Evaluation of the Impacts of Abandoned Mining Areas: A Case Study with Benthic Macroinvertebrate Assemblages

Mi-Jung Bae *, Jeong-Ki Hong and Eui-Jin Kim

Abstract: Mining activities are among the most long-lasting anthropogenic pressures on streams and rivers. Therefore, detecting different benthic macroinvertebrate assemblages in the areas recovered from mining activities is essential to establish conservation and management plans for improving the freshwater biodiversity in streams located near mining areas. We compared the stability of benthic macroinvertebrate communities between streams affected by mining activities (Hwangjicheon: NHJ and Cheolamcheon: NCA) and the least disturbed stream (Songjeonricheon: NSJ) using network analysis, self-organizing map, and indicator species analysis. Species richness was lowest at sites where stream sediments were reddened or whitened due to mining impacts in NHJ and NCA. Among functional feeding groups, the ratio of scrapers was lower (i.e., NHJ) or not observed (i.e., NCA) in the affected sites by mining. The networks (species interactions) were less connected in NHJ and NCA than in NSJ, indicating that community stability decreased in the area affected by mining activity. We identified five groups based on the similarity of benthic macroinvertebrate communities according to the gradients of mining impacts using a self-organizing map. The samples from the reference stream (clusters 1 and 5), sites located near the mining water inflow area (cluster 4), sites where stream sediments acid-sulfated (cluster 2), and sites that had recovered from mining impacts (cluster 3). Among the 40 taxa selected as indicators defined from the five clusters in self-organizing map, only few (Physa acuta, Tipula KUa, and Nemoura KUb) indicator species were selected in each cluster representing the mining-impacted sites. Our results highlighted that the benthic macroinvertebrate community complexity was lower in streams affected by mining activity. Furthermore, the range of disturbed areas in the streams, where conservation and management plans should be prioritized, can be quantified by examining alterations in the benthic macroinvertebrate community.

Keywords: abandoned mining area; self-organizing map; network analysis; indicator species analysis

1. Introduction

Freshwater ecosystems are most vulnerable to anthropogenic disturbances worldwide [1,2]. Hence, understanding the patterns and drivers of biodiversity loss is essential for foreseeing the changes of freshwater ecosystems to environmental alteration and establishing conservation strategies [3]. Among the existing anthropogenic pressures (e.g., industrialization, urbanization, and land use changes), mining activities have the most long-lasting effects on streams and rivers [4]. In streams and rivers near mining areas, physical and chemical factors are vulnerable to changes, such as organic matter breakdown [5], conductivity increase [6], or sediment contamination [7,8], erosion or deposition [9]. In particular, drainage water from abandoned mining areas (or active and historic mining operations) is the main input source of heavy metal pollution in adjacent streams [10,11]. Heavy metal concentration in water often exceeds the permissible limits recommended for drinking or agricultural use, and such toxic contaminants are destined to severely damage important resources in freshwater after exposure [12]. Despite the permissible limit for water quality criteria, accumulated heavy metal in streambeds due to past mining activities...
cannot be removed for several decades [13]. Thus, freshwater ecosystems in streams and rivers near mines or abandoned mining areas are disturbed or even destroyed, resulting in the general modification of the composition and functional attributes of aquatic organisms as well as reduced species richness [14].

Benthic macroinvertebrates play an important role in aquatic food webs [7,15]. Because of their high sensitivity to various contaminants, benthic macroinvertebrates have been generally applied to assess the ecological impacts from various anthropogenic disturbances. Benthic macroinvertebrates are closely related to streambed substrate, which provides a place to rest, shelter, and feed [16] and are the essential material exchangers crossing the sediment-water interface [7,16,17]. Therefore, benthic macroinvertebrates are generally known to be the sensitive organisms to the effects of mining activities [12]; identifying and characterizing alterations of benthic macroinvertebrate communities in streams affected by mining activities is useful for detecting the impacts of abandoned mining areas on aquatic ecosystems.

In general, anthropogenic pressures, such as mining impacts, can cause abrupt changes in the composition of functional feeding groups (FFGs) as well as the structure of the benthic macroinvertebrate community [18,19]. For example, the abundance and diversity of species generally decrease in streams located near abandoned mining areas (e.g., [20–26]). High levels of cadmium and zinc concentration had a negative effect especially on the number of Plecoptera and Trichoptera [27,28]. In particular, the number of sensitive taxa (e.g., stoneflies, mayflies, caddisflies, etc.) sharply decrease, whereas the tolerant taxa (e.g., chironomids, beetles, etc.) become dominant [12,29,30]. Additionally, interactions between FFGs also vary as the available habitat for stream fauna is reduced, the quality of food resources deteriorates, and metal precipitates cause chemical weathering of the streambed [12,31,32]. Consequently, the ratio of scrapers in FFGs, including Psephenidae and Heptageniidae, which usually attach their bodies and scrape algae in boulders and cobbles [33], can rapidly decrease in the affected ecosystem.

To date, when evaluating the changes in stream environments near abandoned mining areas, only the water quality variables or the coarse level of benthic macroinvertebrate data (i.e., family level) have been used (but also see [34,35]), and descriptive or multivariate analyses have been applied to interpret the results. Network analysis [36], which has now been increasingly used to understand the interactions and stability of complex ecological communities, can provide undetectable insights compared with traditional analytical methods that analyze the species separately [37,38]. A self-organizing map [39], which is an unsupervised neural network, is a powerful tool to interpret the relation between ecological community and environment based on visualizing, grouping, and predicting the complex ecological data. The application of self-organizing map has been recently increased especially in the interpretation of the ecological data (e.g., [3,38,40,41]). However, there have been no studies that applied both techniques simultaneously to evaluate the mining impacts on the benthic macroinvertebrate community.

This study aimed to investigate the changes in benthic macroinvertebrate communities in streams near mining areas using modelling approaches (network analysis and self-organizing map). First, we compared the stability of benthic macroinvertebrate communities between two streams within an area affected by mining activities and a reference stream in the least-disturbed area. We applied network analysis to compare the community complexity (the number of interactions among species) among three streams (reference stream and two streams near the mining area). Second, the range of disturbed areas in the streams was quantified by examining alterations in the benthic macroinvertebrate community. We used the self-organizing map, which can classify clusters based on the similarity of community composition to evaluate if there were significant differences of the communities between streams and detect the sites impacted or recovered from mining areas.
2. Materials and Methods

2.1. Study Area

To compare the impacts of abandoned mining areas on benthic macroinvertebrate communities, we monitored the streams near abandoned mining areas (Hwangjicheon: NHJ and Cheolamcheon: NCA) and in a less disturbed area (Songjeonricheon: NSJ). The research area (Taebaek, Korea) presented Monsoon-influenced hot-summer humid continental climate according to the Köppen-Geiger climatic type [42]. During the last 10 years, the mean annual temperature was 9.16 °C and the total amount of annual precipitation was 1233 mm. Approximately 71.6% of the total annual precipitation was concentrated from June to September [43]. The research area is a mountainous highland with a very low rate of arable land with 270.17 km² (89.0%) of forest land, 14.22 km² (4.7%) of agricultural land, and 19.13 km² (6.3%) of other areas. It mainly consists of sedimentary rocks (sandstone and limestone) and anthracite and is the region where Korea’s representative coal mines existed and produced approximately 72.9% of the national coal production.

The NHJ stream length is 29.10 km and the watershed area is 204.10 km². NHJ is the main tributary of Nakdonggang River, which is the longest river in Korea and the main drinking water source in the middle eastern area of Korea. A total of 14 waste coal mines are located near the upstream of NHJ. The amount of waste produced is 697,915 m³, and the volume of mine water is 3765 m³/day. Substrates in NHJ are acid-sulfated in a 950 m section from the minehead, and 8084 people inhabit the area near the stream. NCA has a stream length of 18.4 km, a watershed area of 61.50 km², and four waste coal mines. The waste volume is 13,430 m³, 174 m³/day mined metal volume and approximately 100 m white sediment section due to the mining water. In total, 3850 local people inhabit the NCA basin, and the number of mineheads near the NCA is six. In 2010, the heavy metal concentrations of the mine water, stream water, and groundwater were officially measured, and the results showed that Pb and Fe concentrations in NCA, as well as Cd, Pb, Fe, and Mn concentrations in NHJ, exceeded the water quality limits [44].

