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1    **Using invertebrate functional traits to improve flow variability assessment within European**  
2    **rivers**

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## 25    **Abstract**

26    Rivers are among the most threatened ecosystems worldwide and are experiencing rapid  
27    biodiversity loss. Flow alteration due to climate change, water abstractions and augmentation is a  
28    severe stressor on many aquatic communities. Macroinvertebrates are widely used for  
29    biomonitoring river ecosystems although current taxonomic approaches used to characterize  
30    ecological responses to flow have limitations in terms of generalisation across biogeographical  
31    regions. A new macroinvertebrate trait-based index, Flow-T, derived from ecological functional  
32    information (flow velocity preferences) currently available for almost 500 invertebrate taxa at the  
33    European scale is presented. The index was tested using data from rivers spanning different  
34    biogeographic and hydro-climatic regions from the UK, Cyprus and Italy. The performance of  
35    Flow-T at different spatial scales and its relationship with an established UK flow assessment tool,  
36    the Lotic-invertebrate Index for Flow Evaluation (LIFE), was assessed to determine the  
37    transferability of the approach internationally. Flow-T was strongly correlated with the LIFE index  
38    using both presence-absence and abundance weighted data from all study areas ( $r$  varying from 0.46  
39    to 0.96). When applied at the river reach scale, Flow-T was effective in identifying communities  
40    associated with distinct mesohabitats characterised by their hydraulic characteristics (e.g., pools,  
41    riffles, glides). Flow-T can be derived using both presence/absence and abundance data and can be  
42    easily adapted to varying taxonomic resolutions. The trait-based approach facilitates research using  
43    the entire European invertebrate fauna and can potentially be applied in regions where information  
44    on taxa-specific flow velocity preferences is not currently available. The inter-regional and  
45    continental scale transferability of Flow-T may help water resource managers gauge the effects of  
46    changes in flow regime on instream communities at varying spatial scales.

47

48    **KEYWORDS:** bioassessment, flow velocity preference, traits theory, river ecosystems,  
49    hydrological alteration, functional ecology

## 50    **1. Introduction**

51    Rivers are among the most threatened ecosystems worldwide (Tickner et al., 2020; Albert et al.,  
52    2021) with their flow regimes (discharge) widely altered, and their longitudinal and lateral  
53    connectivity fragmented due to weirs, impoundments and floodplain development (Grill et al.,  
54    2019; Belletti et al., 2020). The natural flow regime acts as major driver in fluvial ecosystems  
55    (Rolls et al., 2018) with hydrological variability and alteration having wide ranging effects  
56    instream, as well as on aquatic–terrestrial linkages (Tockner et al., 2010). The predicted increase in  
57    extreme flow events frequency and intensity for numerous areas worldwide (Prudhomme et al.,  
58    2014; Chang et al., 2018), coupled with the increasing demands of water abstraction for  
59    anthropogenic activities, are likely to exacerbate the effects of flow regime alteration (Tonkin et al.,  
60    2018; Dudgeon, 2019).

61    Streamflow affects many abiotic characteristics, from water temperature and streambed stability to  
62    channel and floodplain morphology (see Richter et al., 1997 and reference inside). Riverine aquatic  
63    communities display a range of life history traits and morphological adaptations that enable  
64    individuals to persist and maintain populations within lotic ecosystems (Statzner and Holm, 1989;  
65    Mims and Olden, 2012). However anthropogenic alteration of flow regimes (e.g., longitudinal  
66    impoundments, abstraction and augmentation) may constrain biological communities (Krajenbrink  
67    et al., 2019; Mellado-Díaz et al., 2019; White et al., 2021) and prevent unpolluted or otherwise  
68    relatively pristine rivers from achieving their maximum ecological capacity (Theodoropoulos et al.,  
69    2020). Although the importance of hydromorphological alterations on rivers and their flow regime  
70    are widely recognised, tools to address this stressor have rarely been integrated in biomonitoring  
71    policies or environmental legislations (Poikane et al., 2020; Theodoropoulos et al., 2021).

72    Macroinvertebrate communities are crucial components of aquatic and riparian ecosystems (Suter  
73    and Cormier, 2015) that are frequently used for biomonitoring of rivers worldwide (Buss et al.,  
74    2015), and have been shown to be sensitive to flow regime variability and alteration (Wood et al.,

2000; Belmar et al., 2019; Doretto et al., 2019). Since the late Twentieth Century, an increasing number of biological indices have been developed to characterise the flow preferences of lotic macroinvertebrates. For example, the Rheo-index (Banning, 1990, 1998) was one of the first to characterise faunal associations with river flow velocity and is one of the core metrics for evaluating the quality status of rivers in Germany (Buchner et al., 2019). Similarly, Timm et al. (2011) proposed an index (MESH), able to assess hydromorphological quality of Estonian surface waters (lakes and rivers) focussing on macroinvertebrate taxonomic composition. Theodoropoulos et al. (2020) proposed a dual-index system, called ELF (Hellenic Flow Index), to assess the effects of hydrological alterations upon Greek streams based on optimal discharge conditions for macroinvertebrates, utilising prior knowledge of the pollution status of the site being considered.

One of the most widely tested macroinvertebrate-based indices for characterising faunal river flow preferences is the Lotic Invertebrate Index for Flow Evaluation (LIFE - Extence et al., 1999). The index was developed using data from UK rivers and was specifically designed to reflect faunal associations with flow velocity and its variability via a scoring system. This index and its derivatives are currently used in countries spanning 3 continents and multiple countries including the UK (e.g., White et al., 2021), Canada (e.g., Armanini et al., 2011), New Zealand (e.g., Greenwood et al., 2016) and Costa Rica (e.g., Quesada-Alvarado et al., 2020). However, all these indices are based on locally derived scores for each macroinvertebrate taxon, typically recorded at family or species level, according to their flow velocity preferences. In general, the determinations of velocity-preference were derived from an information-driven expert panel process, rather than empirical and contemporary measures of State-Pressure-Response. The benefit of this approach is a mechanistic independence of output. There is, however, an inbuilt element of subjectivity. Given natural biogeographical constraints, most historic approaches are often limited in terms of the geographical application to the areas where they were originally developed (e.g., lack of specific taxonomic information, scores not available for local endemics or families unique to a specific

100 region/ country, or presence of new / alien taxa). Therefore, there is the need to increase the  
101 potential transferability and applicability of tools to monitor and characterise hydrological  
102 variability and stress where it occurs (Friberg et al., 2011; Alonso et al., 2013; Poff, 2018).

103 Ecological functional information used to characterise floral and faunal communities has received  
104 increasing attention in recent years, as it potentially overcomes the constraints imposed by  
105 traditional taxonomic approaches (Mouillot et al., 2013; Kunz et al., 2021). Within freshwater  
106 ecosystems, a wide range of faunal traits (e.g., body size, voltinisms, feeding habits, life-history  
107 strategies) have been assigned to macroinvertebrates. These traits are commonly used in ecological  
108 and ecohydrological research to assess the effects of flow alteration on macroinvertebrate  
109 communities, providing complementary information to traditional taxonomic approaches  
110 (Guareschi et al., 2014; Ruhi et al., 2018; but also see limitations in Hamilton et al., 2020). As the  
111 combination of species traits determines their likelihood and ability to withstand disturbance events,  
112 such as flow alteration, a non-random taxa selection is expected along environmental gradients and  
113 associated with specific disturbances (Mouillot et al., 2013).

114 This study aimed to develop a new aquatic macroinvertebrate trait-based index, called Flow-T,  
115 derived from the “current velocity” trait proposed by Tachet et al. (2010) in a comprehensive  
116 international trait dataset encompassing nearly 500 macroinvertebrate taxa. We compared the  
117 performance of the newly developed index with those of an already established index, LIFE, in  
118 different biogeographic and climatic regions and using both family and mixed-level resolution (e.g.  
119 species, genera and families within the same dataset). Flow-T was calculated using: i) the original  
120 UK macroinvertebrate dataset used to develop and test the LIFE index; ii) Mediterranean rivers  
121 from Cyprus focussed on riffle and pool habitats and iii) Italian rivers encompassing 5 different  
122 habitats characterised by different flow velocity characteristics. Flow-T was expected to perform  
123 better than the existing index (LIFE) when applied in geographical areas where local / region  
124 specific flow-sensitivity indexes are currently lacking because of increased taxa coverage. In

125 addition, it overcomes the inherent issue of applying flow-sensitivity taxa lists outside regions they  
126 were developed in. Specifically, we predicted similar responses of both indices in the UK and  
127 better performances of Flow-T over LIFE in Italy and Cyprus because the first includes more taxa  
128 than the second. We also predicted better performance of mixed-level taxonomic resolution  
129 compared to family data because not all flow velocity preferences are preserved or represented at  
130 the family level.

