ELF - A benthic macroinvertebrate multi-metric index for the assessment and classification of hydrological alteration in rivers

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Abstract
Hydrological alteration prevents unpolluted rivers from reaching acceptable ecological conditions. In Europe, 40% of rivers are degraded due to hydrological alteration but the indices currently used in ecological monitoring cannot quantify the degree of hydrological alteration or discriminate between pollution and hydrological alteration. In this article, we demonstrate the development, calculation and validation of the Hellenic Flow Index (ELF), a new macroinvertebrate-based multi-metric index to assess, quantify and classify hydrological alteration in Greek streams and rivers. 1351 samples collected throughout Greece were partitioned in reference and test datasets and were pre-classified in varying levels of pollution and hydrological alteration. Optimal flow ranges for benthic macroinvertebrates were calculated and flow sensitivity metrics were developed. We tested the predictive accuracy of 607 versions of the ELF index, that is, 607 combinations of seven hydrologically sensitive macroinvertebrate metrics. The developed index is a combination of two ELF versions that had the highest predictive accuracy on two validation datasets. The index developed can assess, quantify and classify hydrological alteration and is also capable of discriminating between pollution and hydrological alteration. The ELF’s overall predictive accuracy was 75% and the index was equally accurate in discriminating between clean, polluted and hydrologically altered sites. Within the regular ecological monitoring, water agencies, managers and decision makers can now use a tool that will quickly flag watercourses that specifically need hydrological restoration instead of pollution mitigation measures and thus apply targeted actions towards the ecological restoration of rivers.

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* Both authors contributed equally to this study
1. Introduction

Hydrological alteration can be defined as any anthropogenic or climate-change induced disruption in the magnitude or timing of historically established river flows. The causes of hydrological alteration are twofold: first, dams are flow obstacles that disrupt the longitudinal continuity of rivers (Kondolf, 1997; Nilsson et al., 2005). In addition, they divert water from rivers for domestic, agricultural or industrial uses, thus reducing the amount of water available to aquatic ecosystems. Secondly, climate change, either natural or human-induced, alters flow regimes in the long term. It is anticipated that climate change will intensify dam construction, increasing the degree of flow regulation (Milly et al., 2005; Chang et al., 2018), the severity of flow intermittence and the number of temporary rivers worldwide (Larned et al., 2010). Decreased streamflow and conversion of permanent watercourses to temporary ones have already been observed, especially in many peri-Mediterranean river basins, due to extreme hydrological alteration (Skoulikidis et al., 2017).

Until the early 2010s, environmental management had focused on monitoring and mitigating the impacts of pollution on aquatic ecosystems. In the European Union, within the demands of the Water Framework Directive 2000/60/EC (WFD - European Union Council, 2000), streams and rivers have been monitored to assess their ecological status. National and European ecological indices using fish (Pont et al., 2006, 2007; Schmutz et al., 2007), benthic macroinvertebrates (AQEM Consortium, 2001; Buffagni et al., 2006; Birk et al., 2012; Lazaridou et al., 2018), diatoms (Cemagref, 1982) and aquatic macrophytes (Haury et al. 2006; Camargo, 2018) have been developed and employed to provide WFD complying frameworks for ecological monitoring that would help towards quantifying and ultimately upgrading the ecological status of European rivers. But research showed that streamflow is the major factor that shapes fundamental ecological processes (Poff and Zimmerman, 2010), and is the primary determinant of macroinvertebrate taxonomic richness and diversity in running waters (Karouzas et al., 2019). This became evident in 2012, when the first results of ecological monitoring showed that 40% of European rivers failed to reach acceptable ecological conditions due to hydrological and hydromorphological alteration (EEA, 2012). And it was further confirmed in 2018 (EEA, 2018); unpolluted rivers may not be ecologically safe due to hydrological and hydromorphological alteration.

Despite the knowledge that hydrological alteration prevents almost half of European watercourses from reaching acceptable ecological status, the degree of hydrological alteration cannot be quantified using the ecological indices that are currently in use. Most indices available worldwide cannot discriminate between pollution and hydrological alteration (Schuwirth et al., 2015; Smith et al., 2019). They can accurately quantify ecological degradation but they cannot indicate the cause of the degradation, and thus, they cannot properly guide management and restoration actions. Worldwide, regarding benthic macroinvertebrates, the efforts towards developing hydrologically sensitive ecological indices, that would inform whether a river reach is hydrologically altered or not, are very specific: (i) the LIFE index developed in the United Kingdom (Extenze et al., 1999), later amended by the DEHLI index (Chadd et al., 2017) and (ii) the CEFI index developed in Canada (Armanini et al., 2011). These ecological indices follow a specific concept: benthic macroinvertebrate taxa are scored according to their flow velocity preference; the scores of all taxa are summed, the result is often divided by the number of taxa, other metrics may also be included, and the outcome, either categorized in classes or not, is indicative of the degree of hydrological alteration of a river reach.

In this study, we developed a new macroinvertebrate-based multi-metric index, the Hellenic Flow Index (ELF), to assess, quantify and classify hydrological alteration in Greek streams and rivers. Our purpose was to develop an index that would not only quantify and classify the degree of hydrological alteration but also discriminate between polluted and hydrologically altered sites. For the classification of hydrological alteration we used the same classes with those used in the Water Framework Directive 2000/60/EC to facilitate easy interpretation of the results and enable possible comparisons with other relevant ecological indices. This paper demonstrates the process of ELF development, calculation and validation. It is our intention to provide water agencies, managers and decision makers with a tool that, within the regular, WFD-based ecological monitoring, will quickly flag those watercourses that specifically need hydrological restoration instead of pollution mitigation measures, and thus facilitate targeted efforts towards the ecological restoration of rivers.
2. Materials and methods

2.1. ELF development step 1 - Optimal flow ranges of benthic macroinvertebrates

We assembled data, relating the mean flow velocity (V), mean water depth (D), water discharge (Q) and wetted width (w) to the abundance of benthic macroinvertebrates, mostly identified to family level, from the records of the Greek Surface Water Monitoring Programme (GSWMP; first management cycle; 2012-2015, hereby called ‘the GSWMP-1215 dataset’). The GSWMP-1215 dataset covers 449 sites in Greece sampled during spring and summer from 2012 to 2015. The hydrological variables V and D were recorded using the Swoffer 2100 current velocity meter at evenly spaced points/segments along a perpendicular-to-the-flow cross section. We measured depth-averaged flow velocities at 0.6 x D when D ≤ 0.75 m, and by averaging 0.2 x D and 0.8 x D when D > 0.75 m, following the approach of Nolan and Shields (2000). The water discharge was then calculated as

\[ Q = \sum_{i=1}^{n} w_i \cdot D_i \cdot V_i \]

for the ‘i = 1 to n’ segments of the cross section (Donald and Steven, 2008).

