

1           **European river typologies fail to capture diatom, fish, and**  
2           **macrophyte community composition**

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67 **Abstract**

68 Typology systems are frequently used in applied and fundamental ecology and are relevant  
69 for environmental monitoring and conservation. They aggregate ecosystems into discrete  
70 types based on biotic and abiotic variables, assuming that ecosystems of the same type are  
71 more alike than ecosystems of different types with regard to a specific property of interest. We  
72 evaluated whether this assumption is met by the Broad River Types (BRT), a recently proposed  
73 European river typology system, that classifies river segments based on abiotic variables,  
74 when it is used to group biological communities. We compiled data on the community  
75 composition of diatoms, fishes, and aquatic macrophytes throughout Europe and evaluated  
76 whether the composition is more similar in site groups with the same river type than in site  
77 groups of different river types using Analysis of Similarities, classification strength, typical  
78 species analysis, and the area under zeta diversity decline curves. We compared the  
79 performance of the BRT with those of four region-based typology systems, namely, Illies  
80 Freshwater Ecoregions, the Biogeographic Regions, the Freshwater Ecoregions of the World,  
81 and the Environmental Zones, as well as spatial autocorrelation (SA) classifications.

82 All typology systems received low scores from most evaluation methods, relative to predefined  
83 thresholds and the SA classifications. The BRT often scored lowest of all typology systems.  
84 Within each typology system, community composition overlapped considerably between site

85 groups defined by the types of the systems. The overlap tended to be the lowest for fishes and  
86 between Illies Freshwater Ecoregions.

87 In conclusion, we found that existing broad-scale river typology systems fail to delineate site  
88 groups with distinct and compositionally homogeneous communities of diatoms, fishes, and  
89 macrophytes. A way to improve the fit between typology systems and biological communities  
90 might be to combine segment-based and region-based typology systems to simultaneously  
91 account for local environmental variation and historical distribution patterns, thus potentially  
92 improving the utility of broad-scale typology systems for freshwater biota.

93 **Keywords**

94 Typology systems, ecoregions, freshwater ecosystems, typology evaluation, biological quality  
95 elements, biomonitoring  
96

## 1   **1. Introduction**

2

3   Ecologists use typology systems to assign ecosystems to discrete types (Stoddard 2004,  
4   Soranno et al. 2010). The degree to which real-world ecosystems are discrete entities or  
5   artificial divisions of a continuous gradient is an ongoing debate (Eliot 2011, Liautaud et al.  
6   2019), but the utility of typology systems is less contentious (Leathwick et al. 2011, Ebach  
7   2021). They are used in water quality monitoring to delineate ecosystems with similar natural  
8   conditions (Reynoldson et al. 1997). Conservationists use them to identify areas with high  
9   species richness or endemism (Heiner et al. 2011, Oliveira et al. 2015), to identify ecosystem  
10   types that merit increased protection efforts (Mackey et al. 1988), or to describe desired  
11   ecosystem states (Vynne et al. 2022). In broad-scale analyses, ecosystem typologies provide  
12   spatial units for the comparison of community trait composition (Iversen et al. 2019) or  
13   temporal trends in species abundance (Powell et al. 2022).

14   All typology systems are models (Goodwin 1999, Loveland and Merchant 2004) that simplify  
15   their subjects to permit inference and prediction under a set of assumptions. Ecosystem  
16   typologies bundle ecosystems into a set of discrete groups characterized by selected biotic  
17   and abiotic variables (e.g., altitude, temperature, and bedrock geology). These models are  
18   based on the assumption that ecosystems of the same type are more similar to each other  
19   than to ecosystems of different types, with respect to a specific property of interest. Each  
20   typology system is optimized for one property (e.g., delineating homogeneous communities of  
21   mammals) and might fail to delineate meaningful patterns in other properties (e.g., background  
22   nitrogen concentration) (Loveland and Merchant 2004).

23   Ecologists commonly use ecosystem typologies to delineate ecosystems with similar  
24   biological communities. These typology systems usually defined contiguous areas (regions)  
25   as mapping units and focused on terrestrial ecosystems. Region-based typologies are  
26   appropriate for terrestrial (Olson et al. 2001) and marine ecosystems (Spalding et al. 2007),  
27   as both lack inherent geometry. However, river ecosystems are dendritic networks (Benda et

28 al. 2004, Campbell Grant et al. 2007) and change from headwater to mouth (Vannote et al.  
29 1980, Herlihy et al. 2021). Region-based typologies can not account for these factors, but  
30 segment-based river typologies that classify confluence to confluence sections of rivers can.  
  
