Effects of a nuclear power plant warmwater outflow on environmental conditions and fish assemblages in a very large river (the Danube, Hungary)

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Abstract

Direct or indirect effects of nuclear power plants' (NPPs) warmwater effluents on the structure of biotic assemblages are poorly known in very large rivers. We examined changes in physical habitat structure, temperature condition and their possible effects on the structure of Danubian fish assemblages due to the outflow of the Paks NPP's warmwater channel, in Hungary. Seasonal surveys conducted both upstream and downstream from the outfall showed that its hydromorphological effects were generally local and comparable to natural or other anthropogenic hydromorphological changes. The effect of the returned cooling water was more apparent in the seasonally recorded surface water temperatures and depended highly on the spatial positioning of the sampling sites. However, environmental and spatial variables accounted only for a low amount of variance in case of both shoreline and offshore fish assemblage data. Overall, we found that the outflow exerted only a local scale effect on the structure of Danubian fish assemblages. Rather, fish assemblages varied largely both inshore and offshore, which dynamics overruled any effects of the artificially elevated temperature. Our study highlights the importance of the assessment of hydrogeomorphological variability of rivers and their influence on fish assemblage variability when examining spatial effects of thermal pollution.

Key words: thermal pollution, habitat use, fish, geomorphology, substrate, Danube River

Running head: effects of a warmwater outflow in a very large river

1. Introduction

Anthropogenic stressors influence aquatic ecosystems in a variety of ways. Of these, physical habitat alterations and the human assisted spread of invasive species are among the top cited factors, which threaten the diversity and integrity of aquatic ecosystems (Dudgeon et al., 2006; Reid et al., 2019). Although less studied, but the destruction of the habitat is often accompanied by thermal and chemical pollution, which may magnify the effect of hydrological and/or geomorphological modifications (Teixeira et al., 2009; Erős et al., 2015).

Nuclear energy plays an important role in the electricity production of the world, comprising cca. 14% of the electric energy needs (Karakosta et al., 2013). Beside clearly renewable energy sources (e.g., solar radiation, wind) nuclear power plants (NPPs) contribute significantly to mitigating greenhouse gas emissions (Adamantiades and Kessides, 2009). On the other hand, the public acceptance of nuclear energy is low, which is mainly due to the problems of handling radioactive waste and safety reasons. Although less known by the public, the operation of NPPs needs a significant amount of cold water to control condenser process temperature. The wasted (thus heated) cooling water is then emptied back to the recipient aquatic environment, which may present a thermal pollution for the biota (Raptis et al., 2016).

Studies on the effect of thermal pollution on ecological assemblages have yielded controversial results. For example, detailed studies highlighted the decline of brown macroalgae assemblages to thermal stress from the effluent discharge of NPPs in coastal marine environments (Schiel et al., 2004; Széchy et al., 2017). On the other hand, no significant effect on coral reef fish assemblages was observed around an NPP in southern Taiwan (Jan et al., 2001). Another study in a coastal environment in Southeastern Brasil showed that fish species richness and diversity (Shannon–Wiener index) was negatively influenced by the thermal pollution from an NPP (Teixeira et al., 2009, 2012). However, the effect depended also on the diversity of habitat structure. In complex habitats, where physical habitat diversity was high, fish assemblages were unaffected by thermal pollution (Teixeira et al., 2012). Although studies from freshwater environments are limited, these studies show the controversial effects of thermal pollution from NPPs on ecological assemblages (see e.g., Descy and Mouvet, 1984; Daufresne et al., 2003). Overall, further studies are needed from a variety of aquatic environments for a detailed understanding of the effect of NPP effluents on the community organization of aquatic organisms.

Our study target, the Paks NPP is situated in Hungary directly at the right bank of the very large Danube River. The power plant is operated by four pressurized-water reactor blocks. The cooling water of the NPP is obtained directly from the Danube, and the used (warmed) water is returned to the Danube through the warmwater channel. The channel has an approximate discharge of 100 $m^3 s^{-1}$ in the majority of the year, when all of the four blocks of the NPP are operating (Janovics et al., 2014). Such a discharge is equal to the discharge of a medium sized river, and consequently may substantially influence the environmental conditions of the mainstem Danube (mean discharge at the study reach is 2300 $m^3 s^{-1}$, while the lowest navigable discharge, which may last for weeks, is 1240 $m^3 s^{-1}$). The confluence zones of rivers are generally characterized by complex hydro- and morphodynamic features (e.g., Bradbrook et al., 2000; Rhoads and Sukhodolov, 2001; Baranya et al., 2015). These locally varying

hydrological and morphological conditions can affect the organization of ecological assemblages (Rice et al., 2006). The focus of recent studies (e.g. Czeglédi et al., 2015; Erős and Lowe, 2019) is a thorough understanding of the significance of tributary effects on the mainstem river in a variety of hydrological, geomorphological and topographic conditions (e.g., discharge, substratum composition, relative size of the tributary and the mainstem river, network position in the catchment).

In this study we assessed the effect of the artificial tributary of the NPP on the abiotic and biotic conditions of the Danube River. We addressed the following questions: i) How does the warm water effluent of the Paks NPP influence the thermal and hydromorphological characteristics of the Danube, ii) Do changes in abiotic conditions influence the structure of Danubian fish assemblages?

2. Material and methods

2.1. Study area

The Danube has a drainage area of approximately 800,000 km^2. River regulation, namely the construction of hydroelectric schemes, especially in the Upper Danube (i.e., in Germany and Austria), and channelization have profoundly modified the physical structure of the Danube throughout its course. The Hungarian reach (Fig. 1), referred to as the 'Middle Danube', runs for 417 km and has a mean annual discharge of ~2000 $m^3 s^{-1}$. The main channel has a substratum dominated by gravel and sand, a mean depth of 4 m and a mean flow velocity of 1.3 m s^{-1}. The banks are relatively natural (except the section lying within Budapest, the capital of Hungary), interrupted with embanked rip-rap shorelines of ~ 100-1000 m long sections. The studied segment (i.e., between 1535-1520 rkm) of the Paks NPP is a typical representative of the Hungarian reach of the river. Here, the substrate is dominated by sand and fine gravel. The river bank is partly modified by rip-rap embankments and groins to enhance navigation.

2.2. Data collection

Altogether three sampling campaigns were conducted in the studied 15 km long reach in spring 2020 (19-21 May) in summer 2020 (21-23 July) and autumn 2020 (15-17 September). Within the study reach we selected eight 500 m long subreaches (Fig. 1) to explore the effect of the NPP on the abiotic characteristics and fish assemblages of the Danube. Of these, three subreaches situated upstream of the warm effluent, and consequently were not influenced by the NPP. On the other hand, five subreaches were selected downstream of the outfall. These were thus supposed to be influenced by the thermal water to a varying degree.