NSJ was considered the reference stream, with a stream length of 9.4 km and a watershed area of 49.3 km². It is located in a well-preserved mountainous area, with water temperature below 20 °C even in summer and dissolved oxygen of 9 mg/L. It also provides a habitat for Brachymystax lenok, a cold-water fish species and a second-grade endangered species according to the Ministry of Environment, Korea.

2.2. Ecological Data

Benthic macroinvertebrates were collected from 23 sampling sites (NHJ: 8 sites, NCA: 7, and NSJ: 8) using a Surber sampler (30 × 30 cm², 300 µm mesh) at a depth of 10 cm during April 2017. This month presents normal water flow periods, thus avoiding community variability related to the season (Figure 1). Each site was sampled in a riffle area with three replicates within a 50-m reach. Collected benthic macroinvertebrates were preserved in 95% ethanol in the field, and then the solution was replaced with 70% ethanol in the laboratory. In the laboratory, specimens were sorted, counted, and identified mostly into species level under a microscope (SMZ10; Nikon, Japan). Identification was conducted based on previously published methods [45–50].
A total of 34 environmental factors were measured at the sampling sites or in the laboratory. Geographical factors were extracted using the Spatial Analyst toolbox in ArcGIS 10.6 (ESRI, Redlands, CA, USA), including altitude, slope, distance from the source, and stream order. The ratio of land cover types (e.g., urban, agriculture, forest, grassland, wetland, and bare land), was extracted in an area of 1 km-length and 200 m-wide buffer zone (i.e., 500 m up- and downstream from benthic macroinvertebrate sampling sites) [3]. Hydrological factors, substrate composition, and water quality factors, such as pH, conductivity, dissolved oxygen (DO) and turbidity were measured in each sampling site using YSI ProDSS (YSI Inc./Xylem, Rye Brook, NY, USA), whereas biological oxygen demand (BOD), ammonia-nitrogen (NH$_3$-N), nitrate-nitrogen (NO$_3$-N), total nitrogen (TN), phosphate-phosphorus (PO$_4$-P), total phosphorus (TP), and chlorophyll-a (Chl-a) were measured in the laboratory [51]. In the case of NHJ and NCA, heavy metal concentrations (Pb, Fe, Cd, and Mn) in stream water were also measured using ICP-OES (Optima 7300DV & Avio500; Perkin-Elmer, MA, USA) and ICP-MS (NexION 300X; Perkin-Elmer, MA, USA). However, the metal concentrations did not exceed the Korean water quality limits, and thus these factors were not included in the subsequent analysis.

2.3. Data Analysis

We conducted four analytical approaches (Figure 2). First, as descriptive measures, species richness, abundance, and FFGs were compared between the three streams sampled. Second, network analysis based on species co-occurrence was computed to evaluate the
community structure complexity among the streams. Pairwise correlations for each species were calculated using Spearman’s correlation rank [43,52]. The number of nodes (number of vertices (species) in a network), number of edges (number of connections in a network), average node degree (average number of connections per node), average path length (expected distance between two nodes), transitivity (the tendency of the nodes to cluster together), and edge density (the values dividing existing edges dividing into all the possible edges) were calculated to compare the degree of community structure complexity [36,38]. Network analysis was conducted using the package igraph [53] in R software [54].

Figure 2. The procedure for data analysis.

Third, a self-organizing map (SOM) [40] was used to characterize the sites affected by abandoned mining areas and the patterns of benthic macroinvertebrate community compositions. The SOM is composed of an input and an output layer. The input layer (74 species and 23 sites) was condensed and arrayed into a two-dimensional grid (output layer) for easy interpretation of the community. We determined the number of output units based on the formula recommended by [55]: 5 × sqrt (number of samples). Thus, we used 30 output units (N = 5 × 6). The SOM units were classified by K-means cluster with the Davies-Bouldin index, which finds optimal clustering based on a partition that minimizes distances within and maximizes distances between clusters [56–58]. For the SOM analysis, species abundance data were log-transformed (log (x + 1)), and rare taxa (i.e., the species that occurred only once) were excluded. SOM analysis was conducted using the SOM toolbox in MATLAB version 6.1 [59]. To test significant differences in the community composition among the clusters defined by SOM analysis, we computed the multi-response permutation procedure (MRPP) using the function “mrpp” in the vegan package in R [60]. In addition, the Kruskal-Wallis (KW) test was used to compare the environmental factors and community indices among the clusters. In the cases when the environmental factors or community indices were significantly different among clusters,
the nonparametric Dunn’s multiple comparison test was applied for posthoc comparisons. Kruskal-Wallis and Dunn’s multiple comparison tests were conducted using the package agricolae [61] in R.

Finally, an indicator species analysis was applied to identify the species (or taxa) representing each cluster [62–64]. Indicator species were determined based on the relative abundance and relative occurrence frequency among the defined clusters. The Indicator Value (IndVal) ranges from 0 to 100 (i.e., all the individuals in a certain species are included only in a certain cluster) [38]. A species (or taxa) is determined as an indicator species only if IndVal for a particular cluster is significant and higher than 25% ($p < 0.05$) [62]. Indicator species analysis was performed using the “indval” function in the Labdsv package [65] in R.

3. Results

3.1. Differences in Benthic Macroinvertebrate Community among Streams

The average number of species was the highest between the sites in the NSJ stream (reference stream) with 32 species (five phyla, seven classes, 13 orders, 42 families, and 76 species in total), followed by NHJ (four phyla, six classes, 13 orders, 37 families, and 55 species) and NCA (five phyla, seven classes, 14 orders, 33 families, and 49 species) with 16 species. In the NHJ stream, species richness decreased from eight species in site 2 to two in site 4, where stream sediments consisted of acid sulfate soils but increased from 14 species in site 5 to 30 species in site 8 (the highest number of species in NHJ) (Figure 3). In the NCA stream, only six species were observed at site 2. In the NSJ stream, species richness ranged from 24 (site 7) to 40 (site 2).

Abundance also decreased from site 2 (1237 individual/m$^2$) to site 4 (7) in the NHJ stream, similar to the pattern of species richness. In the NCA stream, abundance was the lowest at site 2 (219). In the NSJ stream, abundance ranged from 3270 individual/m$^2$ (site 4) to 11,270 individual/m$^2$ (site 8).

For FFGs, regardless of the mining impacts, both the abundance and species richness of collector gatherers (CG) were the highest in all streams. However, the percentage of CG was much higher in NCA (SR: 38.1%, abundance: 87.4%) and NHJ (SR: 42.4%, abundance:
88.2%) than in NSJ (SR: 25.2%, abundance: 59.9%) (Figure 4). In the case of scrapers (SC), both the species richness and abundance were higher in NSJ (SR: 33.1%, abundance: 20.1%) than in NCA (SR: 26.9%, abundance: 5.5%) and NHJ (SR: 19.0%, abundance: 1.1%).

Figure 4. Differences in functional feeding group (FFG, %) based on abundance and species richness (SR) in streams in reference sites and abandoned mining area (NSJ: Songjeonrichoen stream, NCA: Cheulamcheon stream and NHJ: Hwangjicheon stream).

Considering the differences between study sites in each stream, the ratio of SC decreased from site 1 (26.3% in abundance and 1.4% in species richness) to site 4 (0.0%) in NHJ. SC and collector-filterers were not observed in site 2 in NCA. The ratio of shredders (0.0–31.1% in abundance and 0.0–16.7% in species richness) and predators decreased, whereas that of SC increased from up- to downstream in NSJ.