31 Segment-based river typologies have been proposed at national (Snelder et al 2004) and  
32 global levels (Ouellet Dallaire et al 2019), but until recently we lacked a unified European  
33 system. Lyche Solheim et al. (2019) filled this gap with the Broad River Types (BRT), which  
34 aggregate the disparate river typology systems created by participating countries of the  
35 European Water Framework Directive (WFD) into twelve broad types. The demand for such a  
36 typology system was demonstrated by a quick adoption from the research community (e.g.,  
37 Birk et al., 2020; Lemm et al., 2021; Posthuma et al., 2020). The BRT were created to  
38 aggregate and compare information on environmental state and relevant pressures acting on  
39 the rivers (Lyche Solheim et al. 2019). As noted above, the usefulness of ecosystem  
40 typologies hinges on the assumption that ecosystems of the same type are more similar than  
41 ecosystems of different types. For the BRT, this crucial assumption remains largely unchecked  
42 for biological communities. Jupke et al. 2022 showed that patterns in the composition of  
43 benthic macroinvertebrate communities are not well captured by the BRT. The concordance  
44 between ecosystem typologies and biological communities differs between taxonomic groups  
45 (Paavola et al. 2003, Infante et al. 2009, Ficetola et al. 2021), and should therefore be  
46 evaluated for multiple taxonomic groups.

47 Diatoms, fishes, and aquatic macrophytes are each commonly used to monitor the status of  
48 freshwater systems (Aguiar et al. 2011, Masouras et al. 2021, Pont et al. 2021). They are  
49 complementary in the stressors they identify (Johnson et al. 2006, Hering et al. 2006,  
50 Cellamare et al. 2012, Marzin et al. 2012), partly as they represent an ecosystem's state at  
51 different trophic levels and spatio-temporal scales (Lainé et al. 2014). Diatoms have short  
52 generation times (days to weeks), disperse via passive drift in water or air (e.g., Liu et al. 2013)  
53 and attached to animals (Maguire 1963, Manning et al. 2021). Their community composition  
54 reflects the current environmental conditions (water conductivity, pH, nutrients, organic

55 pollution). Fishes are long-lived and mobile. Their community composition represents the state  
56 of a riverscape (temperature, connectivity, and hydromorphology) over larger spatio-temporal  
57 scales (Hoeinghaus et al. 2007). Macrophytes are also long-lived but, due to their mostly  
58 sessile nature, respond most strongly to environmental conditions (water chemistry, light  
59 availability, substrate) in their direct vicinity (Alahuhta et al. 2014), and hence integrate  
60 environmental fluctuations over long temporal but fine spatial scales.

61 Here, we aim to evaluate the fit between the BRT and the community composition of diatoms,  
62 fish, and aquatic macrophytes. We evaluated the coherence between community composition  
63 and the BRT with Analysis of similarities, classification strength, typical species analyses, zeta  
64 diversity analyses. To contextualize the BRT's performance, we compared it to those of four  
65 region-based typology systems (Illiies Freshwater Ecoregions (IFE, Illies 1978), Biogeographic  
66 Regions (BGR, EEA 2012), Freshwater Ecoregions of the World (FEoW, Abell et al. 2008),  
67 and Environmental Zones (EnZ, Metzger et al. 2005)), and to spatial autocorrelation (SA)  
68 classifications. The SA classifications are naïve typology systems, consisting of simple  
69 geometric forms spread over Europe (Figure 1). We aim to answer two questions: (Q1) Do the  
70 site groups delineated by the BRT host communities of diatoms, fish, and macrophytes whose  
71 composition is more similar within than among types? (Q2) Are the BRT a better classification  
72 of diatom, fish, and aquatic macrophyte communities, with regard to their composition, than  
73 the four region-based approaches?

## 74 **2. Material and Methods**

### 75 **2.1 The typology systems**

76 The BRT reduce the number of national WFD river types (1,247) to a workable set, which can  
77 be used to compare water body status data across Europe (Lyche Solheim e al. 2019).  
78 National types were combined based on altitude, catchment size, geology, region, and flow  
79 regime. Rare types were merged with the most similar type. The final BRT categorize 12 river  
80 types, as detailed in Table 1. We utilized the digital version of the BRT published by Globenvik  
81 (2019).

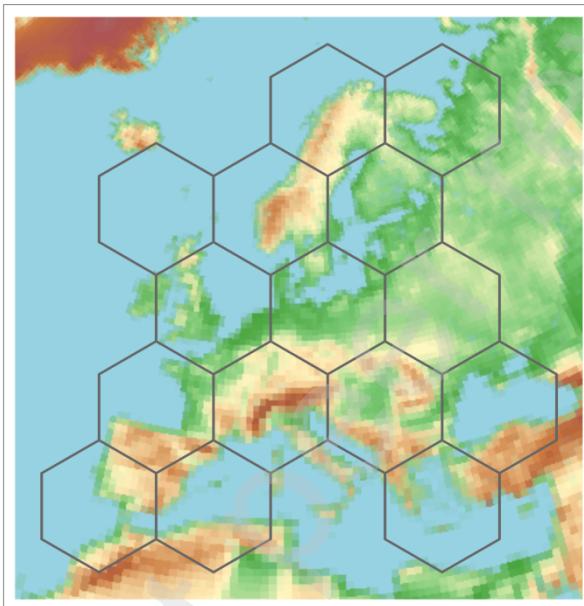
82 Table 1: Codes and names of the twelve Broad River Types proposed by Lyche Solheim et al. (2019). The sizes refer to  
 83 catchment area: very small-small <100 km<sup>2</sup>, medium-large 100–10.000 km<sup>2</sup> and very large >10.000 km<sup>2</sup>. Lowland denotes  
 84 river segments <200 meters above sea level (m.a.s.l.), mid-altitude 200–800 m.a.s.l. and highland >800 m.a.s.l. The geologies  
 85 describe the prevailing lithological or pedological conditions in the catchments. Catchments are calcareous or siliceous if the  
 86 respective soil types or minerals cover >50 % of the catchments area. If coverage is between 40 % and 50 % it is classified as  
 87 mixed. Catchments with >20 % of their area covered by histosols are classified as organic. Mediterranean rivers are treated  
 88 separately. For them the flow regime (perennial/temporary) is considered additionally.