Hydromorphological data and temperature measurements

Water depth and flow velocity were mapped within each subreach along equidistant transects, 50 m apart, by a Teledyne (USA) RDI Rio Grande Workhorse 1200 KHz Acoustic Doppler Current Profiler (ADCP) mounted on a 6 m long measurement vessel. Instantaneous vertical velocity profiles and flow depth values were recorded by the ADCP in every 1.3 seconds. A Stonex (Italy) S9 Real Time Kinematic (RTK) GPS connected to the ADCP recorded the vessel

position simultaneously. For the follow-up analysis, depth-averaged flow velocity values were derived from the vertical profiles. The density of the surveyed transects, 11 within the 500 m long subreach, allowed linear interpolation of both the velocity and depth values. Accordingly, two-dimensional maps of water depth and depth-averaged flow velocity could be derived for each subreach.

The bed material was sampled during the first (spring) campaign at five points along the central cross-section in each subreach, collecting altogether 40 samples. In the four-month time period covering our seasonal measurements, the discharge of the Danube remained under 4500 $m^3 s^{-1}$. Due to the lack of a higher flood during the study period, it could be assumed that the bed material composition did not change significantly after the spring sampling, thus no additional samplings were done. A metal bucket sampler was trawled on the riverbed for a short time and a bulk sample was retrieved. The exact location and the flow conditions were recorded simultaneously by the GPS and ADCP, respectively. The bed material samples were dried and sieved with a Retsch (Germany) AS 450 Basic vibratory sieve shaker to determine grain size distribution.

Water temperature was recorded in two aspects: along cross-sections near the surface by the ADCP and along depth in the verticals of the bed material sampling locations. The vertical samplings were performed by a Teledyne (USA) Digibar S sound velocity profiler. The exact location and the flow conditions were recorded simultaneously by the GPS and ADCP, respectively. The cross-sectional temperature data was also dense enough for linear interpolation in the data processing stage, similarly to water depth and depth-averaged flow velocities. Temperatures along water depth were used to determine spatial variability, particularly near the NPP effluent.

Fish collections

Fish were collected using shoreline electric fishing and offshore benthic trawling. For the sampling of shoreline fish assemblages, 17 (spring 2020) and 18 (summer and autumn 2020) 500 m long rip-rap or natural shoreline stretches were selected (Fig. 1), along the same subreaches where the hydromorphology was revealed. Rip-rap stretches can be characterized as rock and boulder covered artificial banks, while natural stretches lack any apparent shoreline engineering measures and have a general substratum type of sand and fine gravel (Erős et al., 2008). The ratio of natural and rip-rap stretches represented the distribution of these two main shoreline mesohabitat types. The 500 m long units were electrofished from a boat, using a Hans Grassl GmbH (Germany) EL64 IIGI, DC, 7.5 KW device. Fish were caught with a hand held anode (2.5 m long pole with a ring anode of 40 cm diameter and a net mesh size of 6 mm) while slowly moving downstream with the boat as per Wolter and Freyhof (2004). The cathode, a 3 m long copper cable, was floated at the rear of the boat. We used nighttime sampling because former surveys of the Danube (Erős et al., 2008) and other surveys (Wolter and Freyhof, 2004) justified that it is more efficient than daytime sampling of shoreline fish assemblages. At the end of each 500-m unit, captured fish were identified, counted and returned into the water.

Offshore distribution of fish was examined using an electrified benthic framed trawl (Szalóky et al., 2014). In each subreach (Fig. 1), across the width of the main channel, we distributed 5

trawl paths, 500 m long each, excluding the littoral, less than 2 m deep shoreline zone. These paths were approximately equispaced and centered over the approximate place of the main channel centerline (Gutreuter et al., 2009), at the place where abiotic (i.e., bed material) samples were also collected. The trawl consisted of a stainless-steel frame (2 m wide × 1 m high) of which a drift net was attached (mesh size 5 and 8 mm for the inner and outer mesh bag, respectively) (for details see Szalóky et al., 2014). The frame was electrified with a Hans Grassl GmbH (Germany) EL65 IIGI electrofishing device operated with a VANGUARD (USA) HP21 14.9 KW generator. A 6 m long copper cathode cable was connected freely and pulled approx. 2 m before the electrified frame. The fishing team consisted of two people handling the framed net, one handling the electrofishing device and one operating the boat. Trawling was conducted during daytime with a 6.3 m long boat powered by a 50-horsepower outboard Mercury (USA) four stroke engine. Before starting trawling, the operators lowered the frame to the bottom while the boat was slowly moving downstream with the flow. Trawling route was started to be measured by a GARMIN (USA) GPSMAP 60CSx only after the net reached the bottom, which could be easily felt while holding the central rope, and right after electroshocking started. The direct current (approx. 350 V, 33 A) was applied for 5-8 s with 3-5 s breaks between the operations to minimize fright bias and injury of fish. The applied trawling speed was slightly higher than the current velocity of the river (approx. 1.3 m s^{-1}). The collected fish were identified, measured to the nearest mm standard lengths and then released back to the river.

2.3. Statistical analysis

General linear models with normal error distribution and identity link function were used to examine how 1) the bank side (i.e., right vs. left side of the river; the NPP effluent is on the right bank of the Danube), 2) the location of the sampling site (upstream vs. downstream from the effluent) and 3) the season (spring, summer or autumn) were related with temperature values of inshore samples. Similarly, linear models were used to explore how 1) the position of offshore samples (here position of the sample along each subreach across the width of the main channel), 2) the location of the sampling site (upstream vs. downstream from the effluent) and 3) the season (spring, summer or autumn) were related with temperature values of offshore samples. The models were built considering all interaction effects of the three factors. Then, model selections were conducted to identify the minimal adequate models, which hold only significant ($P \leq 0.05$) terms (Crawley 2015). This explorative analysis revealed strong relationships between water temperature and the three tested factors in case of both the inshore and offshore samples (see the Results section). This finding suggested that the effect of temperature could not be disentangled from the effect of the spatial positioning of the sampling sites and the season. Therefore, the factors of spatial positioning (i.e., bank side, location regarding the power plant outflow, and position along the cross-section) were used as effectors to test the influence of the power plant on the habitat and fish assemblage in the further data analyses.

Redundancy analyses were used to test how position and location of the sampling sites, season and the examined environmental variables, such as depth, velocity and substrate composition influenced the species richness and assemblage structure (i.e., relative abundance) of the fish communities for both the shoreline electrofishing and the bottom trawling data. The examined

environmental variables (specifically mean depth, mean velocity, %silt, %sand, %gravel, %pebble, %cobble) were considered as conditional variables in the models to dissect the influence of spatial positioning and related temperature effects on the structure of fish assemblages from those caused by hydrogeomorphological variability. Species richness data (i.e., number of species captured in the 500 m long sampling units) were ln(x+1), while abundance-based fish assemblage data (i.e., number of individuals of each species captured in the 500 m long sampling units) were Hellinger transformed before the analyses. Species occurring in less than three sampling units had been removed and the conditional variables had been normalized to zero mean and unit variance in the abundance-based analyses.

3. Results

3.1. Environmental effects of the warmwater outflow

The flow discharges in the Danube were 1700, 2700 and 1800 $m^3 s^{-1}$ during the spring, summer and autumn campaigns, respectively. Accordingly, larger water depth values were recorded in summer (Fig. 2). Mean water depth varied between 4-6 m between the seasonal surveys. Morphological features typical to a meandering reach can be observed on the depth maps. Cross-section shapes at river bends (subreach No. 3, 5, 6) show asymmetric character, whereas inflexion reaches (e.g., subreach No. 2, 7, 8) are more complex, indicating shallow regions in the middle of the channel.

