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Research Article

Investigating ecological degradation of mining-impacted small river systems: A macrozoobenthic case study from Northern Armenia

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A R T I C L E I N F O

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A B S T R A C T

The degradation and loss of aquatic ecosystems have become environmental challenges facing the world. This study aimed to assess the ecological status of the Shnogh River and its tributaries (the Kharatadzor and Dukanadzor rivers) in the northern mining region of Armenia using macrozoobenthic parameters. Macrozoobenthic samples were collected from six sites between 2013 and 2018, with the exception of two sites that were sampled only during 2013-2015 and 2015-2018, respectively. Parameters analyzed included species' richness, abundance, the Shannon-Wiener diversity index, and the extended biotic index (EBI). Cluster analysis, based on EBI values, and principal component analysis (PCA), based on species' richness, were applied to identify patterns of community structure and ecological deterioration. The upper sections of the Kharatadzor and Dukanadzor rivers, designated as control sites, showed higher taxonomic richness and diversity, including pollution-sensitive taxa, indicating a good to high ecological status. However, species' richness and diversity, total abundance, and ecological quality noticeably declined at the downstream sites of the rivers, which may have been due to mining-related contamination and habitat degradation. These influences, in turn, also extended to the Shnogh River, particularly at sites downstream of the confluences with the Dukanadzor and the Kharatadzor rivers, as bio-indicated by similar patterns of macrozoobenthic parameters. Cluster analysis grouped the sites, separating upstream control sites included in the analysis from downstream impacted sites. While PCA identified two principal components explaining 83.7% of the total variance (PC1=61.5%, PC2=22.2%), with sensitive (e.g., Ephemeroptera, Plecoptera, Trichoptera) and tolerant taxa (e.g., Odonata, Decapoda) contributing to observed patterns. Macrozoobenthic parameters, supported by cluster analysis and PCA, confirmed the noticeable hydroecological deterioration and taxonomic simplification. These findings underscore the need for stricter regulations on mining discharges to safeguard freshwater ecosystems and biodiversity.

1. Introduction

The hydrographic network of any mountain watershed mainly consists of streams and small rivers (Rinaldo *et al.*, 2018; Xingyuan *et al.*, 2023). Small rivers represent the initial links of the hydrographic system, eventually merging into larger river systems. In addition, these parts are the most vulnerable to anthropogenic impacts, both direct and indirect. The formation and characteristics of small rivers are very closely related to the landscape of the basin, which makes them particularly susceptible to the overuse of water resources of rivers and their catchment basins (Foti *et al.*, 2022).

Small rivers play a crucial role in regulating the hydrological regime and ecological balance of landscapes, influencing the hydrological and hydrochemical properties of downstream lowland rivers. However, in recent years, climate change has led to water losses and increased water use, causing the widespread degradation of the hydrological regimes of many rivers (Fritz *et al.*, 2008; Meyer *et al.*, 2007).

Small mountain rivers are widely used in Armenia to provide water for domestic, industrial, recreational, and agricultural needs (Global Water Partnership, 2002). The exponential growth of the mining and energy industries in the mountain river watersheds has caused numerous environmental, hydrological, technological, and organizational problems (Gevorgyan, 2011). In 2017, the European Union (EU) and Armenia signed an agreement under which, among other things, Armenia committed to aligning its laws with EU standards and international environmental instruments, such as the EU Water Framework Directive (WFD) (European Union & Republic of Armenia, 2018).

The main objective of the WFD is to develop and/or maintain good ecological and chemical conditions in all surface water bodies. In the WFD, biological quality elements (BQEs), including fish, macrophytes, and benthic fauna, are the primary determinants of ecological classification, with hydrophysical, hydrochemical, and hydromorphological factors serving as secondary indicators.

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Classification of ecological status involves comparing the current state with typical baseline conditions; the greater the discrepancy, the worse the ecological status (European Parliament and Council, 2000; Pennelli *et al.*, 2006). Among other things, BQEs should be used more frequently as part of the overall ecological quality assessment (Nixon *et al.*, 1996).

The need for a wider use of benthic macroinvertebrates as important indicators for assessing ecological quality has been highlighted by recent hydroecological assessments in Armenia (Asatryan & Dallakyan, 2019; Dallakyan & Asatryan, 2021). As a diverse and widespread group, they are highly sensitive to anthropogenic changes in aquatic environments. Because changes in their community structure caused by pollution can have cascading effects on higher trophic levels, such as fish and bird populations, they provide important information on environmental quality (Onwona Kwakye *et al.*, 2021).

Mining activities have numerous direct and indirect effects on social and ecological systems, encompassing exploration, development, construction, operation, maintenance, expansion, closure, decommissioning, and reuse. These effects, which affect community livelihoods, ecological integrity, and regional development dynamics, can be both positive and negative. However, mining activities have led to several environmental problems, including air, water, and soil pollution with particulate matter and hazardous substances, such as chemical waste and heavy metals. Since it often causes bioaccumulation, habitat destruction, and long-term ecological problems, this pollution poses a significant threat to public health and ecological stability (Gevorgyan *et al.*, 2021).

