Mining salinisation of rivers: its impact on diatom (Bacillariophyta) assemblages

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Abstract: The composition of the diatom assemblages was analysed in four rivers of Upper Silesia, Poland in 2017. The diatom assemblages studied were found to reflect anthropogenic salinization caused by mining activities. The assemblages in those rivers characterised by the highest salinity (Bolina and Mleczna) showed a relatively low taxonomic richness. The diatom assemblages were dominated by species typical of brackish or marine waters. The rivers with a minimal or weak anthropogenic impact (Centuria and Mitręga) supported taxonomically richer diatom assemblages typical of mid–altitude siliceous or calcareous streams (respectively), that have a fine particulate substratum. The presence of a new species, Planothidium nanum sp. nov., was revealed. The new species shows a unique set of morphological characters, including small size; its elliptical outline as well as very widely–spaced central striae on the sternum valve (sinus) and widely–spaced central striae on the raphe valve allow to separate it from other similar Planothidium.

Key words: diatom, mining activities, new species, river salinisation, Upper Silesia, Poland

INTRODUCTION

Aquatic organisms are the first environmental component that reacts to any hydromorphological and physico–chemical modifications of riverine water quality. Studies of aquatic organisms are therefore very useful in assessing the human impact and detecting any changes in lotic ecosystems. The use of various organisms that react to long– or short–term changes in aquatic environments gives a precise outlook on the health of these ecosystems (Ector & Rimet 2005; Kelly et al. 2009; Li et al. 2010; Solak et al. 2012; Poikane et al. 2016). Additionally, the EU Water Framework Directive (WFD 2000) recommended that all biological components be used in order to have a full view of the ecological status of flowing waters. The diatoms used to monitor waters are particularly good indicators (e.g. Kelly 2002; Bąk et al. 2004; Vilbaste et al. 2004; Bąk & Szlauer–Łukaszewska 2012). They can be found in every aquatic environment in both clean springs and in heavily polluted environments. They settle on various substrates such as stones, macrophytes or on anthropogenic substrates. In addition, diatoms form a large part of the benthos, often reaching a value of 90–95% (Ács et al. 2004). However, the strong water pollution that is caused by, for example, salinisation is the main factor that limits the occurrence of fauna and flora (Williams 1987). Therefore, in heavily polluted water environments in which it is impossible to examine all of the recommended biological elements, diatoms are very good indicators, especially when there is a lack of invertebrates and ichthyofauna.

The central part of Southern Poland is the most urbanised and at the same time the most polluted region. Upper Silesia, which is located in this part of the country, is a historic area in Poland and the Czech Republic (the upper Odra basin and the initial course of the Vistula) and is also one of the most industrialised and urbanised regions in Europe. Upper Silesia has no natural water reservoirs, and it has only artificial reservoirs (Lewin et al. 2015). The degradation of many of the rivers in this region is primarily connected with hydromorphological transformations, the inflow of domestic sewage from household sewers and salt water from coal mines. These rivers are characterised by an increased concentration of nutrients and chlorides, water hardness and often a very high conductivity (e.g. Halabowski et al. 2019a, 2019b; Spyra et al. 2015; Lewin et al. 2018; Sowa et al. 2018). The high level of urbanisation of the Upper Silesian region and the use of salts as de–icing agents for roads also contributes to the degradation of the
surrounding aquatic environments (including the spread of alien and invasive species; Halabowski et al. 2019b; Lewin et al. 2015).

The algae of the region were studied in a few surveys and primarily concerned phytoplankton and to a lesser extent microphytobenthos (Starmach 1939; Bucka 1960, 1964, 1966; Hanak–Schmager 1974; Kwandrans 1998, 2002; Wilk–Woźniak et al. 2011). It came to our attention that specific research on diatoms (Bacillariophyta) in the Polish part of Upper Silesia had not yet been performed, although quite a number of papers have been published on the diatoms that inhabit the waters from neighbouring areas. Most of the published results concerned the diatoms in springs and streams (e.g. Kwandrans 2007; Wojtal 2013) and rivers (fewer in water reservoirs), e.g. Wojtal & Kwandrans 2006; Cichoń 2016). According to the EU Water Framework Directive (WFD 2000), the biomonitoring that is carried out in Poland is based on macrophytes, ichthyofauna, benthic macroinvertebrates and phytobenthos, including rivers that are located in the studied area. However, accessible data of the state monitoring lack identified taxa lists and detailed data on species are not usually published.

The aims of the presented research were to compare the biodiversity of the diatom assemblages in rivers that are located in the industrial and urban area of Upper Silesia and adjacent regions that are affected by various degrees of anthropogenic pressure and to determine the most important environmental factors that have a significant impact on their structure. Another aim of the study was to assess the ecological condition of the investigated riverine courses based on microphytobenthos.

MATERIALS AND METHODS

Study area. The study was carried out in four rivers that are located in southern Poland, all of which are part of the Upper Vistula River. Two of the rivers that were studied (the Mleczna and Bolina Rivers) are affected by the coal mining activity in Upper Silesia (Southern Poland). The two remaining study sites are located outside of area of the mining activity (the Centuria and the Mitręga Rivers; Fig. 1). The anthropogenic effects were primarily observed as changes in the physical and chemical parameters of the water and the morphological transformation.
of the riverbeds at the research locations. The Mleczna River is characterised by a high mineralisation of the water due to the discharge of underground salt water from four coal mines, i.e. ‘KWK Boże Dary’, ‘KWK Mysłowice–Wesola’, ‘KWK Ziemowit’ and Experimental Mine ‘Barbara’. The water from the ‘KWK Boże Dary’ coal mine flows through a system of reservoirs before discharging into the river, while the water from the ‘KWK Mysłowice–Wesola’ coal mine flows through the Przyrywa River, which is a tributary of the Mleczna River. Two sites (at the upper and lower courses) that have different degrees of water pollution from the coal mines were selected at the Bolina River for the study. The upper course is contaminated by salt water from the ‘KWK Murcki–Staszic’ coal mine. By contrast, the lower course of the Bolina River is affected by the discharge of underground salt water from the ‘KWK Wieczorek’ coal mine. The anthropopressure of the study sites on the Mitręga River is caused by a dam reservoir that was constructed a few kilometres away. The sampling site of the Centuria River is not affected by anthropopressure. This site is located near a river spring. The general characteristics of the rivers and the sampling sites are summarised in Tables 1 and 2.