ID	Name
RT1	Very large rivers
RT2	Lowland, calcareous or mixed, medium-large
RT3	Lowland, calcareous or mixed, very small-small
RT4	Lowland, siliceous incl. organic, medium-large
RT5	Lowland, siliceous incl. organic, very small-small
RT6	Mid-altitude, calcareous incl. organic, medium-large
RT7	Mid-altitude, calcareous or mixed, very small-small
RT8	Mid-altitude, siliceous incl. organic, medium-large
RT9	Mid-altitude, siliceous incl. organic, very small-small
RT10	Highland and glacial
RT11	Mediterranean, perennial
RT12	Mediterranean temporary and very small

89  
 90 The typology systems we used as reference points are shortly introduced below (for details  
 91 and maps see SI1). All reference typologies are region-based typologies as no other segment-  
 92 based typologies are available for all of Europe. IFE divide between 25 regions based on the  
 93 distribution of macroinvertebrate fauna, the BGR partition Europe into 12 regions based on  
 94 their potential natural vegetation, the FEoW are a global system that classifies catchments  
 95 based on their fish faunas, and the EnZ are 12 zones derived from principal component  
 96 analysis of 22 environmental variables.

97 We created classifications that capture the spatial autocorrelation inherent in community  
 98 composition data but are otherwise uninformed by biogeographic transition zones. These  
 99 spatial autocorrelation (SA) classifications were created by laying a regular, hexagonal grid

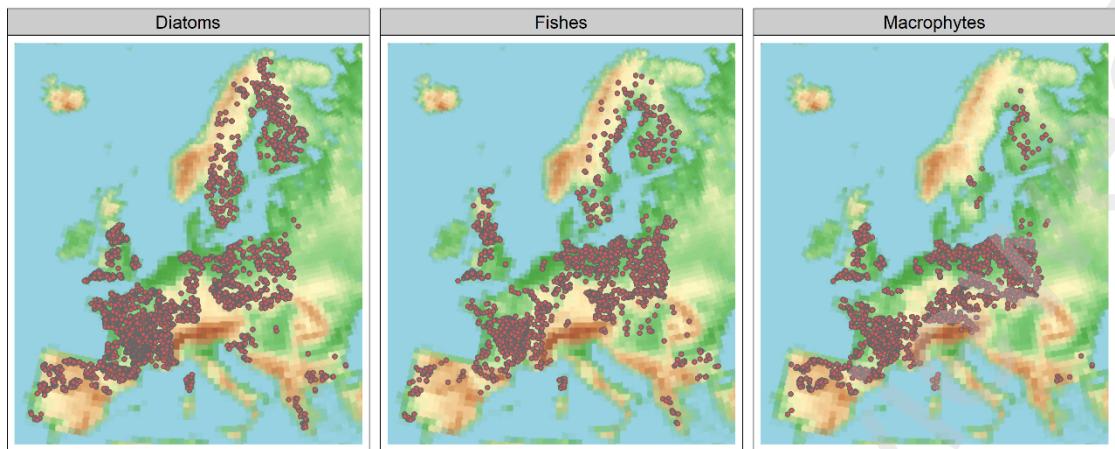
100 with 15 cells (types) over Europe (Figure 1). We chose 15 types as this approximately matches  
101 the average number of types from the other typology systems. To evaluate whether size or  
102 form impact the results, we created multiple SA classifications. Besides the 15-cell hexagonal  
103 classification, we created a square classification with the same cell size (12 types), as well as  
104 hexagonal and square classifications with half the cell size (36 and 33, types, respectively).  
105 Maps of the additional SA classifications are provided in the supplementary information.



106  
107 *Figure 1: Hexagonal spatial autocorrelation (SA) classification with 15 cells. Each cell is a separate type. The SA classification*  
108 *is a naïve approach to classification capturing spatial autocorrelation but uninformed by ecologically relevant variables.*

## 109 **2.2 Data preparation**

110 We compiled 21, 23, and 25 datasets for diatoms, fish, and macrophytes, respectively (Figure  
111 2, Table S1, S2, and S3). All sampling was conducted according to EU norms (EN 13946, EN  
112 14407, and EN 15708 for diatoms, EN 14011 for fishes, and EN 14184 for macrophytes). We  
113 harmonized the datasets by transforming all data to presence-absence. The samples were  
114 taken between 2000 and 2021, 1986 and 2021, and 2006 and 2021 for diatoms, fishes, and  
115 macrophytes respectively. The composition of communities has likely changed during these  
116 periods (Tison-Roseberry et al. 2022). However, the magnitude of this bias is small and it is  
117 unlikely to change our results quantitatively (see SI3 for analysis). We included samples taken  
118 between May and September and only included the most recent sample at each site (Figure  
119 3).