This study aimed to assess changes in the macrozoobenthic population and ecological status of small river systems affected by mining. To assess the hydroecological impact of mining activities, the authors conducted a quantitative and qualitative study of the macrozoobenthic community in the Shnogh River and its tributaries (the Kharatadzor and Dukanadzor rivers) in the Lori Province of Armenia. Mining activities pose a serious threat to the ecological health of surface watercourses in Armenia. The sector plays a significant role in Armenia's economy, especially in the Lori Province of northern Armenia. However, inadequate management of industrial discharges remains a significant environmental challenge. Previous studies have documented heavy metal contamination in the rivers and soils of mining-affected areas within Lori Province (Gevorgyan *et al.*, 2016, 2021; Hovhannisyan *et al.*, 2022), including elevated concentrations of Mo, Cu, Pb, and Zn at the Shnogh River site located downstream of the Teghout Mountain Enrichment Combine (Gevorgyan *et al.*, 2021). However, detailed information on the hydrobiological and hydroecological consequences of this contamination has not yet been published. Observing the macrozoobenthic community in this region, particularly upstream and downstream locations relative to the Teghout Mountain Enrichment Combine, including the copper-molybdenum mine and its tailings dam, is, therefore, a crucial step toward evaluating mining-induced changes in the macrozoobenthic community and the ecological status of small river systems, thereby contributing to broader efforts for regional environmental sustainability. This study provides insight into the ecological impacts of mining on river systems by integrating macrozoobenthic data with multivariate statistical analysis. The results enhance our understanding of how mining-induced stresses impact aquatic ecosystems and community structure, leading to the loss of aquatic biodiversity in the South Caucasus.

2. Materials and Methods

2.1 Study area

The Shnogh River flows through the Lori Province of Armenia and is a right tributary of the transboundary Debed River. It originates on the northern slopes of the Gugarats mountain range and extends for 20 km, draining a mostly forested catchment area of 114 km^2. Cutting through the volcanic plateau, it creates a 1.5 km long canyon and about 100 m deep, with a highly fragmented V-shaped valley. The river has an average annual discharge of 0.8 m^3/s. The Shnogh River and its tributaries, including the Kharatadzor and Dukanadzor rivers, are typical mountain streams and are almost entirely fed by rain and snowmelt. In the downstream basin of the Shnogh River lies the Teghout Mountain Enrichment Combine, covering an area of 1,544.6 hectares, which includes an open copper-molybdenum mine and a tailings dump dam (Nalivaiko, 2021).

2.2 Sample collection, preparation, and analysis

A total of 72 macrozoobenthic samples were collected from different locations of the Shnogh River and its tributaries, the Kharatadzor and Dukanadzor rivers (Fig. 1, Table S1), between 2013 and 2018, except for the upper reach of the Dukanadzor River (sampled only between 2013 and 2015) and the Shnogh River site downstream of the confluence with the Dukanadzor River (sampled only between 2015 and 2018). Analyses were based on comparable hydrological seasons, identical fieldwork protocols, and standardized data processing. To account for temporal variability and minimize bias associated with unequal sampling durations, both range and long-term averages were estimated. This ensured the ecological comparability of datasets collected over different time intervals.

Sampling was conducted according to the requirements of the AQEM project (AQEM, 2002). For each site, a protocol was developed that described river and floodplain morphology, hydrology, and vegetation, ensured accurate geolocation, and documented biological sampling.

The samples were collected in 5-7 replications using a Surber sampler with a 500 µm mesh size and a frame size of 33 cm × 33 cm, covering an area of 0.1 m^2. They were fixed with a 96% ethanol solution and subsequently processed in the laboratory. Laboratory procedures included sieving, sorting, and identification of organisms. Before sorting, samples were carefully rinsed through a set of sieves under running water to remove fine particles. The mesh sizes of the sieves were selected based on the substrate characteristics.

The fixed macrozoobenthic specimens were identified microscopically. Initially, samples were sorted and separated from the substrate under a stereomicroscope (Micromed MS-4-ZOOM LED, St. Petersburg, Russian Federation). Taxonomic identification to finer levels (species or genus) was performed using an upright microscope (Micromed 2, 3-20×, St. Petersburg, Russian Federation). Identification was carried out with the aid of freshwater taxonomic keys. Specimens from each taxonomic group were dried on filter paper and quantified to determine the abundance of each taxon.

2.3 Biodiversity and Water Quality Assessment

The diversity of the macrozoobenthic community in the river ecosystems was assessed using the Shannon-Wiener diversity index (H') (Shannon, 1948),

$$H' = -\sum_{i=1}^{s} p_i \ln p_i,$$

where s is the number of species, and p_i is the proportion of individuals of ith species relative to the total number of individuals.

The diversity index levels were classified according to the scheme used by Baliton *et al.* (2020): very high (H′ ≥ 3.5), high (H′ = 3.00-3.49), moderate (H′ = 2.50-2.99), low (H′ = 2.00-2.49), and very low (H′ ≤ 1.99). River water quality was evaluated using the extended biotic index (EBI) (AQEM, 2002). The EBI levels were classified into water quality categories according to Semenchenko (2004): high (EBI > 9), good (EBI = 8-9), moderate (EBI = 6-7), low (EBI = 4-5), and bad (EBI = 1-3).