**Sampling procedure and taxonomic analyses.** The samples were collected four times per sampling site in 2017 (spring, early and late summer, autumn and winter) in order to determine the physical and chemical parameters of the water. Conductivity, total dissolved solids, pH and the temperature of the water were measured in the field using a HI 9811–5 pH/EC/TDS/°C meter (Hanna Instruments) and dissolved oxygen was measured using a CO–401 oxygen meter (Elmetron). Salinity was measured using a WTW meter and the result was expressed as the total dissolved solids that had been converted from the measurements of electrical conductivity. For the high anthropogenic impact sites (conductivity > 1000 μS.cm⁻¹), a conversion factor of 0.72 × conductivity was applied according to **Piscart et al.** (2005), while for the low anthropogenic impact sites (conductivity < 1000 μS.cm⁻¹), the conversion factor was 0.77 × conductivity. Analyses of the concentrations of chlorides, sulfates, iron, nutrients, calcium and magnesium as well as total hardness and alkalinity were performed in the laboratory according to the standard methods of **Hermanowicz et al.** (1999). The morphometry of the riverbed, i.e. the average width of the channel, the depth of the river and the flow velocity were also performed in the field.

For the taxonomic analyses, diatoms were taken from a 25 cm² surface of the bottom sediments or concrete slabs (at each of the study sites). Permanent slides for the light microscopy were prepared following the standard protocol (**Battarbee 1986; Bodén 1991**). The samples were treated with 10% hydrochloric acid (HCl) to remove any calcium carbonate and after a thorough washing, they were boiled in 37% hydrogen peroxide (H₂O₂) to eliminate any organic contamination. In the laboratory, the diatoms were cleaned and mounted in 4% **KOH.**

**Table 1. General characteristics of the sites that were investigated.**

<table>
<thead>
<tr>
<th>Characteristic</th>
<th>The Bolina River, lower course</th>
<th>The Bolina River, upper course</th>
<th>The Centuria River</th>
<th>The Mitręga River</th>
<th>The Mleczna River</th>
</tr>
</thead>
<tbody>
<tr>
<td>Geographical coordinates</td>
<td>N 50° 14' 43&quot;; E 19° 06' 02&quot;</td>
<td>N 50° 13' 48&quot;; E 19° 05' 08&quot;</td>
<td>N 50° 24' 52&quot;; E 19° 29' 12&quot;</td>
<td>N 50° 26' 02.9&quot;; E 19° 17' 58.3&quot;</td>
<td>N 50° 06' 58.4&quot;; E 19° 04' 30.2&quot;</td>
</tr>
<tr>
<td>Elevation of sampling sites (m a.s.l.)</td>
<td>257</td>
<td>262</td>
<td>343</td>
<td>300</td>
<td>236</td>
</tr>
<tr>
<td>Environment</td>
<td>Build-up area / wasteland</td>
<td>Wasteland / woodland</td>
<td>Woodland</td>
<td>Wasteland</td>
<td>Farmlands</td>
</tr>
<tr>
<td>Location / region</td>
<td>Upper Silesia / Silesian Upland</td>
<td>Upper Silesia / Silesian Upland</td>
<td>Nature monuments „Zróđla Centurii”/ Kraków–Częstochowa Upland</td>
<td>Silesian Upland</td>
<td>Upper Silesia / Silesian Upland</td>
</tr>
<tr>
<td>The part of the river studied</td>
<td>Downstream</td>
<td>Upstream</td>
<td>Upstream / Spring</td>
<td>Downstream</td>
<td>Downstream</td>
</tr>
<tr>
<td>Human pressure</td>
<td>Extreme</td>
<td>Very high</td>
<td>No impact</td>
<td>Low</td>
<td>High</td>
</tr>
<tr>
<td>Hydromorphological transformations and development</td>
<td>Concrete channelled, very heavy pressure of mine water supply</td>
<td>Channelled with concrete plate; heavy pressure of mine water supply</td>
<td>Unaltered site</td>
<td>Channeled, with fascine-reinforced banks and a dam reservoir</td>
<td>Channeled, with fascine-reinforced banks and indirect supply of mine waters</td>
</tr>
<tr>
<td>Type of river / geology</td>
<td>(5) Mid-altitude siliceous streams that have a fine–particulate substratum</td>
<td>(5) Mid-altitude siliceous streams that have a fine–particulate substratum</td>
<td>(5) Mid-altitude calcareous streams that have a fine–particulate substratum on loess</td>
<td>(6) Mid-altitude calcareous streams that have a fine–particulate substratum on loess</td>
<td>(6) Mid-altitude calcareous streams that have a fine–particulate substratum on loess</td>
</tr>
<tr>
<td>Type of bottom sediments</td>
<td>Silt and sand</td>
<td>Coal mine sludge</td>
<td>Sand and silt</td>
<td>Sand and silt</td>
<td>Silt and sand</td>
</tr>
</tbody>
</table>
matter. After washing four times with distilled water, the final suspension was pipetted onto cover slips, left to evaporate and mounted on glass slides using Naphrax® diatom mountant. The slides were examined using Zeiss Axio Scope A1 and Nikon Eclipse E600 light microscopes. The measurements and photographic documentation were performed using AxioVision Rel. 4.8 software. In all of the samples that were studied, a minimum of 400 valves were identified to the species or variety level and their relative abundance was determined. A number of ecological metrics were used to analyse the diatoms – the percentages of particular taxa:

