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Assessment of the ecological status of the river water system in European North under conditions of ultra-fresh humus waters using biotic indices (Suna River, Lake Onega Basin, Russia)

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ABSTRACT

Protecting rivers and lakes from pollution is crucial for maintaining the health of aquatic ecosystems and ensuring the well-being of both wildlife and humans. Present study intends to examine the water quality of the Suna River (Eastern Fennoscandia in the European North of Russia) to assess the ecological risk. Widespread methods for assessing water quality based on macrozoobenthos and phytoperiphoton were applied. It was found that in conditions of ultra-fresh waters with high humus content, biotic indicators may indicate pollution of water bodies that do not actually experience significant anthropogenic impact. Ratings ranging from 'poor' to 'excellent' were obtained for different stations, reflecting the influence of natural features of different sections of the river. 'Good' water quality was noted at the river stones biotopes. In river gravel, sand and silt biotopes, as well as all lake biotopes, 'mediocre' water quality was found. The low anthropogenic income on the river catchment allows us to conclude that most indices for assessing water quality significantly underestimate the results in the natural conditions of Eastern Fennoscandia. Biotic indices EPT, BBI and EBI are most accurate for assessing the water quality of rivers and lakes in Eastern Fennoscandia.

Key words: bioindication, macrozoobenthos, monitoring, phytoperiphyton, pollution

HIGHLIGHTS

- An assessment of the ecological state of the freshwater ecosystem in the North of Russia has been carried out.
- Macrozoobenthos and periphyton can be indicators of the state of the environment.
- The anthropogenic factor is important in the formation of a high-quality aquatic environment.
- Water quality is determined by the degree of the impact on the ecosystem.
- Very low water mineralization and high humus content make it difficult to determine the ecological status of rivers and lakes.

INTRODUCTION

The destruction of river ecosystems is one of the major problems in recent times due to increasing human interference around the world (Wen *et al.* 2017; Rana & Joshi 2023; Mogane *et al.* 2023). Human activity is a significant factor affecting rivers, which often negatively affects the physicochemical parameters of river water quality (Delkash *et al.* 2018). Land clearing, agricultural activities, and livestock waste that can produce sediment, organic matter, nutrients, pathogens, and heavy metals also affect river water quality (Ullah *et al.* 2018; van Vliet *et al.* 2023; Yao *et al.* 2023). Regular monitoring of water quality through biological, chemical and physical assessments can help identify and solve problems early. Incorporating biotic indicators into water quality assessments can enhance the understanding of ecosystem health and improve management decisions (Abbasi & Abbasi 2012). Biotic indicators can provide early warning signs of water quality degradation before physical and chemical indicators may show noticeable changes, and offer a more holistic view of ecosystem health as they reflect the overall biological integrity of the environment. By monitoring the response of biota to these stressors, a more nuanced understanding of the overall health of the ecosystem can be achieved. Many methods have been developed for biological assessment of the state of water bodies – metrics, indices and indicators, which are currently widely used

(AFNOR 1992; Acosta *et al.* 2009; Abbasi & Abbasi 2012; Tamiru 2019; Vshivkova *et al.* 2019; Baryshev 2020). With the aim of bioassessment, most of the biological groups are used, as each group presents specific responses to various pressures, related to their habitat requirements and life-cycle (Johnson & Hering 2009; Johnson & Ringler 2014; Mogane *et al.* 2023). The use of several biological groups at once allows you to obtain more accurate results and a reliable assessment of the level of water pollution (Karaouzas *et al.* 2019).

It should be taken into account that in different landscape and climatic conditions they have different effectiveness, for this reason, many indices and their varieties have been developed, and work is constantly underway to adapt methods for specific regions (Birk *et al.* 2012; Lazaridou *et al.* 2018; Golovatjuk & Zinchenko 2020; Mogane *et al.* 2023; Uddin *et al.* 2023). To assess the level of pollution using biotic indices, it is important to know how regional natural features affect their values and to determine which methods are more suitable for the study area. In European countries, 297 methods were applied to rivers, lakes, coastal and transitional waters; the majority were based on benthic invertebrates (26%) and to a lesser extent to phytobenthos (10%) (Birk *et al.* 2012). Few biotic analysis methods have been developed for northern regions (Golovatjuk & Zinchenko 2020; Mogane *et al.* 2023). The specifics of using known methods in rivers and lakes of the north in conditions with very low water mineralization and high humus content are not well known.

