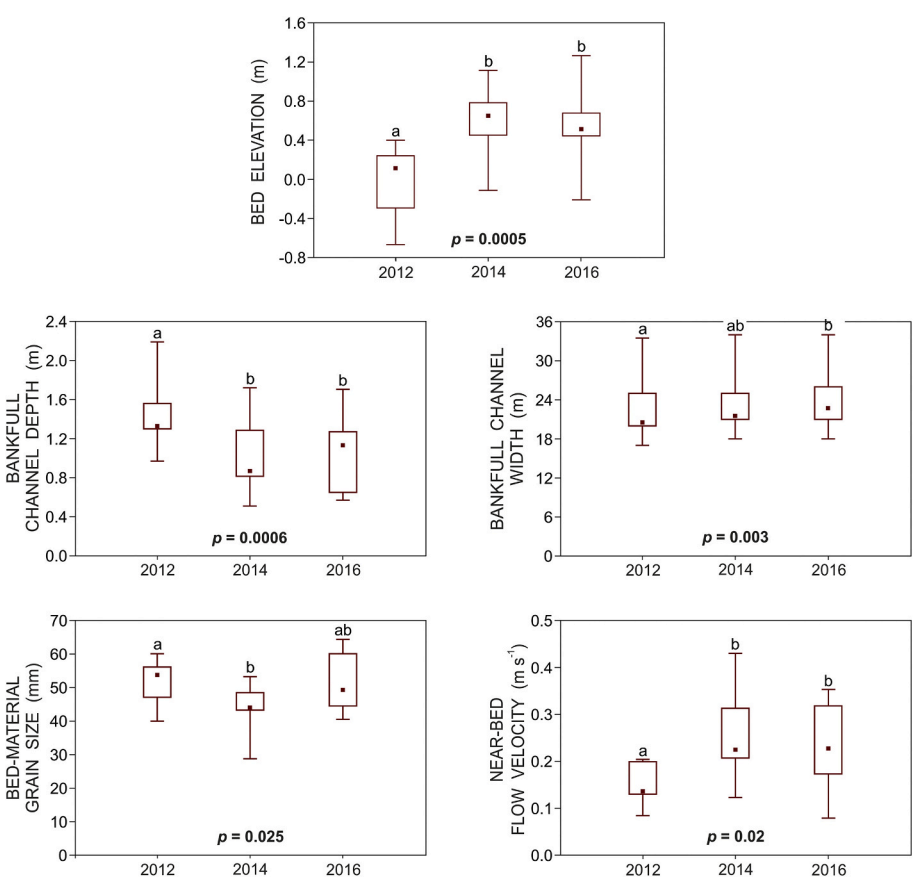

Fig. 3. Changes of selected physical habitat parameters of Krzczonówka Stream in the surveyed cross-sections in the years 2012–2016. Boxplots present median (squares), the first and the third quartiles (bottom and top of boxes) and extreme values (whiskers). Statistical significance of the difference of the parameters between the analysed dates, determined by a Friedman test, is indicated. Different letters signify statistically different samples indicated by a Fischer's LSD *post hoc* test.

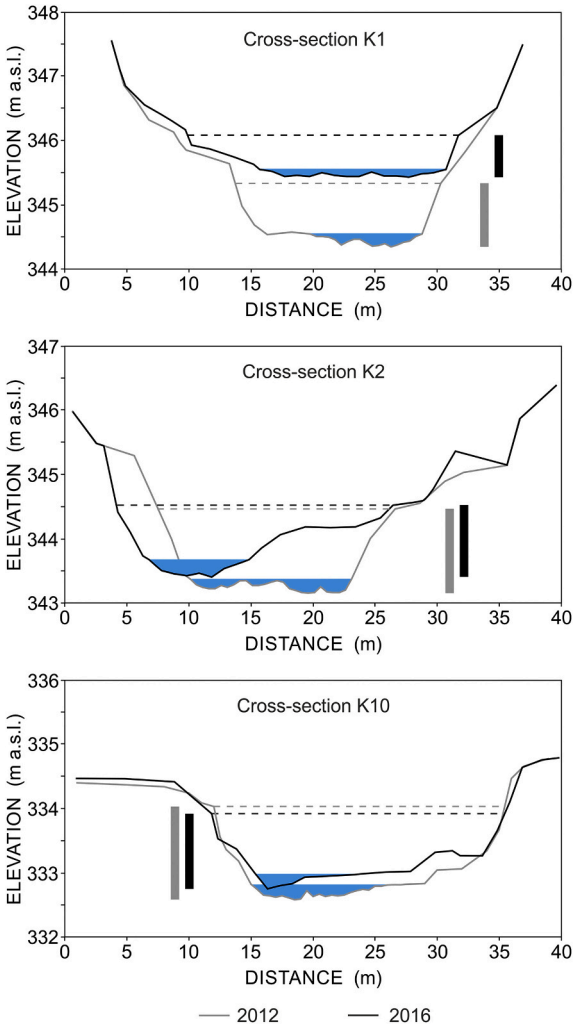

Fig. 4. Examples of changes in channel geometry, vertical position of the channel bed and bankfull channel depth in the study cross-sections of Krzczonówka Stream between the first (2012) and the last year (2016) of the restoration project. Horizontal dashed lines indicate the position of bankfull water stage and vertical columns depict bankfull channel depth in 2012 (grey) and 2016 (black). In cross-section K1, a large increase in bed elevation was associated with a moderate reduction of channel depth. Cross-section K2 experienced moderate bed aggradation coupled with a small reduction of channel depth, whereas cross-section K10 was typified by small changes in bed elevation and channel depth.