2.4 Data analysis and processing

Box plot was created using Statistica (ver. 7, StatSoft, Tulsa, OK, USA). Multivariate analyses (cluster analysis and principal component analysis (PCA)) were performed using Origin (ver. 2018, OriginLab Corporation, Northampton, MA, USA). Cluster analysis was conducted using the group average method with Pearson correlation distance to identify groups of sites with similar ecological quality based on EBI values. PCA was applied to identify major gradients in macrozoobenthic community structure across sites, based on the species' richness of each

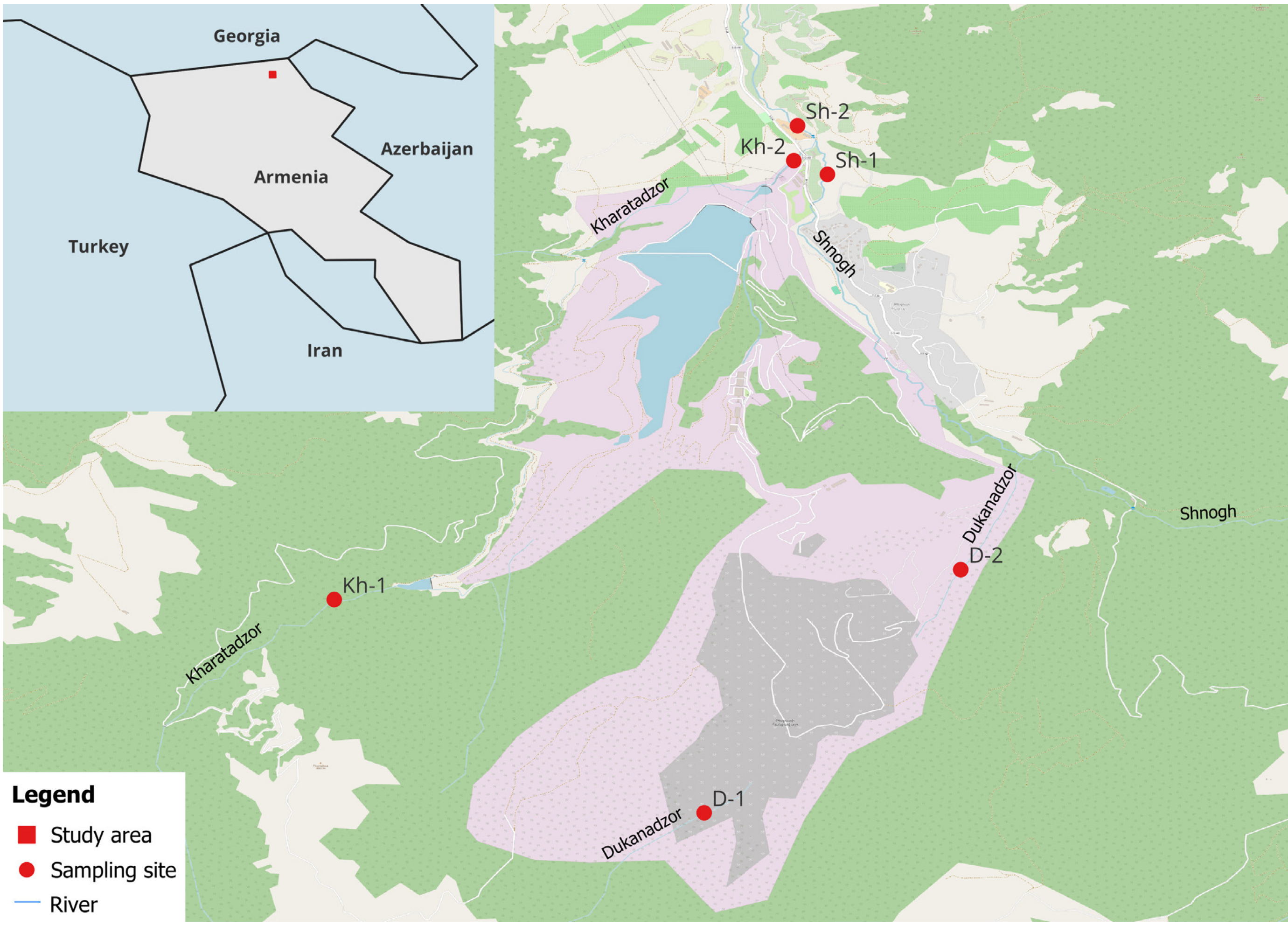

Fig. 1. Map shows the study area in Armenia and the locations of the sampling sites.

taxonomic group. The software automatically standardized data for both analyses before the analysis. All univariate calculations (average, minimum, and maximum values) were done with Excel (ver. 2019, Microsoft, Redmond, USA). Mapping was performed using QGIS (ver. 3.40.7, QGIS Development Team, Grüt, Switzerland).