Depth-averaged flow velocities showed a strongly varying pattern among the subreaches (Fig. 3). Maximum flow velocities varied between 1.4 and 1.6 $m s^{-1}$ between seasons. The warm effluent affected the flow conditions to some extent. Specifically, the confluence of the outflow narrowed the streamline in the subsequent subreach No. 4, compared to other subreaches. Moreover, the groin at the right bank (Fig. 3) further narrowed the channel, resulting in increased flow velocity in the main stream and close to zero flow velocities in the recirculation zone downstream of the groin. Among the studied subreaches, this was the most complex in terms of the flow field.

The bed material composition was dominated by sand, gravel and pebble in each subreach (Fig. 4). The proportion of silt (<0.25 mm) and cobble (>64 mm) were uniformly very small and zero, respectively. The spatial variability of the bed composition was low, the sand content, however, was somewhat higher at the first and last subreaches. Overall, the effluent did not seem to affect the subreach grain size composition of the bed material.

The effect of the returned cooling water was more apparent in the seasonally recorded surface water temperatures than on hydromorphological features of the river (Fig. 5). The operational temperature of the cooling water is constantly 10°C higher than the ambient temperature of the river. The differences we measured, however, remained under 4°C, due to the mixing induced cooling between the outfall and the subreaches. The highest differences (3-4°C) were observed in subreach No. 4, which was nearest and thus most affected, at 200 m distance from the effluent. The temperature gradient continuously decreased downstream, with a weakening rate however, and remained at least 1-2°C within the investigated reach. Although the warmwater

plume somewhat widened, in the last two subreaches only the nearbank regions seemed to be affected. Within the studied ~8 km reach downstream of the warmwater outfall, no complete mixing took place. It is important to note that the warmer water affected the side arms along the right bank, too.

General linear models showed significant relationships between water temperature and the spatial factors and season. Specifically, shoreline temperature (Table 1) depended on the three-way interaction of the bank side, location, and season (adjR^2 = 0.99, $F_{11,41}$=643.9, p<0.001), indicating that the variability of temperature values in a given season could be explained by both the bank side and location (upstream vs. downstream from the outflow) of the sampling sites. In case of the offshore samples (Table 2), temperature values depended on the two-way interaction of the position along the cross-section and location, and also on the two-way interaction of the location and the season (adjR^2 = 0.96, $F_{29,90}$=111.7, p<0.001). These results show that water temperature varied according to the location but its magnitude was highly controlled by the position along the cross-section, and that the variability caused by the location was also driven by the seasons (Fig. 5).

3.2. Fish assemblages

A total of 16,398 fish specimens were collected during the seasonal surveys and identified to 36 species (Table 3). Shoreline electrofishing yielded a larger number of species compared with offshore trawling, independently of the season. The bleak, *Alburnus alburnus* (49.53%); round goby, *Neogobius melanostomus* (11.42%); white-finned gudgeon, *Romanogobio vladykovi* (9.61%) and the white bream, *Blicca bjoerkna* (8.71%) were the most dominant species in the shoreline samples, but beside these, many species proved to be relatively abundant in the shoreline samples. On the contrary, only a few species were dominant in the offshore samples including the white-finned gudgeon, *Romanogobio vladykovi* (41.47%); the round goby, *Neogobius melanostomus* (29.64%) and the Danube streber, *Zingel streber* (13.17%).

Albeit significant, the spatial and environmental variables accounted only for a low amount of variance of species richness in case of the shoreline electrofishing samples (adj R^2 = 0.1854). The covariant environmental variables (i.e., velocity, depth and substrate composition) explained a much higher proportion of the variance (47.5%), than the spatial variables (24.9%). Overall, the three-way interaction of season, location and position variables was responsible for the variance in species richness (Table 4). The variance in species richness explained by environmental and spatial variables was also low for the offshore trawling samples (adj R^2 = 0.2667). However, here the covariant environmental variables explained a lower proportion of the variance (15.3%), than the spatial variables (40.5%). Interestingly, a strong effect of season was observed on species richness (Table 5), but it is important to note, that the three-way interactions of season, location and position was only marginally insignificant (p=0.063).

Similarly to species richness, only a low amount of variance of assemblage structure (Hellinger transformed abundance data) could be explained by environmental and spatial variables (adj R^2 = 0.1730) for the shoreline electrofishing samples. The covariant environmental variables (i.e., velocity, depth and substrate composition) explained a higher proportion of the

variance (45.2%), than the spatial variables (24.7%). Again, a three-way interaction of season, location and position was found to be responsible for the variance in the assemblage structure (Table 6). Environmental and spatial variables accounted only for a very low proportion of variance in the offshore structure of fish assemblages (adj $R^2 = 0.0865$). The covariant environmental variables explained a lower proportion of the variance (17.2%) than the spatial variables (27.5%). Beside the effect of the covariant environmental variables, seasonal effects and inner position within a subreach was the most important determinant of the offshore fish assemblage structure, while the impact of location (i.e., upstream or downstream from the outflow) proved to be insignificant (Table 7).

4. Discussion and conclusions

Several studies have shown that tributary confluences influence hydromorphological conditions of recipient rivers, the extent of which depends on many factors (e.g., Bradbrook et al., 2000; Rhoads and Sukhodolov, 2001; Benda et al., 2004; Rice et al., 2006; Jones and Schmidt, 2017). For example, the flow pattern at confluences is mainly influenced by the horizontal alignment of the merging channels, the local morphological alterations and the unstable shear layer between the two flows at different speed (Bradbrook et al., 2000). The ratio of the flow discharges also plays an important role (e.g., Baranya and Józsa, 2007). The mixing and transport of sediments and pollutants is also strongly determined by the local hydromophological features at confluences (Boyer et al., 2006; Szupiany et al., 2009; Baranya et al., 2015). In our study system the observed local hydromorphological effects of the NPP outflow were comparable to natural or other human induced hydromorphological changes. For example, the natural bending of the river generating secondary flow components has a deterministic role in the spatial variation of the flow velocity field, showing, in fact, a more important factor than the returned cooling water of the NPP. Similarly, both flow depth and velocity maps show that the mesoscale effects of riverbed meandering, the presence of groins, side arms, point and side bars were more important determinants of hydromorphological changes than that caused by the effluent (Fig. 2, 3). In addition, the warm water does not transport sediment, which could alter the substrate composition of the mainstem. Therefore, although deposition bars of debris and shells were observed in the vicinity of the confluence, tributary effects of the NPP outflow were only very local and confined to a not more than 500 m long segment downstream the outfall.

Temperature effects, on the contrary, were larger, and clearly distinguishable from the background natural values. Although the largest difference in temperature was observed in the subreach right below the effluent (subreach No. 4), differences in temperature were clear between the left and right banks of course exclusively downstream of the effluent in each subreach. Although it is clear that temperature effects of the NPP outflow may depend substantially on discharge and temperature differences between the warmwater channel and the recipient river, most studies have showed substantial decrease of temperature along a spatial gradient from the effluent in both coastal areas and rivers (Daufresne et al., 2003; Teixeira et al., 2012) – similarly to our findings.