Benthic macroinvertebrate samples were collected following a semi-quantitative 3-min kick-and-sweep method (Armitage and Hogger, 1994) with an additional 1-min sampling at the bank’s vegetation when present (Wright, 2000). Macroinvertebrates were collected using a 0.25 × 0.25 m² rectangular hand net with a mesh size of 500 μm. Samples were preserved in plastic boxes containing 70% ethanol, transferred to the laboratory and identified using macroinvertebrate identification guides for the Mediterranean region. To avoid pollution-related sources of bias, only samples of no or very minor anthropogenic influence (without upstream sources of pollution or hydrological alteration) were selected for the analysis, following relevant guidelines (MedGIG, 2012; Feio et al., 2014). Thus, from the initial pool of 1702 samples, 497 samples from 229 sites, covering 76 river basins, were finally included in the working dataset (Fig. 1).

Optimal conditions were determined between macroinvertebrate abundance and the hydrological variables for each taxon. The abundance value of each taxon was normalized to the 0-1 range by dividing by the maximum abundance of the taxon observed in the dataset. The hydrological variables included were V (m/s), D (m) and, within the concept that a specific Q will have a different effect on macroinvertebrates depending on the width of the channel, the discharge per

![Fig. 1. Location of the 229 non-impacted sites used for the development of the optimal flow ranges of benthic macroinvertebrates](image-url)
unit width \( (q=Q/w) \) \( (\text{m}^2/\text{s}) \) was used instead of \( Q \). The optimal range of each variable \( (V, D \text{ and } q \text{ against the normalized abundance (A)}) \) was calculated as follows: For the \( A \) values of each macroinvertebrate taxon, we calculated the 25th (P25), 50th (median) and 75th (P75) percentiles. All \( A \) values were afterwards converted to a binary score; \( A \) values that were below P75 were converted to 0 and all \( A \geq P75 \) were converted to 1. The minimum and maximum values of the examined hydrological variable for which the \( A \) score was 1 were calculated and were considered as the optimal range for the specific variable (where optimal actually stands for \( \geq P75 \)) (Fig. 2).

Due to the inherent limitations of exploring the univariate response of macroinvertebrates to each hydrological variable separately (Rydgren et al., 2003; Hirzel et al., 2008; Theodoropoulos et al., 2018), we used the discharge per unit width \( (q) \) to integrate information from the \( V \) and \( D \) variables. Assuming a rectangular channel, \( Q = V \cdot \text{Area} = VDw \) and thus \( Q/w = VD \)  

For each macroinvertebrate taxon, we calculated the relative optimal \( q \) range as follows:

\[
q'_{ri} = \frac{q'_i}{q'_{\text{max}}} \times 100\%
\]

where,

\( q'_{ri} \) is the relative optimal \( q \) range for the \( i \)th macroinvertebrate taxon
\( q'_i \) is the optimal \( q \) range for the \( i \)th macroinvertebrate taxon
\( q'_{\text{max}} \) is the maximum \( q \) range observed

All macroinvertebrate taxa were then categorized into seven flow sensitivity groups, based on their relative optimal \( q \)-range values, ranging from highly generalists \( (q'_{ri} > 95\%) \) to highly specialists \( (q'_{ri} < 5\%) \).

2.2. ELF development step 2 - Reference and test datasets
Macroinvertebrate-abundance taxalists -family level- from three datasets, the GSWMP-1215 and the benthos-GR dataset (Theodoropoulos et al., 2018), which were collected from 76 river basins in Greece, and the WG-0610 dataset, collected from 24 sites in western Greece (Theodoropoulos and Iliopoulou-Georgudaki, 2010; Theodoropoulos et al., 2015), were assembled and partitioned in one reference and two test datasets. Our purpose was to develop (i) a robust reference

![Fig. 2. Schematic representation of the process applied for the calculation of the optimal range of each macroinvertebrate taxon against each hydrological variable (q in this example). P75: 75th percentile; q: discharge per unit width](image-url)
dataset to be used for the establishment of optimal values and class intervals for each ELF metric, (ii) a large, index development dataset, to be used for a robust ELF development process and (iii) a smaller dataset to be used as an additional, external validation dataset. Thus, the reference dataset included 82 samples, selected from the GSWMP-1215 and the benthos-GR datasets based on a multi-factor assessment (MedGIG, 2012; Feio et al., 2014) that identified sites of no hydrological, morphological, physicochemical and land use alterations. Following a similar process, the 1st test dataset (TD1; the large, index development dataset) included 1189 samples of the GSWMP-1215 dataset, which were categorized based on the surrounding land use, the sources of pollution, hydrological alteration, and expert-judgment, into four groups: 1. No pollution, no hydrological alteration (477 samples), 2: Pollution, no hydrological alteration (638 samples), 3: No pollution, hydrological alteration (40 samples), 4: Pollution, hydrological alteration (34 samples). The 2nd test dataset (TD2; the smaller, external validation dataset) included all samples of the WG-0610 dataset (n=80) that were also a-priori categorized, using the same approach as with TD1, in levels of pollution and hydrological alteration.

The stressors-variables included in the multi-factor analysis, based on the evaluation protocol of Feio et al. (2014), were channelization, bank alteration, local habitat alteration, riparian vegetation, general morphology, connectivity, stream flow, upstream dam influence, hydropeaking, general hydrology, dissolved oxygen, oxygen saturation, N-NH4+, N-NO3-, P-Po43-, P-total, % artificial areas, % intensive agriculture, % extensive agriculture, % semi-natural areas, % urbanization, % non-natural land use, % agriculture. All expert-judgment-based assessments were applied by a team of ten members, including the authors, two additional senior researchers and four field experts of the ecological monitoring personnel, which cooperated to accurately evaluate all hydromorphological, physicochemical and land use information.

2.3. ELF development step 3 - Selection of metrics
Our perception was that the potential candidate-metrics for the ELF index should be influenced only by hydrological alteration, being insensitive to pollution. Thus, they could be able to (i) not only highlight and quantify hydrological alteration but also (ii) distinguish it from pollution. Within this concept, we initially used the ASTERICS 4.0.4 software (http://www.fliessgewaesser-bewertung.de) to calculate macroinvertebrate metrics regarding feeding and locomotion types, habitat and flow/current preference that could be potentially included in the index. Within a trial-and-error pre-processing, the Spearman’s coefficient (RS) was calculated between V, D and the macroinvertebrate metrics-traits of the reference and the hydrologically altered samples (group 3 - TD1) to identify possible hydroecological correlations. Those metrics that showed the highest correlation with V and D were finally included in the ELF development. Additional macroinvertebrate metrics were calculated based on the previously described flow sensitivity classification scheme and were also considered as potential ELF candidates. Finally, from the developed pool of candidate-metrics, we selected seven that matched the aforementioned criteria and could be further analyzed for inclusion in the ELF index.