3. Results and Discussion

The upstream sites of the Kharatadzor (Kh-1) and Dukanadzor (D-1) rivers, selected as control sampling sites, demonstrated relatively high macrozoobenthic species diversity. At Kh-1, species richness varied from 16 to 31 species, while at D-1 it varied from 14 to 29 (Fig. 2a). A total of 52 and 43 taxa, representing different taxonomic levels (mainly species and occasionally genus), were recorded at Kh-1 and D-1, respectively (Table S2).

A large number of species found at both sites demonstrated low or very low tolerance to pollution (Table S2), according to the classification of Mandaville (2002). Notable examples identified at Kh-1 included *Serratella ignita*, *Ephemera romantzovi*, and other pollution-sensitive species of Ephemeroptera; seven species of Plecoptera, including *Brachyptera brevipennis* and *Pontoperla teberdinica*; as well as *Rhyacophila armeniaca* (among seven species of Trichoptera) and *Blepharicera fasciata* (Diptera) (Table S2). At D-1, sensitive taxa included *Agapetus oblongatus*, *Glossosoma capitatum*, and *Rhyacophila nubile* (Trichoptera); four taxa of Plecoptera (stonefly larvae); and three species of Ephemeroptera (mayfly larvae) (Table S2). The presence of these pollution-sensitive taxa indicated that ecological conditions were favorable at both sites, with low anthropogenic disturbance in the upstream parts.

Shannon-Wiener diversity index values ranged from 3.0 to 4.0 at Kh-1 and from 3.0 to 3.8 at D-1 (Fig. 2b), indicating high to very high macrozoobenthic diversity at both sites (Baliton *et al.*, 2020). Total macrozoobenthic abundance ranged from 101 to 749 ind./m_2 (including outliers) at Kh-1 and from 77 to 1171 ind./m2 at D-1 (Fig. 2c). Similarly, EBI scores of 9 to 12 at Kh-1 and 9 to 11 at D-1 indicated water quality in the range of good to high ecological status at both sites (Fig. 2d) (Semenchenko, 2004), confirming the biological integrity of the river sites.

Despite the favorable conditions, macrozoobenthic abundance has declined over time. Total abundance at Kh-1 decreased from 281-749 ind./m^2 in 2013-2014 (during mining construction activities) to 101-420 ind./m^2 in 2015-2018 (during mining operations), representing a multi-year average decline of about 52% between the two periods. Total abundance at D-1 decreased noticeably, dropping from 627-1171 ind./m^2 in 2013-2014 to only 77 ind./m^2 in 2015, corresponding to a multi-year average decrease of over 90%. Such a sharp decline at D-1 was most likely the result of significant riverbed disturbance caused by waste rock deposition during the construction of the landfill and quarry in the river floodplain. In contrast, the biological decline at Kh-1 may have been related to the influence of the adjacent tailings dam located in the former river valley. The tailings are dispersed through mechanical, hydraulic, and aeolian processes, while the weathering of ore particles increases the release of heavy metals into the environment (Ritcey, 1989).

However, the temporal trend was different for macrozoobenthic diversity. The Shannon-Wiener diversity index did not fall below three and did not exceed four at both sites during the entire study period. All this indicated that total abundance was more sensitive to short-