131

## 132 **2. Materials and Methods**

### 133 *2.1 Biological and trait datasets*

134 LIFE is a macroinvertebrate-based index for the assessment of flow conditions within streams and  
135 rivers (Extence et al., 1999). It was developed in the UK by the Environment Agency of England  
136 (EA) using historic macroinvertebrate data from 1986 to 1997. The core dataset comprised of multi-  
137 habitat 3-minute kick samples from three chalk streams: River Lark and Waithe Beck (Anglian  
138 region), the R. Kennet (Thames region) and two Midlands rivers: R. Derwent and R. Wreake (detailed  
139 information regarding all sampling methods and study areas is provided in S1). In this study  
140 macroinvertebrate community data from the original sites used by Extence et al. (1999) were  
141 employed to derive the values of LIFE and Flow-T using the original dataset from 1986 to 1997 (for  
142 a total of 133 samples) and an extended dataset from the same rivers spanning 1986 to 2019 (285  
143 samples). The taxonomic resolution was mostly at species level except for some groups (e.g.,  
144 Diptera). Briefly, the LIFE index is calculated as follows. first, each taxon (both family and species  
145 level) is assigned to one of the 6-flow groups (from I: Rapid flow velocity to VI: Drought resistant).  
146 Flow sensitivity scores are then assigned based on a two-entry table where the entries reflect the flow  
147 group and the abundance category of individual taxa on a log-scale. For each sample, the final score  
148 is derived by dividing the sum of individual taxon flow scores by the number of scored taxa. Higher  
149 flow velocities typically result in communities characterised by higher LIFE scores.

150 The second dataset used to compare the LIFE methodology and Flow-T was from the eastern  
151 Mediterranean island of Cyprus. Macroinvertebrates were sampled quantitatively from distinct  
152 habitats (using a Surber sampler) and identified to family level according to the national  
153 biomonitoring method (Water Development Department, 2006). Samples were collected from both  
154 riffle and pool habitats, where the latter included both instream pools and glides (see below for  
155 mesohabitat nomenclature). The Cypriot dataset comprised a total of 872 samples from 77 rivers and  
156 143 sites, spanning the period from 2005 to 2018.

157 LIFE and Flow-T were also calculated using a dataset derived from three Italian braided rivers within  
158 the Po River basin (R. Taro, Trebbia and Enza within the Emilia Romagna Region, Northern Italy).  
159 The three rivers were sampled once during spring and summer 2020, providing a total of 180 samples  
160 (60 macroinvertebrate samples per-river). River reaches of 300 m in length for the Enza River and of  
161 ~1000 m for the wider channels of the Trebbia and Taro Rivers were used and mesohabitats (riffle,  
162 pool, isolated pond, backwater and glide) identified before the collection of macroinvertebrate  
163 samples according to Belletti et al. (2017). Samples were collected from each mesohabitat type  
164 identified trying to encompass the entire range of flow velocity/depth available throughout the reach.  
165 At each sampling point, mean flow velocity and water depth and the mesohabitat type was recorded.

166 The Tachet et al. (2010) database (hereafter TAC) represents the most widely used trait resource for  
167 European freshwaters. It has been successfully applied in previous research at both national and  
168 continental scales (Mellado-Díaz et al., 2008; Bonada and Dolédec, 2017; Laini et al., 2019) and  
169 has been integrated into the online European dataset *freshwaterecology.info* (Schmidt-Kloiber and  
170 Hering, 2015). This later database includes 22 ecological and biological traits available for the  
171 majority of European freshwater taxa and was developed using information gathered from around  
172 6,000 sources (Bonada and Dolédec, 2017). Information regarding flow velocity preferences for  
173 macroinvertebrate taxa is available within the ecological trait “current velocity” which is divided in  
174 4 modalities (lentic, slow, moderate and fast) where each taxon affinity is coded using a fuzzy



175 coding from 0 (no affinity) to 3 (strong affinity). Information is available mostly at genus level, but  
176 species-level (when genera are monospecific, e.g., *Ancylus fluviatilis*), subfamily-level (e.g., family  
177 Chironomidae, for which information at subfamily level is reported) and family-level (e.g.,  
178 Tipulidae and Tubificidae) are also present.

179

## 180 2.2. Flow-T calculation

181 Flow-T is calculated using the current velocity preference categories for each sample using:

$$182 \quad Flow - T = \frac{\sum_i^n \log(A_i + 1) \times (m_i + f_i)}{\sum_i^n \log(A_i + 1)} \times 100$$

183 Where n represents the number of taxa in a sample,  $A_i$  the abundance of the  $i^{th}$  taxon,  $m_i$  and  $f_i$  the  
184 “moderate” and “fast” velocity preference classes of the  $i^{th}$  taxon according to Tachet et al. (2010).  
185 The logarithm of the abundance can be replaced by 1 to obtain the presence-absence version of the  
186 Flow-T index (both tested in the present study). Hereafter, Flow- $T_{abu}$  and Flow- $T_{pa}$  will refer to Flow-  
187 T based on abundance and presence-absence data, respectively. Furthermore, two other versions of  
188 the Flow-T index were calculated for both presence-absence and abundance data (a reduced and a  
189 weighted version respectively). The first version excludes Chironomidae and Oligochaeta from the  
190 calculation because of the absence of a clear association for them at the family level with flow velocity  
191 (due to the large number of species). The second version assigns more weight to the taxa that are low  
192 flow specialists (velocity classes “lentic” and “slow”) or high flow specialists (velocity classes  
193 “moderate” and “fast”). Weights were calculated using the trait specialisation index (TSI) developed  
194 by Mondy and Usseglio-Polatera (2014). TSI calculations were undertaken using two aggregated  
195 velocity classes, the first being the sum of the lentic and slow velocity classes and the second the  
196 moderate and fast velocity classes. Complete outputs for these two latter approaches are presented in  
197 Appendix S2. A summary of the different versions of the Flow-T index used in this research is  
198 presented in Table 1. Flow-T can be calculated with a mixed taxonomic resolution (e.g., family, genus

199 and species resolution in the same dataset) and family level, where family level traits are obtained by  
200 averaging traits at a finer taxonomic level. Trait categories were converted to percentages of affinity  
201 prior to the Flow-T and TSI calculation.

202

### 203 2.3 Data analysis

204 For the UK dataset, the association between LIFE and Flow-T was tested using Pearson  
205 correlation coefficients for family and species level data. Species level LIFE was compared to Flow-  
206 T at the mixed taxonomic level. To investigate how the choice of family level scores may influence  
207 the correlation between Flow-T and LIFE indices, taxa within each family were randomly sampled  
208 and taken as representative of the family scores (e.g., *Ecdyonurus* for Heptageniidae in the first run,  
209 *Heptagenia* in the second). Two different values of both Flow-T<sub>pa</sub> and Flow-T<sub>abu</sub> were thus obtained:  
210 i) considering the mean genus values as a proxy for the family value and ii) considering the random  
211 selection of finer taxonomic levels within a family (e.g., genus, subfamily for some dipterans). This  
212 later process was repeated 1,000 times and the median and the percentile confidence interval (2.5% -  
213 97.5%) calculated.

214 Flow-T and LIFE indices for Cyprus rivers were calculated at the family level. Linear mixed  
215 effect models were used to test the association between both indices and mesohabitat as a fixed effect  
216 and sampling site as a random effect. Indices values were centred and standardized prior to the  
217 analysis to allow comparison of the results. Models with and without the mesohabitat were compared  
218 and the best model selected according to deviance information criterion - DIC (Spiegelhalter et al.,  
219 2002). The greater the difference between the DIC of two competing models the greater the support  
220 for the best model (which is the one with the lowest DIC). Like the Akaike Information Criterion,  
221 DIC differences < 7, between 7 and 14 and > 14 provide plausible, equivocal and implausible support  
222 for the model with larger DIC (Burnham et al., 2011).

223 For Italian data, the LIFE index was calculated at the family level and Flow-T derived at both  
224 the family, the taxonomic resolution used for biomonitoring purposes at the national level (Buffagni  
225 and Erba, 2007), and at a mixed taxonomic level (family and genus). Linear mixed effect models  
226 were used to regress both indices in association with mesohabitat, flow velocity and their interaction  
227 as fixed effects. For each index, two different error structures were used to account for spatial  
228 autocorrelation and the best model selected according to Ventrucchi et al. (2020) (method description  
229 and full results are presented in S3). Models including an interaction and site as the random effect  
230 were compared to additive models to select the best performing model based on the DIC. In addition,  
231 models considering “site” as a fixed effect were derived to validate the results obtained with the linear  
232 mixed model approach because the estimation of the random effect could be affected by the low  
233 number of sites used ( $n = 3$ ). According to Zuur et al. (2009) we used the random effect approach in  
234 the final analysis because it focuses on the variables of interest (flow velocity values and  
235 mesohabitat). Similarly, we tested the response of LIFE and Flow-T to discriminate between the five  
236 mesohabitats, without including flow velocity in the model and to discriminate riffles from pools, by  
237 joining pools and glides.