3.2. Species Co-Occurrence Patterns of the Benthic Macroinvertebrate Community

Co-occurrence networks indicated a clear difference between the streams affected by the abandoned mining area and the reference stream. For example, the networks were less connected in the NHJ and NCA streams than in the NSJ stream (Table 1 and Figure 5). In addition, the average node degree and number of edges were highest in sites from the NSJ stream (38.68 and 1635), followed by those in the NHJ (25.60 and 826) and NCA (22.94 and 622) streams.

Table 1. Parameters of the benthic macroinvertebrate co-occurrence network (NSJ: Songjeonrichoen stream, NCA: Cheulamcheon stream, and NHJ: Hwangjicheon stream).

| Parameters               | Reference Stream | Streams Located Near Abandoned Mining Areas |
|-------------------------|------------------|---------------------------------------------|
|                         | NSJ  | NCA  | NHJ  | NCA  | NHJ  | NCA  | NHJ  |
| Number of nodes         | 76   | 49   | 55   | 49   | 55   | 49   | 55   |
| Number of edges         | 1635 | 622  | 826  | 622  | 826  | 622  | 826  |
| Average node degree     | 38.68| 22.94| 25.60| 22.94| 25.60| 22.94| 25.60|
| Average path length     | 1.48 | 1.52 | 1.53 | 1.52 | 1.53 | 1.52 | 1.53 |
| Transitivity            | 0.70 | 0.73 | 0.78 | 0.73 | 0.78 | 0.73 | 0.78 |
| Edge density            | 0.52 | 0.48 | 0.47 | 0.48 | 0.47 | 0.48 | 0.47 |
3.3. Patterns of Benthic Macroinvertebrate Community

The benthic macroinvertebrate communities were patterned and classified using the SOM learning process based on the similarities of the community composition (Figure 6). The SOM classified output units into five clusters (1–5) based on the K-means cluster with the Davies-Bouldin index. Benthic macroinvertebrate communities among five clusters were significantly different according to MRPP (A = 0.30, p < 0.001). These five clusters represented the differences of the community according to the gradients of mining impacts. Samples from the NSJ stream were all included in clusters 1 (site 1–4) and 5 (site 5–8); samples from sites located near the mining water inflow area were mainly included in cluster 4; samples from sites where stream sediments were mainly acid-sulfated were included in cluster 2; samples from sites that had recovered from mining impacts were included in cluster 3 in NHJ and NCA streams.

Figure 5. Co-occurrence networks among benthic macroinvertebrate species from streams located near abandoned mining area (NCA and NHJ) and reference stream (NSJ). Nodes indicates each species observed in each stream, and lines represent connections among species.

(a) Distribution and classification of sampling sites in the self-organizing map based on the abundance of benthic macroinvertebrates, (b) K-means cluster with Davies-Bouldin index. Numbers 1 to 5 indicate cluster defined by self-organizing map.
Community indices and environmental factors also differed between the five clusters (Tables 2–4). For example, species richness (KW = 18.5, p < 0.05) and Plecoptera species richness (KW = 16.5, p < 0.05) were higher in clusters 1 and 5. In addition, the Shannon diversity index (KW = 16.4, p < 0.05), Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness (KW = 18.7, p < 0.05), and EPT abundance (KW = 16.3, p < 0.05) were higher in clusters 1, 3, and 5. In the FFGs, the ratio of SC (abundance: KW = 15.3, p < 0.05; SR: KW = 13.0, p < 0.05) was higher in cluster 1 (abundance: 17.4% and SR: 26.6%) and 5 (abundance: 66.2% and SR: 28.0). Regarding environmental factors, the ratio of forest (KW = 11.6, p < 0.05) in land use was higher in clusters 1 (92.3%) and 4 (93.8%), although all the clusters showed a high ratio of forest (more than 55% on average). There was no significant difference in substrate composition, except for the size <0.063 mm (KW = 12.6, p < 0.05). In terms of water quality, conductivity (KW = 15.3, p < 0.05) was higher in clusters 2 (360.8 μS/cm), 3 (379.4 μS/cm), and 4 (370.3 μS/cm), respectively. The concentration of T-N (KW = 17.6, p < 0.05) was higher in clusters 2 (2.11 mg/L) and 3 (2.1 mg/L).

### Table 2. Differences in community indices between five clusters in a self-organizing map.

| Clusters | 1 | 2 | 3 | 4 | 5 |
|----------|---|---|---|---|---|
| Species richness | 37 (3) | 12 (6) | 27 (4) | 11 (6) | 28 (3) |
| Abundance | 6815 (3391) | 7917 (1778) | 20,819 (16,946) | 1394 (1324) | 7193 (3431) |
| Ephemeroptera | 1404 (387) | 99 (157) | 1181 (513) | 238 (442) | 2921 (1747) |
| Plecoptera | 1324 (1444) | 4 (6) | 2 (5) | 20 (19) | 7 (5) |
| Trichoptera | 357 (242) | 120 (191) | 957 (933) | 45 (88) | 676 (596) |
| Ephemeroptera | 11 (2) | 2 (2.3) | 9.2 (2.9) | 1.5 (1.7) | 12 (0.8) |
| Plecoptera | 6.3 (0.5) | 0.5 (0.5) | 0.2 (0.4) | 1.3 (0.5) | 1.3 (0.5) |
| Trichoptera | 8.8 (1.0) | 2.5 (2.6) | 7.2 (1.1) | 2 (3.4) | 6.5 (1.9) |
| EPT abundance | 3085 (1258) | 223 (348) | 2141 (925) | 304 (534) | 3605 (2267) |
| EPT richness | 36 (6) | 5 (4.9) | 16.6 (3.7) | 4.8 (4.9) | 19.8 (1.7) |
| Non-insecta abundance | 452 (619) | 185 (165) | 456 (280) | 21 (21) | 150 (70) |
| Non-insecta species richness | 3278 (1916) | 7509 (14,494) | 18,222 (16,269) | 1069 (827) | 3438 (1207) |
| Other-insecta abundance | 3 (0.8) | 4 (2.4) | 5.6 (0.9) | 1.5 (0.6) | 3 (1.2) |
| Other-insecta species richness | 26 (2.2) | 5 (4.9) | 16.6 (3.7) | 4.8 (4.9) | 19.8 (1.7) |
| Dominance index | 0.61 (0.09) | 0.95 (0.04) | 0.82 (0.1) | 0.89 (0.06) | 0.64 (0.06) |
| Shannon diversity | 3.06 (0.34) | 0.78 (0.28) | 1.56 (0.71) | 1.14 (0.63) | 2.69 (0.28) |
| Richness index | 17.71 (1.12) | 27.93 (24.07) | 16.12 (1.57) | 23.03 (4.04) | 17.58 (1.06) |
| Evenness | 0.42 (0.05) | 0.11 (0.04) | 0.21 (0.1) | 0.16 (0.09) | 0.37 (0.04) |

Values represent mean (standard deviation); Lowercase letters represent significant differences in environmental values among the five clusters based on the Kruskal-Wallis test and Dunn’s multiple comparison test (p < 0.05).