Changes in bankfull channel depth were also highly significant statistically (Friedman test; $p = 0.0006$); this reflected a decrease in mean value of the parameter from 1.42 m in 2012 to 1.01 m in 2014 (Fischer's LSD test; $p = 0.003$) followed by its slight increase to 1.03 by 2016 (Fig. 3, Table 1). Notably, changes in bankfull channel depth were not significantly correlated with changes in bed elevation ($p = 0.30$) as a result of variable combination of changes in these parameters among the study cross-sections (Fig. 4, Table S1).

Significant differences between the years were also found for bankfull channel width (Friedman test, $p = 0.003$), bed-material grain size ($p = 0.025$) and near-bed flow velocity ($p = 0.02$) (Fig. 3, Table 1). Mean value of bankfull channel width for the study cross-sections gradually increased from 22.5 m in 2012 to 23.3 m in 2014 and 23.8 m in 2016; consequently, channel widths in 2016 significantly differed from those recorded in 2012 (Fischer's LSD test, $p = 0.03$), whereas no statistically significant differences in width values occurred between the pairs of consecutive surveys (Fig. 3, Table 1). Intense deposition of bed material by the flood of 2014 was associated with a reduction in the average value of median grain size for the study cross-sections from 51.8 mm in 2012 to 44.1 mm in 2014, which was followed by an increase of the median grain size to 51.1 mm during the next two years (Table 1). These changes were reflected in a significant difference of the bed-material grain size between 2012 and 2014 (Fischer's LSD test, $p = 0.04$) and a lack of statistically significant differences between the other pairs of surveys (Fig. 3). In turn, mean value of near-bed velocity for the study cross-sections increased from 0.15 m s^{-1} in 2012 to 0.25 m s^{-1} in 2014 and subsequently slightly decreased to 0.23 m s^{-1} in 2016 (Table 1). The parameter values from 2014 and 2016 were significantly different from that recorded in 2012 (Fischer's LSD test, $p = 0.02$).

3.2. Changes in benthic invertebrate assemblages

During 15 surveys performed over five years of the restoration project, a total of 50 benthic macroinvertebrate taxa were found in 10 study cross-sections of Krzczonówka. Fifteen taxa were identified at the species level, twenty four at the genus level, ten at the family level and one at the phylum level (Tables S2–S16). Total numbers of taxa found in the study cross-sections in individual years varied relatively little between 31 and 36, with the lowest number recorded in 2014. However, yearly averages of the number of taxa recorded in individual cross-sections varied considerably between 10.3 and 22.3 (Table S17, Fig. 5), with the differences among individual years of the restoration project being highly significant statistically (Friedman test, $p = 0.00001$). Similar to the total number of taxa, the lowest yearly average was also recorded in 2014. The highest yearly average was recorded in 2016; it was more than twice greater than in 2014 and exceeded by two-thirds the value recorded in 2012 (22.3 versus 13.4), before the onset of restoration activities, with the difference being statistically significant (Fischer's LSD test, $p = 0.004$) (Fig. 5). Especially large increases in

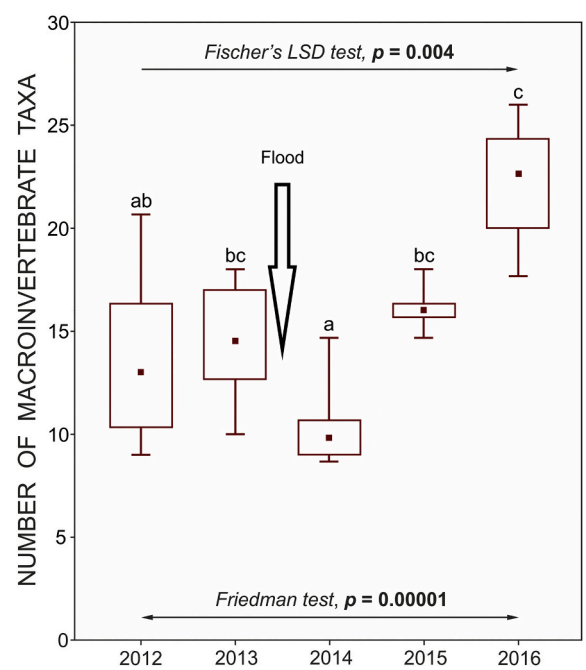

Fig. 5. Changes in the number of benthic macroinvertebrate taxa in the surveyed cross-sections of Krzczonówka Stream in the years 2012–2016. Boxplots present median (square), the first and the third quartiles (bottom and top of boxes), and minimum and maximum values (whiskers). Statistical significance of the differences between all study years is indicated by a Friedman test. Different letters signify statistically different samples indicated by a Fischer's LSD *post hoc* test and the significance of the difference between 2012 and 2016 indicated by this test is shown. Statistically significant differences are indicated in bold. The vertical arrow points to the occurrence of a flood of moderate magnitude preceding the surveys in 2014.

invertebrate richness between 2012 and 2016 were recorded in cross-sections K1, K4 and K9 (Table S17).