238 Marginal and conditional  $R^2_m$  and  $R^2_c$  values were calculated according to Nakagawa and Schielzeth  
239 (2013) for each model for the Cypriot and Italian datasets.  $R^2_m$  represents the variance explained by  
240 the fixed effects, while  $R^2_c$  represents the variance explained by both the fixed and random effects. A  
241 summary of the analysis performed for each dataset is given in Table 2.

242 All models were fitted using a Bayesian approach using the package INLA (Rue et al., 2009) for the  
243 R statistical computing software (R Core Team, 2020). We selected a Bayesian approach because of  
244 its ability to model complex error structure and to provide estimates of uncertainty for all the  
245 parameters involved (e.g.,  $R^2$ ). Models were also computed with a ‘frequentist’ approach and the  
246 results presented in appendix S4. The package biomonitoR

(<https://github.com/alexology/biomonitorR>) was used to calculate both LIFE and Flow-T indices, while the package ggplot2 (Wickham, 2009) was used for plotting the results.

### 3. Results

#### 3.1 Dataset 1, UK: Flow-T and LIFE relationship

A total of 406,576 organisms belonging to 99 families and 411 taxa were identified. All families with a LIFE score also were allocated a score for Flow-T. Flow-T<sub>pa</sub> and Flow-T<sub>abu</sub> displayed strong correlations with family-level LIFE of 0.86 and 0.96 (Fig. 1,  $p < 0.001$ ) for the original dataset and 0.80 and 0.95 for the extended dataset ( $p < 0.001$ ). The random selection of taxa for the family level Flow-T resulted in a median correlation of 0.93 (0.87 – 0.96) for Flow-T<sub>abu</sub> and of 0.80 (0.62 – 0.89) for Flow-T<sub>pa</sub>. The reduced version of Flow-T behaved similarly to the full version while the weighted version displayed lower correlation coefficients (Table 3, appendix S5). The correlation between species level LIFE and Flow-T derived using the mixed taxonomic level of presence and log-abundance data were 0.82 and 0.92 respectively (appendix S6 for full results).

#### 3.2 Dataset 2, Cyprus: Comparison and performance at different spatial scale

A total of 624,958 organisms, belonging to 103 families and 105 taxa were recorded. The number of families allocated a score was 70 (68%) for LIFE and 97 (94%) for Flow-T. Most of the mismatches for the LIFE index were due to absence of specific dipteran families, some of which are not present in the UK, or excluded from LIFE owing to a lack of necessary knowledge (e.g., Athericidae, Blephariceridae and Chironomidae) and oligochaete families, and included rarely occurring families (e.g., Euphaeidae, Bibionidae and Melanopsidae). LIFE displayed a strong correlation with Flow-T<sub>abu</sub> ( $r = 0.72$ ) but not with Flow-T<sub>pa</sub> ( $r = 0.46$ ). The correlation between both versions of Flow-T (abundance-based and presence-absence) was 0.80. The random selection of taxa for family level

Flow-T resulted in a median correlation of 0.55 (0.17 – 0.82) for Flow-T<sub>abu</sub> and of 0.30 (-0.01 – 0.64) for Flow-T<sub>pa</sub> with the family-level LIFE score.

LIFE, Flow-T<sub>pa</sub> and Flow-T<sub>abu</sub> scores based on abundance data were higher for riffles than for pools (Fig. 2). The difference between riffles and pools was most marked when using both Flow-T approaches. The R<sup>2</sup> explained by mesohabitat was low for the LIFE index (median 0.06, Bayesian credible intervals = 0.04 – 0.08) but was higher for Flow-T<sub>pa</sub> (0.17, 0.14 – 0.19) and Flow-T<sub>abu</sub> (0.15, 0.12 – 0.17), respectively. Similarly, the R<sup>2</sup> explained for both mesohabitat and random effects was 0.70 (0.65 – 0.74), 0.66 (0.62 – 0.70) and 0.71 (0.67 – 0.75) for LIFE, Flow-T<sub>pa</sub> and Flow-T<sub>abu</sub>, respectively.

280

### 3.3 Dataset 3, Italy: Comparison and performance at different spatial scale

A total of 32,426 organisms from 64 families and 106 taxa were identified within the Italian dataset. The number of families allocated a score was 45 (70%) for LIFE, with 64 (100%) and 106 (100%) allocated for the family and mixed taxonomy-level Flow-T, respectively. The reduced number of taxa used in the LIFE index was due to the absence of specific dipterans and oligochaetes and because of the absence of the family Oligoneuriidae in the UK. The most frequently occurring mesohabitat was riffle (n = 59), followed by glide (48), pool (35), isolated pool (22) and backwater (16). Median flow velocity was 0.10 ms<sup>-1</sup> and ranged from 0 ms<sup>-1</sup> to 1.68 ms<sup>-1</sup>.

Flow-T<sub>pa</sub> and Flow-T<sub>abu</sub> displayed a correlation with family-level LIFE of 0.73 and 0.90 for family level data and a correlation of 0.79 and 0.82 for mixed- level taxonomy. The random selection of taxa for the family level Flow-T resulted in a median correlation of 0.81 (0.60 – 0.91) for Flow-T<sub>abu</sub> and 0.48 (-0.28 – 0.83) for Flow-T<sub>pa</sub>.

LIFE and both versions of Flow-T were able to differentiate the two main mesohabitats in rivers displaying significantly higher values in riffles than in pools (details in appendix S7) with the same

295 response for both Flow-T approaches. Similarly, LIFE, Flow-T<sub>abu</sub> and flow-T<sub>pa</sub> displayed consistent  
296 patterns when compared to flow velocity and reflected the lentic-lotic gradient between the 5  
297 mesohabitats (from isolated ponds to riffles, Fig. 3). There was a clearer distinction between glides  
298 and the slow flowing and lentic habitats (pools, backwaters, isolated ponds) for both Flow-T  
299 approaches compared to LIFE scores (appendix S8).

300 Models considering the interaction between flow velocity and mesohabitat type performed better than  
301 the additive model for all the combinations of indices and taxonomic levels tested based on the  $\Delta$  DIC  
302 (Table 4). Indices based on abundance data (LIFE, Flow-T<sub>abu</sub>) obtained greater  $\Delta$  DIC than Flow-T<sub>pa</sub>  
303 at the family level suggesting greater differences in the relationship between flow velocity and the  
304 indices among different mesohabitats. In contrast, Flow-T based on presence-absence data displayed  
305 a higher  $\Delta$  DIC compared to the abundance-based Flow-T when considering the mixed taxonomic  
306 level. Indices calculated at the mixed taxonomic level yielded greater  $\Delta$ DIC values compared to those  
307 calculated at family level. Moreover, mixed-level indices generally performed better than those  
308 calculated at the family level, as indicated by the non- or marginally overlapping credible  $R^2_m$   
309 intervals.

310

#### 311 **4. Discussion**

312 To predict the effect of flow variability and alteration on biodiversity and ecosystem functioning is  
313 a major goal for the management of lotic ecosystems (Ruhi et al., 2018; Tonkin et al., 2018). The  
314 development of indices that can be applied across multiple geographic regions or globally would be  
315 ideal (Poff, 2018), but challenging because of the high number of macroinvertebrate taxa involved  
316 and the limited information regarding their flow velocity preferences in some geographical regions.  
317 To address this issue we developed an index, Flow-T, based on an extensive dataset of trait flow  
318 velocity-preferences proposed by Tachet et al. (2010) for European freshwater fauna. Our analyses  
319 demonstrate that the new index can successfully track the effect of flow variation at the both micro

320 and meso-scale and that it displayed consistent responses in different geographical regions (England  
321 - oceanic, Cyprus - hot-summer Mediterranean and Italy – humid sub-tropical) using both presence-  
322 absence and abundance data.

323

## 324 **LIFE and Flow-T comparison**

325 We compared and assessed the new trait-based index for riverine macroinvertebrate communities  
326 Flow-T with the LIFE index, an established flow assessment tool using 3 different datasets. Both  
327 versions of the newly developed Flow-T index (abundance based and presence-absence) displayed  
328 consistently high and statistically significant correlations with the LIFE index calculated on the  
329 original dataset used to establish the LIFE methodology (UK dataset 1986-1997) and on datasets  
330 from different European climatic regions. Stronger correlations were recorded for the abundance-  
331 based version of Flow-T (Flow-T<sub>abu</sub>) compared with the method based on presence only data (Flow-  
332 T<sub>pa</sub>). The reduced versions of Flow-T (excluding Chironomidae and Oligochaeta) displayed similar  
333 responses to versions of Flow-T using all data, while the weighted version (that assigns more  
334 weight to specialist taxa) generally performed less well.