### Table 3. Differences in functional feeding groups between five clusters in the self-organizing map.

| Category | Cluster | 1 | 2 | 3 | 4 | 5 |
|----------|---------|---|---|---|---|---|
| FFG abundance | Predator (%) | 7.3 (2.9) | 9.1 (20.1) | 1.6 (1) | 1.4 (0.8) | 3 (0.9) |
| Parasite (%) | 0 (0) | 0 (0.1) | 0 (0) | 0 (0) | 0 (0) |
| EPT (%) | 17.4 (10.2) | 4.4 (4.5) | 3.9 (2.9) | 0.4 (0.6) | 22.8 (6.2) |
| Collector filterer (%) | 6.8 (8.3) | 1 (1) | 4 (2.8) | 2.7 (2.6) | 7.9 (5.1) |
| Collector gatherer (%) | 53.6 (4) | 85.1 (17.6) | 90.5 (4.2) | 88.6 (9.9) | 66.2 (2.7) |
| Shredder (%) | 14.8 (16.2) | 0.5 (0.7) | 0.1 (0.1) | 6.9 (11.3) | 0 (0.1) |

Values represent mean (standard deviation); Lowercase letters represent significant differences in environmental values among the five clusters based on the Kruskal-Wallis test and Dunn’s multiple comparison test (p < 0.05).
### Table 4. Differences in environmental factors between five clusters in self-organizing map.

| Category                  | Cluster 1 | Cluster 2 | Cluster 3 | Cluster 4 | Cluster 5 |
|---------------------------|-----------|-----------|-----------|-----------|-----------|
| **Geological factors**    |           |           |           |           |           |
| Altitude                  | 738 (85)  | 741 (66)  | 606 (61)  | 857 (47)  | 556 (42)  |
| DFS (km)                  | 0.97 (0.36)| 1.39 (0.51)| 2.76 (0.47)| 0.85 (0.15)| 2.06 (0.33)|
| Slope                     | 8.27 (2.1)| 9.48 (3.37)| 15.11 (6.17)| 11.03 (7.96)| 12.15 (8.73)|
| Stream order              | 2 (1)     | 3 (1)     | 5 (1)     | 2 (0)     | 5 (1)     |
| **Hydrology**             |           |           |           |           |           |
| Depth (cm)                | 9.8 (3.2) | 11.4 (2.4)| 16.3 (4.9)| 10.8 (4.1)| 19.5 (10.8)|
| Velocity (cm/s)           | 0.53 (0.18)| 0.3 (0.16)| 0.59 (0.15)| 0.49 (0.21)| 0.59 (0.26)|
| **Land use (%)**          |           |           |           |           |           |
| Urban                     | 2.9 (3.8) | 6.1 (6.4) | 22.4 (17.1)| 3.2 (2.9) | 4.1 (2.9) |
| Agriculture               | 4.8 (5.8) | 10.5 (9.2)| 12.8 (8.7)| 0.6 (0.8) | 16.7 (16.2)|
| Forest                    | 92.3 (9.4)| 77.6 (16.0)| 55.1 (14.3)| 93.8 (7.6) | 76.2 (19.2)|
| Grassland                 | 0 (0)     | 1.2 (2.4) | 0.9 (1)   | 1.8 (3.4)| 0.8 (1.1) |
| Wetland                   | 0 (0)     | 2.3 (3.1) | 0 (0)     | 0.1 (0.1)| 0.1 (0.1) |
| Bare land                 | 0 (0)     | 4.6 (6.8)| 6.7 (7.1)| 0.6 (0.9)| 2.1 (1.5) |
| **Substrate composition (%)** |       |           |           |           |           |
| <0.063 mm                 | 0.0 (0.0)| 0.2 (0.2)| 0.4 (0.3)| 0.5 (0.5)| 0.0 (0.0)|
| 0.063–2 mm                | 0.9 (0.2)| 0.5 (0.4)| 0.7 (0.2)| 0.9 (0.2)| 1.1 (0.4)|
| 2–4 mm                    | 1.1 (0.3)| 0.9 (0.1)| 0.9 (0.2)| 0.9 (0.1)| 1.2 (0.5)|
| 4–8 mm                    | 1.4 (0.2)| 1.3 (0.6)| 1.0 (2)  | 1.1 (0.4)| 0.9 (0.8)|
| 8–16 mm                   | 1.8 (0.7)| 2.3 (1.6)| 1.9 (0.2)| 2.0 (0.4)| 1.2 (0.8)|
| 16–32 mm                  | 4 (0.6)  | 4.4 (3.2)| 3.2 (0.8)| 4.2 (1.2)| 2.1 (1.8)|
| 32–64 mm                  | 7.9 (2.6)| 7.9 (4.9)| 6.2 (2.4)| 10.1 (4.9)| 6.5 (3.2)|
| 64–128 mm                 | 22.1 (4.4)| 25.5 (9.2)| 18 (6.2)| 21.6 (8.5)| 23.7 (4.6)|
| 128–256 mm                | 55.4 (7.3)| 56.9 (15)| 65.4 (9.4)| 56.1 (12.5)| 52.8 (12.1)|
| >256 mm                   | 5.4 (9.3)| 0 (0)    | 2.5 (4.2)| 2.5 (3.3)| 10.7 (21.4)|
| **Water quality**         |           |           |           |           |           |
| DO (%)                    | 85.9 (1.8)| 100.2 (10.2)| 108.2 (11)| 88.8 (0.4)| 88.1 (2.2)|
| DO (mg/L)                 | 9.59 (0.19)| 10.63 (1.43)| 10.73 (1.44)| 9.82 (0.21)| 9.35 (0.1)|
| pH                        | 7.57 (0.46)| 8.35 (0.5)| 8.73 (0.47)| 7.73 (0.27)| 8.51 (0.23)|
| Conductivity (µS/cm)      | 116.0 (68.8)| 360.8 (113.7)| 379.4 (112.8)| 370.3 (188.1)| 92.3 (23.3)|
| Turbidity (NTU)           | 1.1 (0.6)| 1 (1.5)  | 0.2 (0.3)| 1.1 (1.1)| 1 (0.2)|
| BOD (mg/L)                | 1.13 (0.24)| 0.97 (0.21)| 1.16 (0.29)| 0.7 (0.08)| 1.03 (0.17)|
| NH3-N (mg/L)              | 0.01 (0.003)| 0.016 (0.019)| 0.013 (0.009)| 0.013 (0.004)| 0.008 (0.003)|
| NO2-N (mg/L)              | 0.79 (0.25)| 1.54 (0.41)| 1.18 (0.32)| 0.71 (0.28)| 1.22 (0.13)|
| T-N (mg/L)                | 1.07 (0.36)| 2.11 (0.47)| 2.1 (0.11)| 0.81 (0.28)| 1.48 (0.13)|
| PO4-P (mg/L)              | 0.003 (0)| 0.006 (0.007)| 0.006 (0.004)| 0.03 (0.001)| 0.03 (0)|
| T-P (mg/L)                | 0.008 (0.002)| 0.013 (0.009)| 0.015 (0.008)| 0.006 (0.001)| 0.005 (0)|
| Chl-a (mg/L)              | 0.53 (0.13)| 0.77 (0.53)| 0.5 (0.07)| 0.48 (0.05)| 0.53 (0.05)|

Values represent mean (standard deviation); Lowercase letters represent significant differences of environmental values among the five clusters based on the Kruskal-Wallis test and Dunn’s multiple comparison test ($p < 0.05$).

### 3.4. Indicator Species

Forty taxa were selected as indicator species for the five clusters from the SOM analysis based on IndVal ($p < 0.05$) (Table 5). In cluster 1, indicator species were mainly included in the EPT taxa. In particular, the species in Plecoptera, such as Kamimuria coreana, Stavsolus japonicas, Rhopalopsole mahunkai, and Sweltsa nikkoensis were selected in cluster 1. In cluster 5, 17 species were selected in total, and among them, species such as Ecdyonurus kibunensis and Hydropsyche orientalis, which generally inhabit mid- to downstream, were included. In clusters 2 and 4, only one (Physa acuta) and two species (Tipula KUa and Nemoura KUb) were selected. In cluster 3, 13 species were selected, including Ephemeroptera and Trichoptera species, such as E. levis, Epeorus pellucidus, Hydropsyche kozhantschikovi, and Hydroptila KUa.
Table 5. Indicator species (or taxa) for five clusters from the self-organizing map.