Environmental and spatial variables accounted only for a low amount of variance in case of both shoreline and offshore fish assemblage data. Such a low amount of variability is not surprising, especially offshore, where the relatively short within-segment environmental gradients (deep water, relatively uniform velocity and substrate conditions) make assemblage structure hardly predictable (Wolter and Freyhof, 2004; Szalóky et al., 2021). Similarly to other studies, seasonal changes accounted for a relatively large proportion of variation in the data. These changes can be due to seasonal variations in flow regime and temperature (and related capture efficiency effects), partial migrations of fish between different segments of the river and their side arms and population dynamics of young of the year fish (Matthews, 1998; Wolter and Bischoff, 2001).

Environmental variables were the major determinant of shoreline fish assemblage structure (both richness and relative abundance). This indicates that variations in substrate composition, depth, and velocity were more important variables in structuring fish assemblages than spatial and associated temperature related variability inshore. Our former study in the Hungarian reach of the Danube indicated that species richness of the two major habitat types (i.e., natural sand and gravel dominated shorelines vs. stone and boulder covered rip-raps) did not significantly differ at night, but the composition and relative abundance of the species may differ largely (Erős et al., 2008). Unfortunately, the distribution of natural and rip-rap covered stretches were not ideal in the study system to clearly test and dissect the effects of the environmental and spatial or temperature related factors. This is because following the natural bending of the river, rip-raps covered the right and left bank upstream and downstream of the outflow, respectively, while natural stretches showed the reverse distribution. Nevertheless, the three-way interaction in the data set among season, location and bank side, clearly shows that the structure of fish assemblages can vary largely inshore, which indicates no clearly detectable role of thermal pollution effects. Overall, this result corresponds with the results of Teixeira et al. (2012) who found that structural diversity/complexity of the habitat was an important determinant of fish assemblage structure in the vicinity of an NPP in the coastal area in Southern Brazil. In that study, thermal pollution affected fish assemblages rather in an indirect manner, by decreasing benthic cover of corals, macroalgae and sponges, which in turn decreased structural complexity that exerted a strong effect on fish habitat use. However, fish assemblages remained unaffected at sites with complex physical habitat structure. These results indicate the critical importance of habitat structure in assessing and mitigating the effects of thermal pollution on the structure of fish assemblages.

Contrary to inshore patterns, offshore fish assemblage structure was determined more by spatial variability than merely hydrogeomorphological (i.e., environmental) effects. Nevertheless, the overall explained variance was low, and, importantly, almost zero for the relative abundance data. The seasonally and spatially variable abundance of species confirms our former findings that fish show very elusive habitat responses to the relatively short environmental gradients offshore (Szalóky et al., 2021).

A further step towards applying the results may be assessing the cumulative impact of not just one, but more point source heat effluents. Our findings here (i.e., that the thermal effluent entering from the right only affected the right shoreline zone) suggest that it is important to take

account on whether these point sources are located on the same or on different sides of the river. Naturally, the effect of such point source thermal pollutions also depends largely on the differences in discharge and temperature patterns between the effluents and the recipient river. Although, series of thermal pollution effects are rare in single rivers, there are some notable exceptions. For example, in Europe a large fraction of the flow of the Rhine, the Weser and the Po Rivers are affected by thermal pollution (Raptis et al., 2016). It is thus important to study the cumulative impact of thermal pollution effects in river systems, especially with the acceleration of global warming.

In conclusion, although several studies have shown clear evidence of a variety of effects of NPPs on biotic assemblages, we found that the Paks NPP exerts only a local scale effect on the structure of Danubian fish assemblages. This might be due to the fact, that the outflow does not significantly influence the hydrogeomorphical features and thermal conditions of the river, relative to natural and other human induced variability, or at most only at a very small spatial extent. In fact, we found that the spatiotemporal structure of fish assemblages can vary largely both inshore and offshore, which dynamics overrule the effect of the artificially elevated temperature. This study thus highlights the importance of the examination of hydrogeomorphological variability of rivers and their effects on fish assemblage variability when examining spatial effects of thermal pollution.

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Significance statement to the general public

We examined the effect of a nuclear power plant's warmwater inflow on habitat and temperature conditions and on the structure of fish assemblages in the Danube River, Hungary. We found that the outflow exerted only a local scale effect on hydromorphology and fish assemblage structure. The effect of the returned cooling water was more apparent in the seasonally recorded surface water temperatures and depended highly on the spatial positioning of the sampling sites. Our study highlights the importance of the assessment of hydrogeomorphological variability of rivers and their influence on fish assemblage variability when examining spatial effects of thermal pollution.

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Figure captions

Fig. 1. Map of the study area and location of the sampling units upstream and downstream from the warm water effluent (indicated by red arrow) of the Paks nuclear power plant in the Danube River, Hungary. R and N letters indicate the location of rip-rap (R) and natural (N) shoreline sampling units, while light blue blocks with numbers show the location of surveyed subreaches. The flow direction is shown by a blue arrow on top. Two groins are indicated by two black polygons in subreach No. 1 and 4 (yellow arrows pointing towards).

Fig. 2. Depth maps of the subreaches a) in spring, b) in summer and c) in autumn. Flow direction and the warm effluent are indicated by a blue and a red arrow, respectively. Seasonal differences are closely following the differences in measured flow discharges (1700, 2700 and 1800 m^3s^{-1}, respectively). The local scouring effect of the two groins in subreach No. 1 and 4 is clearly visible. Otherwise the bathymetry of the study area is characterized by the two subsequent bends, with no apparent anthropogenic influence.

Fig. 3. Velocity maps of the subreaches a) in spring, b) in summer and c) in autumn. Flow direction and the warm effluent are indicated by a blue and a red arrow, respectively. The coherence here with seasonal discharges is weaker than by the depth maps. The shading effect of the groins can be seen as lower velocity values near them. The narrowing of the streamline in subreach No. 4 may be the only and also low influence of the warm effluent.

Fig. 4. Substrate composition of the study area determined on subreach-level. Flow direction and the warm effluent are indicated by a blue and a red arrow, respectively. The studied reach is dominated by sand and gravel, while silt and cobble are uniformly minimal or missing, respectively. No apparent influence of the effluent was observed.

Fig. 5. Temperature maps of the subreaches a) in spring, b) in summer and c) in autumn. Flow direction and the warm effluent are indicated by a blue and a red arrow, respectively. A clear heat tail, separated from the ambient temperature is visible in all three subfigures. Note that the left bank is unaffected on the entire reach. A side arm on the right bank diverts the warm flow, which then returns between subreaches No. 5 and 6, hence the higher temperatures in subreach No. 6.

