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# Impact of a small storage reservoir on the hydro-chemical regime of a flysch stream: A case study for the Korzeń stream (Poland)

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**Abstract:** The paper presents in form of a case study the results of 10-year long hydro-chemical studies on the Korzeń stream on which the "Skrzyszów" small storage reservoir was built. Studies aimed at evaluating the impact of the reservoir on the surface water quality in a Flysch stream. The basis for the analysis was results of 21 hydro-chemical water quality parameters, from the following groups of indicators: physical and acidity, oxygen and organic pollution, biogenic, salinity, metals. Indicators were determined in one-month intervals in two periods: 2005–2009 (before the reservoir was built) and 2015–2019 (after the reservoir was built). Obtained results were subjected to a statistical analysis. The trend analysis of changes was performed using the Mann–Kendall test or the seasonal Kendall test; significance of differences between indicator values from two periods was evaluated using the nonparametric Mann–Whitney U test. Results of analysis showed significant change trends of water quality parameters, in case of total iron concentration the trend was downward in both periods. Statistically significant differences between the values of definite majority of indicators were found in two analysed periods, indicating both favourable and unfavourable impact of the reservoir on water quality in the stream. Construction of the storage reservoir resulted in a significant change of physical and chemical indicators of water flowing in the stream. Random variation dynamics as well as tendencies and trends of changes over time have changed. In addition to modifying the stream hydro-chemical regime, the reservoir also affected the social and natural conditions.

**Keywords:** Mann–Kendall test, physical and chemical indicators, storage reservoir, surface water, trend and tendency of changes

## INTRODUCTION

Water is an environment-shaping factor which affects human life and health (Mioduszewski, 2003; Wiatkowski *et al.*, 2021). The importance of water for people and the environment is emphasised in numerous political, social and economic discussions, because water of good quality and in sufficient quantity must be provided to all current and future users. This problem results from limited resources and unfavourable changes in the water balance structure (Kundzewicz *et al.*, 2005; Kanownik and Kowalik, 2010; Mioduszewski, 2014) in relation to the climate change and resulting hazards which are one of the largest challenges for contemporary world (Paruch, Mæhlum and Robertson, 2015; Melo *et al.*, 2016; Degórski, 2018; Gruss and Wiatkowski, 2018; Tokarczyk and Szalińska, 2018; O'Keeffe *et al.*, 2019; Kubicz *et al.*, 2021). In these global environmental conditions, rational water management in drainage basin systems is becoming increasingly important at local, regional and national levels.

Between 2000 and 2019, small reservoirs were built in order to counteract the effects of droughts and floods, within the Provincial Small Water Storage Programmes. Small storage reservoirs perform economic, agricultural, natural and recreational functions, and they improve the water balance in the drainage basin. They also provide important eco-systemic services for aquatic organisms (Baumgartner *et al.*, 2020; Markowska *et al.*, 2020). Before making a decision to build and during the operation of water reservoirs, one should consider not only quantity but also the quality of water for storage and use. Correct use of reservoirs can be often threatened by the pollutants flowing in water (Bogdał and Kowalik, 2015; Bogdał, Kowalik and Witoszek, 2015; Wiatkowski and Rosik-Dulewska, 2015; Wiatkowski *et al.*, 2021) or by silting (Sojka, Jaskuła and Siepak, 2019).

Management of water resources, and hence correct use of storage reservoirs, is the key to maintaining the natural fluvial ecosystem and water for economic purposes (Wiatkowski *et al.*, 2021). The pollutants transported by land from built-on areas are one of the reasons why the quality of waters is deteriorating (Gnecco *et al.*, 2005). In case of drainage basins of lowland and submontane rivers, the water quality is usually affected by agriculture which contributes to an increased content of biogensin rivers. A high concentration of phosphorus and nitrogen has a huge negative impact on the water quality in rivers and regions through which rivers flow, causing eutrophication processes (Policht-Latawiec *et al.*, 2014; Bogdał *et al.*, 2016; Bogdał *et al.*, 2019).

Dammed reservoirs play an important part in shaping the chemical composition and physical and chemical parameters of river systems (Fairchild and Velinsky, 2006; Gao *et al.*, 2013; Soja and Wiejaczka, 2014; Wiejaczka *et al.*, 2017). The chemical composition of water flowing to the reservoir is usually strongly modified during the damming, and the changes can also be observed downstream of the reservoir. The physical and chemical properties of water flowing out of the reservoir depend on the quality of the inflow water, biotic and abiotic processes occurring in the reservoir and on the reservoir parameters. The damming of water in the rivers particularly affects the chemical composition of water in reservoirs with long storage time (Fairchild and Velinsky, 2006; Gao *et al.*, 2013). A relatively slow water flow in reservoirs in comparison to the water speed in rivers allows sedimentation of the suspended matter and initiates the processes reducing the content of nitrogen and phosphorus. Consequently, storage reservoirs on rivers often improve the water quality (Harrison *et al.*, 2009).

The study aimed at determining the impact of the "Skrzyszów" small storage reservoir on the quality of surface water in the Korzeń stream. This aim was achieved by analysing the values and concentrations of 21 physical and chemical water quality indicators which were obtained during systematic tests conducted at the same measurement point in two periods: 2005–2009 – before the reservoir was built, and 2015–2019 – after the reservoir was commissioned. The obtained results were subjected to a statistical analysis – basic descriptive statistics were calculated, trend and tendencies were analysed, and the significance of differences between the indicators values in both periods was evaluated.

## MATERIALS AND METHODS

### RESEARCH AREA

The Korzeń stream drainage basin (Fig. 1) is located in southern Poland – in the eastern part of the Małopolskie Voivodeship, in Tarnów Poviat (21°3'17"–21°6'33" E, 49°56'28"–49°58'57" N).

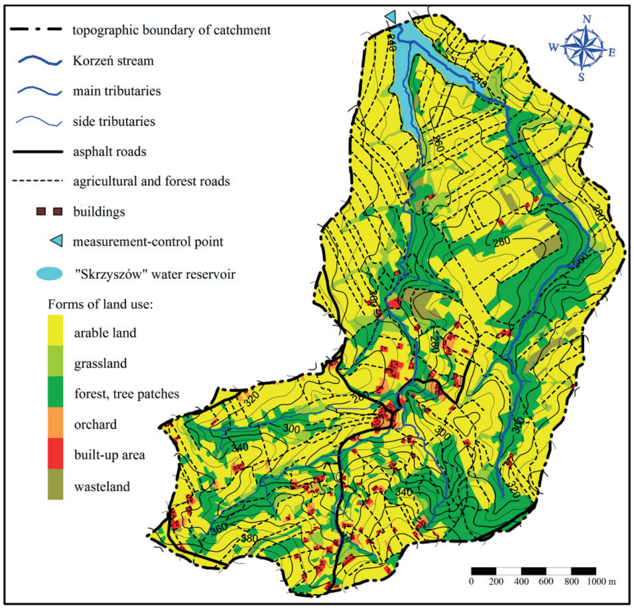


**Fig. 1.** Korzeń stream drainage basin – land use forms (as of 2018) and the measuring point location; source: own study