Analysis of the occurrence of benthic invertebrate taxa in 10 study cross-sections during three samplings in 2012 and 2016 indicated 17 taxa with at least ten occurrences more in the last than in the first year of the restoration project (Table 2). Notably, they encompassed (i) rheophilic taxa characteristic of habitats with coarse substrate and well oxygenated water: winter stoneflies, Capniidae; larvae of Goeridae; and river limpet, *Ancylus fluviatilis*; (ii) limnophilic taxa characteristic of habitats with fine-grained substrate: horse-leach, *Haemopsis sanguisuga*; pond snail, *Lymnaea* sp.; and (iii) rheo–limnophilic taxa: e.g. snail leach, *Glossiphonia complanata*; burrowing mayfly, *Ephemera* sp.; and faucet snail, *Bithynia tentaculata* (Table 2).

3.3. Changes in fish assemblages

Ten electrofishings performed over five years of the project indicated a permanent occurrence of five fish species in the study cross-sections of the stream: stone loach, *Barbatula barbatula*; Carpathian barbel, *Barbus carpathicus*; chub, *Squalius cephalus*; Eurasian minnow, *Phoxinus phoxinus*; brown trout, *Salmo trutta*; completed with a single record of gibel, *Carassius gibelio* (Table S18). The first five species are native and typical of Polish Carpathian streams and rivers, whereas the occurrence of alien and invasive gibel in cross-section K2 in July 2013 was considered incidental and this species was excluded from further analyses. Number of species found in given cross-sections in individual years varied between 2 and 5, with the lowest number recorded three times in cross-section K6 and once in cross-section K7 (Table S18). Maximum species richness was recorded in 14 out of 100 electrofishings conducted and was found in every cross-section except K6, but most often in cross-section K9. Averaged number of subadults and adults recorded in individual cross-sections varied considerably over the five years between 41 and 516.5 (Table S19). The lowest yearly average of the abundance of this age category for all study cross-sections, 134.5, was recorded in 2014, while nearly the same high values in 2012 and 2016 (229.4 and 227.8 individuals, respectively) (Table S19). However, no significant differences in the average abundance of subadults and adults occurred between five years of the restoration project (Friedman test, $p = 0.11$) (Fig. 6). In contrast, yearly averages of the abundance of juveniles for all study cross-sections differed significantly among individual years of the project (Friedman test, $p = 0.0001$). High yearly averages of juvenile abundance were recorded in 2012 and 2015 (344.3 and 368.8, respectively), while relatively low ones in 2013, 2014 and 2016 (136.9–156.5) (Table S19).

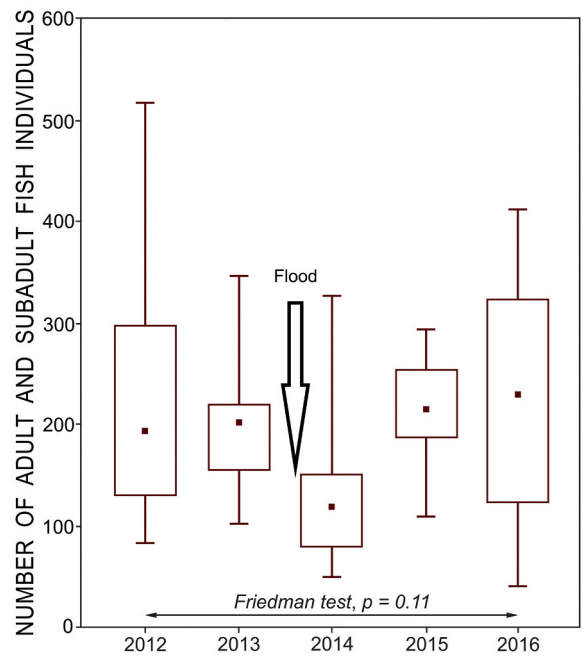

Fig. 6. Changes in the number of subadult and adult fish individuals caught in the surveyed cross-sections of Krzczonówka Stream in the years 2012–2016. Boxplots present median (square), the first and the third quartiles (bottom and top of boxes), and minimum and maximum values (whiskers). Statistical significance of the differences between all study years is indicated by a Friedman test. The vertical arrow points to the occurrence of a flood of moderate magnitude preceding the surveys in 2014.

Data for individual fish species indicate that relatively high abundance of stone loach, chub, Eurasian minnow and brown trout in the study cross-sections of Krzczonówka was found in 2012 and 2015 (Fig. 7). In 2013, only single individuals of chub were caught. Minnow were evident in high numbers during the whole period of the restoration project (Table S18). After completion of the restoration works,

Table 2
Benthic macroinvertebrate taxa with at least ten occurrences in the study cross-sections more in 2016 than in 2012 (against the maximum possible number of occurrences in 10 cross-sections during three samplings in a given year equal 30) and their habitat preferences.

Taxon	Number of occurrences in the study cross-sections		Surplus of occurrences in 2016 above those in 2012	Habitat preferences
	2012	2016		
Dendrocoelum lacteum	12	25	13	Rheo–limnophilic[c]
Dugesia sp.	3	18	15	Rheo–limnophilic
Nematoda	14	29	15	Rheo–limnophilic
Lumbricidae	12	28	16	Rheo–limnophilic
Glossiphonia complanata	8	25	17	Rheo–limnophilic
Haemopis sanguisuga	1	15	14	Limnophilic[b]
Capniidae	14	25	11	Rheophilic[a]
Ephemera sp.	10	23	13	Rheo–limnophilic
Potamanthus luteus	1	14	13	Rheo–limnophilic
Goeridae	8	20	12	Rheophilic
Polycentropus sp.	16	26	10	Rheophilic
Ryacophila sp.	17	29	12	Rheophilic
Chironomidae	16	30	14	Rheo–limnophilic
Ancylus fluviatilis	10	28	18	Rheophilic
Radix sp.	–	17	17	Rheo–limnophilic
Lymnaea sp.	–	15	15	Limnophilic
Bithynia tentaculata	8	25	17	Rheo–limnophilic

[a] Rheophilic taxa – occur in streams, prefer zones with moderate to fast water current.
[b] Limnophilic taxa – prefer standing waters but commonly occur in slowly flowing watercourses.
[c] Rheo–limnophilic taxa – usually occur in streams, prefer slowly flowing watercourses and lenitic zones, but occur also in standing waters.