\[ D_i = \frac{n_i}{N} \cdot 100 \% \]

where \( D_i \) – percentages of particular taxon ‘i’, \( n_i \) – the abundance of a particular taxon ‘i’, \( N \) – the sum of all of the taxa abundances in a sample (Tümpling & Friedrich 1999), the frequency coefficient:

\[ C_i = \frac{k_i}{k} \cdot 100 \% \]

where \( C_i \) – the frequency coefficient of particular taxon ‘i’ [%], \( k_i \) – the number of samples with taxon ‘i’, \( k \) – the number of all of the samples (species were grouped according to their frequency coefficient as euconstants = 75–100%, constants = 75–50%, accessory = 25–50%, incidental = 1–25%; Trojan 1975), and biodiversity indices (i.e. species richness, Shannon index:

\[ H' = \sum_{i=1}^{S} \left( \frac{n_i}{N} \log_2 \frac{n_i}{N} \right) \]

where \( S \) – the number of species (species richness), \( n \) – the abundance of a taxon, \( N \) – the sum of all of the taxa abundances and the Evenness index:

\[ E = \frac{H'}{H'_{\text{max}}} \]

where \( H' \) – the Shannon index, \( H'_{\text{max}} \) – the maximum possible value of the index \( H' \) were calculated (Shannon 1948). OMNIDIA v5 software was used to calculate the percentages of diatoms in specific ecological groups. For scanning electron microscope (SEM) observations, cleaned material was pipetted onto a 25 mm diameter polycarbonate membrane Whatman® Nuclepore filter with a 2 μm mesh, attached to aluminium stubs and sputtered with 20 nm of gold using a Quorum Q150OT ES Turbo–Pumped Sputter Coater. The diatoms were observed using a Hitachi SU 8010 SEM at the Podkarpacie Innovative Research Center of the Environment (PIRCE) at the University of Rzeszów, Poland.

The diatoms that were identified served as the basis for assessing the ecological status of the rivers with the diatom index IO according to the methodology used by the national monitoring system (Picińska–Faltnowicz et al. 2006; Picińska–Faltnowicz & Blachuta 2010; PN–EN 13946 2014; PN–EN 14407 2014; Zgrundno et al. 2018).

The identifications were based on the following

### Table 2. The physical and chemical parameters of the water and the morphology of the riverbed.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>The Bolina River, lower course</th>
<th>The Bolina River, upper course</th>
<th>The Centuria River</th>
<th>The Mitrega River</th>
<th>The Mleczna River</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth of the river bed (m)</td>
<td>0.21–0.25</td>
<td>0.25–0.34</td>
<td>0.10–0.23</td>
<td>0.40–0.59</td>
<td>0.81–1.09</td>
</tr>
<tr>
<td>Flow velocity (m.s⁻¹)</td>
<td>0.40–0.51</td>
<td>0.06–0.42</td>
<td>0.07–0.17</td>
<td>0.01–0.29</td>
<td>0.16–0.21</td>
</tr>
<tr>
<td>Temperature (°C)</td>
<td>15.5–29.1</td>
<td>15.7–28.6</td>
<td>7.5–13.6</td>
<td>10.4–18.4</td>
<td>14.8–25.1</td>
</tr>
<tr>
<td>pH</td>
<td>7.5–7.9</td>
<td>7.5–7.8</td>
<td>7.5–7.8</td>
<td>7.3–8.1</td>
<td>7.4–8.0</td>
</tr>
<tr>
<td>Salinity (PSU)</td>
<td>16.34–33.55</td>
<td>6.57–12.25</td>
<td>0.19–0.21</td>
<td>0.29–0.35</td>
<td>2.97–5.16</td>
</tr>
<tr>
<td>Dissolved oxygen (mg.dm⁻³)</td>
<td>5.13–9.69</td>
<td>4.65–7.09</td>
<td>4.89–6.61</td>
<td>4.62–6.12</td>
<td>0.69–4.82</td>
</tr>
<tr>
<td>Conductivity (μS.cm⁻¹)</td>
<td>22700–44600</td>
<td>9130–17020</td>
<td>250–270</td>
<td>380–460</td>
<td>4120–7160</td>
</tr>
<tr>
<td>Total dissolved solids (mg.dm⁻³)</td>
<td>11360–23300</td>
<td>4570–8510</td>
<td>110–140</td>
<td>180–220</td>
<td>2050–3570</td>
</tr>
<tr>
<td>Chlorides (mg.dm⁻³)</td>
<td>7528–17028</td>
<td>2823–5590</td>
<td>8–20</td>
<td>18–26</td>
<td>1340–1970</td>
</tr>
<tr>
<td>Total hardness (mgCaO₂ dm⁻³)</td>
<td>2268–4858</td>
<td>1072–1920</td>
<td>160–225</td>
<td>175–330</td>
<td>405–560</td>
</tr>
<tr>
<td>Calcium (mg.dm⁻³)</td>
<td>548–1310</td>
<td>328–687</td>
<td>55–60</td>
<td>69–94</td>
<td>101–158</td>
</tr>
<tr>
<td>Magnesium (mg.dm⁻³)</td>
<td>225–670</td>
<td>124–270</td>
<td>4–22</td>
<td>1–30</td>
<td>38–60</td>
</tr>
<tr>
<td>Alkalinity (mgCaO₂ dm⁻³)</td>
<td>230–320</td>
<td>275–380</td>
<td>75–110</td>
<td>165–250</td>
<td>240–275</td>
</tr>
<tr>
<td>Phosphates (mg.dm⁻³)</td>
<td>0.05–0.10</td>
<td>0.02–0.14</td>
<td>0.00–0.11</td>
<td>0.08–0.24</td>
<td>0.16–3.84</td>
</tr>
<tr>
<td>Ammonium (mg.dm⁻³)</td>
<td>1.25–12.12</td>
<td>0.62–1.00</td>
<td>0.00–0.23</td>
<td>0.26–0.36</td>
<td>0.23–1.21</td>
</tr>
<tr>
<td>Nitrates (mg.dm⁻³)</td>
<td>4.43–10.63</td>
<td>0.00–79.74</td>
<td>0.00–11.96</td>
<td>0.89–15.95</td>
<td>5.32–10.19</td>
</tr>
<tr>
<td>Nitrites (mg.dm⁻³)</td>
<td>2.49–9.96</td>
<td>0.68–1.44</td>
<td>0.00–0.01</td>
<td>0.11–0.24</td>
<td>0.20–0.73</td>
</tr>
<tr>
<td>Iron (mg.dm⁻³)</td>
<td>0.12–0.39</td>
<td>0.13–0.88</td>
<td>0.03–0.65</td>
<td>0.25–0.78</td>
<td>0.26–1.46</td>
</tr>
</tbody>
</table>
between the groups of sites with similar diatom populations was aided by cluster analysis. The ranked similarity matrices were constructed using the Bray–Curtis similarity measure with a fourth root transform and group–average sorting, i.e. the average similarity indices for successively created group of sites (LANCE & WILLIAMS 1967).