The Suna River is a large river in Eastern Fennoscandia, that belongs to the Lake Onega basin. The climate in the area of the Suna River basin is moderately continental with marine features. Winter is long and relatively mild; summer is short and cool. The Suna River basin consists of the watershed of Lake Sundozero and the downstream section of the main river. The Sundozero watershed has no significant anthropogenic impact. In the middle course of the Suna River, there is a protected natural area, the Kivach Reserve. Since 1997, a trout farm has been operating on Lake Sundozero, which may have eutrophication effects (Popova & Suhov 2011). Agricultural production is located in the downstream area and contamination with nutrients is quite possible.

The present study is the first that focuses on features of using common indices (both on benthos and phytoperiphyton) for assessing the ecological quality of waters based on conditions of ultra-fresh humus waters in northern Europe. In addition, it was important to assess the ecological quality status of the Suna River, which flows through the territory of the Nature Reserve.

MATERIALS AND METHODS

The Suna River is a large river in Eastern Fennoscandia, that belongs to the Lake Onega basin. The climate is humid continental, characterized by cold winters and mild summers. After the artificial reversal of the river in 1954, its length is 63 km and the drainage basin is 1,830 km^2. The river water is characterized by a high background content of humus and phosphorus (Lozovik *et al.* 2016; Belkina 2019). The Suna River basin consists of the watershed of Lake Sundozero and the downstream section of the main river. The Sundozer watershed has no significant anthropogenic impact. In the middle course of the Suna River, there is a protected natural area, the Kivach Reserve. There are agricultural farms on the banks of the lower course of the river, which can cause eutrophication effects.

The mineralization of water is low – 20–50 mg/L, color is dark due to humus (50–126°), pH is 6.80–7.25 (Lozovik 2013; Lozovik *et al.* 2016). The content of total phosphorus in the water of lakes Pyalozero, Sundozero and Pandozero is close to the regional background for surface waters of Eastern Fennoscandia and does not exceed 50 μg/L (Lozovik *et al.* 2016). In the Suna River and its tributaries within the Kivach Nature Reserve, the concentration of phosphorus in the water also does not exceed these values (Komulajnen 2018). In the lower course of the river, where agricultural pollutants enter, the concentration of total phosphorus is higher and amounts to 96 μg/L (Lozovik *et al.* 2016).

Sampling was performed from 2007 to 2022 at 26 stations in rivers (main river and tributaries) and lakes (Figure 1). All of them, except one station, are located on the territory of the Kivach nature reserve and upstream and are not subject to intense anthropogenic impact.

Most of the material was collected in July and August, since during this season, the water level and flow speed in the rivers of Eastern Fennoscandia are most stable, so the data is most indicative. Along with this, repeating the summer collections, in the autumn period material was collected at two river stations – 10 and 11. The results of autumn collections from Lake Sundozero (st. 20–23) were also analyzed due to the lack of summer material from this lake. The study lasted for quite a long period – from 2007 to 2020 to level out the interannual dynamics.

Sampling was performed in the following water bodies: main rivers, tributary rivers and lakes. A survey of different biotopes, both in rivers and lakes, both GSM (gravel, sand and mud) and stones (bedrock, boulders, cobbles, and pebbles)