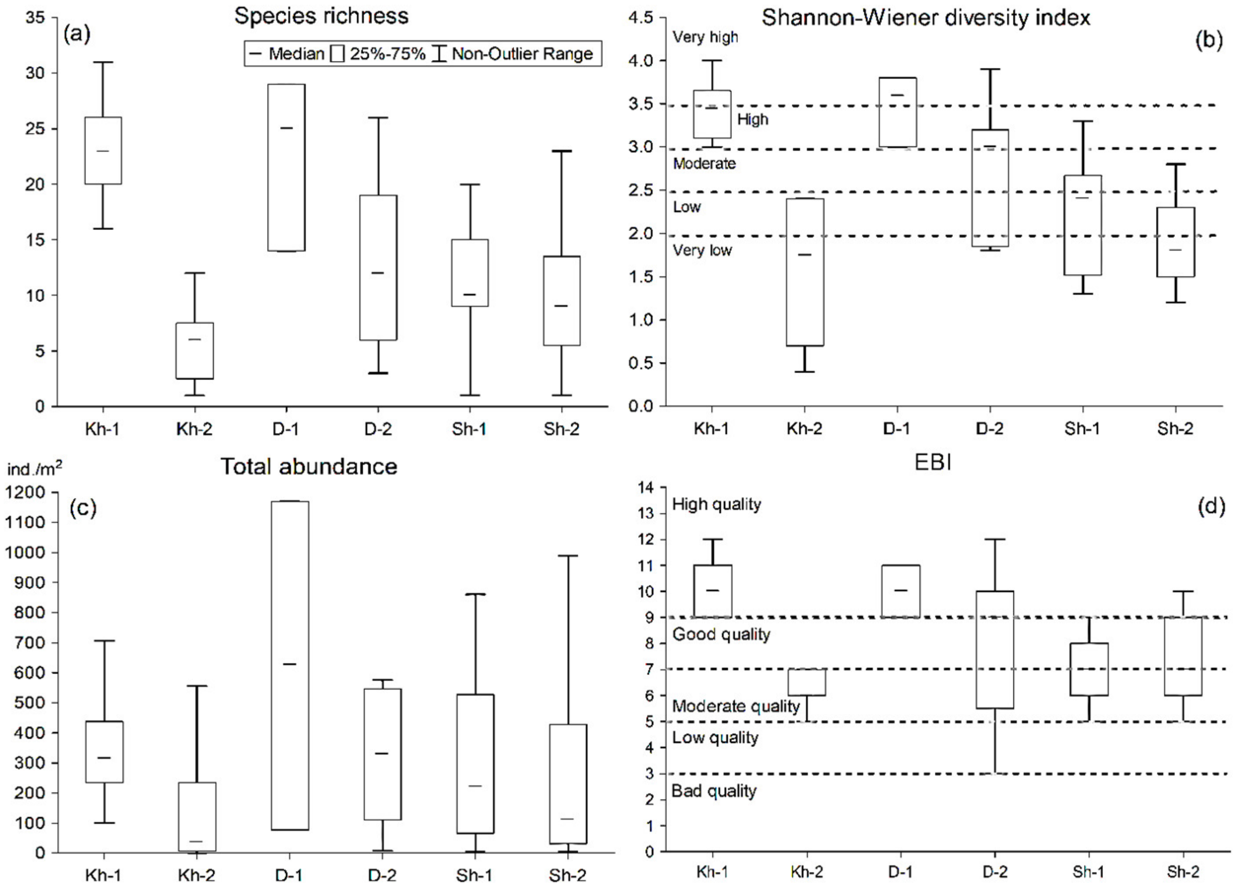

Fig. 2. Macrozoobenthic community metrics in the Kharatadzor, Dukanadzor, and Shnogh rivers: (a) species richness, (b) Shannon-Wiener diversity index, (c) total abundance, and (d) EBI.

term physical disturbances, whereas diversity was more influenced by longer-term ecological stability and resilience.

Downstream sites of the Kharatadzor (Kh-2) and Dukanadzor (D-2) rivers showed clear signs of ecological disturbance. Species richness ranged from 1-12 at Kh-2 to 3-26 at D-2 (Fig. 2a), representing multi-year average decreases of approximately 76% at Kh-2 and about 41% at D-2, compared to their upstream counterparts. Shannon-Wiener diversity index values ranged from 0.4-2.4 at Kh-2 (very low to low diversity (Baliton *et al.*, 2020)) to 1.8-3.9 at D-2 (very low to very high diversity (Baliton *et al.*, 2020) (Fig. 2b). On a multi-year average, diversity decreased by about 57% at Kh-2 and nearly 20% at D-2, relative to their upstream counterparts. EBI showed hydroecological deterioration with scores of 5-7 at Kh-2 (low to moderate water quality (Semenchenko, 2004)) and 3-12 at D-2 (deficient to high water quality (Semenchenko, 2004)(Fig. 2d), representing multi-year average decreases of about 40% and 20%, respectively, compared to their upstream counterparts.

Macrozoobenthic abundances were consistently lower at downstream sites (Kh-2 and D-2), compared to their upstream counterparts (Fig. 2c). Kh-2 showed abundances of 1-556 ind./m^2, excluding outliers and extremes, while D-2 was characterized by abundances of 8-576 ind./m^2 (Fig. 2c). These values were, on a multi-year average, about 75% lower at Kh-2 and approximately 50% lower at D-2, relative to their upstream counterparts. Despite the higher abundance at D-2, the relative decrease in abundance at D-2 was greater than at Kh-2 (Fig. 2c), which contradicted the observed patterns in species richness and diversity (Figs. 2a and b). This discrepancy indicates that total abundance was more affected in the Dukanadzor River, whereas species richness and diversity were more influenced in the Kharatadzor River. This pattern may have been due to the noticeably higher total abundance and simultaneously slightly lower number of species with the lowest tolerance at D-1 compared to Kh-1 (average abundance of 19.4 vs. 6.5 ind./m^2 and average species number of 10 vs. 11, respectively), such as *Leuctra hippopus*, *Perla pallida*, *Protonemura bacurianica*, *Brachyptera transcaucasica*, *Brachyptera brevipennis*, *Pontoperla teberdinica*, *Epeorus caucasicus*, *Rhithrogena iridina*, *Serratella ignita*, *Agapetus oblongatus*, *Glossosoma capitatum*, *Rhyacophila nubila*, *Rhyacophila armeniaca*, and *Blepharicera fasciata* (Mandaville, 2002). The surface waters at the downstream sites may have been affected by Teghout mining activities (copper-molybdenum mine and tailings dam), which have been documented as potential sources of environmental contamination, including that of rivers (Grechko *et al.*, 2021). Mining operations are known to impact benthic communities, especially sensitive taxa, leading to ecological degradation (Allan & Castillo, 2007).