RESULTS

Environmental conditions of the investigated rivers

The salinity, conductivity, total dissolved solids, total hardness, concentration of chlorides, calcium, magnesium, ammonium and nitrates in the water were extremely high at the sampling site with the highest degree of anthropogenic pressure, i.e. in the lower course of the Bolina River (Tables 1 and 2). However, the concentration of sulfates, nitrates and alkalinity were higher in the upper course of the river. Because of the discharge of the coal mine waters into the Bolina River, the salinity ranged from 16.34 to 33.55 PSU (Table 2). The salinity values of the lower course of the Bolina River exceeded the maximum salinity that was recorded for the Baltic Sea (SARBÄLTBYK 2018). By contrast, the lowest values of most of the physical and chemical parameters of the water were recorded in the site that is not impacted by anthropogenic activity (the Centuria River) except for the dissolved oxygen or pH. Both rivers (Bolina and Centuria) represent the abiotic type 5 – mid–altitude siliceous streams that have a fine–particulate substratum. When the two other rivers of the same abiotic type (type 6 – mid–altitude calcareous streams that have a fine–particulate substratum on loess, PIETRUK–FAŁTYNOWICZ & BLACHUTA 2010) but with different anthropogenic pressure were compared, lower values of the physical and chemical parameters were obtained in the Mitręga River (low anthropogenic pressure) than in the Młeczna River (high anthropogenic pressure).

Diatom assemblages in the rivers that were studied

The diatom taxa (species and varieties) that were identified in the microphytobenthos of the rivers that were studied were represented by 55 genera. The centrics were identified by six genera with eight species and varieties, while the 49 pennate genera comprised 150 species and varieties, while the 49 pennate genera comprised 150 species and varieties. Only two of the identified taxa were euconstants (taxa occurring in 75–100% of the samples) and these were Planothidium frequentissimum (Lange–Bertalot) Lange–Bertalot. The constants (occurring in 50–75% of the samples) included seven taxa: Achnanthidium minutissimum (Kützing) Rabenhorst, Ctenophora pulchella (Ralfs ex Kützing) D.M.Williams, Cenococcum granicum (Grunow in Cleve et Grunow). The structures of the diatom flora and the relationships between their assemblages in the rivers that were studied were analysed using the statistical multivariate methods provided by the PRIMER software (Plymouth Routines in Multivariate Ecological Research; CLARKE & WARWICK 2001). Discrimination

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Fig. 2. A Bray–Curtis similarities dendrogram of the sampling sites with respect to the diatom flora and the abundance of taxa.
accessory species (occurrences in 25–50% of the samples) comprised 29 taxa; the largest number of taxa (120) was classified as incidental. Their frequency coefficient was less than 25% of the samples that were analysed.

**Similarities between the rivers with respect to the diatom flora**

An analysis of the similarities between the sampling sites with respect to the diatom assemblages and the abundance of taxa was aided by using the Bray–Curtis similarity measure. The resulting dendrogram shows two groups of rivers with an approximately 30% similarity (Fig. 2).

Although the similarity of diatom assemblages was not very high, the rivers that were studied seem to cluster together predominantly due to anthropogenic pressure. The first group comprised sites with a strong impact – the salinised rivers (Mleczna and Bolina). The diatom assemblages from these sites were predominantly composed of taxa that prefer brackish or brackish–marine water: *Pleurosira laevis* var. *laevis*, *Ctenophora pulchella*, *Achnanthes brevipes* var. *intermedia* (*Kützing*) *Cleve*, *Halamphora coffeiformis* (*C.Agardh*) *Levkov*, *H. luciae* (*Cholnoky*) *Levkov*, *Navicula salinarum* f. *minima* *Kolbe*, *Gyrosigma attenuatum*, *Haslea spicula* (*Hickie*) *Bukhtiyarova* and *Pleurosigma salinarum*. The second distinct group comprised the sites with a minimal or weak anthropogenic impact (Centuria and Mitrega). The taxa that distinguished this group from the previous one primarily prefer fresh–brackish water. The other ecological preferences of the taxa do not significantly differentiate the rivers that were studied and were clustered in two
groups, but some differences were still visible. In the Bolina and Mleczna Rivers, species that are characteristic of eutrophicated waters and for α–mesosaprobic zone dominated. The species that were present in the Centuria River were typical of those that prefer oligotrophic and oligosaprobic waters as well as those that are highly tolerant to trophic conditions, the saprobic status and nutrient content (the dominants included *Planothidium dubium* (Grunow) Round et Bukhtiyarova, *Cocconeis pseudothumensis* Reichardt, *C. neothumensis* Krammer, *Karayevia clevei* (Grunow) Bukhtiyarova, *Amphora inariensis* Krammer and *Achnanthidium minutissimum* var. *minutissimum*). The taxa that were present in the Mitręga River were typical cosmopolitan forms that are found in various types of water, but that quite frequently occur in waters with an elevated trophic and saprobic status (no distinct dominants).