335 When focussing on datasets from Italy and Cyprus from which samples had been collected from  
336 specific mesohabitats, all three indices were able to discriminate between riffle and pool  
337 mesohabitats consistently. However, Flow-T<sub>pa</sub> and Flow-T<sub>abu</sub> demonstrated a clearer differentiation  
338 compared to LIFE, based on the  $R^2_m$  values for the more arid Mediterranean rivers in Cyprus. The  
339 indices performed similarly well when analysing a greater gradient of habitats within the Italian  
340 dataset (from lentic to lotic). However, both Flow-T approaches demonstrated a greater sensitivity  
341 in differentiating between mesohabitats, especially glides and pools, indicating its ability to inform  
342 local scale assessments and its potential application within widely applied meso-scale habitat  
343 models (e.g., Parasiewicz et al., 2013; Vezza et al., 2017). Within the Italian dataset, the significant  
344 interaction between flow velocity and mesohabitat indicated that the association between current

345 speed and faunal communities can vary within individual mesohabitat types. For instance, areas  
346 with low flow velocity within riffles are more likely to be colonized via passive drift from nearby  
347 areas upstream with higher flow velocity and may display higher index values than low flow  
348 velocity areas of lentic mesohabitats. The results therefore support the documented interactive  
349 effects between river flow conditions and local environmental factors (e.g., substratum composition,  
350 temperature, dissolved oxygen) on riverine faunal communities (White et al., 2017; Karaouzas et  
351 al., 2019).

352 At family level, the better performance of Flow-T compared with LIFE in detecting the effect of  
353 meso- and micro-scale flow variable probably reflects the occurrence of taxa that were not  
354 originally considered in the LIFE methodology (e.g., Athericidae, Chironomidae and Oligochaeta  
355 among others, for the Mediterranean dataset, but also Oligoneuriidae - a mayfly family widely  
356 recorded throughout Central and South Europe) and other taxonomic issues (e.g., differences in  
357 taxonomic resolution and classification). For example, the number of families receiving a score, and  
358 so considered in the final index, was different for LIFE and Flow-T, especially within the Cyprus  
359 dataset (70 for LIFE and 97 for Flow-T). This may also partially explain the weaker correlations  
360 recorded between LIFE and Flow-T (based on presence/absence data) for the Mediterranean  
361 dataset, although the use of abundance data appeared to largely mitigate this issue.

362 The taxonomic resolution considered affected the strength of association of Flow-T for both LIFE  
363 and mesohabitat/flow velocity. The taxonomic resolution affected the correlation between LIFE and  
364 Flow-T and was sensitive to the random selection of taxa within families, especially for the Cyprus  
365 and Italian datasets and for Flow-T<sub>pa</sub>. Sensitivity to random selection of taxa could reflect the  
366 relatively poor representation of some eastern European taxa in the TAC database (Usseglio-  
367 Polatera et al., 2000b). For the Italian dataset, the mixed taxonomic resolution version of Flow-T  
368 (family and genus level) performed better than the family level data, being particularly sensitive  
369 when using the presence data for both flow velocity and distinct mesohabitats types. The improved



370 performance of the mixed taxonomy dataset probably reflects among-genus variability of flow  
371 velocity preferences within some macroinvertebrate multi-genus families such as the mayfly larvae  
372 Baetidae and Heptageniidae, the black fly larvae Simuliidae and the caddisfly larvae Limnephilidae  
373 (Dolédec et al., 2007). The increased sensitivity captured when using the mixed taxonomic  
374 resolution may be important when deriving indices where there may be low overall taxa richness or  
375 when using presence only data. Our results suggest that Flow-T performs best when using that  
376 finest taxonomic resolution available on either presence-absence or abundance data. The optimal  
377 taxonomic resolution should be the one that matches the TAC database, where currently velocity  
378 preferences are reported primarily at genus level.

379

#### 380 **On the wider application of macroinvertebrates flow preferences**

381 A wider geographical application of Flow-T would require flow velocity preferences being fixed  
382 (non-labile) rather than labile traits. Labile traits respond directly to the environment through local  
383 adaptation or phenotypic plasticity, leading some to question their geographical generality. For  
384 example, Bonada and Dolédec (2017) reported that some genera within the TAC dataset have a  
385 greater prevalence of multivoltine life cycles in Mediterranean basins and of univoltine/semivoltine  
386 life cycles in Scandinavia because of major differences in thermal regimes. In contrast, non-labile  
387 traits are considered fixed in space and time i.e., they are not malleable within the local  
388 environmental setting or over the lifetime of an individual, and are usually constrained by  
389 phylogeny (Poff et al., 2006). Examples of non-labile traits include respiration and locomotion type.  
390 In the European TAC dataset, traits have been assigned using fuzzy coding, which allows  
391 interpolation of trait variability and trait changes alongside the life history (ontogeny) for some  
392 taxonomic groups (e.g., feeding habits of Coleoptera larvae and adults) (Richoux 1994).  
393 Ecological flow preferences (Usseglio-Polatera et al., 2000a) do not appear to be particularly labile  
394 in the current study. The flow velocity preference information summarized in Flow-T does not

395 behave as a labile functional trait, at least over space (displaying stable patterns across multiple  
396 European climatic regions). In addition, a high degree of consistency in taxon hydraulic preferences  
397 have been identified for benthic macroinvertebrates (e.g., Belmar et al., 2013), but some exceptions  
398 at specific taxonomic levels have also been recognised (Dolédec et al., 2007). An individual taxon  
399 can then have a flow velocity optimum/preference (i.e., high association with a specific flow) but  
400 that in itself is not a labile/non-labile trait, instead that optima can be influenced by other biological  
401 features including body shape, size and mode of respiration or locomotion; representing a mixture  
402 of both labile and non-labile features (see Chessman, 2018). Similarly, according to Verberk et al.  
403 (2013) the preference of a taxon is not a trait or attribute in itself, but rather the result of how a trait  
404 has interacted with environmental conditions. The term “flow velocity preference” has been also  
405 discussed in ecohydrology suggesting that it may only be intended as “association” (Lancaster and  
406 Downes, 2010). Indeed, the latter was also the approach and terminology used within the original  
407 LIFE index (Extence et al., 1999).

408

#### 409 **Novelty and wider applications**

410 Our results indicate that using the original LIFE index within different climatic regions results in a  
411 loss of predictive power in relation to flow variation and mesohabitat types because of the reduced  
412 number of taxa that have been assigned a score. The assignment of scores to these taxa would  
413 potentially address this issue and would be crucial for the wider applicability and improved  
414 performance of the LIFE index beyond the UK. This research also clearly demonstrates how  
415 improving the taxonomic resolution from family level to a mixed taxonomic resolution (family and  
416 genus) improved the response of Flow-T to both flow velocity and mesohabitat type. Flow-T  
417 represents a potentially attractive ecological functional biomonitoring tool because it: i) facilitates  
418 trans-national comparisons in flow ecology associations as it is based on a pre-existing western-  
419 European taxa trait dataset ; ii) has been shown to perform effectively across a gradient of riverine

mesohabitats based on flow characteristics (particularly riffles and pools); iii) it is relatively easy to derive using abundance and presence/absence data; iv) it can be used when LIFE scores for specific families are not considered in the original metric, but are present at sites being studied; v) its range from 0 - 1 (or 0 - 100 if needed) means that it is very easy to interpret. Flow-T is based on a freely available and regularly updated dataset that has become the gold standard for functional analysis of macroinvertebrate communities in Europe. The same approach used in this research could be applied to other biological/ecological traits, leading the way towards the development of a suite of pressure specific indices. Moreover, unlike many taxonomic-based indices, Flow-T can readily integrate new fauna within its calculation by using faunal functional information available for almost 500 invertebrate taxa in the TAC database. This is particularly pertinent considering that colonisation and spread of new alien species seems likely in the future and is increasingly linked with the modification of the riverine flow regime (e.g., Comte et al., 2021). Including species traits in current ecological metrics to achieve broader spatial transferability is still a challenge in environmental flow management (Poff, 2018). Flow-T overcomes this challenge by relying on an extensive database of flow velocity traits and identifying the direction of a mechanistic understanding of flow-ecology relationships, linking hydraulic and hydrological alterations to variation in macroinvertebrate communities. Potential applications of the newly developed index include both routine monitoring and predictive modelling across multiple spatial scales. Flow-T will thus help guide managers in evaluating the effects of flow alterations consistently and for the first time at a truly international scale.