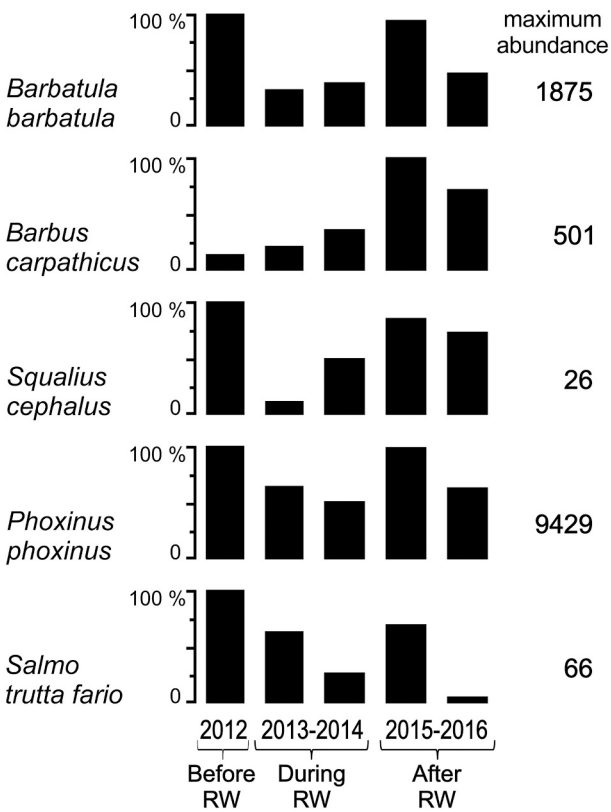

Fig. 7. Changes of total abundance of fish species in the study cross-sections of Krzczonówka Stream between consecutive years of the restoration project. The fish abundance is shown as a percentage of the maximum abundance of a given species during the five years of the project. RW denotes restoration works.

Carpathian barbel became considerably more abundant than before and during these activities, with the highest number of individuals caught in 2015 and somewhat lower one in 2016. In 2016, only single brown trouts were caught (Fig. 7).

3.4. Changes in invertebrate- and fish-based assessments of ecological stream quality

The formula for the invertebrate-based BMWP-PL index of the ecological quality of watercourses takes into account only selected invertebrate families (Dumnicka et al., 2006), therefore, some of the taxa found in the study cross-sections of Krzczonówka were not included in its calculation (Tables S2–S16). A comparison of the average scores of the index calculated on the basis of three surveys in each year (Table S17) indicated significant differences between the five years of the restoration project (Fig. 8; Friedman test, $p = 0.00001$). A Fischer's LSD test indicated that between 2012 and 2015 pairs of the years did not differ significantly in the average index score for the study cross-sections, although in 2014 the average score of 56.5 was markedly lower than those in the remaining three years (73.3–77.5). In these four years, the average score of the index indicated either moderate or good ecological quality of the stream (Fig. 8). However, in 2016 the average index score of 108.4 indicated high ecological quality of the study reach. This score significantly differed from the index scores in the preceding years, including the score obtained in the first year of the project (Fischer's LSD test, $p = 0.01$) (Fig. 8).

The European Fish Index (EFI+) calculated on the basis of summer and autumn electrofishings in each year of the restoration project revealed a different pattern of change in the ecological stream quality. The average index scores for the study cross-sections varied over the project duration between 0.909 and 0.952 (Table S19) and did not differ

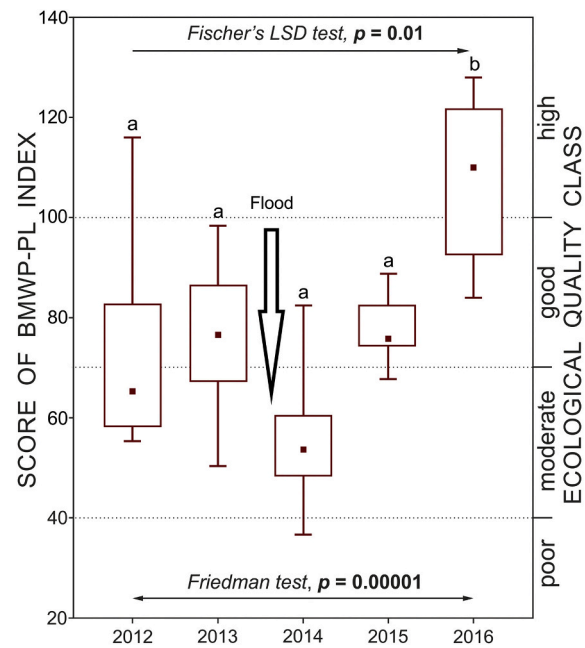

Fig. 8. Changes in the average score of the macroinvertebrate-based BMWP-PL index computed for the surveyed cross-sections of Krzczonówka Stream in the years 2012–2016. Boxplots present median (square), the first and the third quartiles (bottom and top of boxes), and minimum and maximum values (whiskers). Statistical significance of the differences between all study years is indicated by a Friedman test. Different letters signify statistically different samples indicated by a Fischer's LSD *post hoc* test and the significance of the difference between 2012 and 2016 indicated by this test is shown. The boxplots are presented on the background of ecological quality classes ascribed to particular scores of the BMWP-PL index. Statistically significant differences are indicated in bold. The vertical arrow points to the occurrence of a flood of moderate magnitude preceding the surveys in 2014.