**Biodiversity of the diatom flora in the rivers that were studied**

The biodiversity of the flora in the rivers with a minimal or weak anthropogenic impact (Centuria and Mitręga) was high, whereas in those that were strongly affected – the salinised rivers (Mleczna and Bolina) – it was quite low. The Bolina and Mleczna sites, which were characterised by the highest salinity (7–12 PSU in the upstream and 16–34 PSU downstream of the Bolina River and the 3–5 PSU of the Mleczna River), had a relatively low taxonomic richness (23 and 32 taxa in the upstream and downstream courses of the Bolina River, respectively, and 23 taxa in the Mleczna River). The initial stretch of the Centuria River and the downstream section of the Mitręga River had a much higher taxonomic richness (55 and 74 taxa, respectively).

The Shannon (H’ for the logs to the base 2) and the Evenness indices were almost two–fold higher in the rivers that were not salinised (Centuria 4.26 and 0.74, Mitręga 5.36 and 0.87) than in the salinised rivers (2.23 and 0.49 in the upstream and 2.04 and 0.41 in the downstream of the Bolina River and 2.84 and 0.63 in the Mleczna River).

The diatoms that were identified served as the basis for assessing the ecological status of the rivers with the diatom index IO according to the methodology that is used by the state monitoring system. The status of the Bolina River, in both the downstream and upstream course was considered to be poor, whereas the state monitoring system classified the Bolina (both ecological and chemical) as bad. The diatom index that was applied to the remaining rivers confirmed the ecological status classification from the state monitoring – the Mleczna, Mitręga, and Centuria Rivers were classified as representing a poor, moderate and good ecological status, respectively. The lists of the indicator species that were used by the state monitoring system until 2017 did not include the brackish species *Pleurosira laevis*, which was dominant in the Bolina and Mleczna Rivers. After adding *P. leavis* to the list of indicator species in 2018, the results of the assessment of the ecological status of the Bolina and Mleczna Rivers was improved by one level. According to the classification that was developed by Denys (1991) and Van Dam et al. (1994), this species is eutrophic and oligosaprobic. However, as was shown by results of our studies, the species is able to tolerate...
that the first and second axes explain 70.0% of species variance and 78.5% of the variance in the species and environment relationship. Conductivity, alkalinity, river width and depth were most associated (statistically significant according to the forward selection results) with the diatom taxa distribution (Fig. 3). The CCA analysis was significant (Monte Carlo test on the first canonical axis, F–ratio = 11.757, P–value = 0.002; test of significance of all of the canonical axes, F–ratio = 30.897, P–value = 0.002). The CCA analysis showed that *Achnanthes brevipes* var. *intermedia*, *Halamphora coffeiformis*, *Halamphora luciae*, *Haslea spicula*, *Navicula salinarum* var. *minima*, *Pleurosigma salinarum*, *Pleurosira laevis* var. *laevis*, and *P. laevis* var. *polymorpha* were associated with higher riverine conductivity and alkalinity (Fig. 3). By contrast, *Achnanthidium minutissimum*, *Amphora inariensis*, *Cocconeis euglypta* Ehrenberg, *Hippodonta

considerably elevated levels of nutrients and organic pollutants and is capable of forming very abundant populations in rivers with a periodically, and significantly, elevated salinity due to discharges of saline mine water.

During the species identification, one new species for Poland – *Navicula flandriae* Van de Vijver et Mertens in Beauger et al. (2015) and one species that is new to science were found – *Planothidium nanum* sp. nov., a description of which follows. In recent years, the genus *Planothidium* has been in the focus of both morphological and molecular studies. Currently, it comprises about 60 accepted species. Over the last five years (2014–2019), 32 new species were described, of which 31 are freshwater and only one is marine.

**The results of the canonical correspondence analysis (CCA)**

A canonical correspondence analysis (CCA) revealed that the first and second axes explain 70.0% of species variance and 78.5% of the variance in the species and environment relationship. Conductivity, alkalinity, river width and depth were most associated (statistically significant according to the forward selection results) with the diatom taxa distribution (Fig. 3). The CCA analysis was significant (Monte Carlo test on the first canonical axis, F–ratio = 11.757, P–value = 0.002; test of significance of all of the canonical axes, F–ratio = 30.897, P–value = 0.002). The CCA analysis showed that *Achnanthes brevipes* var. *intermedia*, *Halamphora coffeiformis*, *Halamphora luciae*, *Haslea spicula*, *Navicula salinarum* var. *minima*, *Pleurosigma salinarum*, *Pleurosira laevis* var. *laevis*, and *P. laevis* var. *polymorpha* were associated with higher riverine conductivity and alkalinity (Fig. 3). By contrast, *Achnanthidium minutissimum*, *Amphora inariensis*, *Cocconeis euglypta* Ehrenberg, *Hippodonta

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![Fig. 5. Planothidium nanum sp. nov. SEM images. (a–c) raphe valves: (a–b) external view, note the somewhat expanded external proximal central raphe endings; (c) internal view, note the internal proximal raphe endings that are bent in the opposite directions. (d–f) sternum valves: (d–e) external view, (f) internal view, note the presence of a unilateral central area. Scale bars 2 µm.](image-url)
capitata, Meridion constrictum Ralfs, Planothidium lanceolatum (Brébisson ex Kützing) Lange–Bertalot, and Navicula cryptotenella Lange–Bertalot in Krammer & Lange–Bertalot were associated with lower conductivity and alkalinity. Achnanthidium minutissimum, Cocconeis pediculus Ehrenberg, Gomphomena minutum (C.Agardh) C.Agardh, Melosira varians C.Agardh, Nitzschia amphibia Grunow, N. dissipata (Kützing) Rabenhorst, and Navicula viridula (Kützing) Ehrenberg were associated with lower water conductivity and river width (Fig. 3).