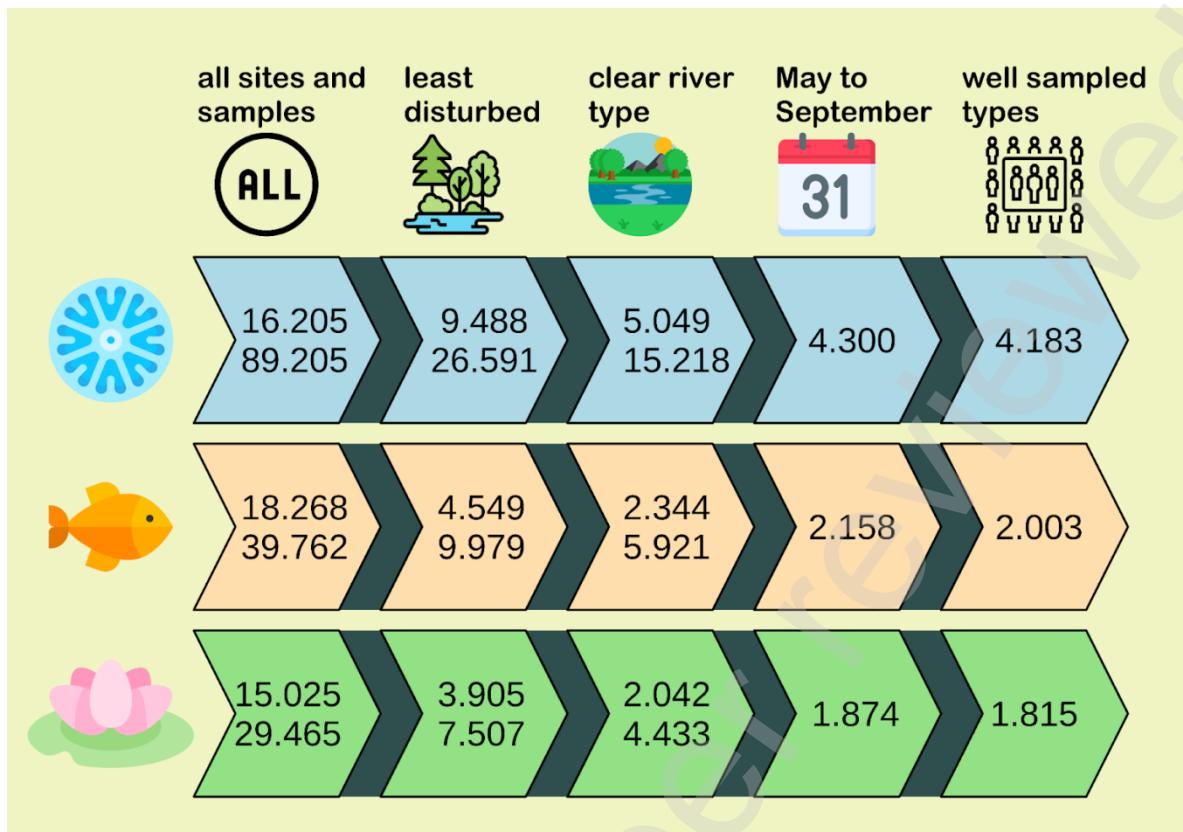


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121 *Figure 2: Spatial distribution of sampling sites for diatoms, fishes, and aquatic macrophytes. The map only shows sampling*  
 122 *sites that we deemed least disturbed and could assign unambiguously to one stream in the digital representation of the Broad*  
 123 *River Types.*

124

125 The diatom data required extensive harmonization because of varying nomenclatures,  
 126 identification errors (Morales et al. 2001, Kahlert et al. 2009), and ongoing changes to the  
 127 accepted nomenclature (e.g., Mann & Vanormelingen 2013). We updated names to current  
 128 synonyms and grouped often misidentified taxa into complexes. We updated names with the  
 129 taxonomic database from the OMNIDIA software (Lecointe et al. 1993) and the algaebase  
 130 website (Guiry et al. 2014). We used Table S2 from Kahlert et al. (2020) to group contentious  
 131 taxa into complexes. For fish and macrophytes, we replaced taxonomic synonyms with  
 132 accepted names as indicated by the Global Biodiversity Information Facility ([www.gbif.org](http://www.gbif.org)).  
 133 We removed taxa that only occurred in one sample. For macrophytes, we included true  
 134 hydrophytes, helophytes, and mosses but no riparian vegetation. Harmonization tables  
 135 providing original names and synonyms for all three taxonomic groups are available as  
 136 supplementary material.