Biological declines were also observed at downstream sites (Kh-2 and D-2) over time, particularly between the mining construction phase (2013-2014) and the operational phase (2015-2018). For example, total macrozoobenthic abundance at Kh-2 decreased from 6-556 ind./m^2 in 2013-2014 to 1-116 ind./m^2 in 2015-2018, excluding outliers and extremes, representing a multi-year average decline of about 82%. Shannon-Wiener diversity index values declined from 0.6-2.4 to 0.4-1.9 over the same periods, corresponding to a multi-year average decrease of approximately 33%. At D-2, total abundance decreased from 110-576 ind./m^2 to 8-568 ind./m^2, representing a multi-year average decline of about 52%, while diversity index values shifted from 2.7-3.2 to 1.8-3.9, corresponding to a multi-year average decrease of approximately 16%.

To assess the broader impact of polluted tributaries on the Shnogh River ecosystem, two sampling stations were studied: one located downstream of the confluence with the Dukanadzor River (Sh-1) and the other downstream of the confluence with the Kharatadzor River (Sh-2). It was reported that an average Mo concentration of approximately 0.07 mg/L in the water at the Shnogh River site, located downstream of the Teghout Mountain Enrichment Combine, exceeded the local norm (ARLIS, 2011; Gevorgyan *et al.*, 2021), specifically the II class of water quality (0.0015 mg/L) by more than 45 times. The corresponding average Mo concentration in macrozoobenthos samples was 21.89 mg/kg, which is over four times higher than that at reference sites (below

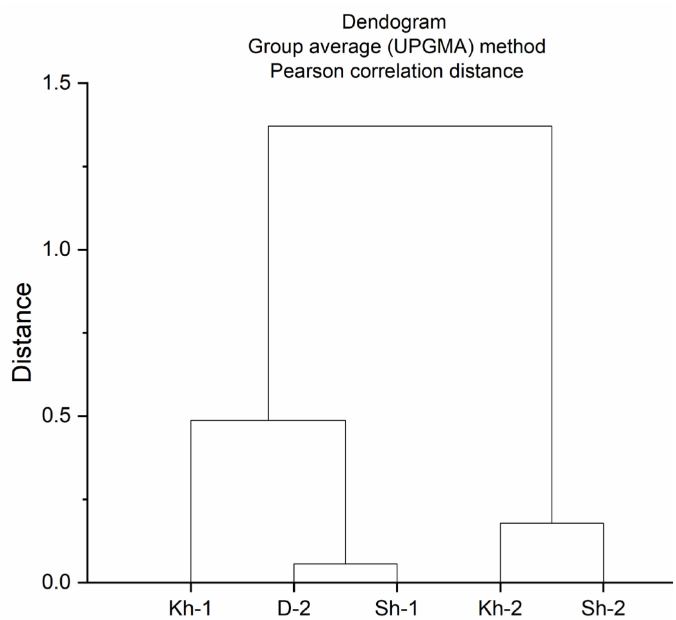

Fig. 3. Hierarchical clustering of sampling sites based on similarity in EBI values. D-1 was excluded due to insufficient data for this analysis.

the detection limit). Similarly, the average Cu concentration in water at this site was approximately 0.03 mg/L, which was more than three times higher than at reference sites (< 0.01 mg/L) (Gevorgyan *et al.*, 2021) and exceeded the local norm of 0.023 mg/L (ARLIS, 2011). In macrozoobenthos, the Cu concentration was 104.24 mg/kg, about 1.4-4.9 times higher than at reference sites (21.13-74.29 mg/kg) (Gevorgyan *et al.*, 2021). These contaminations were attributed to the Teghout copper-molybdenum mining activities, including the associated tailings dump (Gevorgyan *et al.*, 2021). The Shnogh River is mainly connected to these contamination sources through its tributaries, the Kharatadzor and Dukanadzor rivers, which flow through the Teghout mining areas.

Observations of benthic macroinvertebrates at the Shnogh River sites Sh-1 and Sh-2 revealed 1–20 species at Sh-1 and 1–23 at Sh-2 (Fig. 2a). Shannon–Wiener diversity index values ranged from 1.3 to 3.3 at Sh-1 (Fig. 2b), indicating very low to high macrozoobenthic diversity (Baliton *et al.*, 2020) and from 1.2 to 2.8 at Sh-2 (Fig. 2b), showing very low to moderate diversity (Baliton *et al.*, 2020). Macrozoobenthic abundance ranged from 4 to 861 ind./m^2 at Sh-1 and from 4 to 990 ind./m^2 at Sh-2 (Fig. 2c). The corresponding EBI scores ranged from 5 to 9 at Sh-1 (Fig. 2d), indicating low to good water quality (Semenchenko, 2004), and from 5 to 10 at Sh-2 (Fig. 2d), indicating low to high ecological status (Semenchenko, 2004). Macrozoobenthic abundance ranged from 4-861 ind./m^2 at Sh-1 and from 4-990 ind./m^2 at Sh-2 (Fig. 2c). All macrozoobenthic parameters were noticeably lower in the Shnogh River compared to the control sites (Kh-1 and D-1; Figs. 2a-d); for example, species' richness was, on a multi-year average, 52-57% lower, diversity 35-45% lower, EBI scores 26-30% lower, and total abundance 16-58% lower. These decreases can be attributed to the cumulative impact of contamination from the Dukanadzor and Kharatadzor tributaries, with more pronounced degradation observed at Sh-2, likely due to the additive effect of both tributaries. This impact was also evident in the temporal changes in macrozoobenthic parameters of the Shnogh River, particularly biological declines observed at Sh-2 from the mining construction phase (2013-2014) to the operation phase (2015-2018). Total abundance declined from 259-990 ind./m^2 in 2013-2014 to 4-383 ind./m^2 in 2015-2018; species richness decreased from 11-23 to 1-11; Shannon-Wiener diversity index values shifted from 1.6-2.8 to 1.2-2.8; and EBI scores dropped from 7-10 to 5-7, representing multi-year average decreases of 85%, 59%, 16%, and 29%, respectively.