Description of the new species

Planothidium nanum sp. nov. Bąk, Kryk et Halabowski (Figs 4–5)

**Description:** Light microscopy, Figs 4a–au – Valves consistently elliptical with obtusely rounded ends that were never protracted. Length 5.5–11.5 µm, breadth 4.0–5.5 µm. Rapheless valve: Raphe filiform, straight with slightly expanded central pores and distinctly curved distal endings. Axial area very narrow, straight. Central area is asymmetrical due to the two more widely spaced central striae on the primary side than on the secondary side of the valves, where one or two shortened striae are present. Striae slightly radiate throughout the valves, 12–14 in 10 µm.

Scanning electron microscopy, external view (Figs 5a–b): Raphe straight with slightly expanded central ends, terminal fissures unilaterally bent to the mantle. Striae moderately radiate becoming strongly radiate near the ends. Transapical striae composed of 2–3 series of round areolae. Areola density counted in a single series is ca. 55–65 in 10 µm.

SEM internal view (Fig. 5c): Central raphe ends not coxial but deflected to the opposite sides. Distal ends with small helictoglossae. Central nodule together with the virgae features a high profile raised above the troughs containing the bi– or triseriate occluded areolae. Rapheless valve: A depression is visible in LM as a broad space between two central striae on primary side. Cavum structure is absent. Axial area narrow, straight or slightly broader towards the central area.

SEM external view (Figs 5d–e): Striae consisting of 1–2 series of areolae; virgae considerably broader than striae. The surface on the valve is entirely smooth. SEM internal view, see Fig. 5f.

**Type:** Poland, Upper Silesia, Centuria River, 50º24’52”N, 19º29’12”E, 11 July 2017.

**Holotype (assigned here):** Slide no. 25760 in Coll. Andrzej Witkowski, Institute of Marine and Environmental Sciences, University of Szczecin (SZCZ), the holotype is represented by Figs 3d and 3e.

**Type locality:** Centuria River located in southern Poland that belongs to the Upper Vistula River system; epipsammic sample from bottom sediment.

**Etymology:** The Latin epithet means dwarf.

**Distribution:** Known only from the type locality.


**Comparison the new species with morphologically related taxa**

The small size of Planothidium nanum and its elliptical outline combined with the very widely spaced central striae on the sternum valve (sinus) and the widely spaced central striae on the raphe valve should clear up any confusion with other species. However, the taxa that were most similar to the new species are P. frequentissimum, P. dubium, P. rostratoholarcticum Lange–Bertalot et Bąk in Bąk et Lange–Bertalot, H. ulna, and P. rostratoholarcticum by the absence of a cavum (P. nanum has a sinus). It differs from P. reichardtii, P. granum and P. dubium by the shape of its ends–subrostrate vs rounded in P. nanum. In addition, P. dubium has larger valves (length 10–20 µm, width 5.0–7.5 µm) and P. granum is generally narrower, 3.6–4.2 µm (vs length 5.5–11.5 and width 4.0–5.5 µm in P. nanum). New species differs from P. minutissimum in its valve outline – rhombic– to elliptic–lanceolate in P. minutissimum and elliptical in P. nanum and by being smaller in size than P. minutissimum, i.e. length 6.0–8.5 µm and width 3.0–3.5 µm, and also by the striae density – 16–18/10 µm (vs 12–14/10 µm in P. nanum).
collected from the Mleczna and Bolina Rivers revealed the occurrence of *N. flandriae*. The species was described in 2015 from Flanders (Flandria), the northern region in Belgium where this species was observed in several rivers and canals (i.e. Leopoldkanaal, Oostpolderkreek) in alkaline waters with a high conductivity. *Navicula flandriae* was very abundant in the Mleczna River (38% relative abundance) and a few specimens occurred in the downstream of the Bolina River (1.2%). The valves were 35–46 μm long, 7–8 μm wide with 13–14 striae per 10 μm (Fig. 6a–j). According to Van de Vijver and Mertens in Beauger et al. (2015), the valves are 35–65 μm long, 7.0–9.5 μm wide with 12–13 striae per 10 μm. The specimens that were found in Poland were characterised by a higher stria density in both the smaller and larger specimens. The LM and SEM observations (Fig. 6k–p) were consistent with those given by Beauger et al. (2015). *Navicula flandriae* was accompanied by *Pleurosira laevis* var. *laevis*, *Gomphonema parvulum* (Kützing) Kützing, *Cyclotella meneghiniana*, *Bąk et al*.: Mining salinisation: the impact on diatoms
Halymphora coffeiformis, Cocconeis placentula var. placentula Ehrenberg, Navicula perminuta Grunow in Van Heurck, Planothidium delicatum, Conticribra weiss-flogii (Grunow) Stachura–Suchoples et D.M. Williams, Navicula gregaria, Navicula veneta Kützing, Ctenophora pulchella, Haslea spicula, Nitzschia palea var. palea (Kützing) W. Smith – as the most abundant taxa in the Mleczna River.