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138 *Figure 3: Overview of data processing and the resulting number of sites and samples. The rows are the taxonomic groups*  
 139 *diatoms, fishes, and macrophytes. Within each box the upper number gives the number of sites and the lower is the number*  
 140 *of samples. The first column gives the numbers before data processing. The second column gives the numbers after only*  
 141 *least disturbed and not impoverished sites are retained. The third column provides the numbers after removing samples*  
 142 *that could not unambiguously assign a specific broad river type. The fourth column gives the number of samples after*  
 143 *only the most recent sample and only samples between May and September were retained. As only one sample is kept per*  
 144 *site, the number of samples and sites is the same. The last column gives the number of sites after those that were in*  
 145 *ecosystem types with less than 20 samples were removed.*

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147 To prevent anthropogenic stressors from harmonizing communities across river types  
 148 (McKinney and Lockwood 1999, Petsch 2016), we analyzed only the least disturbed sites  
 149 (Stoddard et al., 2006) (Figure 3). We identified the least disturbed sites with a European  
 150 database including the stressors total phosphorus and nitrogen load, alterations of mean and  
 151 base flow, mixture toxic pressure, and extent of agricultural and urban land use at a sub-  
 152 catchment scale (Lemm et al 2021). We standardized each stressor to the range from 0 to 1  
 153 and categorized all sub-catchment units where all scores were  $\leq 0.24$  as least disturbed (see  
 154 SI4 for rationale). In addition, we removed samples with  $\leq 10$  diatom species,  $\leq 2$  fish or  
 155 macrophyte species. Lastly, we visually validated the assignment of samples to BRT by

156 comparing the position of the sampling sites with the digital river network and the  
157 CaroDB.Positron base map through the mapview R package (Appelhans et al. 2021) and  
158 removed samples from erroneous assignments.

159 **2.3 Evaluation of typologies**

160 We evaluated all combinations of taxonomic groups and river types for which we had at least  
161 20 samples using Analysis of Similarities (ANOSIM, Clarke, 1993), Classification Strength  
162 (CS, Van Sickle, 1997), typical species analysis (TSA, Jupke et al. 2022), and the area under  
163 the zeta diversity decline curve (AUC  $\zeta$ ), a method proposed in this paper.

164 ANOSIM compares the ranked similarities within and among types. The resulting R statistic  
165 ranges from -1 to 1. The best classification, in which all within-type similarities are higher than  
166 all among-type similarities, corresponds to an  $R$  of 1. To confirm our first research question,  
167 whether community composition is more similar with BRT-types than between them, they  
168 would need to obtain an ANOSIM R-value  $\geq 0.5$  (Clark & Gorley 2006) and to outperform the  
169 SA classifications. CS is the difference between mean similarity within-types ( $W$ ) and mean  
170 similarity among-types ( $A$ ). The means are weighted by the number of samples per ecosystem  
171 type. Values range between -1 (samples are equal among types but share no taxa within  
172 types) and 1 (samples are equal within types but share no taxa among types). We affirm  
173 research question one if the CS is above 0.1 (Hawkins et al. 2000, Soininen et al. 2004) and  
174 above the CS of the SA classifications. We used TSA to determine typical species/complexes  
175 for the types of each typology system. In TSA, a typical species/complex of a type is one that  
176 occurs in 33% of samples from that type. To test if the TSA communities, i.e. all typical  
177 species/complexes of a type, differ between types, we computed the Jaccard dissimilarities  
178 between TSA communities of the same typology system. These dissimilarities range from 0  
179 (identical communities) to 1 (no taxa in common). A good typology system would have high  
180 dissimilarity between the TSA communities. Finally, we determined the area under the zeta  
181 diversity decline curve. Zeta diversity is the average number of shared species between a

182 given number of sites (Hui and McGeoch 2014). Zeta diversity extends to multi-site  
183 comparisons through the number of considered sites, the zeta degree. For example,  $\zeta_3$  is the  
184 average number of species shared between three sites. Zeta diversity decreases  
185 monotonically with increasing zeta degree and the rate of decline is steeper when fewer  
186 species are shared between sites. Therefore, zeta diversity decline should be slower within  
187 types than among types. To evaluate declines, we used the area under the zeta diversity  
188 decline curve ( $AUC\zeta$ ). This metric is derived by computing the zeta diversities for the zeta  
189 degrees 1 to 10, scaling all zeta diversities so that  $\zeta_1 = 1$  and then computing the area under  
190 the curve that is created by drawing a line through all zeta diversities. Higher  $AUC\zeta$  values  
191 imply a slower decline, i.e., more similar communities and thus a better typology system.

192 **2.4 Software**

193 All analyses were conducted with R 4.2.2 (R Core Team 2022). Data were prepared with  
194 data.table 1.14.2 (Dowle and Srinivasan 2022), tidyverse 1.3.2 (Wickham et al. 2019), and sf  
195 1.0-9 (Pebesma 2018). Analyses were conducted with zetadiv 1.2.0 (Latombe et al. 2018),  
196 vegan 2.6-2 (Oksanen et al. 2022) and parallelDist 0.2.6 (Eckert 2022). Maps and figures were  
197 created with ggplot2 3.4.0 (Wickham 2016), wesanderson 0.3.6 (Ram and Wickham 2018),  
198 maptiles 0.4.0 (Giraud 2022), and tmap 3.3-3 (Tennekes 2018).