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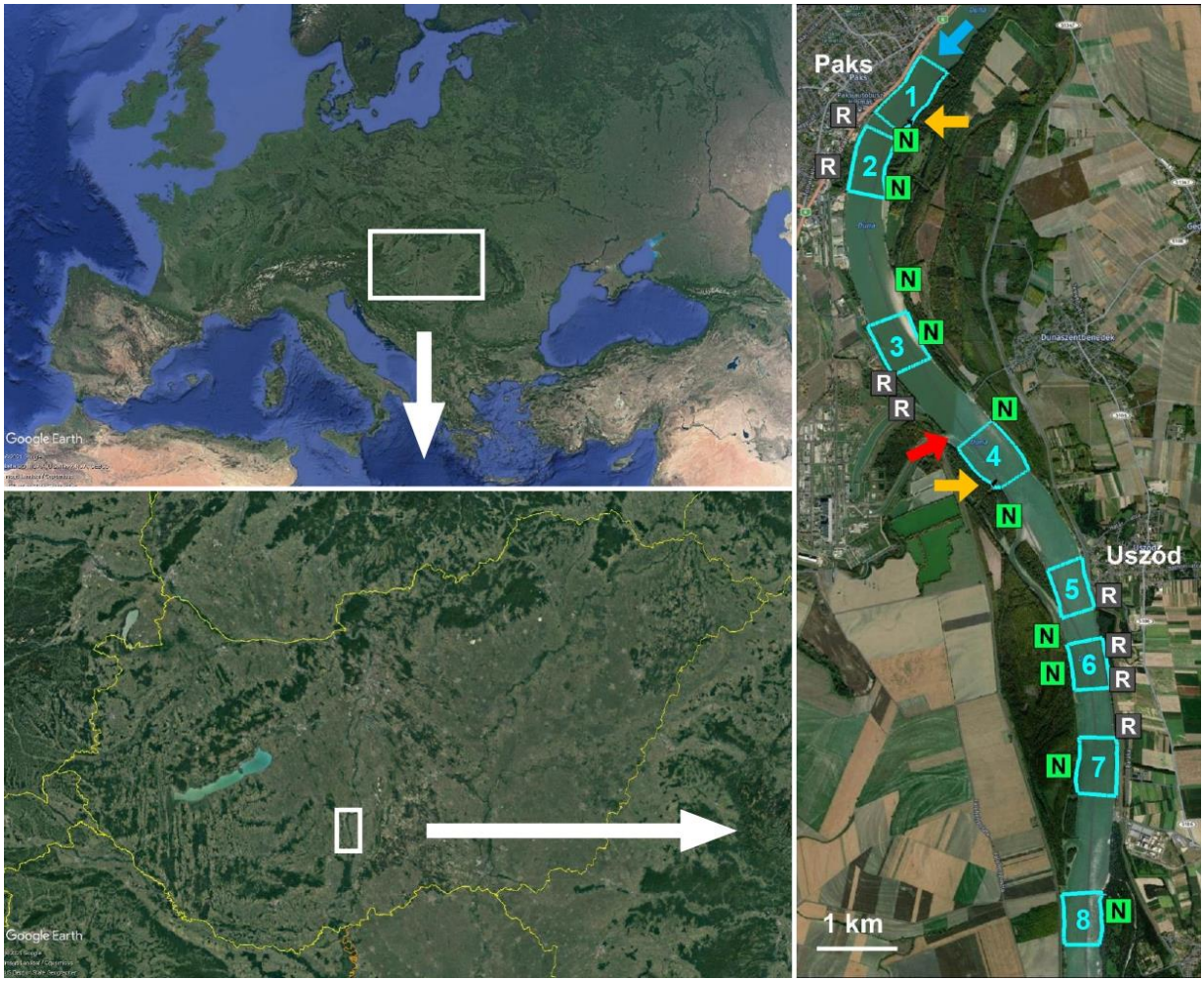
Fig. 1.



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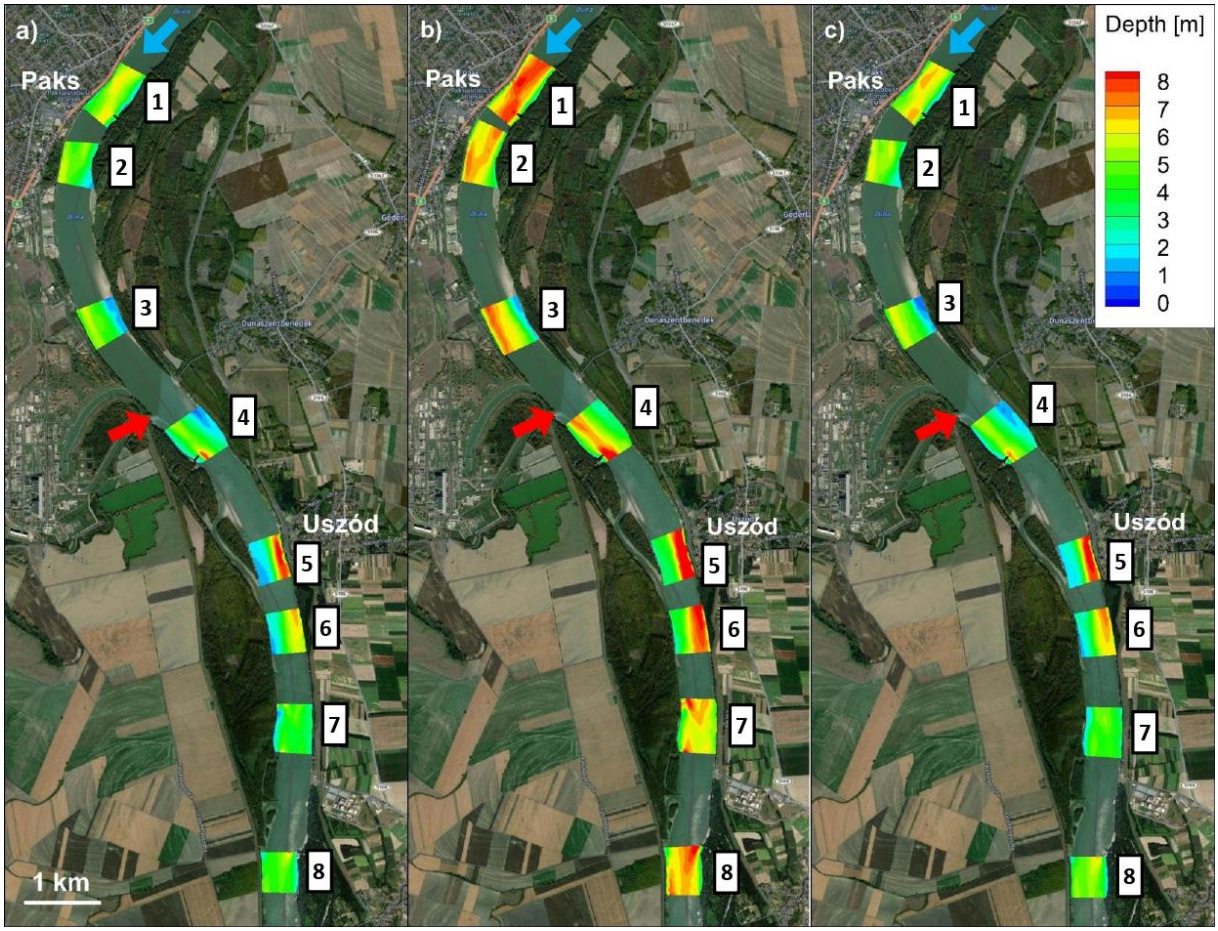