2.4. ELF development step 4 - Development and validation of the index
Following the concept of the Water Framework Directive 2000/60/EC (WFD - European Union Council, 2000), the developmental process of the ELF index was similar to the development of the STAR-ICMi (Buffagni et al., 2004, Buffagni et al., 2006); the ELF index would be an ecological quality ratio, with values ranging from 0 to 1, quantifying the hydrological alteration within a WFD-based five-class system. The ELF development, calculation and validation process were the following:

A. Development:
1. From the 82 reference samples, we calculated the median values (M) for each of the seven metrics.
2. The boundaries for each class were afterwards defined; high/good boundary: 0.8 x M, good/moderate boundary: 0.6 x M, moderate/poor boundary: 0.4 x M, poor/bad boundary: 0.2 x M.
B. Calculation:

1. Using the class boundaries calculated from the reference dataset, the hydrological class based on each metric was calculated for each sample of the test dataset and was assigned one of the following values; bad: 0.1, poor: 0.3, moderate: 0.5, good: 0.7, high: 0.9.

2. The class values of the seven metrics were afterwards combined to a single ELF score ranging from 0 to 1 by calculating (i) the average of all class values, (ii) the minimum of all class values, (iii) the median of all class values, (iv) a majority-vote algorithm, in which the output is either the most frequently occurring class value among the metrics, or the average of the class values in case of equal occurrence and (v) a majority-vote algorithm but replaced by the minimum value in case of equal occurrence. This score was classified into a hydrological quality class based on the following class intervals; 0.00-0.20: bad, 0.21-0.40: poor, 0.41-0.60: moderate, 0.61-0.80: good, 0.81-1.00: high.

3. We tested multiple versions of the ELF index, that is, multiple combinations of the seven metrics (from a single-metric ELF to a seven-metric ELF) with various metric combination alternatives (average, minimum, median, majority-average, majority-minimum). Thus, the final ELF index was selected among 607 candidate ELFs.

C. Validation:

1. The predictive accuracy of each of the 607 ELFs was tested by calculating the percentage of correctly classified instances (CCI) for each of the four groups of TD1.

2. The index (or a combination of indices) that had the highest accuracy at all four groups was selected as the ELF index.

3. The predictive accuracy of the ELF index was further tested by calculating the percentage of CCI for the samples of TD2.

3. Results

3.1. Optimal ranges, means and maxima

Based on the optimal flow velocities (V') for each macroinvertebrate taxon (Fig. A1, A3), twenty taxa (21% of all taxa) had maximum V' > 1 m/s and mean V' > 0.5 m/s. The optimal V range for these taxa was also the highest (> 1 m/s). Forty five taxa (47%) had maximum V' between 0.5 m/s and 1 m/s and mean V' between 0.24 m/s and 0.5 m/s. Among them, Sialidae, Lepidostomatidae and Coenagrionidae had optimal V range lower than 0.5 m/s, while the V range of the rest varied between 0.5 m/s and 1 m/s. Twenty three taxa (24%) had maximum V' between 0.2 m/s and 0.5 m/s, and mean V' between 0.06 m/s and 0.24 m/s. From them, Atyidae, Euphaeidae and Haliplidae had optimal V range < 0.2 m/s, while all other taxa had optimal V range > 0.2 m/s. The maximum V' of 7 taxa (7%) (Ephydridae, Polymitarcidae, Lestidae, Libelulliidae, Valvatidae, Chloroperlidae, Culicidae) was less than 0.2 m/s and their mean V' ranged between 0.005 m/s and 0.6 m/s. These taxa also had similarly low optimal V ranges, varying from 0.01 m/s to 0.11 m/s.

Regarding the optimal water depths (D') (Fig. A2, A3), forty two taxa (44%) had maximum D' ≥ 1 m and mean D' ≥ 0.5 m. The optimal D' range for these taxa was also the highest (≥ 1 m). Twenty three taxa (40%) had maximum D' between 0.5 m and 1 m and mean D' between 0.15 m and 0.5 m. Their optimal D range was also similar (between 0.2 m and 0.5 m). Ten taxa (10%) had maximum D' between 0.25 m and 0.45 m and mean D' between 0.12 m and 0.18 m, with the exception of Planorbidae, Piscicollidae, Isonychiidae and Polymitarcidae, which had lower mean D' values. Five taxa (5%) (Culicidae, Chloroperlidae, Notonectidae, Haliplidae, Ephydridae) had maximum D' between 0.15 m and 0.2 m, and mean D' between 0.02 m and 0.095 m. While all other taxa had an optimal V range > 0.2 m/s. The optimal D range of these taxa was also minimum, varying from 0.05 m to 0.19 m.

3.2. Flow optima and levels of flow sensitivity

The optimal discharge per unit width (q'), which can be approximated by V·D for rectangular channels, ranged from 0 m³/s to 0.56 m³/s. A categorization of q based on the relevant Q and w values was applied (Fig. 3) to facilitate the information integration between q, w, V and D. It has to be noted that low q values (< 0.03 m²/s) include samples of low Q and low w but also samples of low Q and high w. Similarly, high q values (> 0.4 m²/s) include samples of high Q and high
w but also samples of moderate Q and w values. This suggests that the integration suggested, although useful to combine information between Q, w, V and D, needs to be carefully addressed by the reader through simultaneous reference to Fig. A3, Fig. 3 and Fig. 4 in order to find the optimal q, V and D values of each macroinvertebrate taxon.

**Table 1.** Flow-sensitivity classification of macroinvertebrate taxa based on their optimal preference range for discharge-per-unit-width (qr’=q’max-q’min).