According to the Kondracki's division (Kondracki, 2009), the drainage basin is located in the Western Carpathians province, Outer Western Carpathians sub-province. The basin area is 9.65 km$^2$, and the altitude is 231.3–396.8 m a.s.l. The average weighted terrain gradient is 11%. The stream drainage basin is characterised by a low diversity of soil types. The dominant soils are Luvisols, Hamplic Cambisols and Stagnic Cambisols. Before the storage reservoir was built (2005–2009), the arable land, evenly distributed in the basin area, was the dominant land use form, accounting for 60.7% of the basin area. Grassland was 7.7% of the basin area and was mainly located in river valleys. Orchards accounted for a small fraction: 1.2%. Woodlands in the stream drainage basis accounted for 25.9% of its area. In addition, there were areas with scattered development and wasteland: 2.4 and 2.1% of total area, respectively. They were located near water courses, in the central and southern part of the drainage basin. In 2015–2019, the location of individual land use forms (Fig. 1) and their percentage structure changed only very little (Tab. 1). In this time, as a result of construction of the

**Table 1.** Land use in the Korzeń stream drainage basin

| Forms of land use | Use structure in years | | | |
|---|---|---|---|---|
| | 2005–2009 | | 2015–2019 | |
| | ha | % | ha | % |
| Arable land | 586.5 | 60.7 | 585.9 | 60.7 |
| Grassland | 73.9 | 7.7 | 50.5 | 5.2 |
| Forest, tree patches | 250.0 | 25.9 | 255.9 | 26.5 |
| Orchard | 11.3 | 1.2 | 11.3 | 1.2 |
| Built-up area | 23.1 | 2.4 | 23.6 | 2.5 |
| Wasteland | 20.0 | 2.1 | 19.4 | 2.0 |
| Land under standing water | 0.0 | 0.0 | 18.2 | 1.9 |
| Total | 964.8 | 100.0 | 964.8 | 100.0 |

Source: own study.

"Skrzyszów" storage reservoir 18.2 ha (1.9%) of the drainage basin were covered by water. The share of grassland and agricultural wasteland decreased by 2.5 and 0.1%, respectively. The share of woodlands increased by 0.6%, and the share of developed land increased only by 0.1%.

The Korzeń stream is a fifth-order watercourse, a left-bank tributary of the Wątok River, which joins the Biała River – a right-bank tributary of the Dunajec. Its springs are located at the altitude of approximately 310.00 m a.s.l. The length is 5.11 km. In addition to the Korzeń stream, the hydrographical system of the drainage basin includes also the left-bank Dopływ spod Trzemesnej (length 5.67 km) and numerous smaller watercourses (6.96 km). The river network density in the Korzeń drainage basin is 1.84 km·km$^{-2}$. The characteristic and probable flows calculated according to empirical formulas are: $SSq$ = 0.008 m$^3$·s$^{-1}$·km$^{-2}$; $NNq$ = 0.001 m$^3$·s$^{-1}$·km$^{-2}$; $Q_{50\%}$ = 0.631 m$^3$·s$^{-1}$·km$^{-2}$; $Q_{1\%}$ = 3.109 m$^3$·s$^{-1}$·km$^{-2}$ (Bogdał *et al.*, 2014).

In 2012, within the "Malopolskie Voivodeship Small Storage Reservoirs Programme" (Hydroprojekt Kraków Sp. z o.o., 2004) local authorities decided to build a storage reservoir on the Korzeń stream. The reservoir was commissioned in 2014 and is the first completed project under the programme. The main function of the reservoir is to protect the areas in the valley of Korzeń and its recipient the Wątok River against floods and also provide water storage for use in irrigation during droughts and improve the water balance in the catchment. In addition, the reservoir stores water for firefighting purposes, has a tourist and business function, improves biodiversity and landscape quality of the area and increases its resistance to degrading factors (biological development of the reservoir rim). Protection of the Korzeń stream biodiversity downstream of the reservoir is also an important function.

The reservoir comprises the 10.5 m high and 224 m long earth dam made of local clays and protected with a rock riprap. The dam has a tower overflow with bottom sluices (Hydroprojekt Kraków Sp. z o.o., 2007). Typical reservoir parameters are presented in Table 2.

**Table 2.** Basic parameters of the "Skrzyszów" reservoir

| Parameter | Unit | Value |
|---|---|---|
| Compensating capacity | m$^3$ | 469,000 |
| Flood preventing capacity | m$^3$ | 206,000 |
| Capacity at normal damming level | m$^3$ | 553,000 |
| Total capacity | m$^3$ | 759,000 |
| Inundation area at normal damming level (NPP) / max. damming level (max. PP) | ha | 18.2 / 22.8 |
| Damming height at max. PP | m | 9.90 |
| Average / max. depth $h$ at NPP | m | 3.0 / 8.9 |
| Average/ max. reservoir width at NPP | m | 120 / 300 |
| Max. reservoir length | km | 1.2 |
| Main dam length | m | 224 |
| Altitude NPP | m a.s.l. | 238.00 |
| Altitude Max PP | m a.s.l. | 239.00 |
| Dam volume | m$^3$ | 30,000 |

Source: own study.

## ANALYTICAL METHODS

The hydro-chemical examinations of the Korzeń stream water were conducted in once a month in two periods: before the reservoir was built (2005–2009) and after it was commissioned (2015–2019). Water sampling has been done once every month, the exact of water sampling dates were chosen randomly once every month. The water samples were taken in one measuring point located at 1+520 km of the stream – in the first period from natural bed, and from the engineered bed downstream of the dam in the second period (Fig. 1). Twenty one water quality indicators were determined over the entire 10-year period of the study (Tab. 3), using the same methods of sampling and field and laboratory tests.

**Table 3.** Studied water quality indicators

| Indicator group | Indices |
|---|---|
| Physical and acidity | temperature, °C |
| | pH |
| | total suspended solids ($TSS$), mg·dm$^{-3}$ |
| Oxygen and organic pollution | dissolved oxygen ($DO$), mg $O_2$·dm$^{-3}$ |
| | saturation with oxygen ($DO_{sat}$), % |
| | biochemical oxygen demand ($BOD_5$), mg $O_2$·dm$^{-3}$ |
| | chemical oxygen demand ($COD_{Mn}$), mg $O_2$·dm$^{-3}$ |
| Biogenic | ammonia nitrogen ($N–NH_4^+$), mg·dm$^{-3}$ |
| | nitrite nitrogen ($N–NO_2^-$), mg·dm$^{-3}$ |
| | nitrate nitrogen ($N–NO_3^-$), mg·dm$^{-3}$ |
| | phosphate phosphorus ($P–PO_4^{3-}$), mg·dm$^{-3}$ |
| Salinity | electrolytic conductivity at 20°C ($EC$), μS·cm$^{-1}$ |
| | dissolved substances ($DS$), mg·dm$^{-3}$ |
| | sulphates ($SO_4^{2-}$), mg·dm$^{-3}$ |
| | chlorides ($Cl^-$), mg·dm$^{-3}$ |
| | calcium ($Ca^{2+}$), mg·dm$^{-3}$ |
| | magnesium ($Mg^{2+}$), mg·dm$^{-3}$ |
| | sodium ($Na^+$), mg·dm$^{-3}$ |
| Metal concentration | potassium ($K^+$), mg·dm$^{-3}$ |
| | total iron ($Fe^{2+}$, $Fe^{3+}$), mg·dm$^{-3}$ |
| | manganese ($Mn^{2+}$), mg·dm$^{-3}$ |

Source: ow study.