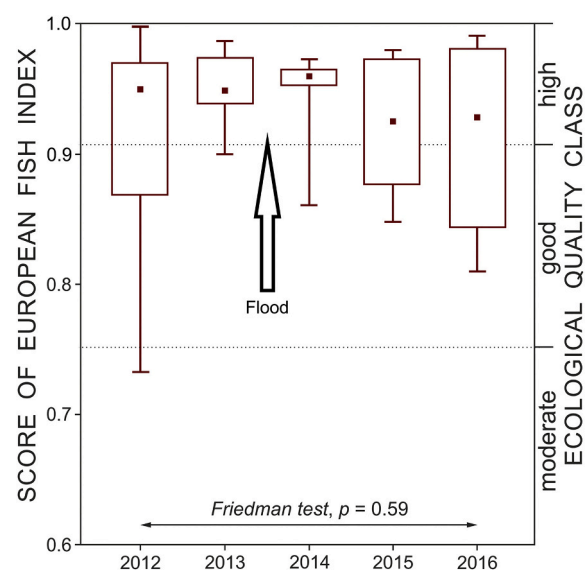

Fig. 9. Changes in the average score of the European Fish Index computed for the surveyed cross-sections of Krzczonówka Stream in the years 2012–2016. Boxplots present median (square), the first and the third quartiles (bottom and top of boxes), and minimum and maximum values (whiskers). Statistical significance of the differences between all study years is indicated by a Friedman test. The boxplots are presented on the background of ecological quality classes ascribed to particular scores of the EFI+ index. The vertical arrow points to the occurrence of a flood of moderate magnitude preceding the surveys in 2014.

significantly between the years (Friedman test, $p = 0.59$). Moreover, they consistently indicated high ecological quality of the stream reach (Fig. 9).

3.5. Explanation of changes of invertebrate taxa richness and BMWP-PL index by changes of physical habitat parameters

With the lack of significant changes of the fish-based index of the ecological quality of Krzczonówka, examination of factors responsible for restoration-induced changes of the stream communities focused on potential relationships between changes of the taxonomic richness of benthic invertebrates and the invertebrate-based index of ecological quality, and changes of physical habitat parameters recorded in the study cross-sections between the first and the last year of the restoration project. A stepwise multiple regression analysis indicated that the change of invertebrate taxa richness was inversely related to a relative change of bankfull channel depth and directly related to changes of low-flow channel width, cross-sectional variation in flow depth, bankfull channel width and near-bed velocity (Table 3). As indicated by its highest standardized regression coefficient, change of the cross-sectional variation of flow depth had the greatest strength in explaining the variation in the change of invertebrate taxa richness among the study cross-sections (Table 3). In turn, the change of the BMWP-PL index was inversely related to a relative change of bankfull channel depth and directly related to changes of bankfull channel width, cross-sectional variation in flow depth and near-bed velocity (Table 3). Out of these explanatory variables, change of bankfull channel width had the greatest influence on the variation in the change of the index scores among the cross-sections (Table 3).

A redundancy analysis was then applied to explain a combined pattern of the variability in changes of invertebrate taxa richness and scores of the BMWP-PL index among the study cross-sections, and a stepwise forward selection of physical habitat parameters in the RDA indicated relative change of bankfull channel depth as the only significant ($p = 0.08$) explanatory variable (Fig. 10). This analysis indicated that the first ordination axis described 32.5% of variance of the dependent variables and 100% of variance of the relationship between changes of invertebrate taxa richness and the BMWP-PL index, and independent variables. The second ordination axis explained 64.9% of the variation in the dependent variables but did not add to the explanation of their relationship with changes of physical habitat parameters.

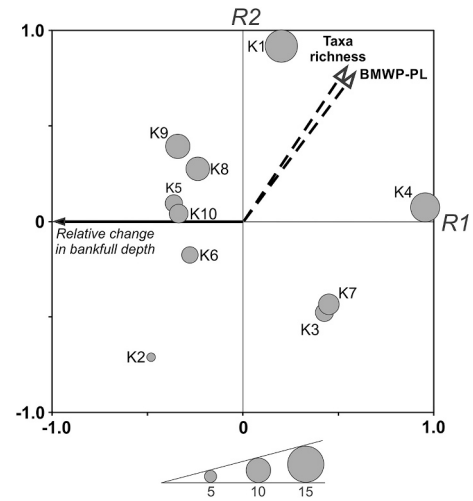

Fig. 10. Results of a redundancy analysis performed on the changes recorded between 2012 and 2016 in macroinvertebrate taxa richness in 10 study cross-sections of Krzczonówka Stream and in the score of the BMWP-PL index ascribed to the cross-sections. Positions of the vectors of the dependent variables on the two first RDA axes are indicated by dashed arrows and that of a significant explanatory variable by a solid line. Circles show the positions of cross-sections on the R1 × R2 plane and circle size is proportional to the increase in macroinvertebrate taxa richness in a given cross-section between 2012 and 2016.