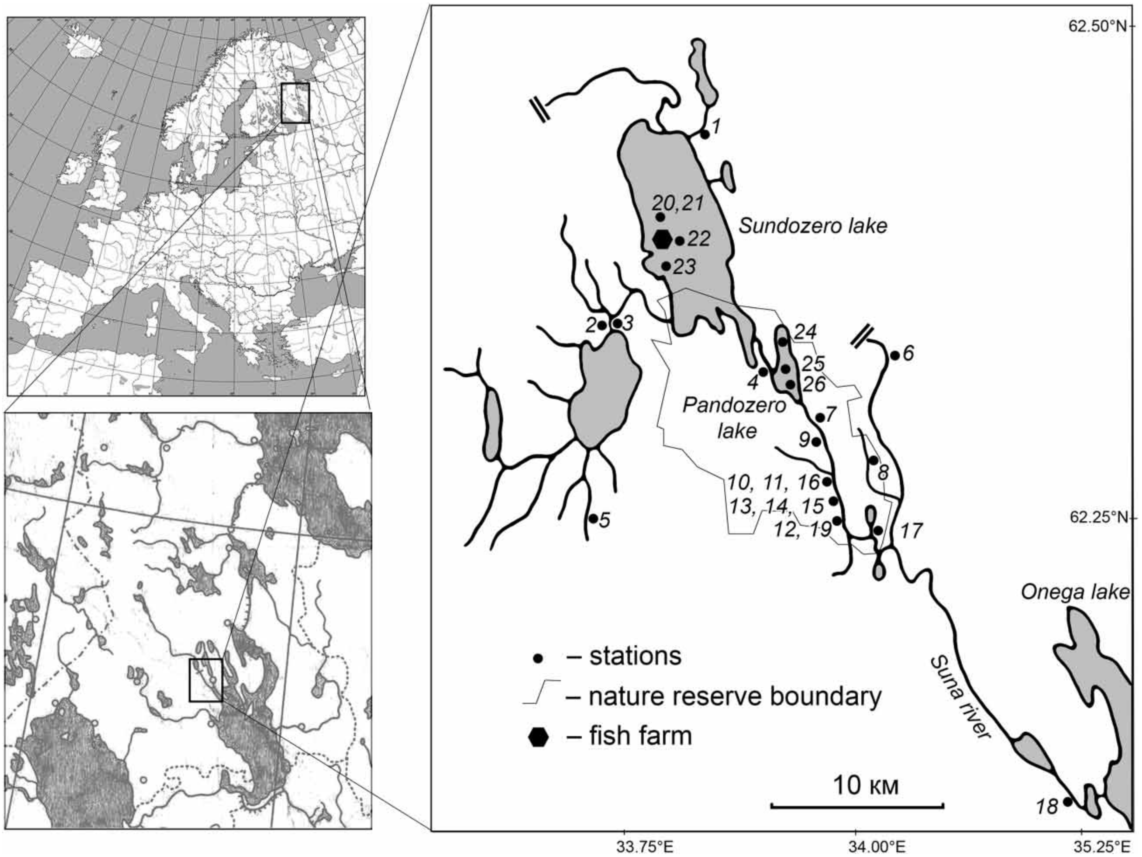

Figure 1 | Map of stations (1–26) where material was collected in the Suna River (2007–2020).

substrates, made it possible to assess the ecological state of the entire river ecosystem. The description of the stations is shown in Table 1.

Water quality biotic assessment was carried out based on phytoperiphyton and macrozoobenthos biotic indices. During phytoperiphyton (periphyton algae) sampling, 3–5 submerged cobbles were collected and brushed to obtain biofilm material. Benthic macroinvertebrates were collected with the quantitative methods. To take samples on STONES biotopes, a Surber-type frame with an area of 0.04 m^2 was used, and three samples were collected at each station. For GSM biotopes at depths more than 0.5 m sampling an Ekman sampler with an area of 0.025 m^2 was used, two catches per sample (0.05 m^2 in total), two samples from each station. For GSM biotopes at shallower depth (less than 0.5 m), sampling was carried out using a bottom scraper, collecting substrate from an area of 0.04 m^2, three were collected at each station. A total of 65 macrozoobenthos samples and 11 phytoperiphyton samples were collected. Phytoperiphyton samples were fixed with formaldehyde (4%), macrozoobenthos samples with ethyl alcohol (70%).