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453

#### 454 **Author Contributions**

455 AL conceived the idea and designed the methodology; AL, IT, RC provided the raw data; AL and  
456 MV analysed the data; AL and SG led the writing of the manuscript. All authors contributed  
457 critically to the drafts and gave final approval for publication.

458

#### 459 **Data availability statement**

460 Data are available in the supplementary material.

461

#### 462 **Declaration of competing interest**

463 The authors have no conflicts of interest to report.

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## 466 REFERENCES

- 467 Albert, J.S., Destouni, G., Duke-Sylvester, S.M., Magurran, A.E., Oberdorff, T., Reis, R.E.,  
 468 Winemiller, K.O., Ripple, W.J., 2021. Scientists' warning to humanity on the freshwater biodiversity  
 469 crisis. *Ambio* 50, 85–94. <https://doi.org/10.1007/s13280-020-01318-8>
- 470 Alonso, C., Arvoviita, J., Baattrup-Pedersen, A., Belletti, B., Brabec, K., Bogestrand, J., Perez, M.C.,  
 471 Dudley, B., Ecke, F., Friberg, N., Gothe, E., Green, S., Gunn, I.D.M., Hajek, O., Hendriks, D.M.D.,  
 472 Jones, J.I., Kairo, K., Kalivodova, M., Komprdova, K., Kohut, L., Kraml, J., Laize, C., Larsen, S.E.,  
 473 Lorenz, A., Lebiezinski, K., Mader, H., Mayr, P., Murphy, J.F., MacDonald, C., Nemethova, S.,  
 474 Noble, R.A., O'Hare, M., Raapysjarvi, J., Segersten, J., Turunen, J., Verdonschot, P., Vink, J., 2013.  
 475 Impacts of hydromorphological degradation and disturbed sediment dynamics on ecological status.  
 476 Deliverable 3.1 of REFORM (REstoring rivers FOR effective catchment Management), a  
 477 collaborative project (large-scale integrating project) funded by the European Commission within the  
 478 7th Framework Programme under Grant Agreement 282656.
- 479 Armanini, D.G., Horrigan, N., Monk, W.A., Peters, D.L., Baird, D.J., 2011. Development of a benthic  
 480 macroinvertebrate flow sensitivity index for Canadian rivers. *River. Res. Appl.* 27 (6), 723–737.  
 481 <https://doi.org/10.1002/rra.1389>
- 482 Banning, M. 1990. Der Rheo-Index—eine Möglichkeit zur Berechnung der Auswirkungen des  
 483 Flußstaus auf die benthische Lebensgemeinschaft. *Erweiterte Zusammenfassungen der Jahrestagung*  
 484 *der DGL* 186–190.
- 485 Banning, M., 1998. Auswirkungen des Aufstaus größerer Flüsse auf das Makrozoobenthos:  
 486 dargestellt am Beispiel der Donau. *Essener ökologische Schriften*, 9, Westarp-Wiss,  
 487 Hohenwarsleben.
- 488 Belletti, B., Garcia de Leaniz, C., Jones, J., Bizzi, S., Börger, L., Segura, G., Castelletti, A., van de  
 489 Bund, W., Aarestrup, K., Barry, J., Belka, K., Berkhuysen, A., Birnie-Gauvin, K., Bussetini, M.,  
 490 Carolli, M., Consuegra, S., Dopico, E., Feierfeil, T., Fernández, S., Garrido, F., Garcia-Vazquez, E.,  
 491 Garrido, S., Giannico, G., Gough, P., Jepsen, N., Jones, P.E., Kemp, P., Kerr, J., King, J., Łapińska,

492 M., Lázaro, G., Lucas, L.M., Martin, P., McGinitty, P., O’Hanley, J., Olivo del Amo, R., Parasiewicz,  
 493 P., Pusch, M., Rincon, G., Rodriguez, C., Royte, J., Schneider, C.T., Tummers, J.S., Vallesi, S.,  
 494 Vowles, A., Verspoor, E., Wanningen, H., Wantzen, K.M., Wildman, L., Zalewski, M., 2020. More  
 495 than one million barriers fragment Europe’s rivers. *Nature* 588, 436–441.  
 496 <https://doi.org/10.1038/s41586-020-3005-2>  
 497 Belletti, B., Rinaldi, M., Bussetini, M., Comiti, F., Gurnell, A.M., Mao, L., Nardi, L., Vezza, P.,  
 498 2017. Characterising physical habitats and fluvial hydromorphology: A new system for the survey  
 499 and classification of river geomorphic units. *Geomorphology* 283, 143–157.  
 500 <https://doi.org/10.1016/j.geomorph.2017.01.032>  
 501 Belmar, O., Velasco, J., Gutiérrez-Cánovas, C., Mellado-Díaz, A., Millán, A., Wood, P.J., 2013. The  
 502 influence of natural flow regimes on macroinvertebrate assemblages in a semiarid Mediterranean  
 503 basin. *Ecohydrology* 6 (3), 363–379. <https://doi.org/10.1002/eco.1274>  
 504 Belmar, O., Bruno, D., Guareschi, S., Mellado-Díaz, A., Millán, A., Velasco, J., 2019. Functional  
 505 responses of aquatic macroinvertebrates to flow regulation are shaped by natural flow intermittence  
 506 in Mediterranean streams. *Freshw. Biol.* 64 (5), 1064–1077. <https://doi.org/10.1111/fwb.13289>  
 507 Bonada, N., Dolédec, S., 2017. Does the Tachet trait database report voltinism variability of aquatic  
 508 insects between Mediterranean and Scandinavian regions? *Aquat. Sci.* 80, 7.  
 509 <https://doi.org/10.1007/s00027-017-0554-z>  
 510 Buchner, D., Beermann, A.J., Laini, A., Rolauffs, P., Vitecek, S., Hering, D., Leese, F., 2019.  
 511 Analysis of 13,312 benthic invertebrate samples from German streams reveals minor deviations in  
 512 ecological status class between abundance and presence/absence data. *PLoS ONE* 14, e0226547.  
 513 <https://doi.org/10.1371/journal.pone.0226547>  
 514 Buffagni, A., Erba, S., 2007. Parte A - Metodo di campionamento per i fiumi guadabili, Istituto di  
 515 Ricerca sulle Acque del Consiglio Nazionale delle Ricerche. Roma, Italy.

516 Burnham, K.P., Anderson, D.R., Huyvaert, K.P., 2011. AIC model selection and multimodel  
 517 inference in behavioral ecology: some background, observations, and comparisons. *Behav. Ecol.*  
 518 *Sociobiol.* 65, 23-35. <https://doi.org/10.1007/s00265-010-1029-6>  
 519 Buss, D.F., Carlisle, D.M., Chon, T.S., Culp, J., Harding, J.S., Keizer-Vlek, H.E., Robinson, W.A.,  
 520 Strachan, S., Thirion, C., Hughes, R.M., 2015. Stream biomonitoring using macroinvertebrates  
 521 around the globe: a comparison of large-scale programs. *Environ. Monit. Assess.* 187, 4132.  
 522 <https://doi.org/10.1007/s10661-014-4132-8>  
 523 Chang, S., Graham, W., Geurink, J., Wanakule, N., Asefa, T., 2018. Evaluation of impacts of future  
 524 climate change and water use scenarios on regional hydrology. *Hydrol. Earth. Syst. Sci.* 22, 4793–  
 525 4813. <https://doi.org/10.5194/hess-22-4793-2018>  
 526 Chessman, B.C., 2018. Dissolved-oxygen, current and temperature preferences of stream  
 527 invertebrates estimated from field distributions: application to assemblage responses to drought.  
 528 *Hydrobiologia* 809, 141–153. <https://doi.org/10.1007/s10750-017-3455-1>  
 529 Comte, L., Grantham, T., Ruhi, A., 2021. Human stabilization of river flows is linked with fish  
 530 invasions across the USA. *Glob. Ecol. Biogeogr.* 30 (3), 725–737. <https://doi.org/10.1111/geb.13258>  
 531 Dolédec, S., Lamouroux, N., Fuchs, U., Mérioux, S., 2007. Modelling the hydraulic preferences of  
 532 benthic macroinvertebrates in small European streams. *Freshw. Biol.* 52 (1), 145–164.  
 533 <https://doi.org/10.1111/j.1365-2427.2006.01663.x>  
 534 Doretto, A., Bo, T., Bona, F., Apostolo, M., Bonetto, D., Fenoglio, S., 2019. Effectiveness of artificial  
 535 floods for benthic community recovery after sediment flushing from a dam. *Environ. Monit. Assess.*  
 536 191, 88. <https://doi.org/10.1007/s10661-019-7232-7>  
 537 Dudgeon, D., 2019. Multiple threats imperil freshwater biodiversity in the Anthropocene. *Curr. Biol.*  
 538 29 (19), R960–R967. <https://doi.org/10.1016/j.cub.2019.08.002>  
 539 Extence, C.A., Balbi, D.M., Chadd, R.P. (1999) River flow indexing using British benthic  
 540 macroinvertebrates: a framework for setting hydroecological objectives. *Regul. Rivers Res. Manage.*