199 **3. Results**

200 For all three taxonomic groups, the results of the different SA classifications differed minimally  
201 and without a discernible pattern. Below, we present the results of the 15-type hexagonal SA  
202 classification (see SI6 for the results of the other SA classifications).

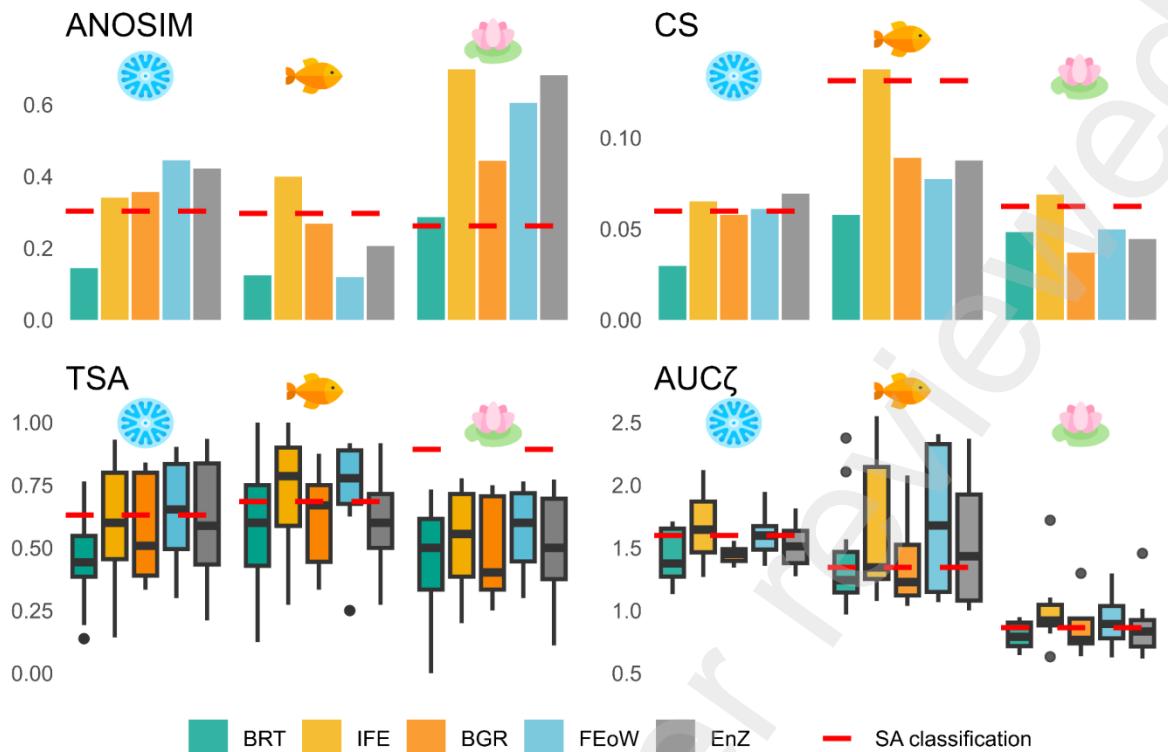


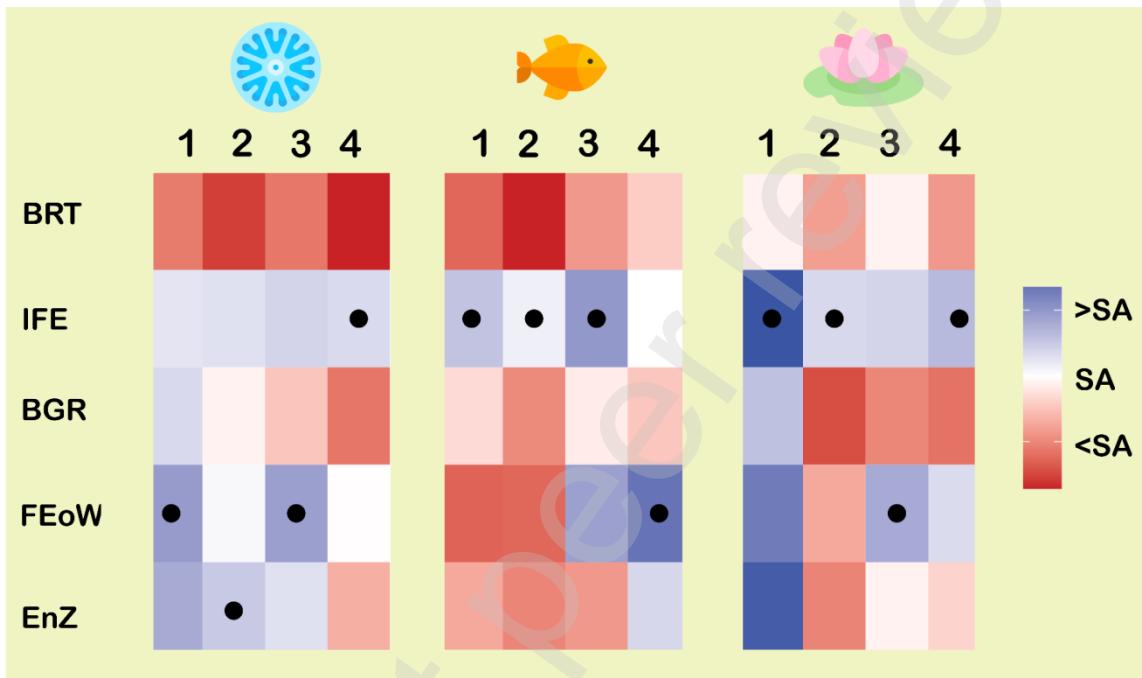
Figure 4: Results of the four cluster analyses, Analysis of similarities (ANOSIM), Classification Strength (CS), Typical Species Analysis (TSA), and area under the zeta diversity decline curve ( $AUC\zeta$ ), for the Broad River Types (BRT), Illies Freshwater Ecoregions (IFE), Biogeographic Regions (BGR), Freshwater Ecoregions of the World (FEoW), and Environmental Zones (EnZ) across diatoms, fishes, and macrophytes. For all four analyses, higher values imply a better classification. The dashed red line indicates the results for the 15-type hexagonal spatial autocorrelation (SA) classification. For ANOSIM, the R-statistic is shown. An R of 1 corresponds to a perfect classification where within-type similarities exceed among-type similarities. For the CS, the classification strength is shown. A high dissimilarity between the TSA communities of two types implies that the different species are common in each of the two types. Therefore, a good typology system would have high dissimilarities between TSA communities. For  $AUC\zeta$ , the area under the zeta diversity decline curve is shown. If communities share fewer species, their zeta diversity decline curves will be steeper and hence enclose a smaller area over a fixed number of orders. A good typology would aggregate ecosystems with similar communities and hence with slowly declining zeta diversity curves and large areas under the decline curve. This analysis returns one area under the curve per type and taxonomic group. The distribution of these areas is shown here.