To assess species diversity at stations, the Shannon index (H) was calculated using the formula: $H = -\Sigma p_i \ln p_i$, where p_i is the proportion of individuals of the ith species; and Simpson's dominance index (D) according to the formula $D = \Sigma\, p_i^2$.

Indices that are widely used in world practice were chosen. For macrozoobenthos of watercourses – Family Biotic Index (FBI), Extended Biotic Index (EBI), Iberian Biological Monitoring Working Party Index (Iberian BMWP), Average Score Per Taxon (ASPT), Belgian Biotic Index (BBI), Indice Biologique Global Normalize (IBGN)), Pantle and Book saprobity index (SI_b), Gutnight-Whitley index (GW), EPT Richness index (EPT) and Mayer index (IM) (Pantle & Buck 1955; Sladecek 1973; Armitage *et al.* 1983; Ghetti 1986; Hilsenhoff 1987; Lenat 1988; Acosta *et al.* 2009); for macrozoobenthos of lakes – IM, GW, SI_b and Balushkina Index (IB) (Balushkina 1987); for phytoperiphyton – saprobity index according to Pantle and Buck (SI_a) and trophic diatom index (TDI) (Kelly & Whitton 1995).

To find a unified assessment of water quality, the average water quality index (AvWQI) was calculated that combined the obtained data. AvWQI was calculated based on the arithmetic average of all indicators, adapted to a single quality scale in

Table 1 | Characterization of the biotopes studied in the Suna River basin (basin of Lake Onega, Russia) in 2007–2020

No.	Date	Water body	Biotope	M	F
1	17.07.2017	Tributary river (Petrolampi)	GSM	3	–
2	31.07.2018	Tributary river (Niva)	STONES	3	+
3	31.07.2018	Tributary river (Niva)	GSM	3	–
4	22.08.2013	Main river (Suna)	STONES	3	+
5	30.07.2018	Tributary river (Pyalya)	GSM	3	+
6	01.08.2009	Tributary river (Sandal)	STONES	3	+
7	04.08.2014	Main river (Suna)	STONES	3	+
8	01.08.2009	Tributary river (Chechkin)	STONES	3	+
9	01.08.2009	Main river (Suna)	STONES	3	+
10	09.10.2007	Main river (Suna)	STONES	3	–
11	09.10.2007	Main river (Suna)	STONES	3	–
12	28.07.2017	Main river (Suna)	GSM	2	–
13	28.07.2017	Main river (Suna)	GSM	2	–
14	25.07.2016	Main river (Suna)	GSM	2	–
15	26.07.2016	Main river (Suna)	GSM	2	–
16	25.07.2016	Main river (Suna)	GSM	2	+
17	01.08.2009	Tributary river (Gimoyoya)	GSM	3	+
18	30.07.2018	Main river (Suna)	GSM	2	+
19	02.08.2009	Main river (Suna)	STONES	3	+
20	24.11.2020	Lake (Sundozero)	GSM	2	–
21	24.11.2020	Lake (Sundozero)	GSM	2	–
22	24.11.2020	Lake (Sundozero)	GSM	2	–
23	24.11.2020	Lake (Sundozero)	STONES	2	–
24	15.07.2017	Lake (Pandozero)	STONES	2	–
25	15.07.2017	Lake (Pandozero)	STONES	2	–
26	15.07.2017	Lake (Pandozero)	STONES	2	–

Note: No. indicates number of stations in accordance with Figure 1; GSM, gravel, sand, and mud; STONES, bedrock, boulders, cobbles, and pebbles; M, macrozoobenthos, number of quantitative samples; F, phytoperiphyton ('+' indicates qualitative data, '–' indicates no data).

points and had five gradations: 1 – very bad, 2 – bad, 3 – mediocre, 4 – good, 5 – excellent. The intervals of index values for conversion to a five-point scale are given in Table 2.