| Cluster | Order | Family | Species (Taxa) | IndVal | p Value |
|---------|-------|--------|---------------|--------|---------|
| 1       | Ephemeroptera | Heptageniidae | Cinygmula KUa | 0.75  | 0.009   |
| 1       | Ephemeroptera | Heptageniidae | Epeorus curatus | 0.64  | 0.007   |
| 1       | Ephemeroptera | Ephemerellidae | Cincticostella lecanidoidae | 0.48  | 0.014   |
| 1       | Plecoptera | Nemouridae | Nemoura KUa | 1.00  | 0.002   |
| 1       | Plecoptera | Perlodidae | Statislulus japonicus | 1.00  | 0.001   |
| 1       | Plecoptera | Perlidae | Kamimuria coreana | 1.00  | 0.001   |
| 1       | Plecoptera | Nemouridae | Amphinemura coreana | 0.75  | 0.006   |
| 1       | Plecoptera | Leuctridae | Rhopalopsole mabunshikai | 0.75  | 0.01    |
| 1       | Plecoptera | Chloroperlidae | Sveltsa nikkoensis | 0.67  | 0.002   |
| 1       | Trichoptera | Philopotamidae | Wormaldia KUa | 0.75  | 0.005   |
| 1       | Trichoptera | Hydrobiosidae | Apsilochorema KUa | 0.62  | 0.021   |
| 1       | Trichoptera | Rhyacophilidae | Rhyacophila shikotsusensis | 0.56  | 0.032   |
| 1       | Trichoptera | Lepidostomatidae | Lepidostoma KUb | 0.47  | 0.044   |
| 1       | Coleoptera | Elmidae | Elmidae sp | 0.80  | 0.004   |
| 1       | Diptera | Tipulidae | Hexatoma KUa | 0.64  | 0.008   |
| 1       | Diptera | Ceratopogonidae | Ceratopogonidae sp | 0.62  | 0.006   |
| 1       | Diptera | Tipulidae | Dicranota KUa | 0.53  | 0.018   |
| 2       | Basomatophomra | Physidae | Physa acuta | 0.41  | 0.039   |
| 3       | Archiogogochaeta | Naididae | Chaetogaster limnaei | 0.72  | 0.003   |
| 3       | Archiogogochaeta | Tubificidae | Limnodrilus gotoi | 0.48  | 0.003   |
| 3       | Mesogastropoda | Pleuroceridae | Semisulcospira libertina | 0.41  | 0.046   |
| 3       | Ephemeroptera | Heptageniidae | Ecdyonurus levis | 0.79  | 0.001   |
| 3       | Ephemeroptera | Heptageniidae | Epeorus pellicidus | 0.43  | 0.037   |
| 3       | Ephemeroptera | Baetidae | Baetis fuscatus | 0.80  | 0.001   |
| 3       | Ephemeroptera | Baetidae | Baetis ursinus | 0.61  | 0.001   |
| 3       | Trichoptera | Hydroptilidae | Hydroptila KUa | 0.87  | 0.001   |
| 3       | Trichoptera | Hydropsychidae | Hydropsyche valvata | 0.71  | 0.002   |
| 3       | Trichoptera | Hydropsychidae | Cheumatopsyche KUa | 0.41  | 0.019   |
| 3       | Trichoptera | Hydropsychidae | Hydropsyche kozhantschikovi | 0.40  | 0.04    |
| 3       | Diptera | Tipulidae | Antochi KUa | 0.39  | 0.002   |
| 3       | Diptera | Chironomidae | Chironomidae spp | 0.24  | 0.033   |
| 4       | Plecoptera | Nemouridae | Nemoura KUb | 0.44  | 0.001   |
| 4       | Diptera | Tipulidae | Tipula KUb | 0.65  | 0.007   |
| 5       | Ephemeroptera | Heptageniidae | Ecdyonurus bajkovae | 0.71  | 0.002   |
| 5       | Ephemeroptera | Heptageniidae | Ecdyonurus kubunensis | 0.47  | 0.012   |
| 5       | Ephemeroptera | Ephemerellidae | Uracanthella rufa | 0.57  | 0.004   |
| 5       | Ephemeroptera | Leptophlebiidae | Paraleptophlebia chocoarata | 0.51  | 0.012   |
| 5       | Ephemeroptera | Ephemeridae | Ephemera strigata | 0.48  | 0.05     |
| 5       | Trichoptera | Glossosomatidae | Glossosoma KUa | 0.60  | 0.006   |
| 5       | Trichoptera | Hydropsychidae | Hydropsyche orientalis | 0.48  | 0.018   |

Only taxa with significant values are shown. IndVal: indicator value.

4. Discussion

Our study showed that the community complexity of benthic macroinvertebrates was lower in streams located near mining areas than in the reference stream. For example, the biodiversity of benthic macroinvertebrates decreased in the study sites (site 2 to 4), less than 1.8 km away from a site to which effluents of mining area directly flow, and the biodiversity was recovered starting from 4.1 km downstream of the effluent site (site 5) in NHJ. Although the effects of the effluents may vary depending on the amount of pollutants and the type of mining industry, the difference in the community and diversity of benthic macroinvertebrates between the study sites reflected the degree of proximity to the mining area (influence zone).

In this study, based on both physicochemical and biological parameters, NHJ and NCA were divided into three sections: the sites not affected by mining (the upper sections in both streams), sites directly affected by mining (e.g., site 2 to 4 in NHJ), and sites that were recovered from mining impacts (e.g., site 5 to 8 in NHJ). In the case of the least-disturbed stream, NSJ was largely divided into two areas: the upper-middle stream (site 1 to 4) and middle-downstream (site 5 to 8).

The co-occurrence networks revealed that the stability (complexity) of the benthic macroinvertebrate community was lower in the streams located near mining areas than
in the reference stream. Network size and connectivity between species were lower in NHJ and NCA than in NSJ, resulting in the loss of biodiversity and biotic integrity [66,67]. Moreover, the average path length representing the expected distance between the two species was higher in NHJ and NCA than in NSJ.

These three streams, as typical mountainous streams, have similar environmental conditions except for the acid mining impacts: high altitude (from 514 to 898 m), high forest ratio (from 44 to 100%) in land use, and a high proportion of large substrate (i.e., cobble and boulder) in the streambed. Among these streams, the major differences were the anthropogenic impacts. In particular, the study sites, where the whitening/reddening of stream sediment was directly observed, showed relatively high values of conductivity, T-N, and T-P, which was caused by the effluent of treated wastewater from mining areas.

The species, such as chironomids, which have short life cycles (1 month) to endure their life in unstable and/or disturbed habitats, could be dominant in NHJ and NCA. In addition, SC, which scrape algae and organic matter from the surface of rocks and stream plants, were not observed in the FFG. According to the database of the National Aquatic Ecological Monitoring Program of South Korea [68], the diversity and ash-free dry mass (the weight of periphyton per unit area, cm$^3$) of periphyton, which is the main food source of SC, was much lower in NHJ and NCA than in NSJ. Owing to a lower quality and amount of food sources, as well as deterioration of water quality, no SC was observed in the sites affected by mining effluents. Our results were similar to those of other studies [69–74] that showed lower species richness and abundance of SC owing to the increased osmoregulation stress [75–77] and heavy metal bioaccumulation due to heavy metal contaminated biofilm, which is the main food source of SC [21,78–80]. The loss of certain FFGs and/or taxa of benthic macroinvertebrates is critical in freshwater ecosystems because each of them is responsible for certain roles, such as food sources of higher trophic organisms, nutrient cycling, leaf litter decomposition and/or bioremediation from disturbed habitats [81–83]. Furthermore, some of Ephemeroptera (e.g., Baetis) and Trichoptera (e.g., Hydropsyche and Polycentropus) groups are relatively tolerant to mining impacts [7,84,85]. For example, B. rhodani was relatively abundant in the Nent River, where high concentrations of Zn were continuously observed even though mining ceased at the beginning of the 1900s. Non-cased Trichoptera are tolerant to moderately polluted areas (<500 mg Zn/L) [13]. Although these two taxa were not included as indicator species in sites 2 and 4, which were directly affected by effluents, Baetid groups such as Baetis fiscatus and Baetis usrsinus, and Trichoptera species such as Hydropsyche valvata, Cheumatopsyche KUa, and Hydropsyche kozhantschikovi, were selected as indicator species in the recovered sites (cluster 3 in SOM). Furthermore, the abundance and species richness of SC, such as Ecdyonurus levis, Epeorus pellucidus, and Semisulcospira libertine, were significantly higher in the recovered sites.