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218 For diatoms and fish, all ANOSIMS indicated weak separation between the types ( $R < 0.5$ ,  
219 Figure 4) and most typology systems performed similar to the SA classifications. For  
220 macrophytes, IFE, FEoW, and EnZ were good classifications ( $R > 0.5$ ) and all typology systems  
221 outperformed the SA classifications, though the BRT only slightly. CS was low ( $< 0.1$ ) for all  
222 analyses, except for IFE and fish (Figure 4). The SA classification outperformed most typology  
223 systems for fish and macrophytes and always performed better than the BRT. The dissimilarity  
224 between TSA communities was lowest in macrophytes but the SA model outperformed most  
225 combinations of taxonomic group and typology systems, including the BRT for all the  
226 taxonomic groups. The complete list of typical communities is available in the supplementary

227 materials. The median AUC $\zeta$  was lowest for macrophytes and the median AUC $\zeta$ s of the SA  
 228 classification generally had a similar magnitude as those of the actual typology systems. The  
 229 BRT's AUC $\zeta$  was always lower than that of the SA classifications. The typologies performance  
 230 relative to the SA classifications and the best performing typology systems for each evaluation  
 231 method and taxonomic groups are shown in Figure 5.



232  
 233 *Figure 5: Summary of all results relative to the 15-type hexagonal spatial autocorrelation (SA) classification. The rows indicate*  
 234 *the typology systems: Broad River Types (BRT), Illies Freshwater Ecoregions (IFE), Biogeographical Regions (BGR), Freshwater*  
 235 *Ecoregions of the World (FFeoW), and Environmental Zones (EnZ). The columns are the different analyses: Analysis of*  
 236 *similarities (1), classification strength (2), Jaccard dissimilarity between typical species analysis communities (3), and area*  
 237 *under the zeta diversity decline curve (4). All results are relative to the SA classification. Blue cells indicate that the typology*  
 238 *received a higher score than the SA classification, red cells indicate the opposite. White cells show that the performance is*  
 239 *similar to the SA classification. Black dots highlight the typology system that performed best for a given combination of*  
 240 *taxonomic group and evaluation method.*

## 241 4. Discussion

242 We compiled datasets of diatom, fish, and macrophyte occurrences from rivers throughout  
 243 Europe to determine whether spatial patterns in community composition follow broad-scale  
 244 environmental covariates as captured by different ecosystem typologies. Our first research  
 245 question was whether the site groups delineated by the BRT host communities of diatoms,  
 246 fish, and macrophytes whose composition is more similar within than among types. This is not  
 247 the case for any of the three taxonomic groups. The BRT failed to meet the predefined quality  
 248 threshold for ANOSIM and CS and performed worse than the SA model in 11 out of 12 tests.

249 Our second research question was whether the communities within BRT were more  
250 homogenous and distinct than those of the types of IFE, BGR, FFeW or EnZ. Here, the  
251 opposite was the case. In most analyses, the BRT were least reflected by the community  
252 compositions. Our analyses were based on presence-absence data. The results therefore  
253 pertain solely to the composition of communities and not their structure, i.e., the abundance  
254 of different taxa. Since patterns in community structure can deviate from those in community  
255 composition (Mueller et al. 2013) we advise against generalizing our results to community  
256 structure. Further, we wish to emphasize that our sampling sites are not uniformly distributed  
257 across types or within types. As such data do not exist on broad-scales we cannot evaluate  
258 the magnitude or direction of bias this might induce.