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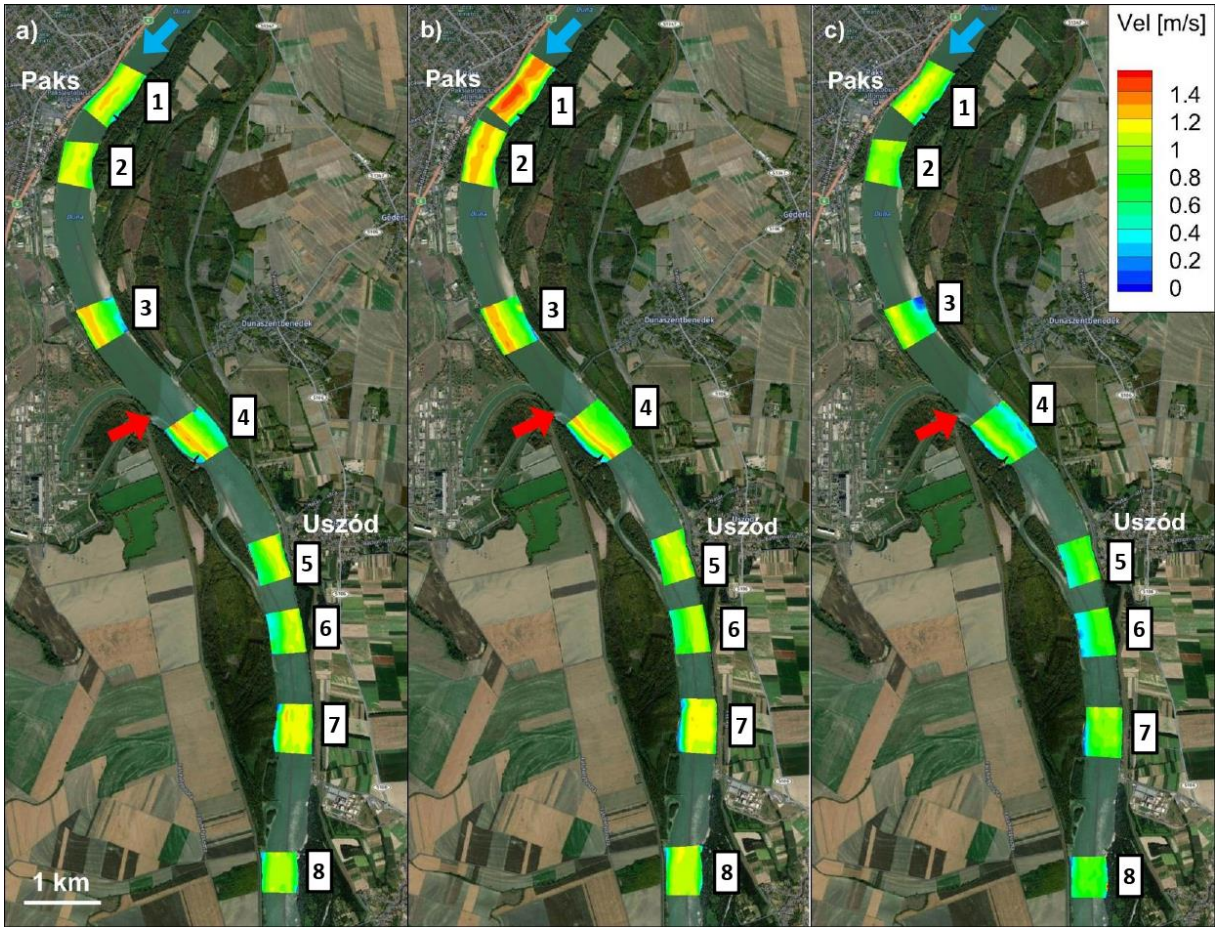

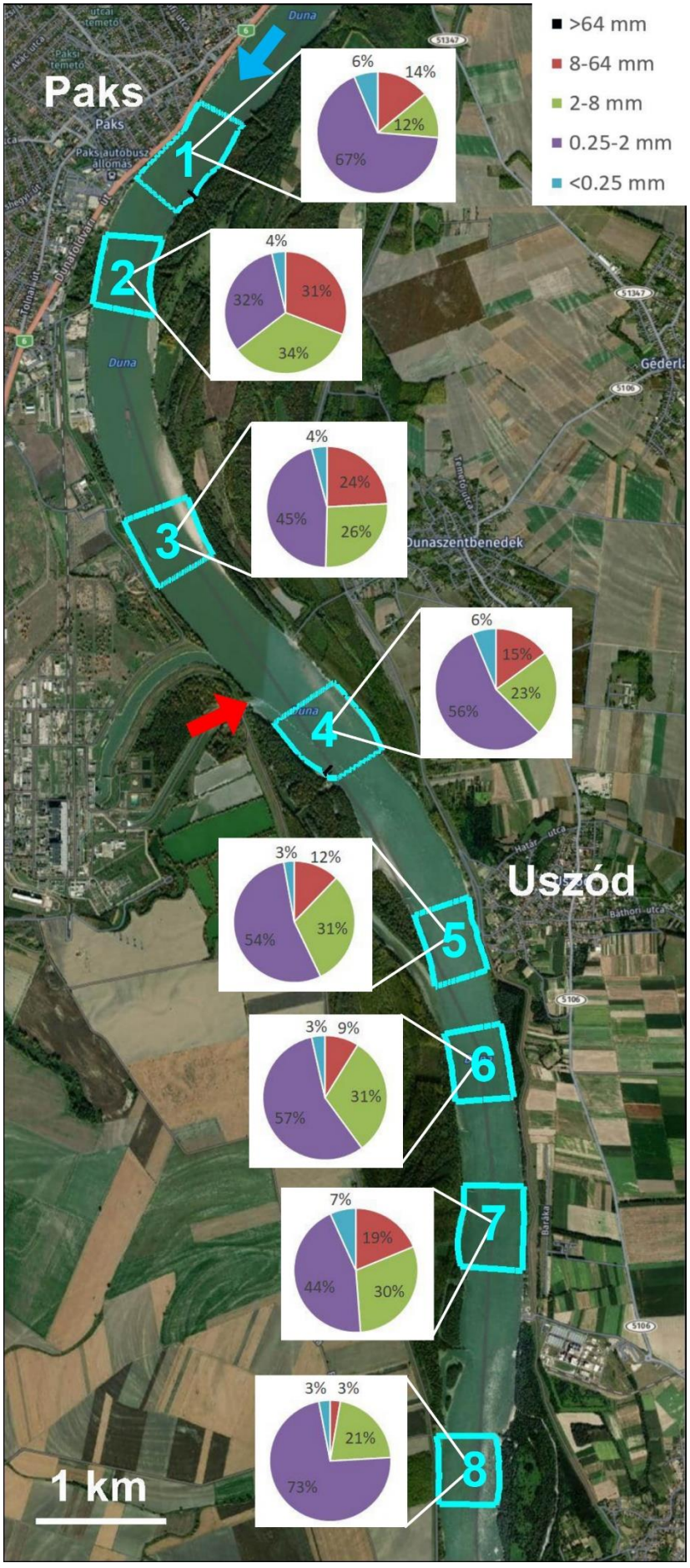
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Fig. 4.