Fig. 3 shows a hierarchical clustering of sampling sites based on the similarity of EBI values, measured using the Pearson correlation distance metric. Rather than focusing on absolute EBI values, this

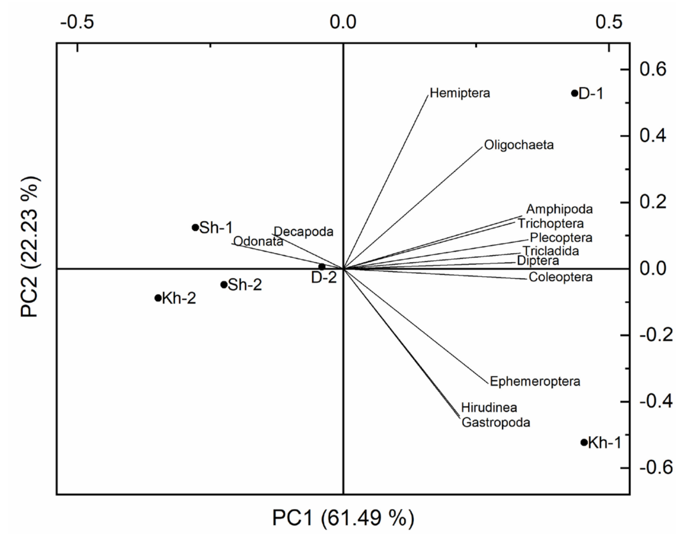

Fig. 4. PCA of macroinvertebrate community composition based on species' richness in sampling sites.

approach grouped sites with comparable biological water quality trends. The D-2 and Sh-1 sites were the closest to each other (Fig. 3), suggesting that they had similar ecological conditions or were subject to similar anthropogenic influences. Their proximity within the cluster suggested identical patterns of contamination, likely from Teghout mining activities. Kh-1 was included at a higher hierarchical level in this cluster (Fig. 3), indicating a slightly different biological water quality profile, presumably due to lower levels of human activity.

On the other hand, Kh-2 and Sh-2 formed a distinct cluster with notable internal similarities (Fig. 3), suggesting the presence of different river sections with similar environmental or contaminant characteristics, most likely related to Teghout mining activities. This classification confirms the regional pattern of hydroecological degradation and highlights the similarities between the affected areas in different river sections.

Spatial changes in species' richness were primarily evident for the following taxa: Hemiptera, Oligochaeta, Amphipoda, Trichoptera, Plecoptera, Tricladida, Diptera, Coleoptera, Ephemeroptera, Hirudinea, and Gastropoda, which were more closely associated with Kh-1 and D-1, the control sites sampled (Fig. 4). These sites showed a broader taxonomic spectrum (Fig. 4), including both sensitive (e.g., Ephemeroptera, Trichoptera, and Plecoptera) and tolerant (e.g., Oligochaeta, Hemiptera) taxa (Mandaville, 2002), indicating relatively stable ecological conditions and higher species richness.

In contrast, the remaining sampling sites (Kh-2, Sh-1, Sh-2, and D-2) showed significant losses in species' richness and a shift toward communities dominated by tolerant taxa such as Odonata and Decapoda (Fig. 4). These groups often imply disturbed ecosystems, which are typically associated with habitat modification or deterioration in water quality (Rosenberg & Resh, 1993). The dominance of tolerant taxa indicated an increase in anthropogenic pressure, particularly at Kh-2 and D-2, where nearby Teghout mining activities likely contributed to environmental deterioration compared to the control sites. The consequences of this pressure spread downstream to the Sh-1 and Sh-2 sites in the Shnogh River, resulting in a reduction in the diversity of macroinvertebrates due to environmental filtration of sensitive species and the development of tolerant taxa (Fig. 4).

Thus, the control sites (Kh-1 and D-1) were clustered in association with a broader range of macrozoobenthic taxa, while the Kh-2, D-2, Sh-1, and Sh-2 sites were spatially separated along the PCs, which was consistent with their reduced taxonomic richness and prevalence of tolerant taxa (Fig. 4). The clustering of affected sites revealed homogeneous communities (Fig. 4), indicating hydroecological degradation, confirming the EBI, abundance results, and diversity metrics.