The following parameters were measured directly in the field: pH with a CP–104 pH-meter, temperature and dissolved oxygen content ($DO$), water saturation with oxygen ($DO_{sat}$) using a CO–411 oxygen meter, and electrolytic conductance ($EC$) with a CC–102 conductometer. The following parameters were determined in the laboratory: total suspended solids ($TSS$) and total dissolved solids ($DS$) using the gravimetric method; five-day biochemic oxygen demand ($BOD_5$) using the Winkler titration method; chemical oxygen demand ($COD_{Mn}$) using the titration method; nitrogen concentration: ammonia nitrogen ($N–NH_4^+$), nitrite nitrogen ($N–NO_2^-$) and nitrate nitrogen ($N–NO_3^-$);

phosphorus in the form of phosphate ($P–PO_4^{3−}$) and chlorides ($Cl^−$) using the flow colorimetric analysis with a FIAstar 5000 instrument; concentration of sulphates ($SO_4^{3−}$) using the precipitation-gravimetric method ($Ca^{2+}$), concentration of magnesium ($Mg^{2+}$), sodium ($Na^+$), potassium ($K^+$), total iron ($Fe^{2+}$, $Fe^{3+}$) and manganese ($Mn^{2+}$) using the atomic absorption spectrometry on a UNICAM SOLAR 969 spectrometer.

## STATISTICAL METHODS

Basic descriptive statistics were determined separately for each period of the study: minimum and maximum value, arithmetic mean, median and coefficient of variation (*CV*). The *CV* was used to evaluate variation of measurement datasets, using four ranges: low variation (0–20%), medium variation (21–40%), high variation (41–60%) and very high variation (>60%) (Bogdał *et al.*, 2019).

The tendency (trend over time) analysis for water quality indicators for each period was performed using the nonparametric Mann–Kendall test and the seasonal Kendall test. The Mann–Kendall test outdistances other nonparametric trend tests due to its simplicity and wide scope of application (Tosunoglu and Kisi, 2017). It is widely used in hydrological analyses (Rutkowska, 2013; Stasik, Kęsicka and Korytowski, 2020; Tadesse and Dinka, 2022], climate change analyses (Azizzadeh and Javan, 2015; Skowera *et al.*, 2015; Bougara *et al.*, 2021; Gadedjisso-Tossou, Adjegan and Kablan, 2021), and water (Rogora *et al.*, 2015; Zeleňáková, Purcz and Oravcová, 2015; Dąbrowska *et al.*, 2016; Kanownik, Policht-Latawiec and Fudała, 2019) and air quality analyses (Wang *et al.*, 2022).

The *S* Mann-Kendall statistics is calculated according to the Equation (1) (Helsel *et al.*, 2020):

$$S = \sum_{i=1}^{n-1} \sum_{j=i+1}^{n} \operatorname{sgn}(x_j - x_i) \quad (1)$$

where: $n$ = dataset length, $x_i$ and $x_j$ = data values at time $i$ and $j$,

$$\operatorname{sgn}(x_j - x_i) = \begin{cases} 1;\ \text{if } (x_j - x_i) > 0 \\ 0;\ \text{if } (x_j - x_i) = 0 \\ -1;\ \text{if } (x_j - x_i) < 0 \end{cases}$$

The null hypothesis assumes that the data are independent random variables of identical distribution (no trend), and if $n > 10$ it is expected that $S$ statistics will have a normal distribution with mean equal to zero and variance calculated from the Equation (2):

$$Var(S) = \frac{n(n-1)(2n+5)}{18} \quad (2)$$

where: $Var(S)$ = variance of $S$.

The normalised test statistics $Z$ is calculated from the Equation (3):

$$Z = \begin{cases} \frac{S-1}{\sqrt{Var(S)}};\ \text{if } S > 0 \\ 0;\ \text{if } S = 0 \\ \frac{S+1}{\sqrt{Var(S)}};\ \text{if } S < 0 \end{cases} \quad (3)$$

If the calculated $Z$ value is greater than the critical value of standard normal distribution at the significance level $\alpha$, the null hypothesis is rejected and indicates the existence of statistically significant positive or negative trend in the data series. The positive values of the statistics indicate an upward tendency, and the negative values indicate a downward tendency. If a significant trend is found, the rate of change is determined using the Theil–Sen slope estimator $\beta$ (Eq. 4) calculated taking into account all $i < j$ ($i$ = 1, 2, ..., $n$–1 and $j$ = 2, 3, ..., $n$) (Helsel *et al.*, 2020):

$$\beta = \operatorname{med} \frac{x_j - x_i}{j - i} \quad (4)$$

The rate of change was expressed in units of the studied water quality indicator per month. The Mann–Kendall test should be used when no seasonality is expected or when trends occur in various directions (upward or downward) in various seasons of the year. The test does not discover that there are seasonal patterns; a pattern is rather recorded as random noise in the process. In such a situation, seasonality may hinder noticing the real trend, because the trend tests (parametric and nonparametric) generally involve seeing how the trend signal stands out above the noise. The water quality data often show strong seasonal patterns and require a different trend analysis technique, e.g. seasonal Kendall test. This test accounts for seasonality, by making the Mann–Kendall test calculations for each from $m$ seasons separately and then combining the results. The season in this test may be a month, a quarter of the year or another unit of time. The presence of seasonality means that data have different distributions for various seasons (e.g. months). A monotonic upward tendency can for example occur for years in January, but not in June. The data are compared only between season, e.g. February is compared to February and so on (Meals *et al.*, 2011; Helsel *et al.*, 2020). The statistics is calculated according to the Equation (5) (Helsel *et al.*, 2020):

$$S_k = \sum_{i=1}^{m} S_i \quad (5)$$

where: $m$ = number of e.g. months for which the data were collected over the years; $S_i$ = Mann–Kendall $S$ statistics.

Similarly to the Mann–Kendall test, the resultant normalised value of seasonal Kendall test $S_k$ statistics is evaluated in relation to the standard normal distribution table. The null hypothesis (no trend) is rejected at the significance level $\alpha$ if $|Z_{Sk}| > Z_{\text{crit}}$ – where $Z_{\text{crit}}$ is the value of standard normal distribution with probability of exceeding $\alpha/2$. The visual examination of time series plots developed earlier during the exploratory data analysis allowed determination of indicators which have a strong seasonal component and these indicators were additionally subjected to the trend analysis using the seasonal Kendall test. The statistical inference on significance of difference using the nonparametric Mann–Whitney U test was performed in order to verify whether the values of water quality indicators differ from each other in analysed periods. The database of all the statistical tests analysed in the work were monthly values of water quality indicators. The significance level $\alpha$ = 0.05 was used for all tests. Nonparametric tests were used due to the absence of normal distribution in majority of analysed parameters according to the results of Shapiro–Wilk test. The statistical analyses were performed with the use of Statistica 13 and XLSTAT package.

## RESULTS AND DISCUSSION

Statistically significant differences were found between the values of all analysed physical and acidity indicators determined in 2005–2009 and 2015–2019 (Tab. 4).