<table>
<thead>
<tr>
<th>Flow sensitivity</th>
<th>q’</th>
<th>qr’</th>
<th>Macroinvertebrate taxa</th>
</tr>
</thead>
<tbody>
<tr>
<td>Highly generalists</td>
<td>&gt; 0.532</td>
<td>&gt; 95%</td>
<td>Chironomidae, Ephemerellidae, Gammaridae, Leptoceridae, Limoniidae, Potamonidae, Psychodidae</td>
</tr>
<tr>
<td>Generalists</td>
<td>0.42-0.532</td>
<td>75%-95%</td>
<td>Aeshnidae, Aelligidae, Atyidae, Ceratopogonidae, Curculionidae, Elmidae, Gyriidae, Helophoridae, Hydrobiidae, Hydrophilidae, Hydroptilidae, Lymnaeidae, Nemouridae, Oligoneuriidae, Perlidae, Physidae, Platycnemididae, Polycentropodidae, Scirtidae, Simuliidae, Stratiomyidae</td>
</tr>
<tr>
<td>Meso-generalists</td>
<td>0.28-0.42</td>
<td>50%-75%</td>
<td>Anthomyiidae, Athericidae, Baetidae, Brachycentridae, Caenidae, Cordulegasteridae, Dryopidae, Empididae, Glossiphonidae, Glossosomatidae, Heptageniidae, Hybracarina, Hydropsychidae, Lepidostomatidae, Leuctridae, Limnephilidae, Oligochaeta, Perlodidae, Philopotamidae, Psychomyiidae, Rhyacophilidae, Sericostomatidae, Tabanidae, Tipulidae</td>
</tr>
<tr>
<td>Intermediate</td>
<td>0.168-0.28</td>
<td>30%-50%</td>
<td>Ancylidae, Blephariceridae, Bythiniidae, Calopterygidae, Dixidae, Dugesiidae, Dytiscidae, Ephemeroptera, Erpobdellidae, Gerridae, Goeridae, Gomphidae, Hydraenidae, Leptophlebiidae, Lestidae, Notonectidae, Osmylidae, Ostracoda, Taenioptrerygidae, Veliidae</td>
</tr>
<tr>
<td>Meso-specialists</td>
<td>0.084-0.168</td>
<td>15%-30%</td>
<td>Aphelocheiridae, Coenagrionidae, Corduliidae, Corixidae, Dolichopodidae, Mesoveliidae, Muscidae, Palaemonidae, Rhagionidae, Sialidae</td>
</tr>
<tr>
<td>Specialists</td>
<td>0.028-0.084</td>
<td>5%-15%</td>
<td>Chloroperlidae, Euphaeidae, Hydrochidae, Hydrometridae, Isonychiidae, Piscicolidae, Planorbidae, Valvatidae</td>
</tr>
<tr>
<td>Highly specialists</td>
<td>0-0.028</td>
<td>&lt; 5%</td>
<td>Culicidae, Ephydridae, Haliliidae, Libellulidae, Polymitarcidae, Potamanthidae</td>
</tr>
</tbody>
</table>

Within the aforementioned, 14 taxa (15% of all taxa) were characterized as specialists, with their optimal q ranging from 0 to 0.168 m²/s, being < 30% of the maximum optimal range (Fig. 4, Table 1). Among them, Culicidae, Ephydridae, Haliliidae, Libellulidae, Polymitarcidae and Potamanthidae were highly specialists (q’ between 0 m²/s and 0.028 m²/s, < 5% of q’ maximum). Twenty eight taxa (29%) were characterized as generalists, with q’ between 0.42 m²/s and 0.56 m²/s (> 75% of q’ maximum). Among them, Chironomidae, Ephemerellidae, Gammaridae, Leptoceridae, Limoniidae, Potamonidae, Psychodidae were highly generalists (q’ between 0.532 m²/s and 0.56 m²/s, > 95% of q’ maximum). Twenty four taxa (25%) were characterized as meso-generalists (q’ between 0.28 m²/s and 0.42 m²/s, between 50% and 75% of q’ maximum) and 20 taxa (21%) were grouped into the intermediate category, with q’ between 0.168 m²/s and 0.28 m²/s (30% to 50% of q’ maximum).
Fig. 4. Optimal discharge-per-unit-width (q) values for the benthic macroinvertebrates recorded in the dataset. The optimal range represents the q values in which the normalized abundance (A) of each taxon was ≥ 75th percentile of all A values.
Fig. 4 (continued). Optimal discharge-per-unit-width (q) values for the benthic macroinvertebrates recorded in the dataset. The optimal range represents the q values in which the normalized abundance (A) of each taxon was ≥ 75th percentile of all A values.
3.3. The ELF index

3.3.1. Candidate metric 1: Relative abundance of highly generalist, generalist, meso-generalist and intermediate taxa

The median values and the relevant class boundaries were calculated for (i) the relative abundance (R_{A}) of highly generalists, (ii) the R_{A} of generalists, (iii) the R_{A} of meso-generalists and (iv) the R_{A} of intermediate taxa (Table 2). The metrics were afterwards combined into a single metric using the following equation:

$$CR_A = \frac{CR_{A,HG} + CR_{A,G} + CR_{A, MG} + CR_{A, I}}{4}$$

where,

CR_{A,HG} the class value (see section 2.4) of the relative abundance of the highly generalist taxa; CR_{A,G} the class value of the relative abundance of the generalist taxa; CR_{A, MG} the class value of the relative abundance of the meso-generalist taxa; CR_{A, I} the class value of the relative abundance of the intermediate taxa

<p>| Table 2. Median values and class boundaries for the first candidate metric of the ELF index. |
|---------------------------------|-----|-----|-----|-----|-----|-----|-----|-----|-----|</p>
<table>
<thead>
<tr>
<th>P/B</th>
<th>M/P</th>
<th>G/M</th>
<th>H/G</th>
<th>Median</th>
<th>H/G</th>
<th>G/M</th>
<th>M/P</th>
<th>P/B</th>
</tr>
</thead>
<tbody>
<tr>
<td>R_{A,HG}</td>
<td>0.036</td>
<td>0.072</td>
<td>0.108</td>
<td>0.144</td>
<td>0.181</td>
<td>0.217</td>
<td>0.253</td>
<td>0.289</td>
</tr>
<tr>
<td>R_{A,G}</td>
<td>0.021</td>
<td>0.042</td>
<td>0.062</td>
<td>0.082</td>
<td>0.103</td>
<td>0.123</td>
<td>0.144</td>
<td>0.164</td>
</tr>
<tr>
<td>R_{A,MG}</td>
<td>0.120</td>
<td>0.241</td>
<td>0.361</td>
<td>0.482</td>
<td>0.602</td>
<td>0.723</td>
<td>0.843</td>
<td>0.964</td>
</tr>
<tr>
<td>R_{A,I}</td>
<td>0.004</td>
<td>0.009</td>
<td>0.013</td>
<td>0.018</td>
<td>0.022</td>
<td>0.027</td>
<td>0.030</td>
<td>0.035</td>
</tr>
</tbody>
</table>

R_{A}: Relative abundance, H: High class, G: Good class, M: Moderate class, P: Poor class, B: Bad class, HG: Highly generalists, G: Generalists, MG: Meso-generalists, I: Intermediate

3.3.2. Candidate metric 2: Relative richness of highly generalist, generalist, meso-generalist and intermediate taxa

Within the same concept, the median values and the relevant class boundaries were calculated for (i) the relative richness (R_{R}) of highly generalists, (ii) the R_{R} of generalists, (iii) the R_{R} of meso-generalists and (iv) the R_{R} of intermediate taxa (Table 3). The metrics were afterwards combined into a single metric using the following equation:

$$CR_R = \frac{CR_{R,HG} + CR_{R,G} + CR_{R, MG} + CR_{R, I}}{4}$$

where,

CR_{R,HG} the class value of the relative richness of the highly generalist taxa; CR_{R,G} the class value of the relative richness of the generalist taxa; CR_{R, MG} the class value of the relative richness of the meso-generalist taxa; CR_{R, I} the class value of the relative richness of the intermediate taxa