541 15 (6), 545–574. [https://doi.org/10.1002/\(SICI\)1099-1646\(199911/12\)15:6<545::AID-](https://doi.org/10.1002/(SICI)1099-1646(199911/12)15:6<545::AID-)

542 [RRR561>3.0.CO;2-W](https://doi.org/10.1002/(SICI)1099-1646(199911/12)15:6<545::AID-RRR561>3.0.CO;2-W)

543 Friberg, N., Bonada, N., Bradley, D.C., Dunbar, M.J., Edwards, F.K., Grey, J., Hayes, R.B., Hildrew,  
544 A.G., Lamouroux, N., Trimmer, M., Woodward, G., 2011. Adv. Ecol. Res. 44, 1–68.  
545 <https://doi.org/10.1016/B978-0-12-374794-5.00001-8>

546 Greenwood, M.J., Booker, D.J., Smith, B.J., Winterbourn, M.J., 2016. A hydrologically sensitive  
547 invertebrate community index for New Zealand rivers. Ecol. Indic. 61 (2), 1000–1010.  
548 <https://doi.org/10.1016/j.ecolind.2015.10.058>

549 Grill G, Lehner B, Thieme M, Geenen, B., Tickner, F., Antonelli, F., Babu, S., Borrelli, P., Cheng,  
550 L., Crochetiere, H., Ehalt Macedo, H., Filgueiras, R., Goichot, M., Higgins, J., Hogan, Z., Lip, B.,  
551 McClaini, M.E., Meng, J., Mulligan, M., Nilsson, C., Olden, J.D., Opperman, J.J., Petry, P., Reidy  
552 Liermann, C., Sáenz, L., Salinas-Rodríguez, S., Schelle, P., Schmitt, R.J.P., Snider, J., Tan, F.,  
553 Tockner, K., Valdujo, P.H., van Soesbergen, A., Zarfl, C., 2019. Mapping the world’s free-flowing  
554 rivers. Nature 569, 215–221. <https://doi.org/10.1038/s41586-019-1111-9>

555 Guareschi, S., Laini, A., Racchetti, E., Bo, T., Fenoglio, S., Bartoli, M., 2014. How do  
556 hydromorphological constraints and regulated flows govern macroinvertebrate communities along an  
557 entire lowland river? Ecohydrology 7 (2), 366–377. <https://doi.org/10.1002/eco.1354>

558 Hamilton, A.T., Schäfer, R.B., Pyne, M.I., Chessman, B., Kakouei, K., Boersma, K.S., Verdonschot,  
559 P.F.M., Verdonschot, R.C.M., Mims, M., Khamis, K., Bierwagen, B., Stamp, J., 2020. Limitations of  
560 trait-based approaches for stressor assessment: The case of freshwater invertebrates and climate  
561 drivers. Glob. Chang. Biol. 26 (2), 364–379. <https://doi.org/10.1111/gcb.14846>

562 Karaouzas, I., Theodoropoulos, C., Vourka, A., Gritzalis, K., Skoulidakis, N.T., 2019. Stream  
563 invertebrate communities are primarily shaped by hydrological factors and ultimately fine-tuned by  
564 local habitat conditions. Sci. Total Environ. 665, 290–299.  
565 <https://doi.org/10.1016/j.scitotenv.2019.02.134>



566 Krajenbrink, H.J., Acreman, M., Dunbar, M.J., Hannah, D.M., Laize, C.L., Wood, P.J., 2019.  
 567 Macroinvertebrate community responses to river impoundment at multiple spatial scales. *Sci. Total*  
 568 *Environ.* 650, 2648–2656. <https://doi.org/10.1016/j.scitotenv.2018.09.264>  
 569 Kunz, S., Kefford, B.J., Schmidt-Kloiber, A., Matthaei, C.D., Usseglio-Polatera, P., Graf, W., Poff,  
 570 N.L., Metzeling, L., Twardochleb, L., Hawkins, C.P., Schäfer, R.B., 2021. Tackling inconsistencies  
 571 among freshwater invertebrate trait databases: harmonising across continents and aggregating  
 572 taxonomic resolution. *Freshw. Biol.* <https://doi.org/10.1111/fw.13840>  
 573 Laini, A., Viaroli, P., Bolpagni, R., Cancellario, T., Racchetti, E., Guareschi, S., 2019. Taxonomic  
 574 and Functional Responses of Benthic Macroinvertebrate Communities to Hydrological and Water  
 575 Quality Variations in a Heavily Regulated River. *Water* 11 (7), 1478.  
 576 <https://doi.org/10.3390/w11071478>  
 577 Lancaster, J., Downes, B.J. 2010. Linking the hydraulic world of individual organisms to ecological  
 578 processes: Putting ecology into ecohydraulics. *River Res. and Appl.* 26 (4), 385–403.  
 579 <https://doi.org/10.1002/rra.1274>  
 580 Mellado-Díaz, A., Sánchez-González, J.R., Guareschi, S., Magdaleno, F., Velasco, M.T., 2019.  
 581 Exploring longitudinal trends and recovery gradients in macroinvertebrate communities and  
 582 biomonitoring tools along regulated rivers. *Sci. Total Environ.* 695, 133774.  
 583 <https://doi.org/10.1016/j.scitotenv.2019.133774>  
 584 Mellado-Díaz, A.M., Alonso, M.L.S., Vidal-Abarca Gutiérrez, M.R., 2008, Biological traits of stream  
 585 macroinvertebrates from a semi-arid catchment: patterns along complex environmental gradients.  
 586 *Freshw. Biol.* 53 (1), 1–21. <https://doi.org/10.1111/j.1365-2427.2007.01854.x>  
 587 Mims, M.C., Olden, J.D., 2012. Life history theory predicts fish assemblage response to hydrologic  
 588 regimes. *Ecology* 93(1), 35–45. <https://doi.org/10.1890/11-0370.1>  
 589 Mondy, C.P., Usseglio-Polatera, P., 2014. Using fuzzy-coded traits to elucidate the non-random role  
 590 of anthropogenic stress in the functional homogenisation of invertebrate assemblages. *Freshw. Biol.*  
 591 59 (3), 584–600. <https://doi.org/10.1111/fw.12289>

592 Mouillot, D., Graham, N.A.J., Villéger, S., Mason, M.H.W., Bellwood, D.R., 2013. A functional  
 593 approach reveals community responses to disturbances. *Trends Ecol. Evol.* 28 (3), 167–177.  
 594 <https://doi.org/10.1016/j.tree.2012.10.004>

595 Nakagawa, S., Schielzeth, H., 2013. A general and simple method for obtaining R<sup>2</sup> from generalized  
 596 linear mixed-effects models. *Methods Ecol. Evol.* 4 (2), 133–142. [https://doi.org/10.1111/j.2041-](https://doi.org/10.1111/j.2041-210x.2012.00261.x)  
 597 [210x.2012.00261.x](https://doi.org/10.1111/j.2041-210x.2012.00261.x)

598 Parasiewicz, P., Rogers, J.N., Vezza, P., Gortázar, J., Seager, T., Pegg, M., Wiśniewolski, W.,  
 599 Comoglio, C., 2013. Applications of the MesoHABSIM Simulation Model. In: *Ecohydraulics*. John  
 600 Wiley & Sons, Ltd, pp 109–124

601 Poikane, S., Herrero, F. S., Kelly, M. G., Borja, A., Birk, S., van de Bund, W., 2020. European aquatic  
 602 ecological assessment methods: A critical review of their sensitivity to key pressures. *Sci. Total*  
 603 *Environ.* 740, 140075. <https://doi.org/10.1016/j.scitotenv.2020.140075>

604 Poff, N.L., Olden, J.D., Vieira, N.K.M., Finn, D.S., Simmons, M.P., Kondratieff, B.C., 2006.  
 605 Functional trait niches of North American lotic insects: traits-based ecological applications in light  
 606 of phylogenetic relationships. *J. North Am. Benthol. Soc.* 25 (4), 730–755.  
 607 [https://doi.org/10.1899/0887-3593\(2006\)025\[0730:FTNONA\]2.0.CO;2](https://doi.org/10.1899/0887-3593(2006)025[0730:FTNONA]2.0.CO;2)