After the construction of the reservoir, the temperature and pH values were higher than in the previous period, and the values of *TSS* were lower. It is generally known that a slow flow through storage reservoirs (in comparison to rivers and streams) allows the sedimentation of suspension (Harrison *et al.*, 2009; Rigacci *et al.*, 2013). Most river load transported to the reservoir is collected in its lower part, while the suspended and dissolved load is only partially retained. After the "Skrzyszów" reservoir was built, the average monthly value of *TSS* in the Korzeń stream water was about 2 times less than in the previous period, indicating a significant reduction. The water taken downstream of the "Skrzyszów" reservoir was warmer – on the average by 2.8°C. The water temperature in the downstream part of the watercourse may be affected by the method of water outflow from the reservoir, that is whether the outflow is from the surface for from near the bottom (Ignatius and Rasmussen, 2016). Usually, increased water temperature downstream of the damming is observed in the summer half year as a result of shallow reservoirs in which thermal stratification does not occur (Jankowski, 2017), and also can be caused by surface discharge of water from the reservoirs with the use of overflow structures (Ignatius and Rasmussen, 2016). In case of reservoirs with thermal stratification, when water flows out through bottom sluices, higher water temperatures downstream of the damming may occur in winter (Jankowski, 2017). The outflow from the "Skrzyszów" reservoir is mainly via a 0.5-m wide cut-out in the top part of tower overflow. Such surface water discharge and slow water flow in the reservoir extend the time during which the water is exposed to sunlight, increasing the water temperature in the Korzeń stream. In addition, in the previous period water flowed in a shaded and well-developed and on the average about 2-m deep bed of an unregulated stream which banks were overgrown with trees and shrubs. The coefficients of variation in excess of 60% indicate a very high diversity of water temperature in both periods. The largest dispersion of values in empirical datasets was found in *TSS*, for which the *CV* before the reservoir was built was 142%, and in the earlier period – 185%. The pH values had the least random variation in both periods ($CV < 20\%$). The values of this acidity indicator changed in the 7.36–8.41 range in the first period, and in the 7.28–8.96 range in the second period of the study (Tab. 4).

**Table 4.** Selected descriptive statistics and results of the significance test of changes between the values of water quality indicators determined in 2005–2009 and 2015–2019

| Indices | 2005–2009 | | | | | 2015–2019 | | | | | Mann–Whitney *U* test | Test probability values (*p*) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | range | | mean | median | *CV* (%) | range | | mean | median | *CV* (%) | | |
| | min. | max. | | | | min. | max. | | | | | |
| Temperature | 0.0 | 19.0 | 8.8 | 8.1 | 65 | 0.4 | 24.3 | 11.6 | 11.3 | 62 | –2.13 | 0.03 |
| *TSS* | 0.8 | 85.2 | 9.0 | 4.5 | 142 | 0.0 | 37.1 | 4.8 | 1.3 | 185 | 4.51 | <0.01 |
| pH | 7.36 | 8.41 | 7.86 | 7.86 | 3 | 7.28 | 8.96 | 7.97 | 7.93 | 4 | –2.05 | 0.04 |
| *DO* | 5.7 | 14.8 | 9.9 | 9.6 | 22 | 5.7 | 15.5 | 9.8 | 9.7 | 22 | 0.12 | 0.90 |
| $DO_{sat}$ | 52 | 118 | 86 | 86 | 15 | 62 | 118 | 90 | 88 | 12 | –1.46 | 0.14 |
| $BOD_5$ | 0.1 | 2.6 | 0.8 | 0.6 | 64 | 0.1 | 10.0 | 2.5 | 2.1 | 63 | –7.55 | <0.01 |
| $COD_{Mn}$ | 1.9 | 9.6 | 4.6 | 4.3 | 40 | 2.5 | 16.5 | 8.9 | 8.3 | 31 | –6.86 | <0.01 |
| $N–NH_4^+$ | 0.000 | 0.195 | 0.010 | 0.000 | 302 | 0.000 | 0.887 | 0.127 | 0.050 | 161 | –6.48 | <0.01 |
| $N–NO_2^-$ | 0.000 | 0.088 | 0.010 | 0.007 | 128 | 0.000 | 0.090 | 0.016 | 0.013 | 110 | –3.14 | <0.01 |
| $N–NO_3^-$ | 0.021 | 3.140 | 0.903 | 0.698 | 75 | 0.000 | 2.180 | 0.525 | 0.333 | 103 | 3.49 | <0.01 |
| $P–PO_4^{3-}$ | 0.004 | 0.156 | 0.027 | 0.019 | 93 | 0.000 | 0.718 | 0.036 | 0.012 | 278 | 2.83 | <0.01 |
| *EC* | 209 | 514 | 414 | 427 | 15 | 191 | 486 | 361 | 364 | 15 | 5.02 | <0.01 |
| *DS* | 200 | 424 | 305 | 309 | 15 | 112 | 262 | 171 | 169 | 16 | 9.34 | <0.01 |
| $SO_4^{2-}$ | 11.9 | 63.7 | 46.7 | 46.7 | 19 | 9.4 | 69.6 | 32.5 | 32.5 | 37 | 6.38 | <0.01 |
| $Cl^-$ | 7.3 | 21.0 | 12.2 | 11.1 | 26 | 0.8 | 14.2 | 10.8 | 11.0 | 21 | 1.87 | 0.06 |
| $Ca^{2+}$ | 33.2 | 92.0 | 69.6 | 68.4 | 17 | 8.1 | 68.9 | 55.1 | 58.9 | 21 | 6.29 | <0.01 |
| $Mg^{2+}$ | 6.8 | 11.6 | 9.8 | 10.2 | 11 | 4.2 | 54.7 | 10.1 | 9.4 | 62 | 1.82 | 0.07 |
| $Na^+$ | 7.6 | 15.3 | 11.5 | 11.2 | 18 | 6.4 | 32.6 | 9.8 | 9.3 | 34 | 5.51 | <0.01 |
| $K^+$ | 1.1 | 4.9 | 2.4 | 2.3 | 35 | 0.2 | 5.0 | 2.8 | 2.9 | 24 | –3.71 | <0.01 |
| $Fe^{2+}$, $Fe^{3+}$ | 0.08 | 1.18 | 0.47 | 0.41 | 54 | 0.04 | 7.78 | 0.66 | 0.50 | 182 | –2.04 | 0.04 |
| $Mn^{2+}$ | 0.01 | 0.26 | 0.10 | 0.09 | 65 | 0.02 | 2.77 | 0.40 | 0.23 | 145 | –4.70 | <0.01 |

Explanations: *CV* = coefficient of variation (%), other explanations and units as in Tab. 3; the statistically significant values are marked in red for $p \leq \alpha = 0.05$.
Source: own study.