<p>| Table 3. Median values and class boundaries for the second candidate metric of the ELF index. |
|---------------------------------|-----|-----|-----|-----|-----|-----|-----|-----|-----|</p>
<table>
<thead>
<tr>
<th>P/B</th>
<th>M/P</th>
<th>G/M</th>
<th>H/G</th>
<th>Median</th>
<th>H/G</th>
<th>G/M</th>
<th>M/P</th>
<th>P/B</th>
</tr>
</thead>
<tbody>
<tr>
<td>R_{R,HG}</td>
<td>0.032</td>
<td>0.064</td>
<td>0.096</td>
<td>0.128</td>
<td>0.160</td>
<td>0.192</td>
<td>0.224</td>
<td>0.256</td>
</tr>
<tr>
<td>R_{R,G}</td>
<td>0.043</td>
<td>0.087</td>
<td>0.130</td>
<td>0.174</td>
<td>0.217</td>
<td>0.261</td>
<td>0.304</td>
<td>0.348</td>
</tr>
<tr>
<td>R_{R,MG}</td>
<td>0.094</td>
<td>0.187</td>
<td>0.281</td>
<td>0.374</td>
<td>0.468</td>
<td>0.561</td>
<td>0.655</td>
<td>0.748</td>
</tr>
<tr>
<td>R_{R,I}</td>
<td>0.027</td>
<td>0.055</td>
<td>0.082</td>
<td>0.109</td>
<td>0.136</td>
<td>0.164</td>
<td>0.191</td>
<td>0.218</td>
</tr>
</tbody>
</table>

R_{R}: Relative richness, H: High class, G: Good class, M: Moderate class, P: Poor class, B: Bad class, HG: Highly generalists, G: Generalists, MG: Meso-generalists, I: Intermediate

3.3.3. Candidate metric 3: Degree of deviation of the mean community q preference from the optimal q preference

The optimal mean q preference (O_{q}), calculated from the 82 samples of the reference dataset, was 0.204 m/s^2. The degree of deviation from O_{q} was afterwards calculated as follows:

$$D_{oq} = \frac{|C_q - O_q|}{O_q}$$
where,
\[ C_q \] the mean community q preference at the site; \[ O_q \] the optimal mean q preference
Thus, the relevant class boundaries for the Doq metric were H/G: 0.2, G/M: 0.4, M/P: 0.6, P/B: 0.8.

3.3.4. Candidate metrics 4 and 5: Relative abundance of rheobiont-rheophilic-rheolimnophilic and limnobiont-limnophilic-limnorheophilic taxa

Based on the Spearman’s coefficient, only two groups of metrics-traits showed statistically significant and moderate correlation with either V and/or D: The relative abundance of rheobiont (RB), rheophilic (RP) and rheolimnophilic (RLP) taxa (RS: 0.138 for D-RB, 0.135 for V-RP, -0.2 for V-RLP, p<0.01) and the relative abundance of limnobiont (LB), limnophilic (LP) and limnorheophilic (LRP) taxa (RS: -0.167 for V-LP, -0.19 for V-LRP, p<0.01) (RB-RP-RLP and LB-LP-LRP attributes were assigned based on the EUROLIMPACS ‘freshwaterecology.info’ database (Schmidt-Kloiber and Hering, 2015; 2019). These metrics were included as potential candidates for the ELF index based on the following equations:

\[
F_{1A} = \frac{A_{RB} + A_{RP} + A_{RLP}}{A_{EUR}}
\]

where,
\[ A_{RB} \] the abundance of rheobiont taxa; \[ A_{RP} \] the abundance of rheophilic taxa; \[ A_{RLP} \] the abundance of rheolimnophilic taxa; \[ A_{EUR} \] the abundance of all taxa that have been assigned to one of the current preference classes of the EUROLIMPACS database

\[
F_{2A} = 1 - \frac{A_{LB} + A_{LP} + A_{LRP}}{A_{EUR}}
\]

where,
\[ A_{LB} \] the abundance of limnobiont taxa; \[ A_{LP} \] the abundance of limnophilic taxa; \[ A_{LRP} \] the abundance of limnorheophilic taxa; \[ A_{EUR} \] the abundance of all taxa that have been assigned to one of the current preference classes of the EUROLIMPACS database

The median value for the \( F_{1A} \) metric was 0.812 and the class boundaries were H/G: 0.650, G/M: 0.487, M/P: 0.325, P/B: 0.162. The median value for the \( F_{2A} \) metric was 0.96 and the class boundaries were H/G: 0.769, G/M: 0.577, M/P: 0.385, P/B: 0.192.

3.3.5. Candidate metrics 6 and 7: Relative richness of rheobiont-rheophilous-rheolimnophilous and limnobiont-limnophilous-limnorheophilous taxa

Within the concept of section 3.3.4. the relative richness of RB, RP, RLP and LB, LP, LRP taxa were also included as potential candidates for the ELF index based on the following equations:

\[
F_{1R} = \frac{r_{RB} + r_{RP} + r_{RLP}}{r_{EUR}}
\]

where,
\[ r_{RB} \] the richness of rheobiont taxa; \[ r_{RP} \] the richness of rheophilic taxa; \[ r_{RLP} \] the richness of rheolimnophilic taxa; \[ r_{EUR} \] the abundance of all taxa that have been assigned to one of the current preference classes of the EUROLIMPACS database

\[
F_{2R} = 1 - \frac{r_{LB} + r_{LP} + r_{LRP}}{r_{EUR}}
\]
where,

\( r_{LB} \) the richness of limnobiont taxa; \( r_{LP} \) the richness of limnophilic taxa; \( r_{LRP} \) the richness of limnorheophilic taxa; \( r_{EUR} \) the abundance of all taxa that have been assigned to one of the current preference classes of the EUROLIMPACS database.

The median value for the \( F1_{R} \) metric was 0.479 and the class boundaries were H/G: 0.383, G/M: 0.287, M/P: 0.192, P/B: 0.096. The median value for the \( F2_{R} \) metric was 0.8 and the class boundaries were H/G: 0.640, G/M: 0.480, M/P: 0.320, P/B: 0.160. A descriptive summary of the metrics included in the ELF index and their relevant class intervals is shown in Table A1.