608 Poff, N.L., 2018. Beyond the natural flow regime? Broadening the hydro-ecological foundation to  
 609 meet environmental flows challenges in a non-stationary world. *Freshw. Biol.*, 63 (8), 1011-1021.  
 610 <https://doi.org/10.1111/fwb.13038>

611 Prudhomme, C., Giuntoli, I., Robinson, E.L., Clark, D.B., Arnell, N.W., Dankers, R., Fekete, B.M.,  
 612 Franssen, W., Gerten, D., Gosling, S.N., Hagemann, S., Hannah, D.M., Kim, H., Masaki, Y., Satoh,  
 613 Y., Stacke, T., Wada, Y., Wisser, D., 2014. Hydrological droughts in the 21st century, hotspots and  
 614 uncertainties from a global multimodel ensemble experiment. *PNAS* 111 (9), 3262–3267.  
 615 <https://doi.org/10.1073/pnas.1222473110>

616 Quesada-Alvarado, F., Umaña-Villalobos, G., Springer, M., Picado-Barboza, J., 2020. Classification  
 617 of aquatic macroinvertebrates in flow categories for the adjustment of the LIFE Index to Costa Rican  
 618 rivers. *Ecohydrology & Hydrobiology*, 21 (2), 368–376.  
 619 <https://doi.org/10.1016/j.ecohyd.2020.08.005>

620 R Core Team, 2020. A language and environment for statistical computing. R Foundation for  
 621 Statistical Computing. Vienna, Austria.

622 Richter, B., Baumgartner, J., Wigington, R., Braun, D., 1997. How much water does a river need?  
 623 *Freshw. Biol.* 37 (1), 231–249. <https://doi.org/10.1046/j.1365-2427.1997.00153.x>

624 Rolls, R.J., Heino, J., Ryder, D.S., Chessman, B.C., Growns, I.O., Thompson, R.M., Gido, K.B.,  
 625 2018. Scaling biodiversity responses to hydrological regimes. *Biol. Rev.* 93 (2), 971–995.  
 626 <https://doi.org/10.1111/brv.12381>

627 Richoux, P., 1994. Theoretical habitat templates, species traits and species richness: aquatic Coleoptera  
 628 in the Upper Rhône River and its floodplain. *Freshw. Biol.* 31 (3), 377–395.  
 629 <https://doi.org/10.1111/j.1365-2427.1994.tb01747.x>

630 Rue, H., Martino, S., Chopin, N. 2009. Approximate Bayesian inference for latent Gaussian models  
 631 by using integrated nested Laplace approximations. *J. R. Stat. Soc. Series B Stat. Methodol.* 71 (2),  
 632 319–392. <https://doi.org/10.1111/j.1467-9868.2008.00700.x>

633 Ruhi, A., Dong, X., McDaniel, C.H., Batzer, D.P., Sabo, J.L., 2018. Detrimental effects of a novel  
 634 flow regime on the functional trajectory of an aquatic invertebrate metacommunity. *Glob. Chang.*  
 635 *Biol.* 24 (8), 3749–3765. <https://doi.org/10.1111/gcb.14133>

636 Schmidt-Kloiber, A., Hering, D., 2015. [www.freshwaterecology.info](http://www.freshwaterecology.info) – An online tool that unifies,  
 637 standardises and codifies more than 20,000 European freshwater organisms and their ecological  
 638 preferences. *Ecol. Indic.* 53, 271–282. <https://doi.org/10.1016/j.ecolind.2015.02.007>

639 Spiegelhalter, D.J., Best, N.G., Carlin, B.P., Linde, A.V.D., 2002. Bayesian measures of model  
 640 complexity and fit. *J. R. Stat. Soc. Series B Stat. Methodol.* 64 (4), 583–639.  
 641 <https://doi.org/10.1111/1467-9868.00353>

642 Statzner, B., Holm, T.F., 1989. Morphological adaptation of shape to flow: Microcurrents around  
 643 lotic macroinvertebrates with known Reynolds numbers at quasi-natural flow conditions. *Oecologia*  
 644 78, 145–157. <https://doi.org/10.1007/BF00377150>  
 645 Suter, G.W., Cormier, S.M., 2015. Why care about aquatic insects: Uses, benefits, and services.  
 646 *Integr. Environ. Assess. Manag.* 11 (2), 188–194. <https://doi.org/10.1002/ieam.1600>  
 647 Tachet, H., Richoux, P., Bournaud, M., Usseglio-Polatera, P., 2010. Invertébrés d'eau douce:  
 648 Systématique, biologie, écologie, édition revue et augmentée. CNRS EDITIONS, Paris.  
 649 Theodoropoulos, C., Karaouzas, I., Stubbington, R., 2021. Biotic indices of hydrological variability  
 650 as tools to inform dynamic ecological status assessments in river ecosystems. *J. Environ. Manage.*  
 651 295, 113124. <https://doi.org/10.1016/j.jenvman.2021.113124>  
 652 Theodoropoulos, C., Karaouzas, I., Vourka, A., Skoulikidis, N., 2020. ELF – A benthic  
 653 macroinvertebrate multi-metric index for the assessment and classification of hydrological alteration  
 654 in rivers. *Ecol. Indic.* 108, 105713. <https://doi.org/10.1016/j.ecolind.2019.105713>  
 655 Tickner, D., Opperman, J.J., Abell, R., Acreman, M., Arthington, A.H., Bunn, S.E., Cooke, S.J.,  
 656 Dalton, J., Darwall, W., Edwards, G., Harrison, K., Hughes, K., Jones, T., Leclère, D., Lynch, A.J.,  
 657 Leonard, P., McClain, M.E., Muruven, D., Olden, J.D., Ormerod, S.J., Robinson, J., Tharme, R.E.,  
 658 Thieme, M., Tockner, K., Wright, M., Young, L., 2020. Bending the Curve of Global Freshwater  
 659 Biodiversity Loss: An Emergency Recovery Plan. *BioScience* 70 (4), 330–342.  
 660 <https://doi.org/10.1093/biosci/biaa002>  
 661 Timm, H., Käiro, K., Möls, T., Virro, T. 2011. An index to assess hydromorphological quality of  
 662 Estonian surface waters based on macroinvertebrate taxonomic composition. *Limnologica* 41 (4),  
 663 398–410. <https://doi.org/10.1016/j.limno.2011.09.006>  
 664 Tockner, K., Pusch, M., Borchardt, D., Lorang, M.S. 2010. Multiple stressors in coupled river–  
 665 floodplain ecosystems. *Freshw. Biol.* 55 (1), 135–151. <https://doi.org/10.1111/j.1365-2427.2009.02371.x>  
 666

667 Tonkin JD, Merritt DM, Olden JD, et al (2018) Flow regime alteration degrades ecological networks  
 668 in riparian ecosystems. *Nature Ecology & Evolution* 2:86–93. [https://doi.org/10.1038/s41559-017-](https://doi.org/10.1038/s41559-017-0379-0)  
 669 [0379-0](https://doi.org/10.1038/s41559-017-0379-0)  
 670 Usseglio-Polatera, P., Bournaud, M., Richoux, P., Tachet H., 2000a. Biological and ecological traits  
 671 of benthic freshwater macroinvertebrates: relationships and definition of groups with similar traits.  
 672 *Freshw. Biol.* 43 (2), 175–205. <https://doi.org/10.1046/j.1365-2427.2000.00535.x>  
 673  
 674 Usseglio-Polatera, P., Bournaud, M., Richoux, P., Tachet, H. 2000b. Biomonitoring through  
 675 biological traits of benthic macroinvertebrates: how to use species trait databases? In: Jungwirth M.,  
 676 Muhar S., Schmutz S. (eds) *Assessing the Ecological Integrity of Running Waters*. Developments in  
 677 *Hydrobiology*, vol 149. Springer, Dordrecht. [https://doi.org/10.1007/978-94-011-4164-2\\_12](https://doi.org/10.1007/978-94-011-4164-2_12)  
 678 Ventrucci, M., Cocchi, D., Burgazzi, G., Laini, A. 2020. PC priors for residual correlation parameters  
 679 in one-factor mixed models. *Stat. Methods Appl.* 29 (4), 745–765. [https://doi.org/10.1007/s10260-](https://doi.org/10.1007/s10260-019-00501-w)  
 680 [019-00501-w](https://doi.org/10.1007/s10260-019-00501-w)  
 681 Verberk, W.C.E.P., van Noordwijk, C.G.E., Hildrew, A.G., 2013. Delivering on a promise:  
 682 integrating species traits to transform descriptive community ecology into a predictive science.  
 683 *Freshw. Sci.* 32 (2), 531–547. <https://doi.org/10.1899/12-092.1>  
 684 Vezza., P., Zanin, A., Parasiewicz, P., 2017. Manuale tecnico-operativo per la modellazione e la  
 685 valutazione dell'integrità dell'habitat fluvial. ISPRA – Manuali e Linee Guida 154/2017, Roma  
 686 Water Development Department, 2006. Tender for supplying services for sampling from river water  
 687 bodies for the purpose of participation in the Intercalibration Exercise of European Community.  
 688 White, J.C., Hannah, D.M., House, A., Beatson, S.J.V., Martin, A., Wood, P.J., 2017.  
 689 Macroinvertebrate responses to flow and stream temperature variability across regulated and non-  
 690 regulated rivers. *Ecohydrology* 10 (1), e1773. <https://doi.org/10.1002/eco.1773>