In the 2005–2009 period, a statistically significant downward trend was found for *TSS* and an upward trend for pH (Tab. 5). The slope coefficient $\beta$ indicates a monthly reduction of *TSS* in the Korzeń water on the average by 0.177 mg·dm$^{-3}$, at the multiyear mean of 9.0 mg·dm$^{-3}$ and the range from 0.8 to 85.2 mg·dm$^{-3}$. The pH in this period of the study increased on the average by 0.004 a month in this period of the study. After the reservoir was built, no statistically significant trends for physical indicators and acidity were found. The *S* statistics only suggests an upward trend for pH and a downward trend for *TSS* (unverified statistically). The visual examination of plots during the data analysis confirmed strong seasonal component in water temperature. Yearly water temperature curves for the stream show two periods of relatively stabilised thermal conditions (winter and summer) and two transitional periods (spring and autumn) with dynamic changes of the measured parameter. The seasonal Kendall test was used to analyse the variability of this indicator over time, and the test did not verify a statistically significant trend in data series. Negative values of *S* statistics for the data from the first period of the study indicate only a downward trend for water temperature (unverified statistically) during one or more months. The results of studies aiming at determination of the impact of storage reservoirs on the water quality in watercourses, in terms of physical indicators and acidity, are inconclusive. In Shim, Yoon and Yoon (2018), the Mann–Whitney U test showed that the water pH before and after the construction of a dam on the Geum River in Korea did not differ significantly. In addition, after the dam construction the authors noticed lower, but unverified statistically, *TSS*. In similar studies on the Çine River in Turkey, Bor and Elçi (2022) found statistically lower pH values in the watercourse downstream of the dam. The lower pH downstream of the dam affected the concentrations of sulphates, nitrates and aluminium. The authors observed reduced values of these parameters when the storage reservoir was punt into operation. Other authors found higher pH of water flowing out of reservoirs which they attribute to the growth of algae reducing the $CO_2$ content in water (Ignatius and Rasmussen, 2016; Dębska *et al.*, 2021).

Statistically significant differences between the results from two periods were obtained for $BOD_5$ and $COD_{Mn}$ (Tab. 4). After the reservoir was built, both organic pollution indicators were significantly higher – $BOD_5$ was 3 times higher on the average (increase by 212.5%), and $COD_{Mn}$ about 2 times higher (increase by 93.5%) in relation to the previous period. A slight increase of water oxygenation was observed at the same time, but was not statistically verified, which may have been affected by the surface water outflow from the reservoir. Monthly saturation with oxygen ($DO_{sat}$) of the Korzeń stream waters was in the 62–118% range, with the mean of 90%. The values were similar before the reservoir was built – 52–118%, with the mean of 86%. The dispersion of values was similar in both periods. The $DO_{sat}$ had a low, the *DO* and the $COD_{Mn}$ a medium, and the$BOD_5$ a very high random variation (Tab. 4). A similar impact of the storage reservoir on oxygen conditions was found by Shim, Yoon and Yoon (2018) – the *DO* concentrations did not differ significantly before and after a dam was built on the Geum River in Korea. However Bor and Elçi (2022) reached different conclusions than in case of the Korzeń stream: in their study, according to the Mann–Whitney U test, the values of $BOD_5$ and $COD_{Mn}$ did not change significantly when the dam was built.

The results of the Mann-Kendall test indicate an upward *DO* and $DO_{sat}$ trend between 2005 and 2009. In that period, the *DO* in Korzeń waters increased on the average by 0.036 mg $O_2$·dm$^{-3}$ a month (Tab. 6), with the multiyear mean of 9.9 mg $O_2$·dm$^{-3}$ and range from 5.7 to 14.8 mg $O_2$·dm$^{-3}$ (Tab. 4). The $DO_{sat}$ values grew on the average by 0.359% a month. No statistically significant trends were found in case of organic water pollution indicators. Negative values of *S* statistics indicate only a downward trend for $BOD_5$ and $COD_{Mn}$ in that period of the study. In the next period (2015–2019), no statistically significant change trend of the analysed oxygen and organic pollution indicators was observed. Contrary to the earlier period, the *S* statistics for *DO* and $DO_{sat}$ indicates a downward trend, and for $BOD_5$ and $COD_{Mn}$ an upward trend in the Korzeń stream (Tab. 6). To summarise, while with time the biochemical and chemical oxygen demand increased, the *DO* content and saturation decreased (after the reservoir was built) and the other way around (before the reservoir was built). Dębska, Rutkowska and Szulc (2021), in a paper presenting the impact of the Komorów storage reservoir on the water quality in the Utrata River showed that $COD_{Mn}$ and the *DO* concentration were higher in the measuring point downstream of the reservoir in each month of the study. The *DO* increase in the Utrata downstream of the damming may have been affected by a lower water temperature at this point and the type of water discharge which mechanically aerated the water, and the $COD_{Mn}$ increase may have been affected by a large amount of agricultural or municipal pollutants and high content of organic substances in the reservoir. The type of discharge structure played a dominant part in changing *DO* values in the water flowing out from nine reservoirs located in the urban-rural environment with a different share of arable land, woodland and urbanised area. The dams with top

**Table 5.** Trend test results for physical and acidity indicators

| Indices | 2005–2009 | | | | 2015–2019 | | | |
|---|---|---|---|---|---|---|---|---|
| | $\beta$ | $S$ | $S_k$ | $p$ | $\beta$ | $S$ | $S_k$ | $p$ |
| Temperature | –0.011** | – | –11** | 0.49** | 0.029** | – | 0** | 1.00** |
| *TSS* | –0.177* | –819* | – | <0.01* | –0.019* | –199* | – | 0.18* |
| pH | 0.004* | 379* | – | 0.02* | 0.001* | 123* | – | 0.41* |

Explanations: *TSS* = total suspended solids, * = Mann–Kendall test, ** = seasonal Kendall test, $\beta$ = Theil–Sen slope, $S$ = statistics of the Mann–Kendall test, $S_k$ = statistics of the seasonal Kendall test, $p$ = test probability value; the statistically significant trend is marked in red for $\alpha$ = 0.05.
Source: own study.

water outflow significantly increased the *DO* concentration (Ignatius and Rasmussen, 2016). An adverse impact of the reservoir on the water quality, as indicated by higher $COD_{Mn}$ values in the outflowing water in comparison to the inflowing water, was shown by Kubicz and Cieślar (2016). It was determined that the likely reason was the release of organic pollutants by sediments accumulated on the bottom of the reservoir. Bor and Elçi (2022) noticed that after the dam was built the values of $BOD_5$ and $COD_{Mn}$ increased over the years due to an intensive agriculture in the Çine River drainage basin. The aforementioned, adverse change tendencies of oxygen indicators during the operation of the "Skrzyszów" reservoir may be a result of higher water temperatures than in the earlier period which reduces the ability to dissolve oxygen in water, which oxygen is necessary, inter alia, in mineralisation of organic substances. On the other hand, the upward tendency of $BOD_5$ and $COD_{Mn}$ may be a result of organic pollutants from decomposition of aquatic plants which increasingly overgrow the surface of the reservoir, absence of natural barriers and agricultural activity on the reservoir rim (reduced area of tree stands and grassland as a result of reservoir construction; intensive use of arable land adjoining the reservoir).