3.3.6. Combination of the metrics into a single multi-metric index and ELF index validation

From the 607 versions of the ELF index denoting the multiple combinations of the abovementioned seven metrics (from a single-metric ELF to a seven-metric ELF) with various metric combination alternatives (average, minimum, median, majority-average, majority-minimum), two versions showed increased performance over others: (i) the ELF version that included the \( CR_{A} \), \( F1_{A} \), \( F2_{A} \), \( F1_{R} \), \( F2_{R} \) metrics, combined by selecting the minimum value of the metrics included (ELF-MIN-118) and (ii) the ELF version that included the \( CR_{A} \), \( Doq \), \( F1_{A} \), \( F2_{A} \), \( F2_{R} \), combined using a majority vote rule, and deriving the average only when all ‘votes’ were different (ELF-MAJAV-111). These two ELFs had varying accuracy in discriminating between clean, polluted, clean/hydrologically altered and polluted/hydrologically altered sites (Fig. 5). ELF-MIN-118 was very accurate in discriminating between clean and hydrologically altered sites. This means that ELF-MIN-118 could accurately classify a clean site as clean (75% accuracy) and a hydrologically altered site as hydrologically altered (85% accuracy). However, it failed to discriminate between pollution and hydrological alteration, classifying a polluted but not hydrologically altered site as hydrologically altered. ELF-MAJAV-111 was also accurate in classifying a clean site as clean (91%), had low accuracy in classifying hydrologically altered sites as hydrologically altered (25%) but was able to discriminate between pollution and hydrological degradation, classifying 67% of polluted sites as not hydrologically altered and 64% of polluted/hydrologically altered sites as hydrologically altered. Thus, the ELF index was a combination of the two versions based on the prior knowledge (or not) on the pollution status of a site, and according to the scheme depicted in Fig. 6.

![Fig. 5. Predictive accuracy of the two most accurate ELF versions, calculated as the percentage of correctly classified instances for four pre-defined categories of sites; clean (n=477), polluted (n=638), clean/hydrologically altered (n=40), polluted/hydrologically altered (n=34)](image)

![Fig. 6. Schematic representation of the ELF-application process, based on the prior knowledge of the pollution status of a site and depending on the predictive accuracy of the two most accurate versions of ELF](image)
The predictive accuracy of ELF-MAJAV-111, ELF-MIN-118 and the combined ELF index was additionally tested against a dataset of 80 macroinvertebrate samples collected from 24 sites in western Greece from 2006 to 2010 (Fig. 7, 8). The lowest accuracy of the two ELF versions ranged from 10.6% (ELF-MAJAV-111; clean/hydrologically altered) to 47.8% (ELF-MIN-118; polluted). However, the lowest accuracy of the combined ELF index was 53.4% (ELF; polluted/hydrologically altered). Overall, ELF ranked first, with 75% accurate predictions, ELF-MIN-118 was 70% accurate and ELF-MAJAV-111 was 66.5% accurate. It has to be noted that, although the ELF-MAJAV-111 and ELF-MIN-118 often misclassified clean/hydrologically altered and polluted samples, respectively, the ELF index correctly classified most instances; the median values lay within the correct categories with low percentile variance (Fig. 8).

**Fig. 7.** Predictive accuracy of the ELF-MIN-118, ELF-MAJAV-111 and the combined ELF index, on the two test datasets (TD1 and TD2) of 1189 and 80 samples, respectively, with a-priori known (pre-defined) varying levels of pollution and hydrological alteration.

**Fig. 8.** Medians, 25th and 75th percentiles calculated from the classification of samples of the test datasets 1 and 2 using the ELF-MIN-118, ELF-MAJAV-111 and the combined ELF index. Green color indicates correct average classification, red color indicates average misclassification. The y-axis denotes the hydrological status class; 0.9: high, 0.7: good, 0.5: moderate, 0.3: poor, 0.1: bad. 1: Clean sites, 2: Polluted sites, 3: Clean/hydrologically altered, 4: Polluted/hydrologically altered.
4. Discussion

4.1. The ELF index system

The ELF index can quantify and classify hydrological alteration. Quantification is based on the deviation-from-reference-conditions concept (Gordon et al., 2004); index scores vary from 0 to 1 and are classified into five classes of hydrological status, ranging from high to bad, in accordance with other ecological indices (Armitage et al., 1983; Buffagni et al., 2004; Lazaridou et al., 2018) and following the demands of the Water Framework Directive 2000/60/EC (European Commission, 2000). The ELF index can also discriminate between pollution and hydrological alteration. ELF is a dual-index system; a multi-metric index (ELF-MIN-118) that can accurately identify hydrological alteration in unpolluted sites (clean: TD1: 75%; TD2: 65%, clean/hydrologically altered: TD1: 84%; TD2: 85%) and a multi-metric index (ELF-MAJAV-111) that is fairly accurate in identifying hydrological alteration in polluted sites (polluted: TD1: 67%; TD2: 96%, polluted/hydrologically altered: TD1: 64%; TD2: 55%) are combined based on the prior knowledge of the pollution status of a site to increase the accuracy of the final ELF index. Therefore, the ELF index system requires two entries: (i) the macroinvertebrate taxalist of a site and (ii) the pollution status of a site (1: Unpolluted, 2: Polluted, 3: Unknown).

4.2. What to expect from an ecological index of hydrological alteration

Potential ELF users should interpret the ELF predictions in the following concept:

A. Within prior knowledge of the pollution status of a site:

1. There is a 65%-75% certainty that the hydrological status of a clean site, without pollution or hydrological alteration, will be correctly classified as good or high.
2. There is a 67%-96% certainty that the hydrological status of a polluted but not hydrologically altered site, will be correctly classified as good or high.
3. There is a 79%-85% certainty, that the hydrological status of a clean but hydrologically altered site, will be correctly classified as moderate, poor or bad.
4. There is a 53%-64% certainty, that the hydrological status of a polluted and hydrologically altered site, will be correctly classified as moderate, poor or bad.
5. Overall, there is a 73%-75% certainty that all sites in a dataset will be categorized by ELF in the classes of hydrological status that they actually are.

B. Without prior knowledge of the pollution status of a site, the predictive accuracy matches that of ELF-MAJAV-111 and the degrees of certainty are (1) 91%, (2) 67%-96%, (3) 11%-25%, (4) 53%-64% and (5) 62%-66%, respectively. In this case, ELF will probably fail to correctly classify a clean but hydrologically altered site, but will be fairly accurate in discriminating and correctly classifying polluted/hydrologically altered sites and polluted/not hydrologically altered sites. Although we suggest the use of ELF-MAJAV-111, the user could also apply the ELF-MIN-118, compare both predictions and further investigate whether the hydrological status of the site is degraded or not, by additionally looking at other non-ecological properties of the site, e.g. surrounding land use, presence of dams in the upstream etc.

ELF in case B (no prior knowledge of the pollution status of a site) would serve as an initial alarm for further investigation, in contrast to case A (a-priori knowledge of the pollution status), in which a moderate, poor or bad status prediction of ELF should trigger hydrological restoration actions, such as environmental flows delivery, catchment-scale land use changes, periodical reduction of irrigational activities, creation of buffer strips (Stanford et al., 1996; Ayres et al., 2014) or the potential addition of artificial structures (wood logs) to facilitate increased and longer-lasting water pools during dry periods (Hafs et al., 2014).