RESULTS AND DISCUSSION

In the phytoperiphyton of the lower course of the Suna River, rock-dwelling algae species common to the region predominate in number – *Tabellaria flocculosa* (Roth) Kützing, 1844, *Eunotia pectinalis* (Kützing) Rabenhorst, 1864, *Cocconeis placentula* Ehrenberg, 1838 and *Achnanthes minutissima* Kützing, 1833. At the same time, the largest part of the fouling biomass is formed by filamentous green algae *Cladophora glomerata* (L.) Kützing, 1843, *Mougeotia* sp., *Zygnema* sp. and *Bulbochaete* sp. The abundance and biomass of algae varied across stations within the range of 0.1–1,301 $\times$ 10^4 cells/cm^2 and 0.01–34.70 mg/cm^2, respectively. More detailed data on the species composition and abundance of benthic communities of the lower course of the Suna River have been published previously (Komulajnen 2018; Kuchko *et al.* 2018; Baryshev & Kuchko 2019; Baryshev 2020).

Macrozoobenthos of the STONES biotopes was represented mainly by the larvae of amphibiotic insects: Trichoptera – *Hydropsyche pellucidula* (Curtis, 1834), *Rhyacophila nubila* Zetterstedt, 1840; Ephemeroptera – *Baetis fuscatus* (Linnaeus, 1761), *B. rhodani* (Pictet, 1843); Plecoptera – *Leuctra fusca* (Linnaeus, 1758), *Diura bicaudata* (Linnaeus, 1758); Diptera of the families Chironomidae and Simuliidae. In the lower course of the Suna River (st. 19), bivalve mollusks (Bivalvia) of the

Table 2 | Intervals of index values for conversion to AvWQI

Index	AvWQI score 1	2	3	4	5	Reference
FBI	>7.2	6.1–7.2	4.6–6.1	3.9–4.6	<3.9	Semenchenko & Razluckij (2010)*
EBI	1–3	4–5	6–7	8–9	10–12	Semenchenko & Razluckij (2010), Ghetti (1986)
BMWP	<15	15–35	36–60	61–100	>100	Jumaat & Hamid (2021)
ASPT	<1.9	1.9–3.3	3.3–4.2	4.2–4.9	>4.9	Semenchenko & Razluckij (2010)*
BBI	1–2	3–4	5–6	7–8	9–10	Semenchenko & Razluckij (2010)
IGBN	4–0	8–5	12–9	16–13	17–20	Semenchenko & Razluckij (2010), AFNOR (1992)
SI_b	>3.5	2.5–3.5	1.5–2.5	0.5–1.5	<0.5	Sladecek (1973)*
SI_a	>3.5	2.5–3.5	1.5–2.5	0.5–1.5	<0.5	Sladecek (1973)*
GW	>80	70–80	60–70	30–60	<30	Balushkina (1987)
EPT	0–6	7–13	14–20	21–27	>27	NC Department of Environment, Health, and Natural Resources (1997)
IM	<4	4–10	11–16	16–22	>22	Balushkina (1987)
IB	>9.5	7.0–9.5	4.5–7.0	1.0–4.5	<1	Balushkina (1987)*
TDI	>3.5	2.5–3.5	1.5–2.5	0.5–1.5	<0.51	Kelly & Whitton (1995)

Note: '*' indicates proportional conversion to a 5-point scale has been made.

family Sphaeriidae, among which *Sphaerium corneum* (Linnaeus, 1758) predominates, reached large abundance (300–19,500 ind./m^2 with an average value of 4,300 ind./m^2) and biomass (0.6–150.0 g/m^2 with an average of 24.3 g/m^2).

The family Chironomidae was the most representative in GSM biotopes – *Microtendipes pedellus* (De Geer, 1776), *Polypedilum nubeculosum* (Meigen, 1804), *P. scalaenum* (Schrank, 1803), *Tanytarsus* sp., *Procladius* sp., followed by bivalve mollusks of family Sphaeridae (*Euglesa* sp.). The average number was 2,762 ind./m^2, varied from 60 to 39,000 ind./m^2. Biomass (excluding large bivalves) varied from 0.1 to 20.6 g/m^2 and averaged 5.1 g/m^2. In some stations in the large course of the Suna River (st. 12 and 16), large bivalve mollusks of the Unionidae family – *Anodonta cygnea* (Linnaeus, 1758), *Unio pictorum* (Linnaeus, 1758) and *U. tumidus* Philipsson 1788 reached a large abundance (up to 120 ind./m^2 and 1,370 g/m^2).