The concentrations of all studied biogenic indicators were statistically very significantly different between the periods of 2005–2009 and 2015–2019 – in each case the test probability values were $p < 0.01$(Tab. 4). In the period after the reservoir was built in relation to the period prior to the construction, the concentrations of ammonia nitrogen ($N–NH_4^+$) and nitrite nitrogen ($N–NO_2^-$) were significantly higher, and concentrations of nitrate nitrogen ($N–NO_3^-$) and phosphate phosphorus ($P–PO_4^{3-}$) were lower. The concentration values of all biogenic indicators had a very high random variation ($CV > 60\%$). In the second period, the more differentiated values were for $N–NO_3^-$ and $P–PO_4^{3-}$, and the concentration change dynamics for $N–NH_4^+$ and $N–NO_2^-$ was lower (Tab. 4). In studies by Bor and Elçi (2022), for nitrogen compounds ($N–NH_4^+$ and $N–NO_2^-$) the Mann–Whitney U test did not show any significant concentration changes after the dam was built, but values of $N–NO_3^-$ and $P–PO_4^{3-}$ were significantly reduced, similarly to this study. The $P–PO_4^{3-}$ concentrations in the Çine waters decreased because the phosphorus compounds were trapped in bottom sediments deposited in the reservoir. The authors also observed a very intensive discharge of municipal and industrial wastewater to the watercourse which increased the concentrations of $N–NH_4^+$ and $N–NO_2^-$ in the river after the dam was built, but only during two years out of the ten-year period of the study.

The values of analysed biogenic indicators from 2005–2009 showed a downward trend for one indicator only – $N–NH_4^+$ (Tab. 7). In case of remaining parameters, the positive values of *S* statistics indicate upward trends. No statistically significant trends were observed in this period of study. After the reservoir was built, a statistically significant downward trend was determined for $N–NO_2^-$ and $N–NO_3^-$ (Fig. 2) and, similarly to the earlier period, a downward tendency for $N–NH_4^+$ and an upward tendency for $P–PO_4^{3-}$ (Tab. 7). The $N–NO_3^-$ changes over time were analysed using the seasonal Kendall test because yearly $N–NO_3^-$ concentrations had two stabilised periods (high concentrations in the summer and low in the winter) and two transitional periods (dynamic changes in the spring and autumn) (Fig. 2). This variation conformed to the typical yearly cycle presented in the literature (Richards *et al.*, 2021; Stępniewski and Łaszewski, 2021). Between 2015 and 2019, the concentration of $N–NO_3^-$ was reduced in the Korzeń stream waters on the average by 0.0129 $mg \cdot dm^{-3}$ a month, with the multiyear mean of 0.525 $mg \cdot dm^{-3}$ and the range of changes from 0 to 2.180 $mg \cdot dm^{-3}$. The $N–NO_2^-$ concentration in water was reduced on the average by 0.0002 $mg \cdot dm^{-3}$ a month, with the mean of 0.016 $mg \cdot dm^{-3}$ and the range of changes equal to 0.090 $mg \cdot dm^{-3}$ (Tab. 4 and 7). The downward trends or tendencies for nitrate nitrogen downstream of the water outflow from the reservoir can mainly be attributed to absorption of the $N–NO_3^-$ nitrogen by aquatic plants and nitrogen accumulation in bottom sediments (Ignatius and Rasmussen, 2016; Stępniewski and Łaszewski, 2021). In case of reservoirs analysed by Ignatius and Rasmussen (2016), the nitrates concentrations in waters downstream of reservoirs were constantly decreasing, and the drop was most visible in agricultural drainage basins with high level of nitrates in the upstream river water.

From among all analysed salinity indicators, only the concentrations of chlorides ($Cl^-$) and magnesium ($Mg^{2+}$) were

**Table 6.** Trend test results for oxygen and organic pollution indicators

| Indices | 2005–2009 | | | 2015–2019 | | |
|---|---|---|---|---|---|---|
| | *β* | *S* | *p* | *β* | *S* | *p* |
| *DO* | 0.036* | 342* | 0.04* | –0.016* | –155* | 0.30* |
| $DO_{sat}$ | 0.359* | 574* | <0.01* | –0.069* | –143* | 0.34* |
| $BOD_5$ | 0.000* | –129* | 0.29* | 0.000* | 7* | 0.99* |
| $COD_{Mn}$ | –0.013* | –28* | 0.73* | 0.010* | 87* | 0.56* |

Explanations as in Tab. 5.
Source: own study.

**Table 7.** Trend test results for biogenic indicators

| Indices | 2005–2009 | | | | 2015–2019 | | | |
|---|---|---|---|---|---|---|---|---|
| | *β* | *S* | $S_k$ | *p* | *β* | *S* | $S_k$ | *p* |
| $N–NH_4^+$ | 0.0000* | –139* | – | 0.29* | –0.0002* | –95* | – | 0.53* |
| $N–NO_2^-$ | 0.0000* | 20* | – | 0.91* | –0.0002* | –387* | – | <0.1* |
| $N–NO_3^-$ | 0.0053** | – | 24** | 0.10** | –0.0129** | – | –42** | 0.02** |
| $P–PO_4^{3-}$ | 0.00005* | 106* | – | 0.52* | 0.00004* | 75* | – | 0.62* |

Explanations as in Tab. 5.
Source: own study.

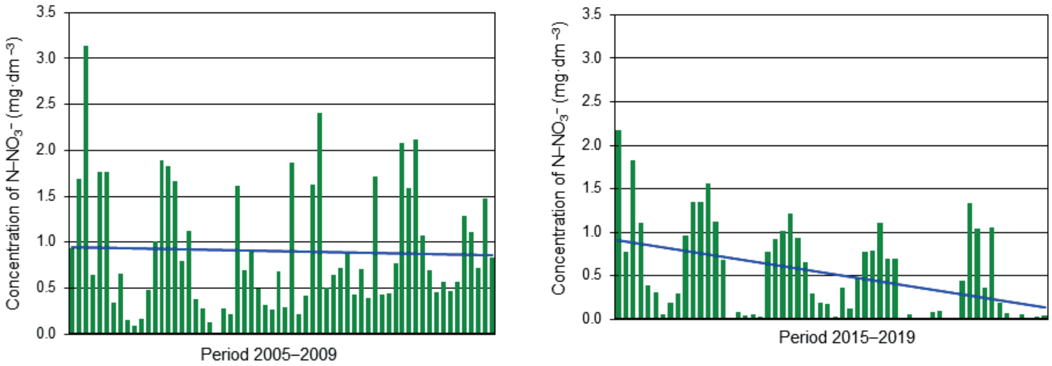


**Fig. 2.** Example of the course of value changes over time for an indicator characterised by seasonality (N–$NO_3^-$); source: own study

not statistically significantly different in both periods of the study. For the remaining indicators, the *p*-value of the Mann–Whitney test was $p < 0.01$ (Tab. 4). After the reservoir was built, the values of electrolytic conductance (*EC*), dissolved solids (*DS*) and concentrations of calcium ($Ca^{2+}$), sulphates ($SO_4^{2-}$) and sodium ($Na^+$) decreased significantly. The *EC* was lower than in the previous period by 53 μS·cm$^{-1}$ (reduction by12.8%), *DS* by 134 mg·dm$^{-3}$ (43.9%), $Ca^{2+}$ concentration by14.5 mg·dm$^{-3}$ (20.8%) and $SO_4^{2-}$ concentration by 14.2 mg·dm$^{-3}$ (30.4%). The average $Na^+$ concentration was lower by 1.7 mg·dm$^{-3}$ (reduction by 14.8%) than the mean before the reservoir was built. The majority of salinity indicators had a low or medium random variation. The largest dispersion of values in the empirical datasets was observed for $Mg^{2+}$, for which *CV* before the reservoir was built was 11%, and when the reservoir was in use it increased to62% (very high variation) (Tab. 4). The results obtained by Bor and Elçi (2022) were mostly conforming. After the construction of the Çine reservoir in Aydin, Turkey, the values of *EC*, *DS*, $Ca^{2+}$, $SO_4^{2-}$ and $Na^+$ were significantly reduced. Based on the Mann–Whitney U test, the authors found a significant drop of $Cl^-$ and $Mg^{2+}$ concentrations downstream of the damming.