Our results evaluated the impact range of the mining area and the differences in community complexity between two streams located near mining areas and one stream in the least-disturbed area. However, a high-efficiency mining water treatment plant was recently constructed (3000 m$^3$ of wastewater per day) near site 2 in NHJ in 2019; therefore, long-term monitoring should be performed to assess the recovery of the macroinvertebrate community from the mining impacts and to further understand the recovery processes and periods of freshwater ecosystem. The community recovery of benthic macroinvertebrates after the remediation or restoration treatment can be different according to the degree of contamination, the removal of metal contaminated soils, the closeness of upstream sources of colonization, hydrologic conditions, the effectiveness of remediation, etc. [86]. For example, in the upper Arkansas River near mining area in Colorado, USA, after mine drainage treatment, macroinvertebrate community became similar with the community reference sites (upstream of mining-impacted area) within two years, and especially increased EPT taxa [20]. In contrast, 20–29 years of long-term monitoring in four mining-impacted watersheds in the western USA revealed that benthic macroinvertebrate species richness increased within 10.25 years on average after remediation activity from mining, including water treatment, construction of contaminated ponds, revegetation of riparian areas [35].
In the Nent catchment, where active mining ceased in the 1900s, high concentrations of Zn are still detected even 100 years later, causing the benthic macroinvertebrate community to remain low compared with that of unpolluted tributaries [13].

In 1 year survey, network analysis and SOM were used to quantify the area influenced by the mining activities which needs to be managed to enhance the benthic macroinvertebrate diversity. These two analytical methods can be further applied to evaluate whether the diversity of benthic macroinvertebrates will recover after the completion of mining treatment. For instance, SOM can be used to estimate the time required for the benthic macroinvertebrate community near mining area to become similar to that of the reference sites and to evaluate if its stability (or complexity) increases after the completion of mining treatment using network analysis (e.g., comparison of the number of edges and average node degree).

5. Conclusions

This study demonstrates that the benthic macroinvertebrate community was less diverse and complex in streams located near mining areas than in the reference stream, with different functions (e.g., no scrapers in the stream nearly located in a mining area) and structures (e.g., the lowest species richness). Our approaches of network analysis and SOM could provide analytical methods for quantifying the impacts of mining activities on the benthic macroinvertebrate community. We revealed the endpoints of the impacts of the mining area (e.g., 4.1 km downstream of the effluent site in NHJ) and a reduced interaction among benthic macroinvertebrates in mining-impacted areas. We showed that those two analytical methods are useful to quantify the area influenced by the mining, which should be prioritized for establishing and implementing conservation and management plans to enhance the community diversity of benthic macroinvertebrates.

Author Contributions: Conceptualization, M.-J.B.; Field survey: M.-J.B. and J.-K.H.; Writing, Review and Editing, M.-J.B., E.-J.K.; Supervision, M.-J.B. All authors have read and agreed to the published version of the manuscript.

Funding: This work was supported by a grant (NNIBR202101107) from the Nakdonggang National Institute of Biological Resources (NNIBR) funded by the Ministry of Environment (MOE), Republic of Korea, as well as by the National Research Foundation of Korea (NRF) grants funded by the Korean government (MSIT) (No. 2019R1A2C2089870).

Acknowledgments: We appreciate anonymous reviewers providing helpful and constructive comments to improve the early version of the manuscript.

Conflicts of Interest: The authors declare no conflict of interest.

References
1. Sala, O.E.; Chapin, F.S., III; Armesto, J.J.; Berlow, E.; Bloomfield, J.; Dirzo, R.; Huber-Sanwald, E.; Huenneke, L.F.; Jackson, R.B.; Kinzig, A.; et al. Global biodiversity scenarios for the year 2100. Science 2000, 287, 1770–1774. [CrossRef]
2. Dudgeon, D.; Arthington, A.H.; Gessner, M.O.; Kawabata, A.-I.; Knowler, D.J.; Lévêque, C.; Naiman, R.J.; Prieur-Richard, A.-H.; Soto, D.; Stiassny, M.L.J.; et al. Freshwater biodiversity: Importance, threats, status and conservation challenges. Biol. Rev. Camb. Philos. Soc. 2006, 81, 163–182. [CrossRef]
3. Bae, M.J.; Li, F.; Kwon, Y.S.; Chung, N.; Choi, H.; Hwang, S.J.; Park, Y.S. Concordance of diatom, macroinvertebrate and fish assemblages in streams at nested spatial scales: Implications for ecological integrity. Ecol. Indic. 2014, 47, 89–101. [CrossRef]
4. Marquès, M.J.; Martínez-Conde, E.; Rovira, J.V.; Ordóñez, S. Heavy metals pollution of aquatic ecosystems in the vicinity of a recently closed underground lead-zinc mine (Basque Country, Spain). Environ. Geol. 2001, 40, 1125–1137. [CrossRef]
5. Niyogi, D.K.; Harding, J.S.; Simon, K.S. Organic matter breakdown as a measure of stream health in New Zealand streams affected by acid mine drainage. Ecol. Indic. 2013, 24, 510–517. [CrossRef]
6. Pond, G.J.; Passmore, M.E.; Borsuk, F.A.; Reynolds, L.; Rose, C.J. Downstream effects of mountaintop coal mining: Comparing biological conditions using family-and genus-level macroinvertebrate bioassessment tools. J. North Am. Benthol. Soc. 2008, 27, 717–737. [CrossRef]
7. Bere, T.; Dalu, T.; Mwedzi, T. Detecting the impact of heavy metal contaminated sediment on benthic macroinvertebrate communities in tropical streams. Sci. Total Environ. 2016, 572, 147–156. [CrossRef]
8. Jarsjö, J.; Chalov, S.R.; Pietroń, J.; Alekseenko, A.V.; Thorslund, J. Patterns of soil contamination, erosion and river loading of metals in a gold mining region of northern Mongolia. *Reg. Environ. Chang.* **2017**, *17*, 1991–2005. [CrossRef]

9. Mol, J.H.; Oboutet, P.E. Downstream effects of erosion from small-scale gold mining on the instream habitat and fish community of a small neotropical rainforest stream. *Conserv. Biol.* **2004**, *18*, 201–214. [CrossRef]

10. Smolders, A.J.P.; Lock, R.A.C.; Van der Velde, G.; Hoyos, R.M.; Roelofs, J.G.M. Effects of mining activities on heavy metal concentrations in water, sediment, and macroinvertebrates in different reaches of the Pilcomayo River, South America. *Arch. Environ. Contam. Toxicol.* **2003**, *44*, 0314–0323. [CrossRef][PubMed]

11. Jacobsen, D. Tropical high-altitude streams. In *Tropical Stream Ecology*; Dudgeon, D., Ed.; Academic Press: San Diego, CA, USA, 2008; pp. 219–256.

12. Loayza-Muro, R.A.; Elias-Letts, R.; Marticorena-Ruiz, J.K.; Palomino, E.J.; Duivenvoorden, J.F.; Kraak, M.H.; Admiraal, W. Metal-induced shifts in benthic macroinvertebrate community composition in Andean high altitude streams. *Environ. Toxicol. Chem.* **2010**, *29*, 2761–2768. [CrossRef]

13. Armitage, P.D.; Bowes, M.J.; Vincent, H.M. Long-term changes in macroinvertebrate communities of a heavy metal polluted stream: The river Nent (Cumbria, UK) after 28 years. *River Res. Appl.* **2007**, *23*, 997–1015. [CrossRef]

14. Romero, A.; Flores, S.; Medina, R. Estudio de los metales pesados en el relave abandonado de Ticapampa. *Rev. Inst. Inv. Fisgmrng.* **2008**, *11*, 13–16.