The macrozoobenthos of lakes were dominated by chironomid larvae (*Chironomus* sp., *Tanytarsus* sp., *Procladius* sp., *Stictochironomus* sp.) and oligochaetes (Oligochaetae) of the families Lumbriculidae (*Lamprodrilus isoporus* Michaelsen, 1901) and Naididae (*Nais simplex* Piguet, 1906). The number and biomass varied from 40 to 660 ind./m^2 and from 0.16 to 2.36 g/m^2, respectively.

The number of species in the macrozoobenthos varied across stations from 5 to 34, the Shannon index ranged from 0.40 to 2.71, and the Simpson index from 0.09 to 0.83. The highest rates of species diversity were found in communities of river STONES (Table 3). There were fewer species in lake biotopes, but species diversity indices here were comparable to river ones.

The integral water quality index AvWQI varied significantly from 2 (poor) to 5 (excellent), as well as the original indexes (Table 4).

Table 3 | Estimation of macrozoobenthos species diversity for bitopes of the Suna River (mean and standard error are presented)

Characteristic	River GSM	River STONES	Lake GSM	Lake STONES
Number of species, *n*	17.1 ± 2.15	27.7 ± 1.50	11.0 ± 0.82	9.3 ± 1.63
Shannon index	1.2 ± 0.14	2.2 ± 0.13	2.2 ± 0.10	1.9 ± 0.27
Simpson index	0.5 ± 0.06	0.2 ± 0.02	0.1 ± 0.02	0.2 ± 0.08

Table 4 | Water quality indices and the integral index (AvWQI) in the Suna River basin

No.	FBI	EBI	BMWP	ASPT	BBI	IGBN	SI_b	GW	EPT	IM	IB	SI_a	TDI	AvWQ
1	6.2	4	15	2.5	3	4	2.7	5	0	11	–	–	–	2
2	5.6	8	54	6.0	6	8	1.5	0	33	11	–	0.8	2.7	4
3	6.7	7	21	4.2	4	8	1.2	17	7	9	–	–	–	3
4	5.1	12	124	5.9	9	14	1.9	0	53	21	–	1.5	2.6	4
5	6.0	7	33	4.1	6	5	1.2	0	5	14	–	1.6	3.4	3
6	4.7	12	76	5.4	8	11	2.0	1	57	20	–	0.6	3.0	4
7	5.3	11	98	5.4	8	12	2.0	2	53	20	–	1.3	2.4	4
8	4.0	9	51	6.4	7	11	2.1	2	60	12	–	0.4	2.3	4
9	4.6	11	98	6.1	8	12	1.7	0	59	18	–	1.1	1.9	4
10	4.1	12	150	6.8	9	15	2.0	0	67	17	–	–	–	5
11	4.9	12	128	6.7	9	14	1.9	16	65	18	–	–	–	4
12	5.8	5	10	2.5	4	3	2.2	4	6	12	–	–	–	2
13	6.0	8	32	4.6	5	8	1.9	3	11	11	–	–	–	3
14	5.9	3	10	5.0	4	3	2.4	0	0	6	–	–	–	2
15	5.9	7	14	3.5	4	5	1.4	5	6	11	–	–	–	3
16	6.0	7	13	4.3	4	7	2.3	2	7	8	–	0.8	1.7	3
17	6.1	7	28	4.7	6	8	1.4	4	20	12	–	0.6	1.2	3
18	6.1	9	75	5.4	5	13	2.0	0	25	18	–	1.3	2.5	4
19	5.8	11	111	6.5	8	13	1.9	0	64	18	–	1.3	2.4	4
20	–	–	–	–	–	–	3.3	7	–	5	2.8	–	–	3
21	–	–	–	–	–	–	2.5	–	–	1	1.9	–	–	3
22	–	–	–	–	–	–	–	33	–	2	–	–	–	3
23	–	–	–	–	–	–	2.5	17	–	5	1.1	–	–	4
24	–	–	–	–	–	–	3.1	7	–	5	2.8	–	–	3
25	–	–	–	–	–	–	2.8	7	–	11	1.0	–	–	4
26	–	–	–	–	–	–	2.6	26	–	8	2.1	–	–	3

Note: No. – station number in accordance with Figure 1 and Table 1; '–' indicates no data.