No statistically significant trends in change of salinity indicators were found in the 2005–2009 period (Tab. 8). In case of *DS*, $Cl^-$ and $Na^+$, negative *S* statistics indicate only the presence of downward tendencies, and for *EC*, $SO_4^{2-}$, $Ca^{2+}$ and $Mg^{2+}$ positive *S* indicate upward tendencies. In the 2015–2019 period, statistically significant downward trends were observed only for *DS* (Fig. 3) and $Ca^{2+}$, and an upward trend was noticed for $Mg^{2+}$. The slope coefficient $\beta$ indicates a monthly reduction of *DS* in the Korzeń water when the reservoir was in operation on the average by 0.5 mg·dm$^{-3}$, with the multiyear mean of 171 mg·dm$^{-3}$ and the range from 112 to 262 mg·dm$^{-3}$. The $Ca^{2+}$ concentration was reduced on the average by 0.178 mg·dm$^{-3}$ a month, and the $Mg^{2+}$ concentration increased by 0.062 mg·dm$^{-3}$ a month in this period of the study, with the multiyear means of 55.1 and 10.1 mg·dm$^{-3}$, respectively (Tab. 4 and 8). Similarly to the period before the reservoir, the *S* values for 2015–2019 indicate an upward trend for *EC* and a downward trend for $Na^+$ concentration in the stream water. However, contrary to the earlier period, the *S* values suggest a downward tendency for $SO_4^{2-}$ and an upward tendency for $Cl^-$ (Tab. 8). Neissi, Tishehzan and Albaji (2019), in their paper presenting the impact of the Jare reservoir (south-west Iran) on quality of surface waters used for irrigation showed that when the dam was built the *EC* values downstream of the structure increased. In natural waters the *EC* depends mainly on the concentration of main ions, such as: $Ca^{2+}$, $Mg^{2+}$, $Na^+$, $Cl^-$, $SO_4^{2-}$ and $HCO_3^-$. The probable reason for the *EC* increase downstream was determined to be transport of soluble salts accumulated in the reservoir to the watercourse downstream of the dam. Increased concentrations of $Ca^{2+}$ and $SO_4^{2-}$ indicated dominating weathering processes of calcite and gypsum, the main substrate minerals in the studied area. In addition, after the construction the authors found lower concentrations of $Mg^{2+}$, $Na^+$, $Cl^-$ and *DS* downstream of the dam. Bor and Elçi (2022) in similar studies on the Çine River in Turkey found that that after the dam was built, the decrease of sulphates concentrations downstream of the structure may be a result of lower pH values than those determined in the earlier period. Increased concentrations of sulphates in water flowing out of the reservoir resulting from release of pollutants by bottom sediments were found in studies by Kubicz and Cieślar (2016).

**Table 8.** Trend test results for salinity indicators

| Indices | 2005–2009 | | | 2015–2019 | | |
|---|---|---|---|---|---|---|
| | $\beta$ | *S* | *p* | $\beta$ | *S* | *p* |
| *EC* | 0.048* | 20* | 0.91* | 0.393* | 126* | 0.40* |
| *DS* | –0.118* | –48* | 0.78* | –0.500* | –330* | 0.03* |
| $SO_4^{2-}$ | 0.103* | 249* | 0.13* | –0.026* | –45* | 0.77* |
| $Cl^-$ | 0.000* | –1* | 1.00* | 0.003* | 37* | 0.81* |
| $Ca^{2+}$ | 0.104* | 215* | 0.19* | –0.178* | –392* | 0.01* |
| $Mg^{2+}$ | 0.011* | 264* | 0.11* | 0.062* | 498* | <0.01* |
| $Na^{2+}$ | –0.019* | –214* | 0.20* | –0.003* | –45* | 0.77* |

Explanations as in Tab. 5.
Source: own study.

The concentrations of metals: potassium ($K^+$), total iron ($Fe^{2+/3+}$) and manganese ($Mn^{2+}$) were statistically different between the periods 2005–2009 and 2015–2019 and were higher in the stream water after the reservoir was built (Tab. 4). The average monthly $K^+$ concentration in the Korzeń water was about 17% higher, and the $Mn^{2+}$ concentration was 4 times higher than the values observed in the earlier period. The $Fe^{2+/3+}$ concentration increased on the average by 0.19 mg·dm$^{-3}$, which is 40.4% of the mean value determined for the period before the reservoir was built. In the first period of the study, the coefficients of variation

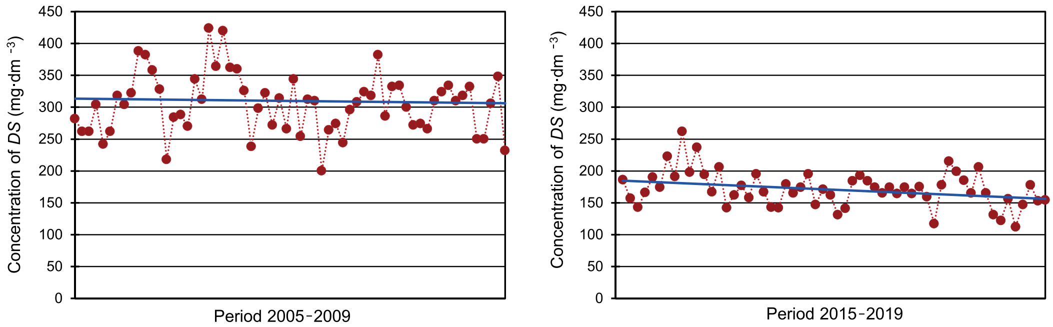


**Fig. 3.** Example of the course of value changes over time for a non-seasonal indicator (dissolved substances *DS*); source: own study

(*CV*) for $Fe^{2+/3+}$ and $Mn^{2+}$ indicate a high and very high random variation, respectively, and in the second period they indicate a very high variation of concentrations of these two metals. The dispersion of values in datasets in case of $K^+$ was similar in both periods – the *CV* values indicate a medium dynamics of concertation changes of this metal (Tab. 4). In studies by Bor and Elçi (2022), the Mann–Whitney U test did not show a significant change of $Fe^{2+/3+}$ and $Mn^{2+}$ concentrations in the Çine River waters after the storage reservoir was built.