4. Discussion

This study has analysed how the lowering of a closed check dam and the construction of block ramps in the deeply incised downstream reach of mountainous Krzczonówka Stream affected physical habitat conditions, stream communities and ecological quality of the stream during the time span of the restoration project. Changes expected to arise from the applied restoration measures were dependent on the release of impounded gravels from the reservoir of the decommissioned check dam, which typically occurs during high flows able to mobilize coarse-grained sediments (e.g., Pizzuto, 2002; Wang and Kuo, 2016). In Krzczonówka, such a release took place in the middle of the project

Table 3
Results of multiple regression analysis for the relationships between changes in (i) macroinvertebrate taxa richness of the study cross-sections of Krzczonówka Stream and in (ii) the score of the macroinvertebrate-based BMWP-PL index computed for these cross-sections, and changes in physical habitat parameters recorded in the cross-sections between the first (2012) and the last year (2016) of the restoration project. The regression models were obtained with a stepwise analysis with forward selection of explanatory variables.

Explained variable and significance of the final model	Explanatory variable	Regression coefficient	Standardized regression coefficient	Explained variance after including the variable in the model	Significance of variable in the model
Change in taxa richness $p = 0.002$	Intercept	2.78			$p = 0.03$
	Relative change in bankfull channel depth	−15.1	−0.47	$R^2_{adj.} = 0.22$	$p = 0.003$
	Change in low-flow channel width	0.43	0.63	$R^2_{adj.} = 0.35$	$p = 0.006$
	Change in the variation of flow depth	12.7	0.78	$R^2_{adj.} = 0.78$	$p = 0.002$
	Change in bankfull channel width	1.17	0.61	$R^2_{adj.} = 0.87$	$p = 0.008$
	Change in near-bed flow velocity	12.1	0.35	$R^2_{adj.} = 0.95$	$p = 0.04$
Change in the score of BMWP-PL index $p = 0.005$	Intercept	−17.4			$p = 0.06$
	Relative change in bankfull channel depth	−72.6	−0.55	$R^2_{adj.} = 0.35$	$p = 0.007$
	Change in bankfull channel width	12.3	1.04	$R^2_{adj.} = 0.50$	$p = 0.002$
	Change in the variation of flow depth	54.5	0.55	$R^2_{adj.} = 0.71$	$p = 0.01$
	Change in near-bed flow velocity	127.9	0.61	$R^2_{adj.} = 0.93$	$p = 0.01$

duration, when the works leading to the lowering of the check dam were still underway. Gravels flushed out from the dam reservoir by the flood of May 2014 were entrapped by block ramps, which resulted in deposition in the downstream reach of 8.3 m^3 of bed material per 1 m of channel length on average (Wyżga et al., 2021b), marked bed aggradation and transformation of the previous bedrock–alluvial or bedrock bed into a gravelly bed. Although the amount of flood-induced bed aggradation generally decreased with increasing distance from the dam (cf. Wang et al., 2014; Magilligan et al., 2016b), block ramps must have played an important role in forcing sediment deposition as indicated by a small increase in bed elevation in the middle part of the study reach with the largest span between sequential ramps. Pools formed on the upstream side of block ramps apparently facilitated deposition of the sediments released from the dam reservoir, similarly to pools between natural riffles (cf. Wohl and Cenderelli, 2000), whereas in the middle part of the reach—on a moderately steep run (cf. Jowett, 1993) not impounded by the downstream ramp—sediment retention could hardly occur in the incised channel.

Although fish-passable grade-control structures may considerably enlarge aquatic habitats in incised streams through the formation of backwatering pools (e.g., Shields et al., 1995, 1998; Newbury, 2013), changes in physical habitat conditions recorded over the project duration reflected an overwhelming influence of the release of sediments from the dam reservoir and their entrapment in the downstream reach because the sediments buried the block ramps along most of the reach length. Our surveys identified significant changes in five physical habitat parameters (Table 1). An increase in bed elevation downstream from the decommissioned check dam was a direct effect of the deposition of sediments released from the dam reservoir, as observed also in other dam removals (Wohl and Cenderelli, 2000; Magilligan et al., 2016b). Bed aggradation was also an important reason for the reduction in bankfull channel depth by 29% on average between 2012 and 2016; however, as the latter change was also brought about by sediment accretion on channel banks (increasing channel depth) and bank retreat (causing a vertical shift of bank edges), changes in bed elevation and bankfull channel depth were not significantly correlated. Bulk deposition of the sediments released from the dam reservoir was associated with marked fining of the bed material, but subsequent outwashing of finer grains caused its coarsening, so that by 2016 sediment calibre almost returned to the values recorded prior to the dam lowering. Similar opposed tendencies of grain-size changes were also recorded over several months after check-dam removal on a mountain river in Taiwan (Wang and Kuo, 2016). Such differing tendencies may explain why studies presenting snapshots of river adjustments after dam removal reported either fining (Magilligan et al., 2016b) or coarsening of the bed material (Kibler et al., 2011) in the downstream reach. With the entrapment of relatively fine bed material in the downstream reach of Krzczonówka, near-bed velocity at base-flow conditions appeared to be substantially higher than in 2012, but the subsequent remarkable coarsening of the bed material was accompanied by merely a slight reduction in near-bed velocity. This indicates that changes in grain roughness exerted only a secondary impact on the recorded changes in near-bed velocity, which should be rather attributed to channel gradient steepening caused by greater bed aggradation near the check dam than in more distant cross-sections. Finally, a statistical significance of the increase in bankfull channel width between 2012 and 2016 reflected a consistency of this change among the study cross-sections, whereas its scale was rather small, 6% on average. Importantly, the monitoring carried out in Krzczonówka did not identify significant increases in the cross-sectional variation of physical habitat parameters that were associated with channel widening and the resultant increase in the complexity of flow pattern of a mountain river restored by establishing an erodible corridor (Wyżga et al., 2013, 2014).