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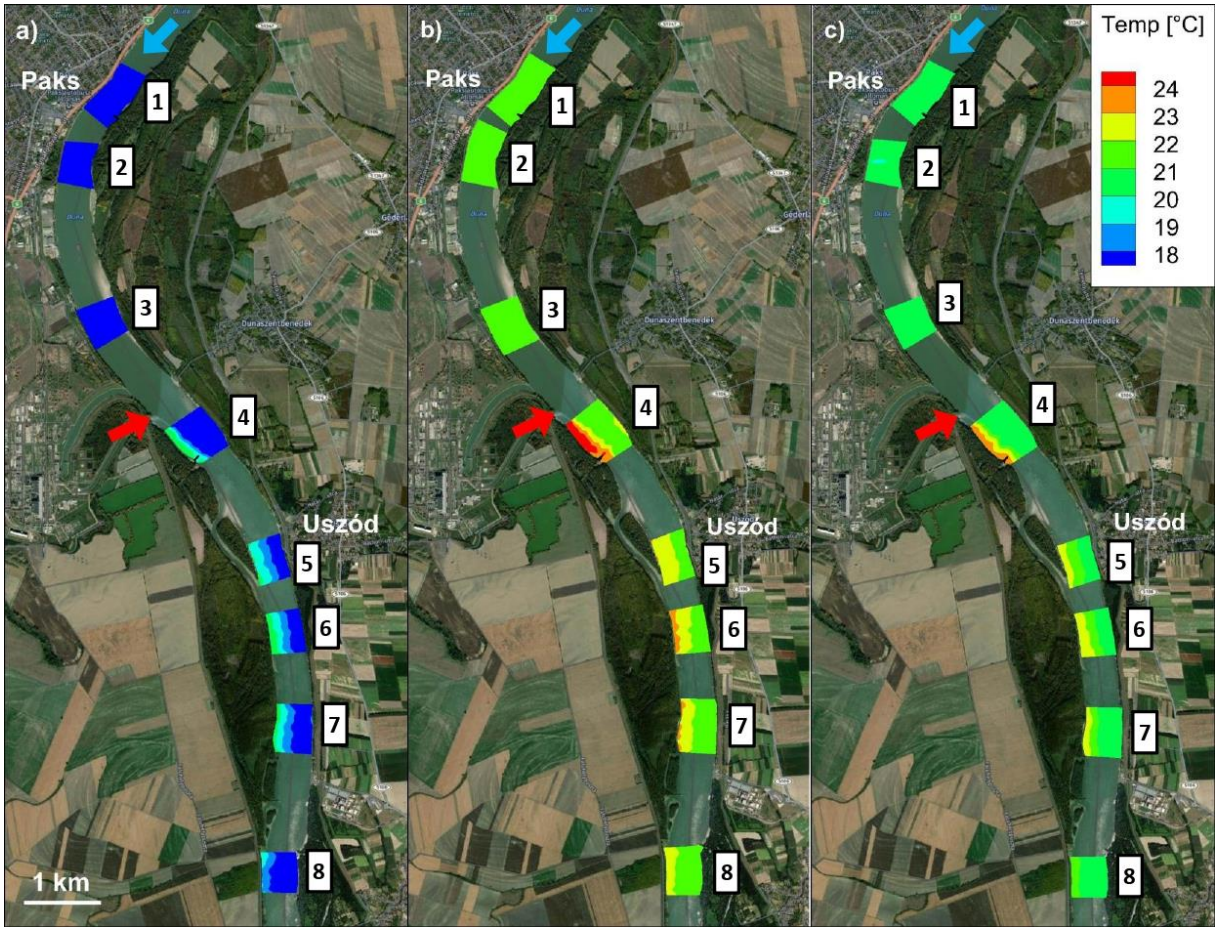
Fig. 5.



Tables

Table 1. Anova table of the minimal adequate GLM model used to explore the relationships between shoreline water temperature (response variable) and spatial location of the sampling sites and seasons (explanatory factors). Bank, two-leveled factor coding the right and left side of the river. Location, two-leveled factor coding the upstream or downstream location of the sampling sites to the outflow of the power plant. Season, a three leveled factor coding the spring, summer, and autumn sampling sessions. Colons denote interaction between the factors.

	Df	Sum of Sq	F statistics	p-value
bank	1	7.7	252.3	< 0.001
location	1	28.5	931.2	< 0.001
season	2	161.9	2641.5	< 0.001
bank:location	1	14.2	463.8	< 0.001
bank:season	2	0.8	13.1	< 0.001
location:season	2	3.0	48.5	< 0.001
bank:location:season	2	0.9	14.9	< 0.001

Table 2. Anova table of the minimal adequate GLM model used to explore the relationships between offshore water temperature (response variable) and spatial location of the sampling sites and seasons (explanatory factors). Position, five-leveled factor coding the location along the cross-section of the river. Location, two-leveled factor coding the upstream or downstream location of the sampling sites to the outflow of the power plant. Season, a three leveled factor coding the spring, summer, and autumn sampling sessions. Colons denote interaction between the factors.

	Df	Sum of Sq	F statistics	p-value
position	4	8	15.06	< 0.001
location	1	25	184.83	< 0.001
season	2	399	1491.38	< 0.001
position:location	4	5	9.82	< 0.001
location:season	2	2	5.69	0.005

585
586 **Table 3.** The fish species collected in the Danube River and their relative abundance in the inshore and offshore samples.