Navicula flandriae was recently observed during routine monitoring in the province of Zuid–Holland (in: Hoofdwatergang Boutweg and Gemaal de Biersum Schuddebeursdijk) and the province of Zeeland (in: Boezem K van Steelandponder and Goese Polder; Adrienne Mertens, personal communication) in the Netherlands. In 2015 and 2016, N. flandriae was recorded in Egyptian inland waters (the Damietta Branch of the Nile River), which area N and P enriched and eutrophic (Cantonati et al. 2016). These data indicate that it may be a widely distributed species that occurs in nutrient–rich waters that are usually accompanied by a high–conductivity. Navicula flandriae has similarities to several quite commonly occurring taxa such as N. triplunctata (O.F. Müller) Bory in Bory de Saint–Vincent, N. recens (Lange–Bertalot) Lange–Bertalot in Krammer et Lange–Bertalot, N. marginalithi Lange–Bertalot in Krammer et Lange–Bertalot, N. korzeniewskii Witkowski, Lange–Bertalot et Metzeltin or N. radiosa Kützing. It is highly probable that it was wrongly identified in Polish waters and confused with the above–mentioned species. Detailed descriptions of its morphological differences are provided by Beauger et al. 2015.

**DISCUSSION**

Decrease in diatom biodiversity and taxonomic richness

The diatom flora at all the five sampling sites closely reflected the extent of the anthropogenic alteration of the environment and habitat. Our results show a decrease in both the biodiversity and taxonomic richness of the diatom assemblages that occurred in the rivers that are affected by mining activity, which was more than two–fold lower than in the rivers with a weak activity. Although it is well known that salinity can have a strong influence on the distribution of diatom species (Cholnoky 1968), the actual importance of this factor has not yet been precisely circumscribed at regional scales with a sufficient number of samples (Pota nova & Charles 2003). To date, there have been no studies that compare diatom assemblages and their biodiversity in salt–polluted rivers with the more natural rivers in the region of Upper Silesia. However, some research was recently conducted in adjacent regions. In the industrial water biotopes of three small water reservoirs that have a high salinity in southern Poland that are connected to the Bobrza River, Kielce Upland, that had conductivity that ranged from 711 to 865 μS.cm⁻¹ (seven samples), only 36 diatom taxa from 24 genera were identified, of which only 16 taxa were recognised as being species that are common in central Europe inland waters and the assemblages were characteristic for alkaliphilous, mesotraphentic and eutraphentic and β–mesosaprobous waters (Malinowska–Gniewosz et al. 2018). In our study, the salt–polluted rivers were characterised by a similarly low number of identified diatom taxa (23 to 32 taxa in the upstream and downstream sites of the Bolina River, respectively), and 23 taxa in the Mleczna River and the diatom flora was characteristic for eutrophic and α–mesosaprobic waters, whereas the Centuria and Mitrega Rivers were characterised by a higher taxonomic richness (55 and 74 taxa, respectively). It is worth mentioning that our research showed a similar decreasing trend in the biodiversity of the salt–polluted rivers compared to the rivers with weak pollution, i.e. the Shannon index decreased from 4.26 and 5.36 (Centuria River and Mitrega River) to only 2.23 and 2.84 (Bolina River and Mleczna River). Based on this, elevated salinity in rivers can cause a decrease in the aquatic biodiversity of diatom assemblages. Like Poland, the Czech part of Upper Silesia has a long mining history. Čecháková et al. (2014) analysed selected groups of plants and animals such as amphibians, aquatic molluscs, other macroinvertebrates, diatoms and aquatic plants in polluted (including post–industrial salinisation) subsidence reservoirs. The conductivity of the reservoirs that were studied at specific sites was significantly higher than in our studies (exceeding 1500 μS.cm⁻¹) and the conclusion was that increased levels of salts in the affected areas result in a decrease in aquatic biodiversity and have a negative effect on different groups of plants and animals (Čecháková et al. 2014). More studies that compared changes in diatom assemblages in terms of land use were carried out in South Africa (Pan et al. 2004; Taylor et al. 2005, 2007a, 2007b). The most comprehensive results, which were published by Walsh & Wepener (2009), showed that 72% of the diatom taxa that were identified in streams with a significant agricultural impact were salt tolerant. The study was conducted in the Crocodile and Magalies Rivers in South Africa, where in some of the samples (conductivity over 600 μS.cm⁻¹), the diatom index scores had a low to moderate integrity. Among the impacted sites, agricultural sites had a somewhat worse ecological status than urban sites according to the diatom indices. The reference sites were dominated by diatom species that preferred clean, freshwater and the indices that were calculated indicated a good ecological status, while the sites that had been impacted by an agricultural or urban influence had a poor to moderate status (Walsh & Wepener 2009). Lavoie et al. (2018) analysed the diatom assemblages from several streams and creeks of the Greater Sudbury River (Ontario, Canada). The results of these studies are another piece of evidence for the decrease in diatom biodiversity in water bodies that have been affected by an anthropogenic impact. The calculated Eastern Canadian Diatom Index (IDEC) had a
better biological integrity at the reference sites (Lavoie et al. 2018) that had had weak anthropogenic influences that was similar to our results. Husteed (1957) recorded that it is not the concentration of a particular salt in the water that influences freshwater diatoms the most but rather the osmotic pressure. Later experiments confirmed the fact that osmotic pressure is an important factor that limits the occurrence of sensitive freshwater diatom species (Cleave et al. 1981) by limiting their ability to absorb nutrients (Tuchman et al. 1984).