The obtained results, particularly higher biodiversity, abundance, and ecological stability at the upstream sites of the rivers, are common patterns for small rivers under low or no anthropogenic pressures (Pond *et al.*, 2022), while the patterns observed at the downstream parts (decreased biodiversity, abundance, and ecological quality), particularly near mining-impacted zones, align well with other findings on small river systems in mining-impacted areas (Clements, 2004; Drover, 2018; Gevorgyan *et al.*, 2016, 2021; Mercado-Garcia *et al.*, 2022). Low flow volumes, narrow channels, and direct connectivity between contamination sources and biological communities increase the vulnerability of small river systems to anthropogenic influences and even short-term or site-specific disturbances, as demonstrated in our study by observed declines in macrozoobenthic parameters resulting from mining operations and construction activities. In this regard, small rivers can serve as early detectors for hydroecological degradation. Conversely, large rivers may exhibit somewhat different patterns due to higher dilution capacity, more diverse microhabitats, and more complex hydrological regimes, which increase their self-purification capacity and resistance to similar pressures, as reported in different studies (Ciszewski & Grygar, 2016; Dennis *et al.*, 2009; Macklin *et al.*, 2006).

4. Conclusions

This study investigated the ecological degradation of mining-impacted small river systems (the Kharatadzor, Dukanadzor, and Shnogh rivers) using macrozoobenthos as a bioindicator. The results showed distinct spatial trends in macrozoobenthic abundance, richness, and diversity, as well as ecological quality in the studied rivers. Control sites in the upper reaches of the Kharatadzor and Dukanadzor rivers demonstrated relatively high levels of ecological health, abundance, species richness, and diversity, featuring sensitive taxa. However, the downstream sites of the Kharatadzor and Dukanadzor rivers exhibited lower macrozoobenthic parameters and a dominance of tolerant taxa, indicating a worsening hydroecological situation and degradation, likely resulting from mining-related activities. These impacts also extended to the Shnogh River, with similar hydroecological consequences.

Multivariate analysis confirmed these findings, with a clear indication of taxonomic homogeneity and a gradient of hydroecological degradation. The results demonstrate the usefulness of macrozoobenthos in detecting hydroecological changes and highlight the need for enhanced environmental protection and continuous monitoring to mitigate the impact of human activities in the region. The results obtained here show that the Teghout Mountain Enrichment Combine activities, including the operation of the copper-molybdenum mine and its associated tailings, have a measurable impact on the macrozoobenthos community both in the tributaries and in the Shnogh River. The observed reduction in species' richness, shift towards tolerant taxa, and decrease in abundance indicate a significant change in ecological integrity, suggesting that even small river systems can serve as early indicators of hydroecological degradation.

Such changes have broader ecological consequences, since macrozoobenthos play a key role in nutrient cycling, organic matter decomposition, and serve as a food source for higher trophic levels. Thus, the overall impact not only reflects local pollution but also emphasizes potential cascading effects on the structure and stability of aquatic ecosystems in the region. In light of these findings, more stringent regulatory measures are needed to limit contaminant discharges and ensure compliance with environmental standards. At the same time, ecological restoration and conservation measures should be implemented to restore affected ecosystems. These measures may include riparian zone restoration, aquatic habitat enhancement, sediment and water remediation, and the development of long-term biomonitoring programs to track ecosystem responses. The combination of regulatory measures with proactive restoration strategies will be crucial for preserving biodiversity and maintaining the ecological resilience of river systems affected by mining activities. From a policy standpoint, establishing a coordinated framework that requires all mining operations to be managed based on cumulative ecological impacts rather than individual site assessments, coupled with mandated real-time environmental monitoring and adaptive management plans to align with ecosystem thresholds for resilience. Furthermore, a policy framework might require mining projects to demonstrate measurable biodiversity offsets or restoration actions that fully compensate for ecological degradation within the same river system, coupled with incentives for restoration, reforestation, and rehabilitation to enhance resilience.

CRediT authorship contribution statement

Susanna Hakobyan: Definition of intellectual content, concepts, literature search, design, experimental studies, data acquisition, data analysis, writing—original draft preparation; **Karen Jenderedjian:** Literature search, statistical analysis, data analysis, data acquisition; **Termine Khachikyan:** Data analysis, data acquisition, statistical analysis; **Ashok Vaseashta:** writing—review and editing, conceptualization; **Hermine Yepremyan:** Data acquisition, experimental studies; **Ruzan Hovhannisyan:** Literature search, Data analysis; **Gor Gevorgyan:** Conceptualization, design, definition of intellectual content, literature search, data acquisition, data analysis, statistical analysis, writing—original draft preparation.

Declaration of competing interest

The authors declare that they have no competing financial interests or personal relationships that could have influenced the work presented in this paper.

Data availability

Most of the data is contained in the article. Additional information may be requested from the authors.

Declaration of generative AI and AI-assisted technologies in the writing process

The authors confirm that there was no use of Artificial Intelligence (AI)-Assisted Technology for assisting in the writing or editing of the manuscript and no images were manipulated using AI.

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Supplementary data

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