Species name	Common name	Code	**Total**	Offshore samples				Inshore samples			
				Sum	Spring	Summer	Autumn	Sum	Spring	Summer	Autumn
Alburnus alburnus (Linnaeus, 1758)	Bleak	albalb	**42.05**	1.14	1.52	0.61	1.47	49.53	32.35	57.82	54.77
Romanogobio vladykovi (Fang, 1943)	White-finned gudgeon	romvla	**14.54**	41.47	36.87	51.27	34.72	9.61	3.10	12.27	11.82
Neogobius melanostomus (Pallas, 1814)	Round goby	neomel	**14.24**	29.64	22.98	23.40	37.22	11.42	25.37	2.72	8.09
Blicca bjoerkna (Linnaeus 1758)	White bream	blibjo	**7.74**	2.48	2.78	1.93	2.85	8.71	10.63	7.76	8.13
Leuciscus idus (Linnaeus, 1758)	Ide	leuidu	**3.88**	0.39	0.25	0.10	0.69	4.52	9.24	3.43	2.52
Chondrostoma nasus (Linnaeus, 1758)	Common nase	chonas	**2.89**	1.30	1.77	1.83	0.69	3.18	5.38	4.48	1.41
Sander lucioperca (Linnaeus, 1758)	Pike-perch	sanluc	**2.09**	0.47	0.25	0.61	0.43	2.38	2.75	2.19	2.27
Zingel streber (Siebold, 1863)	Danube streber	zinstr	**2.04**	13.17	13.38	16.28	10.45	0.00	0.00	0.00	0.00
Ponticola kessleri (Günter, 1861)	Bighead goby	ponkes	**1.27**	0.32	0.00	0.10	0.60	1.44	1.39	0.80	1.77
Rutilus pigus (Lacepède, 1803)	Danube Roach	rutpig	**1.24**	0.51	1.26	0.00	0.69	1.38	0.71	0.62	2.09
Ballerus sapa (Pallas, 1814)	White-eye bream	balsap	**1.10**	1.18	1.26	0.61	1.64	1.08	2.04	0.99	0.62
Gymnocephalus schraetser (Linnaeus, 1758)	Schraetzer	gymsch	**1.10**	2.21	9.60	0.20	1.38	0.89	0.73	1.02	0.92
Abramis brama (Linnaeus, 1758)	Freshwater bream	abrbra	**1.08**	1.77	2.27	0.81	2.42	0.95	1.28	2.32	0.14
Leuciscus aspius (Linnaeus, 1758)	Asp	leuasp	**0.68**	0.04	0.00	0.00	0.09	0.79	0.73	1.05	0.71
Babka gymnotrachelus (Kessler, 1857)	Racer goby	babgym	**0.67**	0.35	0.00	0.20	0.60	0.73	0.16	0.37	1.19
Vimba vimba (Linnaeus, 1758)	Vimba bream	vimvim	**0.46**	0.55	2.53	0.20	0.17	0.45	1.28	0.09	0.17
Zingel zingel (Linnaeus, 1766)	Zingel	zinzin	**0.41**	0.59	1.01	0.41	0.60	0.38	0.11	0.06	0.68
Squalius cephalus (Linnaeus, 1758)	Chub	squcep	**0.41**	0.00	0.00	0.00	0.00	0.48	0.35	0.49	0.55
Sander volgensis (Gmelin, 1789)	Volga pikeperch	sanvol	**0.35**	0.08	0.00	0.00	0.17	0.40	0.08	0.15	0.69
Barbus barbus (Linnaeus, 1758)	Barbel	barbar	**0.35**	1.46	2.27	0.61	1.90	0.15	0.14	0.12	0.17
Silurus glanis (Linnaeus, 1758)	Wels catfish	silgla	**0.31**	0.39	0.00	0.41	0.52	0.30	0.41	0.37	0.20
Lota lota (Linnaeus, 1758)	Burbot	lotlot	**0.30**	0.00	0.00	0.00	0.00	0.36	0.76	0.00	0.32
Rutilus rutilus (Linnaeus, 1758)	Roach	rutrut	**0.19**	0.04	0.00	0.00	0.09	0.22	0.24	0.46	0.09
Neogobius fluviatilis (Pallas, 1814)	Monkey goby	neoflu	**0.13**	0.00	0.00	0.00	0.00	0.16	0.05	0.09	0.24
Perca fluviatilis (Linnaeus, 1758)	European perch	perflu	**0.12**	0.00	0.00	0.00	0.00	0.14	0.05	0.12	0.19

Proterorhinus semilunaris (Heckel, 1837)	Western tubenose goby	prosem	**0.08**	0.12	0.00	0.10	0.17	0.07	0.03	0.00	0.13
Carassius gibelio (Bloch, 1782)	Prussian carp	cargib	**0.07**	0.04	0.00	0.10	0.00	0.08	0.30	0.00	0.00
Cyprinus carpio (Linnaeus, 1758)	Common carp	cypcar	**0.05**	0.04	0.00	0.10	0.00	0.05	0.11	0.06	0.01
Esox lucius (Linnaeus, 1758)	Northern pike	esoluc	**0.04**	0.00	0.00	0.00	0.00	0.05	0.08	0.00	0.06
Gymnocephalus baloni (Holčik & Hensel, 1974)	Danube ruffe	gymbal	**0.03**	0.00	0.00	0.00	0.00	0.04	0.08	0.00	0.03
Gymnocephalus cernua (Linnaeus, 1758)	Ruffe	gymcer	**0.02**	0.00	0.00	0.00	0.00	0.02	0.05	0.03	0.00
Hypophthalmichthys molitrix (Valenciennes, 1844)	Silver carp	hypmol	**0.02**	0.00	0.00	0.00	0.00	0.02	0.00	0.09	0.00
Acipenser ruthenus (Linnaeus, 1758)	Sterlet sturgeon	acirut	**0.02**	0.12	0.00	0.10	0.17	0.00	0.00	0.00	0.00
Pelecus cultratus (Linnaeus, 1758)	Sichel	pelcul	**0.01**	0.08	0.00	0.00	0.17	0.00	0.00	0.00	0.00
Eudontomyzon mariae (Berg, 1931)	Ukrainian brook lamprey	eudmar	**0.01**	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.01
Ballerus ballerus (Linnaeus, 1758)	Zope	balbal	**0.01**	0.04	0.00	0.00	0.09	0.00	0.00	0.00	0.00
	Number of specimens collected		**16398**	**2537**	**396**	**983**	**1158**	**13861**	**3678**	**3236**	**6947**

587

588 **Table 4.** Permutation test (n=2000) by terms under the reduced model of the partial
589 redundancy analysis modelling logarithm transformed ($x' = \ln(x+1)$) species richness of the
590 shoreline samples (response). Colons indicate interaction between the factors.

	Df	Variance	F statistics	p-value
season	2	0.00811	6.275	0.004
location	1	0.00027	0.417	0.524
bank	1	0.00002	0.032	0.870
season:location	2	0.00220	1.702	0.209
season:bank	2	0.00138	1.068	0.348
location:bank	1	0.00122	1.886	0.175
season:location:bank	2	0.00603	4.669	0.016

591

Table 5. Permutation test (n=2000) by terms under the reduced model of the partial redundancy analysis modelling the logarithm transformed ($x' = \ln(x+1)$) species richness of the offshore samples (response). Colons indicate interaction between the factors.

	Df	Variance	F statistics	p-value
season	2	0.03622	14.255	< 0.001
location	1	0.00201	1.583	0.214
position	4	0.00854	1.681	0.173
season:location	2	0.00766	3.016	0.054
season:position	8	0.01766	1.737	0.104
location:position	4	0.00461	0.907	0.464
season:location:position	8	0.02006	1.974	0.063

Table 6. Permutation test (n=2000) by terms under the reduced model of the partial redundancy analysis modelling the Hellinger transformed composition of fish abundances in the shoreline samples. Colons indicate interaction between the factors.

	Df	Variance	F statistics	p-value
season	2	0.02947	5.543	< 0.001
location	1	0.00635	2.389	0.021
bank	1	0.00282	1.062	0.367
season:location	2	0.00407	0.765	0.718
season:bank	2	0.01119	2.104	0.012
location:bank	1	0.00413	1.555	0.136
season:location:bank	2	0.01384	2.603	0.002

Table 7 Permutation test (n=2000) by terms under the reduced model of the partial redundancy analysis modelling the Hellinger transformed composition of fish abundances in the offshore samples. Colons indicate interaction between the factors.

	Df	Variance	F statistics	p-value
season	2	0.01648	3.297	< 0.001
location	1	0.00395	1.580	0.108
position	4	0.02720	2.720	<0.001
season:location	2	0.00940	1.879	0.015
season:position	8	0.01796	0.898	0.729
location:position	4	0.01172	1.172	0.214
season:location:position	8	0.01635	0.818	0.861