**Brackish and marine species in the analysed rivers**

Based on the cluster and the canonical correspondence analysis (CCA) analyses, we were able to indicate the differences between the diatom assemblages in the rivers with a strong and weak anthropogenic impact. The first populations were rich in taxa that preferred brackish or brackish–marine water and the second populations were dominated by taxa that preferred more fresh–brackish water. In the rivers with a strong anthropogenic salinisation, *Pleurosigma laevis* was the dominant species. It is typically a halophilic, sometimes rheophilic species, known for its ability to live in freshwater environments (Compère 1982). In the beginning, *P. laevis* was recorded in freshwater, oligohaline environments of North America, e.g. the Colorado River in the Grand Canyon region (Crayton & Sommerfeld 1979), the Great Lakes (Wuëk & Welling 1981) or the Maumee River in Ohio (Kociolek et al. 1983). Later, the species was reported in Europe (Sims et al. 1996; Gómez 2008; Perez et al. 2009), Asia (Joh et al. 2010; Kärthick & Kociolek 2011; Sharifinia et al. 2016) and South America (Metzelten & García-Rodríguez 2003). Recently, *P. laevis* was found in the Zarafshan River, which is one of the largest rivers in Uzbekistan (Mamanazarov & Gololobova 2017). *Pleurosigma laevis* is mainly associated with brackish or marine environments where its populations grow along with an increase in the chloride levels of the water (Wuëk & Welling 1981; Kociolek et al. 1983). In Sharifinia et al. (2016), this species occurred in sites that were characterised by conductivity that ranged from slightly over 600 to 1000 μS cm⁻¹. In our study, *P. laevis* comprised 73% of all of the identified diatom taxa in the Bolina River and 21% in the Mleczna River, although the values of their salinity (periodically exceeded 40 000 and 7000 μS cm⁻¹, respectively) and noticeable concentration of sulfates (550 and almost 300 mg dm⁻³, respectively). During a microscopic observation, we did not observe any significant teratologies of the *P. laevis* frustule, which usually suggests a response to ecological stress or metal pollution (Lavoie et al. 2018). Schröder et al. (2015) analysed the benthic diatom assemblages of a Western Germany lowland river – the Lippe River. Its lower and middle courses had increased levels of salinity that were caused by coal mining activities (Petruck & Stöffler 2011). The authors found that 31% of all of the diatom species were reliable salinity indicators and that five species were characterised by a positive relationship to increasing salinities: *Amphora libyca* Ehrenberg, *Bacillaria paxillifera* (O.F. Müller) T. Marsson, *Navicula subhamulata* Grunow in Van Heurck, *Nitzschia inconspicua* Grunow, *Rhioscpenisia abbreviata* (C. Agardh) Lange–Bertalot, of which *Bacillaria paxillifera* and *Rhioscpenisia abbreviata* also occurred in our studies. In the upper course of the Bolina River, *Ctenophora pulchella* constituted 63% of all of the identified taxa. This species primarily occurs in brackish waters in coastal regions but can also be found in freshwater bodies that have an increased conductivity, for example, in a polluted urban stream – Collins Channel, California, where the conductivity was almost 3000 μS cm⁻¹ (Jones 2013). Except for *P. laevis* and *C. pulchella* in the aforementioned rivers, we identified more diatoms that prefer brackish and marine environment such as *Achnanthes brevipes var. intermedia* (3% of all of the taxa in the Bolina River), *Halamphora coffeiformis* (2% of the taxa in the Mleczna River), *Halamphora luciae* (2% of the taxa in the Bolina River), *Navicula salinarum* (1% of the taxa in the Bolina River), *Gyrosigma attenuatum* (4% of the taxa in the Bolina River) and *Pleurosigma salinarum* (1% of the taxa in the Bolina River). In addition, *Tabularia fasciculata* (C. Agardh) D.M. Williams et Round comprised 6% of all of the identified species in the entire Bolina River diatom population. It is a cosmopolitan species that has a worldwide distribution and a tolerance for wide ranges of salinity (Snoeijis 1992). It has been found in marine, brackish as well as freshwater habitats (Davidovich et al. 2010). The relationship between benthic diatom assemblages and conductivity has been well studied in the USA where Potapova & Charles (2003) have analysed 3239 diatom samples from 1109 river sites located in different parts of North America, which have conductivity that ranges from 10 to almost 15 000 μS·m⁻¹. The authors considered a value of 1000 μS·cm⁻¹ as the threshold between brackish and freshwater habitats. However, the results may have been underestimated in terms of the lack of a brackish water dataset, which caused species that prefer brackish waters to have their optima set below the aforementioned threshold (e.g. *Ctenophora pulchella*). Compared to our study, *Pleurosigma laevis*, which dominated the downstream sites of the Bolina and Mleczna Rivers occurred in a conductivity of 44 600 μS·cm⁻¹ and 7090 μS·cm⁻¹, respectively, during the inflow of polluted post–mining waters, while Potapova & Charles (2003) reported an optimum of 573 μS·cm⁻¹ with a maximum conductivity value of 1122 μS·cm⁻¹ for the same species. Our results show that *P. laevis* is able to withstand periodic higher conductivity levels and grow (it was the dominant species in the Bolina River). A similar conclusion was drawn for other species, e.g. *Ctenophora pulchella* or *Tabularia fasciculata*. Determining the salinity preferences of individual species seems to be a very difficult task taking into account the multitude of water factors that affect individual diatom populations besides salinity. The occurrence of halophilous species is not always associated with an...
increased salinity. At the beginning of the 20th century, *Actinocyclus normanii* (W. Gregory ex Greville) Hustvedt, which was considered to be a marine species, appeared in rivers and at the end of that century, it occurred in many European inland water bodies (Kiss et al. 1990; Geissler et al. 2006; Kástovský et al. 2010). Later, a similar spread of *Didymosphenia geminata* (Lyngbye) Mart. Schmidt in A. Schmidt as well as *Pleurosigma laevis* and *Bacillaria paxillifera* (Fránková–Kozáková et al. 2007; Hindaková et al. 2010) was observed in the Czech Republic (Gágyorová & Marván 2002). In terms of the periodic inflow of salinised mine waters into the Bolina and Mleczna Rivers, it is difficult to establish whether we observed a case of a marine species invasion or only examples of the short appearance of these organisms into unfavourable environmental conditions. There is a great need to carry out detailed studies on the diatom assemblages of saline rivers, especially on the ecological preferences of the dominant species in them.

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