691 White, J.C., Fornaroli, R., Hill, M.J., Hannah, D.M., House, A., Colley, I., Perkins, M., Wood, P.J.  
 692 (2021). Long-term river invertebrate community responses to groundwater and surface water  
 693 management operations. *Water Res.* 189, 116651. <https://doi.org/10.1016/j.watres.2020.116651>  
 694 Wickham, H. 2009. *ggplot2: Elegant Graphics for Data Analysis*. Springer-Verlag, New York.  
 695 Wood, P.J., Agnew, M.D., Petts, G.E., 2000. Flow variations and macroinvertebrate community  
 696 responses in a small groundwater-dominated stream in south-east England. *Hydrol. Process.* 14 (16-  
 697 17), 3133–3147. [https://doi.org/10.1002/1099-1085\(200011/12\)14:16/17<3133::AID-](https://doi.org/10.1002/1099-1085(200011/12)14:16/17<3133::AID-HYP138>3.0.CO;2-J)  
 698 [HYP138>3.0.CO;2-J](https://doi.org/10.1002/1099-1085(200011/12)14:16/17<3133::AID-HYP138>3.0.CO;2-J)  
 699 Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A., Smith, G.M., 2009. *Mixed effects models and*  
 700 *extensions in ecology with R*. Springer Nature, New York, NY.

Table 1 Different versions of the Flow-T index used in this research. The equation of Flow-T<sub>sub</sub> is the same of Flow-T<sub>pa</sub> and Flow-T<sub>abu</sub> for presence-absence and abundance data but excludes Chironomidae and Oligochaeta from the computation. Flow-T<sub>tsi</sub> flow velocity preferences are weighted by tsi, calculated according to the reported equation. Weighted flow velocity preferences are standardized to sum 1 for each species prior to the calculation of Flow-T<sub>tsi</sub>. The presence-absence equation is reported for Flow-T<sub>tsi</sub> for illustrative purposes, but the abundance-based equation can also be used.

ACRONYM	MEANING	EQUATION
Flow-T <sub>pa</sub>	Presence-absence data using all the taxa receiving a score.	$Flow - T_{pa} = \frac{\sum_i^n (m_i + f_i)}{n} \times 100$
Flow-T <sub>abu</sub>	Abundance data using all the taxa receiving a score.	$Flow - T_{abu} = \frac{\sum_i^n \frac{\log A_i \times (m_i + f_i)}{\log A_i}}{\sum_i^n \log A_i} \times 100$
Flow-T <sub>sub</sub>	Excluding Chironomidae and Oligochaeta.	-
Flow-T <sub>tsi</sub>	More weight assigned to flow-specialist taxa.	$Flow - T_{tsi} = \frac{\sum_i^n (m_i + f_i) * tsi_i}{n} \times 100; tsi = \frac{\sum_i^k F_k \times \frac{1}{k}}{1 - \frac{1}{k}}$

n = number of taxa in a sample; A<sub>i</sub> = abundance of the i<sup>th</sup> taxon; m<sub>i</sub> and f<sub>i</sub> = “moderate” and “fast” velocity preference classes of the i<sup>th</sup> taxon according to Tachet et al. (2010); k is the number of flow velocity classes; F the k<sup>th</sup> scores transformed to relative frequencies for the i<sup>th</sup> taxon.

Table 2. Dataset used in this work with an indication of the sample size, taxonomic resolution, available metadata and the type of analysis performed on each dataset (N = number of samples).

COUNTRY	TIME PERIOD	N	TAXONOMIC LEVEL	METADATA	ANALYSIS
UK	1986-1997	133	Family, mixed-level	-	Correlation with the LIFE index,
	1986-2019	285			Sensitivity of family level index to random selection of finer taxonomic levels
CYPRUS	2005-2018	872	Family	mesohabitat	Correlation with the LIFE index, Sensitivity of family level index to random selection of finer taxonomic levels, Mixed model of index vs mesohabitat_1
ITALY	2020	180	Family, mixed-level		Correlation with the LIFE index,



	Sensitivity of family level index
	to random selection of finer
	taxonomic levels,
mesohabitat,	Mixed model of index vs
current	mesohabitat_1 and
velocity	mesohabitat_2,
	Mixed model of index vs
	mesohabitat*current velocity

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\*mesohabitat\_1 refers to the subdivision of mesohabitat in riffles and pools used for biomonitoring in Italy and Cyprus. Mesohabitat\_2 refers to the subdivision in mesohabitat according to Belletti et al. (2017), frequently used in habitat modelling.

Table 3 Correlation between LIFE and flow-T index for each dataset and typology (pa = presence-absence, abu = abundance; sub = reduced Flow-TI, tsi = weighted flow-T). Data are shown for family level information.

<b>Index</b>	<b>UK</b>	<b>Cyprus</b>	<b>Italy</b>
Flow-T <sub>pa</sub>	0.86	0.46	0.73
Flow-T <sub>abu</sub>	0.96	0.72	0.90
Flow-T <sub>sub_pa</sub>	0.87	0.52	0.82
Flow-T <sub>sub_abu</sub>	0.96	0.73	0.89
Flow-T <sub>tsi_pa</sub>	0.54	0.23	0.51
Flow-T <sub>tsi_abu</sub>	0.86	0.40	0.68

Table 4:  $R^2$  of the mixed model analysis performed using LIFE and the two versions of Flow-T as dependent variables and mesohabitat, velocity and their interaction as independent variables for the Italian dataset. LIFE was calculated at family level while Flow-T at both family and mixed taxonomic level for presence absence (pa) and abundance (abu) data. Both median values and credible intervals are reported, together with the Deviance Information Criteria (DIC) for the models with and without interaction and their differences ( $\Delta$  DIC). See main text and S2 for further results about Cyprus and other versions of Flow-T.

Index	Taxonomic level	$R^2_{m\_median}$	$R^2_{m\_ci}$	$R^2_{c\_median}$	$R^2_{c\_ci}$	DIC <sub>interaction</sub>	DIC <sub>additive</sub>	$\Delta$ DIC
LIFE	Family	0.54	0.21-0.63	0.66	0.58-0.86	207.8	222.6	14.8
Flow-T <sub>pa</sub>	Family	0.49	0.42-0.56	0.50	0.42-0.56	1197.5	1205.2	7.7
Flow-T <sub>abu</sub>	Family	0.67	0.61-0.71	0.70	0.65-0.74	1200.9	1218.8	17.9
Flow-T <sub>pa</sub>	Mixed	0.74	0.70-0.77	0.77	0.72-0.79	1294.0	1333.8	39.8
Flow-T <sub>abu</sub>	Mixed	0.74	0.70-0.77	0.76	0.72-0.79	1237.9	1266.1	28.2

## FIGURE CAPTION

Fig. 1. Relationship between LIFE values and both Flow-T based on the UK using the original dataset from 1986 to 1997.

Fig. 2. Median effect of mesohabitat in LIFE and both Flow-T indices for the Cyprus dataset. Both standardized (left y-axis) and raw (right y axis) of the indices are displayed to ease the comparison among indices. Bayesian credible intervals are also displayed.

Fig. 3. a) Relationships of the LIFE and Flow-T indices with flow velocity for the Italian dataset (pa = presence-absence, abu = abundance). The trend line was computed with a loess function and the grey band represents the 95% confidence intervals; b) boxplot of the association between LIFE and Flow-T indices and mesohabitats for the same dataset (i\_p= isolated pond, b=backwater, p=pool, g=glide, r=riffle). Boxplots report median and interquartile ranges and the whiskers represent the maximum/minimum value of the data that lie within 1.5 times the interquartile range. Box plots with the same letter indicate that no significant difference between groups were recorded according to a model with mesohabitat as a fixed effect and collection site as a random effect. Statistical significance was determined via credible intervals: full details of the analysis are reported in supplementary material 8.