4.3. On the ecological response to hydrological alteration. When is a site hydrologically altered?

Aquatic communities, and the benthic macroinvertebrates in particular, have developed morphological and behavioral adaptations to withstand certain degrees of pollution or hydrological alteration (resistance mechanisms) and recolonize
formerly hostile aquatic habitats after the re-establishment of suitable conditions (resilience mechanisms) (Bogan et al., 2017). On one hand, this means that a river reach may be characterized by experts as hydrologically altered due to the presence of a dam in the upstream, but the degree of hydrological alteration may not be significant enough to surpass the resistance mechanisms of macroinvertebrates and thus induce ecological response. Moreover, tributaries that often merge with the main stem downstream of the dam may compensate for the hydrological alteration caused by the dam. Thus, the presence of a dam or water abstractions in the upstream, does not necessarily imply macroinvertebrate community degradation in the downstream. On the other hand, a river reach may be free from dams or water abstractions, but seasonal flow variation, i.e. prolonged low flows, probably due to a long no-rainfall period, or within a climate-change induced prolonged dry period, may trigger macroinvertebrate response.

The aforementioned were observed during the ELF index development. Spring samples of sites that were affected by upstream dams were not assessed as hydrologically altered, probably due to the increased, rainfall-induced water availability. In contrast, the status of summer samples in sites without dams or water abstractions was classified as moderate, poor or bad, probably due to reduced water levels resulting from a prolonged, summer low-flow period. Finally, hydrological alteration does not necessarily mean ‘flows lower than natural’ but also ‘flows higher than natural’. This kind of hydrological alteration can be also identified by the metrics used in the ELF index, and thus, after a flood or a heavy rainfall event, the ELF index will probably indicate hydrological alteration. Consequently, a seasonal application of the ELF index, within the regular ecological monitoring, may be necessary to incorporate seasonal/interannual hydrological variation, and provide the user with a sequence of hydrological statuses, from which a robust decision will be reached on the human-induced part of the hydrological alteration.

4.4. On the predictive accuracy of ecological indices
The response of aquatic communities to hydrological variation is characterized by high intra- and inter-annual variability (Poff et al., 2010; Fukuda et al., 2013). Benthic macroinvertebrates respond to a multitude of interacting environmental factors that vary over seasons and years (Jowett et al., 1991; Leclerc et al., 2003). Literature indicates that the predictive accuracy of ecological models-indices varies roughly between 50% and 75% (Van Broekhoven et al., 2006; Mouton et al., 2009, Mouton et al., 2011; Vezza et al., 2015; Theodoropoulos et al., 2018), suggesting that there are inherent limitations to what can be achieved. Reflecting the above, the ELF’s overall predictive accuracy was 73%-75%, ranging from 64% to 85% for the first test dataset and from 53% to 96% for the second. In addition, these performance metrics should be cautiously interpreted, within the concept that the pre-defined groups based on which they were calculated, had been developed using expert-judgment. This means that a site that was pre-classified as hydrologically altered due to the presence of a dam in the upstream, may actually not have been hydrologically altered, e.g. due to a former period of increased rainfall. The ELF index would probably correctly classify it as good or high (not altered) but the performance of this classification would be zero since the site was pre-classified as altered.

4.5. Do we need two types of ecological indices?
Considering that the direct measurement of stressors is resource demanding, easy-to-assess stressor-specific ecological indices that could be applied within the regular ecological monitoring process, would be of much help to water managers (Schuworth et al., 2015). However, few stressor-specific indices have been developed so far (Hilsenhoff, 1988; Rolauffs et al., 2004); most ecological metrics/indices currently in use, can only indicate general degradation. Moreover, there is uncertainty in all model/index predictions (Barry and Elith, 2006; Loga et al., 2017; Lazaridou et al., 2018) due the highly variable ecological response that they attempt to approximate. The stressor-specific ELF index, in combination with an already existing index of general degradation (HESY2 - Lazaridou et al., 2018 for the Greek watercourses), could be used to either confirm a good/high status prediction of a general-degradation index (thus reducing its uncertainty), or flag sites that need hydrological restoration and guide targeted actions for ecological restoration as follows:

1. HESY2 status: good/high; ELF status: good/high -> No action
2. HESY2 status: good/high; ELF status: moderate/poor/bad -> Hydrological restoration
5. Conclusions and future work

We developed a macroinvertebrate index that (i) assesses, quantifies and classifies hydrological alteration and (ii) discriminates between hydrological alteration and pollution. Being stressor specific, the ELF index will accurately classify the status of a clean but hydrologically altered site as moderate, poor or bad, and the status of a polluted but not hydrologically altered site as good or high. The predictive performance of the ELF index ensures fairly accurate ecological assessments. As ELF may be influenced by seasonal/interannual hydrological variation, a seasonal application of the index, within the regular ecological monitoring process, is considered essential to facilitate robust decision-making on a site’s hydrological status. Moreover, the ELF development should not be perceived as an endpoint but rather as a process of continuous update towards an increasingly accurate ELF. Class boundaries may be further adjusted or new hydrology-sensitive and pollution-insensitive ecological metrics may be introduced, as the ELF index is further validated on new reference and test datasets from Greece and around the world. Considering that macroinvertebrate taxa are similar across countries and continents, ELF can be locally adjusted by using country-specific reference datasets, expanding its applicability to other countries that need an effective ecological tool to quantify hydrological alteration in streams and rivers.
References


Schmidt-Kloiber A., Hering D. (Eds.). www.freshwaterecology.info - the taxa and autecology database for freshwater organisms, version 7.0 (accessed on 01.06.2019).


The optimal range represents the flow velocity values in which the normalized abundance ($A$) of each taxon was $\geq 75$th percentile of all $A$ values.