The biotic indices were used to yield divergent outcomes. Thus, excellent water quality at all stations is indicated by the Gutnight-Whitley (GW) index (5 points). At the same time, the IGBN index in 12 out of 16 cases differs from the AvWQI to a lesser extent by one or two points.

Comparison of indices with AvWQI made it possible to evaluate their effectiveness and find the most and least informative of them (Table 5).

The most informative indices are those that have a high correlation with AvWQI and at the same time a small average deviation from AvWQI – EPT, BBI and EBI. Several indices are highly correlated with AvWQI, but have a significant mean difference from AvWQI, so they should be used with a correction factor, these are BMWP, ASPT, IGBN and IB. Other indices have a low correlation with AvWQI and are prone to errors. Thus, EPT, BBI and EBI indices are most accurate for assessing the water quality of river systems in Eastern Fennoscandia with low salinity and high humus concentration.

Based on the integral index, the overall score varies from mediocre to good water quality. Station 1 is located on a tributary flowing from a small lake, with no settlements in the watershed area and experiencing no significant anthropogenic impact. Stations 12 and 14 are in the main river within the nature reserve, where anthropogenic influence is minimal. Consequently, the low AvWQI at these stations is not caused by river pollution. The main reason can be considered the natural features of the stations. It is noteworthy that the stations located near the cages of the trout farm (stations 20–23) are characterized by the average AvWQI (3 and 4 points). Therefore this study did not reveal any negative impact of aquaculture on the lake ecosystem.

Table 5 | Comparison of indices with AvWQI and evaluation of their effectiveness

Index	Average deviation from AvWQI	Correlation with AvWQI
FBI	–0.53	–0.68
EBI	0.30	0.91
BMWP	–0.47	0.86
ASPT	0.8	0.86
BBI	–0.16	0.84
IGBN	–0.84	0.92
SIb	–0.40	–0.20
GW	1.60	–0.17
EPT	–0.11	0.87
IM	–0.46	0.55
IB	0.80	–0.89
SIa	0.27	0.04
TDI	–0.64	0.29

The lowest AvWQI (2, bad) was obtained at three stations – 1, 12 and 14. All of them were located in rivers and characterized by a large amount of large-fragmented detritus on the bottom (GSM biotopes). Many of the stations (11) were rated 3 (mediocre), and most of them were GSM biotopes. High values of AvWQI (4, 5) were observed in STONES biotopes (stations 4, 6–11, 19, 23 and 25). This proves the significant influence of the sampling location on the assessment of water quality: in STONES biotopes the values of the integral index are higher (Table 6).

Regular water quality assessments help identify potential contaminants, track changes over time and inform decision-making for effective water resource management and conservation efforts. Many methods have been developed for assessing water quality based on biological indicators and their local adaptations (Bezmaternyh 2007; Semenchenko & Razluckij 2010; Introduction to Biomonitoring 2019).

This work confirms that the result of water quality assessment depends on indices and their modifications that are chosen, and specifically on how suitable they are for the area being assessed. Using widespread and accessible methods, in the course of our work we identified 'mediocre' and 'good' ecological water quality on average for water bodies in the lower course of the Suna River basin. This result is close to that obtained earlier for the Shuya River, which is located in similar natural conditions and has waters with a similar chemical composition (predominantly 'mediocre' and 'good' water quality were noted) (Baryshev 2021). On the one hand, the index values we obtained should be interpreted as a consequence of pronounced anthropogenic pollution. On the other hand, a small percentage of the populated area and a small number of industrial enterprises in the river catchment do not bring a heavy anthropogenic impact on aquatic ecosystems (State report 2020).