Significant alterations in the taxonomic richness of benthic invertebrate community occurred during the restoration project. In 2014, after the passage of a flood and the bed aggradation with sediments released from the lowered check dam, the lowest average number of invertebrate taxa in the surveyed cross-sections and the lowest total number of taxa in the study reach were recorded. Floods markedly increase the drift of benthic invertebrates (Tockner and Waringer, 1997; Kändler and Seidler, 2013) and usually result in considerable decreases in abundance and diversity of the animals (Lake, 2000; Piniewski et al., 2016). These decreases may be especially large where live-bed conditions contribute to invertebrate mortality and where channel reconfiguration during a flood results in the formation of new streambed sections that have to be colonized by invertebrates after the event (Nislow et al., 2012; Hajdukiewicz et al., 2018). Such decreases were also recorded after dam removals in response to the burial of a previous bed surface by the sediments released from the upstream impoundment (Orr et al., 2008; Chiu et al., 2013; Chang et al., 2017). In Krzczonówka, substantial bedload transport and bed aggradation during the flood of May 2014 resulted in very small numbers of benthic invertebrate taxa found shortly after the event (Table S8), whereas subsequent surveys performed in the years 2014–2016 indicated colonization of the aggraded streambed by progressively greater numbers of taxa (Fig. 5, Table S17).

As a result of this post-flood trend, in 2016 the average number of invertebrate taxa in the study cross-sections significantly exceeded not only that of 2014, but also the one recorded in 2012, prior to the onset of restoration activities. In 2016 numerous benthic invertebrate taxa with various habitat preferences occurred in the surveyed cross-sections considerably more often than in 2012, despite nearly the same total number of taxa found in the study reach (35 and 36, respectively) and a lack of statistically significant increases in the cross-sectional variation of physical habitat parameters between these years (Table 1). Before the onset of restoration activities, conditions suitable for a variety of invertebrate taxa with specialized habitat requirements (Beisel et al., 1998) must have already occurred in the study reach, but the relevant habitats were scarce as bedrock covered about half of the streambed area. Because the deeply incised stream conveyed relatively high discharges within the channel, flood flows exerted high shear forces on the streambed (Wyżga et al., 2021b), which facilitated flushing out the animals from the reach. A hyporheic zone may provide refugia for benthic invertebrates during flow pulses (Brunke and Gonser, 1997), but in Krzczonówka Stream the occurrence of only a thin cover of coarse-grained alluvium over the bedrock prevented the animals from finding stable refugia under the bed surface. Transformation of the bedrock–alluvial bed into an alluvial bed with the entrapment of sediments released from the lowered check dam increased the abundance of habitats that could be colonized by the invertebrates with varied requirements, that were already present in the reach pool of taxa. Importantly, a considerable reduction in bankfull channel depth decreased shear forces exerted on the streambed by given flood discharges (Wyżga et al., 2021b) and this, together with the formation of a thicker layer of gravelly alluvium over the bedrock allowing invertebrates to find flow refugia in the hyporheic zone, must have reduced the intensity of flushing out the animals from the study reach during flow pulses.

A significance of the relative reduction in bankfull channel depth as an environmental factor driving changes in invertebrate taxa richness and the invertebrate-based index of ecological stream quality in the study cross-sections was confirmed by statistical analyses. The reduction was the first variable entering multiple regression models for the two ecological parameters and the only variable significantly explaining a combined pattern of their variability in the redundancy analysis. Results of the RDA demonstrated that cross-sections with a large reduction in channel depth (such as K4 or K1) were also typified by relatively large increases in the taxonomic richness of benthic invertebrates and score of the BMWP-PL index, whereas in cross-section K2 the smallest reduction in channel depth was associated with the smallest increases in taxa richness and index score (Fig. 10).

No significant changes in the species composition of fish community and the abundance of subadult and adult fish individuals were recorded

in the study reach of Krzczonówka between the first and the last year of the restoration project. On one hand, the scores of the EFI+ index indicating high ecological quality of the stream reach at the beginning of the restoration project suggest that available habitats were already fully occupied before the onset of restoration activities. On the other hand, the lack of significant changes in low-flow channel width and the cross-sectional variation of water depth and flow velocity indicates that the restoration did not create new habitats that might be used by new species and/or additional fish individuals. However, the period of engineering works in the stream was reflected in significantly lower abundance of YOY individuals in 2013 and 2014. Prolonged periods of the construction-induced high turbidity of stream water may have harmful effects on aquatic fauna (Courtice and Naser, 2020), and in Krzczonówka increased suspended-sediment concentrations occurred during the construction of block ramps and the check-dam lowering. In 2014 this effect must have been aggravated by direct and indirect consequences of the May flood—sweeping away of fish, especially younger individuals (cf. Harvey, 1987), by floodwaters conveyed in the incised channel and the post-flood scarcity of invertebrates for fish feeding caused by the aggradation of streambed (cf. Hajdukiewicz et al., 2018).