**Fig. A1.** Optimal flow velocity values for the benthic macroinvertebrates recorded in the dataset.

<table>
<thead>
<tr>
<th>Taxon</th>
<th>Flow velocity (m/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ephemeroptera</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Limonidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Potamonidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Psychomyidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Heleophoridae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Elmidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Heptagenidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Oligoneuriidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Perlidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Nemouridae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Baetidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Empididae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Stenonemidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Hydropsychidae</td>
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</tr>
<tr>
<td>Rhyacophilidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Sericostomatidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Tipulidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Perlonidae</td>
<td>0.1 - 1.5</td>
</tr>
<tr>
<td>Chironomidae</td>
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</tr>
<tr>
<td>Hydracarina</td>
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</tr>
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<td>Polycentropodidae</td>
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<tr>
<td>Athericidae</td>
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<tr>
<td>Hydropsyllidae</td>
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<td>Simulidae</td>
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<td>Limnephilidae</td>
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<tr>
<td>Ceratopogonidae</td>
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<tr>
<td>Psychodidae</td>
<td>0.1 - 1.5</td>
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<tr>
<td>Erpobdellidae</td>
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<td>Sialis</td>
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<td>Gyriidae</td>
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<td>Leuctridae</td>
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<td>Potamanthidae</td>
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<td>Tabanidae</td>
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<td>Aeshnidae</td>
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<td>Notonectidae</td>
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<td>Ephemereida</td>
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<td>Ocypodidae</td>
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<tr>
<td>Brachycentridae</td>
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<tr>
<td>Scirtidae</td>
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</table>
Optimal flow velocity values for the benthic macroinvertebrates recorded in the dataset. The optimal range represents the flow velocity values in which the normalized abundance (A) of each taxon was ≥ 75th percentile of all A values.
Fig. A2. Optimal water depth values for the benthic macroinvertebrates recorded in the dataset. The optimal range represents the depth values in which the normalized abundance (A) of each taxon was \( \geq 75\text{th} \) percentile of all A values.
Fig. A2 (continued). Optimal water depth values for the benthic macroinvertebrates recorded in the dataset. The optimal range represents the depth values in which the normalized abundance (A) of each taxon was ≥ 75th percentile of all A values.
Fig. A3. Two-dimensional plot of the mean optimal water depth values (D) against the mean optimal flow velocity values (V) of the benthic macroinvertebrates recorded in the dataset.
Table A1. Description of the metrics included in the ELF index and their relevant class intervals. Rₐ: Relative abundance, Rᵦ: Relative richness, ¹Included in the ELF-MIN-118, ²Included in the ELF-MAJAV-111

<table>
<thead>
<tr>
<th>ELF Metric</th>
<th>Metrics</th>
<th>Bad</th>
<th>Poor</th>
<th>Moderate</th>
<th>Good</th>
<th>High</th>
<th>Good</th>
<th>Moderate</th>
<th>Poor</th>
<th>Bad</th>
</tr>
</thead>
<tbody>
<tr>
<td>CRₐ²</td>
<td>Rₐ of highly generalist taxa</td>
<td>≤ 0.036</td>
<td>(0.036, 0.072)</td>
<td>(0.072, 0.108)</td>
<td>(0.108, 0.144)</td>
<td>(0.144, 0.217)</td>
<td>(0.217, 0.253)</td>
<td>(0.253, 0.289)</td>
<td>(0.289, 0.325)</td>
<td>&gt; 0.325</td>
</tr>
<tr>
<td></td>
<td>Rₐ of generalist taxa</td>
<td>≤ 0.021</td>
<td>(0.021, 0.042)</td>
<td>(0.042, 0.062)</td>
<td>(0.062, 0.082)</td>
<td>(0.082, 0.123)</td>
<td>(0.123, 0.144)</td>
<td>(0.144, 0.164)</td>
<td>(0.164, 0.185)</td>
<td>&gt; 0.185</td>
</tr>
<tr>
<td></td>
<td>Rₐ of meso-generalist taxa</td>
<td>≤ 0.120</td>
<td>(0.120, 0.241)</td>
<td>(0.241, 0.361)</td>
<td>(0.361, 0.482)</td>
<td>(0.482, 0.723)</td>
<td>(0.723, 0.843)</td>
<td>(0.843, 0.964)</td>
<td>(0.964, 0.999)</td>
<td>&gt; 0.999</td>
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<tr>
<td></td>
<td>Rₐ of intermediate taxa</td>
<td>≤ 0.004</td>
<td>(0.004, 0.009)</td>
<td>(0.009, 0.013)</td>
<td>(0.013, 0.018)</td>
<td>(0.018, 0.027)</td>
<td>(0.027, 0.030)</td>
<td>(0.030, 0.035)</td>
<td>(0.035, 0.040)</td>
<td>&gt; 0.040</td>
</tr>
<tr>
<td>CRᵦ²</td>
<td>Rᵦ of highly generalist taxa</td>
<td>≤ 0.032</td>
<td>(0.032, 0.064)</td>
<td>(0.064, 0.096)</td>
<td>(0.096, 0.128)</td>
<td>(0.128, 0.192)</td>
<td>(0.192, 0.224)</td>
<td>(0.224, 0.256)</td>
<td>(0.256, 0.288)</td>
<td>&gt; 0.288</td>
</tr>
<tr>
<td></td>
<td>Rᵦ of generalist taxa</td>
<td>≤ 0.043</td>
<td>(0.043, 0.087)</td>
<td>(0.087, 0.130)</td>
<td>(0.130, 0.174)</td>
<td>(0.174, 0.261)</td>
<td>(0.261, 0.304)</td>
<td>(0.304, 0.348)</td>
<td>(0.348, 0.391)</td>
<td>&gt; 0.391</td>
</tr>
<tr>
<td></td>
<td>Rᵦ of meso-generalist taxa</td>
<td>≤ 0.094</td>
<td>(0.094, 0.187)</td>
<td>(0.187, 0.281)</td>
<td>(0.281, 0.374)</td>
<td>(0.374, 0.561)</td>
<td>(0.561, 0.655)</td>
<td>(0.655, 0.748)</td>
<td>(0.748, 0.842)</td>
<td>&gt; 0.842</td>
</tr>
<tr>
<td></td>
<td>Rᵦ of intermediate taxa</td>
<td>≤ 0.027</td>
<td>(0.027, 0.055)</td>
<td>(0.055, 0.082)</td>
<td>(0.082, 0.109)</td>
<td>(0.109, 0.164)</td>
<td>(0.164, 0.191)</td>
<td>(0.191, 0.218)</td>
<td>(0.218, 0.245)</td>
<td>&gt; 0.245</td>
</tr>
<tr>
<td>Doq²</td>
<td>Degree of deviation from the optimal discharge-per-unit-width preference</td>
<td>[0, 0.200]</td>
<td>(0.200, 0.400)</td>
<td>(0.400, 0.600)</td>
<td>(0.600, 0.800)</td>
<td>&gt; 0.800</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

F₁a²,²       | Rₐ of rheobiont, rheophilous and rheolimnophilous taxa                   | ≤ 0.162      | (0.162, 0.325) | (0.325, 0.487) | (0.487, 0.650) | (0.650, 1.000) |
F₂a²,²       | 1 - Rₐ of limnobiont, limnophilous and limnorheophilous taxa            | ≤ 0.192      | (0.192, 0.385) | (0.385, 0.577) | (0.577, 0.769) | (0.769, 1.000) |
F₁h¹         | Rᵦ of rheobiont, rheophilous and rheolimnophilous taxa                  | ≤ 0.096      | (0.096, 0.192) | (0.192, 0.287) | (0.287, 0.383) | (0.383, 1.000) |
F₂h¹,²       | 1 - Rᵦ of limnobiont, limnophilous and limnorheophilous taxa           | ≤ 0.160      | (0.160, 0.320) | (0.320, 0.480) | (0.480, 0.640) | (0.640, 1.000) |

Rₐ: Relative abundance, Rᵦ: Relative richness, ¹Included in the ELF-MIN-118, ²Included in the ELF-MAJAV-111

The ELF index calculation table