Species richness has a significant impact on the value of the integral index for assessing water quality since most of the indices are based on the count of species of certain taxa (AFNOR 1992; Acosta *et al.* 2009; Abbasi & Abbasi 2012). It has

Table 6 | AvWQI values for biotopes of the Suna River basin

Type of water body	Biotope	AvWQI	Average score	Water quality
Tributary	GSM	2, 4, 3, 3	3	Mediocre
	STONES	4, 4, 3	4	Good
Main river	GSM	2, 3, 2, 3, 3, 4	3	Mediocre
	STONES	4, 4, 4, 5, 4, 4	4	Good
Lake	GSM	3, 3, 3	3	Mediocre
	STONES	4, 3, 4, 3	3–4	Mediocre and good

been shown that in the case of rivers, where natural variability (i.e. seasonality, hydrology) is pronounced, the selection of the appropriate indices in monitoring programs in order to detect impacts of anthropogenic pressures is essential (Karaouzas *et al.* 2019). Several factors contribute to the low species diversity in the rivers of Eastern Fennoscandia. The cold climate and short summers of this territory are one of the reasons for the low diversity of aquatic communities. The last glacier retreated recently – 10,000–12,000 years ago, so there was not enough time for the formation of a rich freshwater fauna (Baryshev 2020, 2021). The extremely low mineralization of water in the rivers and lakes of Eastern Fennoscandia significantly limits species richness (Alimov 2008). It has already been proven that high humus concentration in water also can impact the species composition in the benthos and complicate the assessment of water quality (Kalinkina *et al.* 2020; Ozolins *et al.* 2021). Dark water color caused by humus reduces the photic zone and suppresses primary production (Bergström & Karlsson 2019). Despite the fact that humus in natural concentrations is not generally considered a toxic agent, humic acids have been shown to have a significant effect on the enzyme activity of aquatic organisms of various taxonomic groups, resulting in reduced lifespan or survival (Steinberg *et al.* 2003). The influx of natural humic acids from wetlands and forests significantly acidifies the rivers of Eastern Fennoscandia (Moiseenko & Gashkina 2011). In boreal lakes with dark water, the proportion of chironomid larvae is significantly increased, while the abundance of stoneflies and mayflies, which indicate high water quality, is reduced (Kesti *et al.* 2022).

The water quality indicators used in the work are aimed primarily at assessing eutrophication caused by anthropogenic impact and the flow of nutrients and organic substances into rivers and lakes. However, using the example of the Shuya River, located nearby, it was shown that high concentrations of phosphorus can be caused by an increase in the runoff of this element from swamps and forests due to climate change. Relatively warm winters with long thaws cause increased winter water flow, which brings humus, phosphorus and iron into the rivers (Kalinkina *et al.* 2018).

There is no doubt that relatively low 'background' values of water quality assessment indices, observed due to the natural features of the catchment area, make it very difficult to determine the level of anthropogenic pollution using biotic indices. In this case, a hasty interpretation of the data obtained, especially during express research, and without taking into account the chemical composition of the water, can lead to erroneous conclusions about the level of human pollution of lakes and rivers.

CONCLUSION

When assessing water quality, it is important to select more than one biological monitoring method with clearly identifiable responses because multiple pressures affect river ecosystems at different spatial and temporal scales. The results of this work indicate that monitoring programs in countries with dark waters with low salinity can rely on the use of EPT, BBI and EBI biotic indices. In addition, it should be taken into account that biotic indices may reveal some pollution in the absence of negative human influence. When assessing water quality, it is important to take into account the influence of the biotope: examine different habitats, and for comparison choose stations that are identical in hydrological parameters.

AUTHOR CONTRIBUTIONS

I.A.B. collected, processed, and analyzed macrozoobenthos of rivers, and assessed the ecological state of waters using biotic indices. S.F.K. collected, processed, and analyzed the periphyton of water bodies and assessed their condition using specialized indices. E.S.S. collected, processed, and analyzed the macrozoobenthos of lakes Pandozero and Sundozero using indicator species. All authors read and approved the final manuscript.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

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