Despite the stable species composition of the fish community in Krzczonówka, its structure did respond to the implemented restoration measures. Re-establishing a gravel bed on the whole channel area must have favoured effective spawning by lithophilic species—brown trout, chub, Carpathian barbel and Eurasian minnow—that require a gravelly substrate for laying eggs. Free flow of water through gravel interstices is essential for the successful incubation of eggs of lithophilic fish (e.g., Crisp, 2000; Kukuła and Bylak, 2020) and such conditions were facilitated by the winnowing of fines from the sediments deposited by the 2014 flood. Consequently, effective spawning of lithophilic fish was manifested in the relatively high numbers of YOY individuals of chub and brown trout found in 2015, whereas no young-of-the-year chubs were recorded before implementation of the restoration measures. Carpathian barbel is a Natura 2000 species (Council of the European Communities, 1992) which is considered a valuable indicator of the naturalness of riverine ecosystems in the Carpathians (Kukuła, 2003; Prus et al., 2016). A marked increase in the abundance of this species after the implementation of restoration measures (Fig. 7) seems to be a result of restoring free movement of fish through the check dam and the increased spatial extent of gravelly spawning grounds in the study reach of the stream (cf. Foley et al., 2017). The increase in the Carpathian barbel abundance represents a change towards a more natural structure of the fish communities of Carpathian streams, with the abundance of Carpathian barbel dominating over that of stone loach (Kukuła, 2003; Bylak and Kukuła, 2018). The restoration of free movement of fish through the check dam probably prompted brown trout to migrate upstream to more shaded and cooler sections of the stream. Upstream of the dam and the reach analysed in this study, numerous individuals of brown trout were observed in 2016 (J. Jeleński, personal communication).

Evaluation of restoration-induced changes in ecological stream quality with benthic invertebrate- and fish-based indices yielded different results, with a significant improvement in this quality over the project duration indicated by the BMWP-PL index and a lack of quality change shown by the EFI+ index. These differing patterns of quality change most likely reflected different sets of environmental factors controlling the two groups of stream biota (cf. Wyżga et al., 2014), as restoration activities re-established a gravelly substrate in the stream and reduced hydraulic forces exerted by floodwaters on the streambed—with these changes positively influencing benthic invertebrate community—but did not change the variation in water depth and flow velocity, which might affect the fish community in the stream. However, the European Fish Index (EFI+) appeared to be insensitive to year-on-year fluctuations in the abundance of fish, particularly YOY individuals, induced by the flood passage (cf. Hajdukiewicz et al., 2018) and engineering works in the channel, and could not detect positive alterations in the structure of fish community induced by the restoration measures. This agrees with other studies showing low sensitivity of fish-based metrics to changes in hydromorphological conditions (Hering et al., 2006; Dahm et al., 2013) and suggests that a proper evaluation of effects of restoration activities on fish fauna may require interpretation of the species structure of fish community and the abundance of the most sensitive species.

5. Concluding remarks

Lowering of a high check dam, installation of block ramps in the deeply incised downstream reach of mountainous Krzczonówka Stream and the entrapment by these ramps of the sediments released from the dam reservoir during a moderate flood caused diverse alterations in the stream ecosystem during the time span of the restoration project. Significant increases in bed elevation, bankfull channel width and near-bed flow velocity as well as a reduction in bankfull channel depth constituted permanent changes in physical habitat conditions. After the deposition of dam-released sediments, the average taxonomic richness of benthic macroinvertebrates progressively increased and was significantly higher in the last year of the restoration project than before the onset of restoration activities. This change was reflected in a significant improvement of ecological stream quality indicated by the invertebrate-based BMWP-PL index. Changes in the taxonomic richness of benthic macroinvertebrates and the scores of the index between the first and the last year of the restoration project were significantly related to a relative decrease in bankfull channel depth, suggesting that benthic invertebrates benefited from the restoration-driven reduction in hydraulic forces exerted on the channel bed by flood flows. In contrast, the restoration did not increase species richness of fish and the average abundance of subadult and adult fish individuals, which was reflected in a lack of change in ecological stream quality indicated by the fish-based EFI+ index. However, after the implementation of restoration measures, the structure of fish community changed towards more natural one, with the increased proportion of Carpathian barbel.

Despite the diverse effects on different groups of aquatic fauna, the combination of the applied measures appeared useful in restoring the overall quality of the ecosystem of an incised mountain stream. A few factors may provide—possibly complementary—explanation of the differing responses of benthic macroinvertebrate and fish communities to the restoration. First, the restoration-driven changes of stream habitats were beneficial for benthic invertebrates but did not favour an improved condition of fish community that might require increases in the cross-sectional variation of physical habitat parameters (cf. Wyżga et al., 2009). Second, as fish have considerably longer life cycles than aquatic invertebrates, they respond slower to changes induced by restoration measures and thus require longer-term monitoring to detect changes in their communities (Closs et al., 2015). This emphasizes a need to continue the monitoring of restoration effects after the completion of restoration projects (Lorenz et al., 2018; Wyżga et al., 2021a). Third, analysis of metrics other than the abundance and species richness of fish may be required to demonstrate effects of the restoration action on these biota and the associated change in ecological stream quality.

Glossary

Channel incision Channel deepening that leads to increased cross-sectional area and flow capacity of a channel. It occurs if bed degradation is not (fully) compensated for by channel narrowing and increase in channel sinuosity.

Block ramp Low-head transversal hydraulic structure lacking a vertical step that is passable for fish and other riverine biota, while enabling dissipation of the energy of flowing water.

Check dam Transversal hydraulic structure built to control stormwater runoff, stabilize channel bed and reduce the flux of coarse sediment in a stream by storing it in the dam reservoir.

Electrofishing Scientific survey method used to determine fish

abundance and species composition and structure of ichthyofauna. Electrofishing results in no permanent harm to fish which return to their natural state after being caught and released.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2021.106365.

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