15. Vannote, R.L.; Minshall, G.W.; Cummins, K.W.; Sedell, J.R.; Cushing, C.E. The river continuum concept. *Can. J. Fish. Aquat. Sci.* **1980**, *37*, 130–137. [CrossRef]

16. Hussain, Q.A.; Pandit, A.K. Macroinvertebrates in streams: A review of some ecological factors. *Int. J. Fish. Aquacult.* **2012**, *4*, 114–123. [CrossRef]

17. Hawkins, C.P.; Murphy, M.L.; Anderson, N.H. Effects of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. *Ecology* **1982**, *63*, 1840–1856. [CrossRef]

18. Minshall, G.W.; Cummins, K.W.; Petersen, R.C.; Gushing, C.E.; Bruns, D.A.; Sedell, J.R.; Vannote, R.L. Developments in stream ecosystem theory. *Can. J. Fish. Aquat. Sci.* **1985**, *42*, 1045–1055. [CrossRef]

19. Grubaugh, J.W.; Wallace, J.B.; Houston, E.S. Longitudinal changes of macroinvertebrate communities along an Appalachian stream continuum. *Can. J. Fish. Aquat. Sci.* **1996**, *53*, 896–909. [CrossRef]

20. Nelson, S.M.; Roline, R.A. Recovery of a stream macroinvertebrate community from mine drainage disturbance. *Hydrobiologia* **1996**, *339*, 73–84. [CrossRef]

21. Solà, C.; Burgos, M.; Plazuelo, Á.; Toja, J.; Plans, M.; Prat, N. Heavy metal bioaccumulation and macroinvertebrate community changes in a Mediterranean stream affected by acid mine drainage and an accidental spill (Guadiamar River, SW Spain). *Sci. Total Environ.* **2004**, *333*, 109–126. [CrossRef][PubMed]

22. Bataglia, M.; Hose, G.C.; Turak, E.; Warden, B. Depauperate macroinvertebrates in a mine affected stream: Clean water may be key to recovery. *Environ. Pollut.* **2005**, *138*, 132–141. [CrossRef][PubMed]

23. Simmons, J.A.; Lawrence, E.R.; Jones, T.G. Treated and untreated acid mine drainage effects on stream periphyton biomass, leaf decomposition, and macroinvertebrate diversity. *J. Freshw. Ecol.* **2005**, *20*, 413–424. [CrossRef]

24. Tripole, S.; Gonzalez, P.; Vallania, A.; Garbagnati, M.; Mallea, M. Evaluation of the impact of acid mine drainage on the chemistry and the macrobenthos in the Carolina stream (San Luis, Argentina). *Environ. Monit. Assess.* **2006**, *114*, 377–389. [CrossRef][PubMed]

25. MacCausland, A.; McTammany, M.E. The impact of episodic coal mine drainage pollution on benthic macroinvertebrates in streams in the Anthracite region of Pennsylvania. *Environ. Pollut.* **2007**, *149*, 216–226. [CrossRef][PubMed]

26. Bott, T.L.; Jackson, J.K.; McTammany, M.E.; Newbold, J.D.; Rier, S.T.; Sweeney, B.W.; Battle, J.M. Abandoned coal mine drainage and its remediation: Impacts on stream ecosystem structure and function. *Ecol. Appl.* **2012**, *22*, 2144–2163. [CrossRef]

27. Malmqvist, B.; Hoffsten, P.O. Influence of drainage from old mine deposits on benthic macroinvertebrate communities in central Swedish streams. *Water Res.* **1999**, *33*, 2415–2423. [CrossRef]

28. Lefcort, H.; Vancura, J.; Lider, E.L. Long-term changes in macroinvertebrate communities of a heavy metal polluted stream: The river Nent (Cumbria, UK) after 28 years. *River Res. Appl.* **2007**, *23*, 997–1015. [CrossRef]

29. Kiffney, P.M.; Clements, W.H. Structural responses of benthic macroinvertebrate communities from different stream orders to zinc. *Environ. Toxicol. Chem.* **1994**, *13*, 389–395. [CrossRef]

30. Gerhardt, A.; Janssens de Bisthoven, L.; Soares, A.M.V.M. Macroinvertebrate response to acid mine drainage: Community metrics and on-line behavioural toxicity bioassay. *Environ. Pollut.* **2004**, *130*, 263–274. [CrossRef]

31. Clements, W.H. Metal tolerance and predator–prey interactions in benthic macroinvertebrate stream communities. *Ecol. Appl.* **1999**, *9*, 1073–1084. [CrossRef]

32. O’Halloran, K.; Cavanagh, J.-A.; Harding, J.S. Response of a New Zealand mayfly (Deleatidium spp.) to acid mine drainage: Implications for mine remediation. *Environ. Toxicol. Chem.* **2008**, *27*, 1135–1140. [CrossRef]

33. Salinas, G.; Marin, R. Impact of sediment releases on water chemistry and macroinvertebrate communities in clear water Andean streams (Bolivia). *Arch. Hydrobiol.* **2001**, *1*, 33–50. [CrossRef]

34. Herbst, D.B.; Medhurst, R.B.; Black, N.J. Long-term effects and recovery of streams from acid mine drainage and evaluation of toxic metal threshold ranges for macroinvertebrate community reassembly. *Environ. Toxicol.* **2018**, *37*, 2575–2592. [CrossRef][PubMed]
35. Clements, W.H.; Herbst, D.B.; Hornberger, M.I.; Mebane, C.A.; Short, T.M. Long-term monitoring reveals convergent patterns of recovery from mining contamination across 4 western US watersheds. *Freshw. Sci.* 2021, 40, 407–426. [CrossRef]

36. Freeman, L.C. Centrality in social networks conceptual clarification. *Soc. Netw.* 1978, 1, 215–239. [CrossRef]

37. Bluthgen, N. Why network analysis is often disconnected from community ecology: A critique and an ecologist’s guide. *Basic Appl. Ecol.* 2010, 11, 185–195. [CrossRef]

38. Bae, M.J.; Park, Y.S. Evaluation of precipitation impacts on benthic macroinvertebrate communities at three different stream types. *Ecol. Indic.* 2019, 102, 446–456. [CrossRef]

39. Kohonen, T. *Self-Organizing Maps*; Springer: Berlin/Heidelberg, Germany, 2001.

40. Bae, M.J.; Park, Y.S. Biological early warning system based on the responses of aquatic organisms to disturbances: A review. *Sci. Total Environ.* 2014, 466, 635–649. [CrossRef][PubMed]

41. Qu, X.; Peng, W.; Liu, Y.; Zhang, M.; Ren, Z.; Wu, N.; Liu, X. Networks and ordination analyses reveal the stream community structures of fish, macroinvertebrate and benthic algae, and their responses to nutrient enrichment. *Ecol. Indic.* 2019, 101, 501–511. [CrossRef]

42. Beck, H.E.; Zimmermann, N.E.; McVicar, T.R.; Vergopolan, N.; Berg, A.; Wood, E.F. Present and future Köppen-Geiger climate classification maps at 1-km resolution. *Sci. Data* 2018, 5, 180214. [CrossRef]

43. Korea Meteorological Administration. Available online: http://www.kma.go.kr/ (accessed on 8 October 2021).

44. MIRECO. Yearbook of Mireco Statistics. Mine Reclamation Corporation, 2018. Available online: www.mireco.or.kr (accessed on 5 March 2018).

45. Quigley, M. *Invertebrates of Streams and Rivers: A Key to Identification*; Edward Arnold, Ltd.: London, UK, 1977.

46. Pennak, R.W. *Freshwater Invertebrates of the United States*; John Wiley and Sons, Inc.: New York, NY, USA, 1978.

47. Brigham, A.R.; Brigham, W.U.; Gnilka, A. *Aquatic Insects and Oligochaetes of North and South Carolina*; Midwest Aquatic Enterprises: Mahomet, IL, USA, 1982; 837p.

48. Brinkhurst, R.O. Guide to the freshwater aquatic microdrile Oligochaetes of North America. *Can. Spec. Publ. Fish. Aquat. Sci.* 1986, 84, 259.

49. Yoon, I.B. *Illustrated Encyclopedia of Fauna and Flora of Korea*; Ministry of Education: Seoul, Korea, 1988; Volume 30. (In Korean)

50. Merritt, R.W.; Cummins, K.W. *An Introduction to the Aquatic Insects of North America*; Hunt Publishing Company: Dubugue, IA, USA, 2006.