Between 2005 and 2009, a statistically significant downward trend was found only for $Fe^{2+/3+}$. The concentration of this metal decreased in the stream water on the average by 0.004 mg·dm$^{-3}$ a month, with the multiyear mean of 0.47 mg·dm$^{-3}$ and the range from 0.08 to 1.18 mg·dm$^{-3}$. The positive value of *S* indicates an upward tendency in $K^+$ concentrations, and absence of tendency ($S = 0$) in case of $Mn^{2+}$ (Tab. 4 and 9). After the reservoir was built, a statistically significant downward trend was observed for $Fe^{2+/3+}$ and $Mn^{2+}$. In case of $K^+$, the previously observed upward tendency was maintained. The rate of $Fe^{2+/3+}$ concentration change was faster in this period than in the previous period. The concentration was reduced on the average by 0.007 mg·dm$^{-3}$, with the multiyear mean of 0.66 mg·dm$^{-3}$ and the range from 0.04 to 7.78 mg·dm$^{-3}$. The $Mn^{2+}$ concentrations decreased on the average by 0.005 mg·dm$^{-3}$ a month in this period of study (Tab. 4 and 9). The iron and manganese compounds get into the water both as a result of natural processes and anthropogenic pollution (Czaplicka *et al.*, 2017). The iron content in rivers comes usually from natural sources and depends on the construction and mineral composition of the soil and subsoil. Industrial wastewater can also be a source of iron in waters. Manganese is widely present in surface waters and its content depends mainly on intensity of eluviation from the subsoil. High manganese and iron concentrations are observed particularly in underground waters (Matysiak and Słowik, 2020). Manganese migrates between sediments and water at varying pH and oxidation easier than water. In favourable hydrodynamic conditions, iron and manganese compounds associated with suspensions travel passively with water flow and undergo sedimentation in the reservoir (Czaplicka *et al.*, 2017). Czaplicka *et al.* (2017) observed that the most important factor for accumulation of iron and manganese compounds in the bottom sediments was the content of the silty fraction and organic substances (strong interrelations were found). In the Korzeń drainage basin, the topsoil horizons are mainly built of common and silty loess rich in iron and manganese which may affect an increased deposition of these elements in the sediments built largely of silty particles. The observed downward trend of $Fe^{2+/3+}$ and $Mn^{2+}$ in the "Skrzyszów" reservoir outflow waters may be a result of accumulation of these metals in bottom sediments of the reservoir, while significantly higher $Fe^{2+/3+}$ and $Mn^{2+}$ concentrations between periods 2005–2009 and 2015–2019 may be related to the release of these metals accumulated in sediments to the Korzeń waters which takes place during periodic outflows of water through bottom sluices. This thesis is supported by incidentally very high concentrations of $Fe^{2+/3+}$ and $Mn^{2+}$ which increase the mean values and cause very high coefficients of variation in the period after the "Skrzyszów" reservoir was built.

**Table 9.** Test trend results for metals

| Indices | 2005–2009 | | | 2015–2019 | | |
|---|---|---|---|---|---|---|
| | *β* | *S* | *p* | *β* | *S* | *p* |
| $K^+$ | 0.002* | 58* | 0.73* | 0.003* | 71* | 0.64* |
| $Fe^{2+/3+}$ | –0.004* | –375* | 0.02* | –0.007* | –473* | <0.01* |
| $Mn^{2+}$ | 0.000* | 0* | 1.00* | –0.005* | –426* | <0.01* |

Explanations as in Tab. 5.
Source: own study.

## CONCLUSIONS

The conducted studies and the literature review indicate that water reservoirs play an important part in shaping water quality in river systems. The chemical composition of water flowing to the reservoir is usually strongly modified during the damming, and the changes can also be observed downstream of the reservoir. A relatively slow water flow allows sedimentation of the suspended matter and initiates the processes reducing the content of nitrogen and phosphorus. Storage reservoirs on watercourses often improve the water quality, although the results of studies by various authors are inconclusive. This is caused by a lot of factors affecting the physical and chemical properties of water flowing out of the reservoir, e.g. quality of the inflow water, biotic and abiotic processes occurring in the reservoir and the reservoir parameters. The quality of outflow water is also affected by the method of water discharge. In the "Skrzyszów" reservoir, the surface outflow type (overflow) is the dominant method, and sporadically the water is discharged through bottom sluices.

The studied data values from the period before the "Skrzyszów" reservoir was built had a downward tendency in case of 9 out of 21 physical and chemical water indicators. Statistically significant downward trends in this period were observed for concentrations of *TSS* and $Fe^{2+/3+}$, and upward trends for pH, *DO* and $DO_{sat}$. When the reservoir was in operation, a downward tendency was observed for 12 studied water quality indicators, and statistically significant downward trends were observed for $N–NO_2^-$, $N–NO_3^-$, *DS*, $Ca^{2+}$, $Fe^{2+/3+}$ and $Mn^{2+}$, and an upward trend was observed for $Mg^{2+}$. An adverse effect of the reservoir was the change of the upward trend to a downward tendency in case of oxygen indicators (*DO* and $DO_{sat}$) and appearance of an upward tendency for organic pollution indicators ($BOD_5$ and $COD_{Mn}$). On the other hand, a favourable phenomenon is that the downward trend for $Fe^{2+/3+}$ was maintained and new downward trends appeared, particularly for $Mn^{2+}$, $N–NO_2^-$ and $N–NO_3^-$.

Statistically significant differences were found between the values of 17 out of 24 water quality parameters determined in two periods of the study. In the period after the reservoir was built, the values of water temperature and pH, $BOD_5$, $COD_{Mn}$ and concentrations of biogenic substances ($N–NH_4^+$, $N–NO_2^-$) and metals ($K^+$, $Fe^{2+/3+}$ and $Mn^{2+}$) were higher than in the earlier period. The concentrations of *TSS*, $P–PO_4^{3-}$ and $N–NO_3^-$ and the values of most water salinity indicators (*EC*, *DS*, $SO_4^{2-}$, $Ca^{2+}$ and $Na^+$) were however significantly lower. An adverse change was the increase of water temperature and concentrations of $N–NH_4^+$ and $N–NO_2^-$, which however were at a relatively low level during the entire 10-year period of the study. The favourable impact of the reservoir on water quality in the Korzeń stream lies mainly in a significant reduction of *TSS* and decreased concentrations of biogenic indicators causing eutrophication of waters, that is $P–PO_4^{3-}$ and $N–NO_3^-$.

As already mentioned in this paper, the small storage reservoir "Skrzyszów" had a significant impact on the hydrochemical regime of the Korzeń stream: physical and chemical water parameters have changed, and in case of some indicators the variation dynamics has also changed or the tendencies and trends over time have been modified or changed. With so many studied indicators, it is not however possible to clearly say if the reservoir has a positive or negative impact on the water quality. Therefore, other aspects related to the reservoir construction need to be also considered, inter alia social and natural aspects. Before the reservoir was built, the water flow sometimes periodically ceased which adversely affected biological life in the Korzeń stream bed. This problem was solved by ensuring a steady, at least at the level of environmental flow, water outflow from the reservoir. In addition, the reservoir area was previously used as extensive grassland, agricultural wasteland and shrub areas. The reservoir improved the biodiversity: numerous species of aquatic and water-dependent plants appeared, as well as animal species (mainly ichthyofauna and birds). The local microclimate has become milder as a result of increased air humidity. The reservoir several times has saved the downstream agricultural areas and buildings in the valleys of the Korzeń stream and its recipient the Wątok River from permeation and floods. Its landscape and touristic function are also important. The "Skrzyszów" reservoir has brought numerous social and natural benefits.

In order to minimise the impact of catchment development on the quality of water in the reservoir and the stream, it would be necessary to: create buffer zones along all watercourses in the form of trees, bushes and extensively used grassy vegetation; in areas with steeper slopes, convert arable land to permanent grassland, and in some cases to protective forests; increase the permeability of compact soils through agrotechnical and agromelioration measures to reduce surface runoff and erosion processes, etc.

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