51. APHA; AWW; WPCF. *Standard Methods for the Examination of Water and Wastewater*, 21th ed.; American Public Health Association: Washington, DC, USA, 2005.

52. Febria, C.M.; Hosen, J.D.; Crump, B.C.; Palmer, M.A.; Williams, D.D. Microbial responses to changes in flow status in temporary headwater streams: A cross-system comparison. *Front. Microbiol.* 2015, 6, 522. [CrossRef][PubMed]

53. Csardi, G.; Nepusz, T. The igraph software package for complex network research. *Inter. J. Complex Syst.* 2006, 1695, 1–9.

54. R Core Team. *R: A Language and Environment for Statistical Computing; R Foundation for Statistical Computing*: Vienna, Austria, 2017; Available online: https://www.R-project.org/ (accessed on 11 December 2017).

55. Vesanto, J.; Himberg, J.; Alhoniemi, E.; Parhankangas, J. *SOM Toolbox for Matlab 5*; Technical Report A57; Neural Networks Research Centre, Helsinki University of Technology: Helsinki, Finland, 2000.

56. Céréghino, R.; Park, Y.S. Review of the self-organizing map (SOM) approach in water resources: Commentary. *Environ. Model. Softw.* 2009, 24, 945–947. [CrossRef]

57. Legendre, P.; Legendre, L.F. *Numerical Ecology*, 3rd ed.; Elsevier Science BV: Amsterdam, The Netherlands, 2012.

58. García, H.L.; González, I.M. Self-organizing map and clustering for wastewater treatment monitoring. *Eng. Appl. Artif. Intell.* 2004, 17, 215–225. [CrossRef]

59. The Mathworks, Inc. *MATLAB Version 6.1*; The Mathworks, Inc.: Natick, MA, USA, 2001.

60. Oksanen, J.; Kindt, R.; Legendre, P.; O’Hara, B.; Stevens, M.H.H.; Oksanen, M.J.; Suggests, M.A.S.S. The Vegan Package: Community Ecology Package. 2007. Available online: http://cran.r-project.org (accessed on 11 December 2017).

61. De Mendiburu, F. *Agricolae: Statistical Procedures for Agricultural Research; R Package Version*. 2014. Available online: http://cran.r-project.org/package=agricolae (accessed on 11 December 2017).

62. Garie, H.L.; McIntosh, A. Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. *Water Resour. Bull.* 1986, 22, 447–455. [CrossRef]

63. Freeman, P.L.; Schorr, M.S. Influence of watershed urbanization on fine sediment and macroinvertebrate assemblage characteristics in Tennessee Ridge and Valley Streams. *J. Freshw. Ecol.* 2004, 19, 353–362. [CrossRef]

64. Garie, H.L.; McIntosh, A. Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. *Water Resour. Bull.* 1986, 22, 447–455. [CrossRef]

65. Freeman, P.L.; Schorr, M.S. Influence of watershed urbanization on fine sediment and macroinvertebrate assemblage characteristics in Tennessee Ridge and Valley Streams. *J. Freshw. Ecol.* 2004, 19, 353–362. [CrossRef]

66. Water Environment Information System. Available online: http://water.nier.go.kr/ (accessed on 8 August 2020).
69. Pond, G.J.; Passmore, M.E.; Pointon, N.D.; Felbinger, J.K.; Walker, C.A.; Krock, K.J.; Fulton, J.B.; Nash, W.L. Long-term impacts on macroinvertebrates downstream of reclaimed mountaintop mining valley fills in central Appalachia. *Environ. Manag.* **2014**, *54*, 919–933. [CrossRef] [PubMed]

70. Zhao, Q.; Guo, F.; Zhang, Y.; Yang, Z.; Ma, S. Effects of secondary salinisation on macroinvertebrate functional traits in surface mining-contaminated streams, and recovery potential. *Sci. Total Environ.* **2018**, *640*, 1088–1097. [CrossRef]

71. Timpano, A.J.; Schoenholtz, S.H.; Soucek, D.J.; Zipper, C.E. Benthic macroinvertebrate community response to salinization in headwater streams in Appalachia USA over multiple years. *Ecol. Indic.* **2018**, *91*, 645–656. [CrossRef]

72. Timpano, A.J.; Schoenholtz, S.H.; Soucek, D.J.; Zipper, C.E. Salinity as a limiting factor for biological condition in mining-influenced central Appalachian headwater streams. *J. Am. Water Resour. Assoc.* **2015**, *51*, 240–250. [CrossRef]

73. Erasmus, J.H.; Malherbe, W.; Zimmermann, S.; Lorenz, A.W.; Nachev, M.; Wepener, V.; Sures, B.; Smit, N.J. Metal accumulation in riverine macroinvertebrates from a platinum mining region. *Sci. Total Environ.* **2020**, *703*, 134738. [CrossRef]

74. Drover, D.R.; Schoenholtz, S.H.; Soucek, D.J.; Zipper, C.E. Multiple stressors influence benthic macroinvertebrate communities in central Appalachian coalfield streams. *Hydrobiologia* **2020**, *847*, 191–205. [CrossRef]

75. Frick, K.G.; Herrmann, J. Aluminium and pH effects on sodium ion regulation in mayflies. In *The Surface Waters Acidification Program*; Mason, B.J., Ed.; Cambridge University Press: London, UK, 1990; pp. 409–412. [CrossRef]

76. Frick, K.G.; Herrmann, J. Aluminium accumulation in a lotic mayfly at low pH-A laboratory study. *Ecotox. Environ. Saf.* **1990**, *19*, 81–88. [CrossRef]

77. Clements, W.H.; Kotalik, C. Effects of major ions on natural benthic communities: An experimental assessment of the US Environmental Protection Agency aquatic life benchmark for conductivity. *Freshw. Sci.* **2016**, *35*, 126–138. [CrossRef]

78. Maret, T.R.; Cain, D.J.; MacCoy, D.E.; Short, T.M. Response of benthic invertebrate assemblages to metal exposure and bioaccumulation associated with hard-rock mining in northwestern streams, USA. *J. North Am. Benthol. Soc.* **2003**, *22*, 598–620. [CrossRef]

79. Conley, J.M.; Funk, D.H.; Cariello, N.J.; Buchwalter, D.B. Food rationing affects dietary selenium bioaccumulation and life cycle performance in the mayfly Centropilum triangulifer. *Ecotoxicology* **2011**, *20*, 1840–1851. [CrossRef]

80. Arnold, M.C.; Bier, R.L.; Lindberg, T.T.; Bernhardt, E.S.; Di Giulio, R.T. Biofilm mediated uptake of selenium in streams with mountaintop coal mine drainage. *Limnologica* **2017**, *65*, 10–13. [CrossRef]

81. Stagliano, D.M.; Whiles, M.R. Macroinvertebrate production and trophic structure in a tallgrass prairie headwater stream. *J. North Am. Benthol. Soc.* **2002**, *21*, 97–113. [CrossRef]

82. Wallace, J.B.; Webster, J.R. The role of macroinvertebrates in stream ecosystem function. *Annu. Rev. Entomol.* **1996**, *41*, 115–139. [CrossRef]

83. Reinhart, K.O.; VandeVoort, R. Effect of native and exotic leaf litter on macroinvertebrate communities and decomposition in a western Montana stream. *Divers. Distrib.* **2006**, *12*, 776–781. [CrossRef]

84. Qu, X.; Wu, N.; Tang, T.; Cai, Q.; Park, Y.S. Effects of heavy metals on benthic macroinvertebrate communities in high mountain streams. *Ann. Limnol. Int. J. Lim.* **2010**, *46*, 291–302. [CrossRef]

85. Liu, L.; Li, W.; Song, W.; Guo, M. Remediation techniques for heavy metal-contaminated soils: Principles and applicability. *Sci. Total Environ.* **2018**, *633*, 206–219. [CrossRef] [PubMed]