259

260 Differences between diatom assemblages in ecoregions have been shown on a national  
261 (Mykrä et al. 2009, Rimet & Bouchez 2012; Soininen et al. 2004; Tison et al. 2005; Tornés et  
262 al. 2007) and international level (Kelly et al 2012), but are often small. Our CS and ANOSIM  
263 values for diatoms are comparable to literature values between 0.03 and 0.09 for CS (Soininen  
264 et al. 2004, Mykrä et al. 2009) and 0.34 to 0.43 (Kelly et al. 2012, Soininen et al. 2016). None  
265 of the tested typology systems performed well or considerably better than the others.

266 Fish assemblages often exhibit a spatial structure (Jackson and Harvey 1989, Kilgour and  
267 Barton 1999) and accordingly several studies indicated that fish assemblages are well  
268 described by *a priori* typology systems at fine (Hoeinghaus et al. 2007, Vehanen et al. 2020)  
269 and broad scales (Frimpong and Angermeier 2010). The IFE capture fish community  
270 composition best of the evaluated typology systems. The bad performance of the BRT is  
271 surprising, given that longitudinal patterns, which only segment-based typology systems  
272 capture, are common in fish assemblages (Vila-Gispert et al. 2002). The influence of broad-  
273 scale factors and historical distribution patterns seems to override these longitudinal types.

274 Macrophyte community composition tended to be more similar within types (high ANOSIM R)  
275 but the absolute differences in similarity were small (low CS) and the most common taxa

276 tended to occur across types (similar TSA communities). This is in agreement with previous  
277 studies that found considerable overlap between the macrophyte communities of different river  
278 types (Baatrup-Pedersen et al. 2006, Alahuhta and García-Girón 2021). However, the  
279 biogeography of aquatic macrophytes is less well-studied than that of diatoms or fish. Existing  
280 studies tend to focus on lentic systems (Alahuhta et al. 2021) and the driving factors of  
281 macrophyte community composition are known to differ between lentic and lotic systems  
282 (Iversen et al. 2019, Gillard et al. 2020). As for fish, the IFE were the best typology system for  
283 macrophytes in our study.

#### 284 **Relevance of typology systems and ways forward**

285 The availability of well-fitting typology systems matters. Typology systems are not merely  
286 useful heuristics for researchers but crucial to the practical implementation of environmental  
287 policy. Policies need to define quantifiable standards and targets which requires distinct  
288 groups (Mau 2017) to render complex matters legible to institutions (Scott 2008). For example,  
289 a river type-specific risk assessment of chemicals is not achievable on a river by river basis  
290 but would be feasible with a workable set of river types. If we neglect the potentially  
291 international nature of environmental issues, e.g. by relying on national tools to implement  
292 international policies, we risk being blindsided by transboundary harm (Knox 2002) and  
293 transboundary crises (Boin 2009). Together with Jupke et al. (2022), we show that European  
294 river typology systems fail to capture patterns in the community composition of aquatic biota  
295 considerably better than arrangements of simple geometric forms. It is important to note that  
296 the typology systems evaluated here were not designed for the specific purpose we evaluated  
297 them on. Hence, we do not wish to argue against the use of typology systems in general.  
298 Rather, we wish to highlight this research gap and encourage future efforts to close it.

299 To improve the concordance between community composition and river typologies, we  
300 suggest combining segment-based and region-based approaches. Region-based systems do  
301 not capture taxonomic turnover along a rivers course (Vannote et al 1990, Baattrup-Pedersen  
302 et al. 2006), while segment-based systems may fail to account for regional climatic and

303 geological differences (Omernik and Bailey 1997). The BRT include a region variable, but  
304 additional regional differences may be relevant in Europe. Watson et al. (2021) and Jupke et  
305 al. (2022) have previously suggested integrating the BRT into a region-based system. Based  
306 on our results, we recommend combining the BRT with IFE, which were superior to other  
307 typology systems for fish and macrophytes.

308 Further avenues are alternative type descriptors and different ways to derive the types. None  
309 of the typology systems we considered integrate human influences (e.g., land use, pollution,  
310 or hydromorphological alterations). While this would be counterproductive for the delineation  
311 of ecosystems with similar least disturbed conditions, they might improve the fit, if the purpose  
312 is to predict differences in actually occurring communities. Another class of potential type  
313 descriptors are biotic data. They can be used as type descriptors, as is done in the IFE, BGR,  
314 and FEoW. While these three systems are based on expert knowledge about species ranges,  
315 others have created typology systems directly from observations. Several authors have  
316 derived diatom typologies for the evaluation of ecological indices (e.g., Goldenberg Vilar et al.  
317 2014, Grenier et al. 2006, Lavoie et al. 2011, Tang et al. 2016). However, these typologies are  
318 at the subnational scale and were not meant to be generalized beyond their specific studies.  
319 For both macrophytes and fish, biotypes or -regions are commonly derived based on  
320 community composition and structure (e.g., Alahuhta & García-Girón 2021, Holmes et al.  
321 1998, Loewen et al. 2021, Pont et al. 2007, Riis et al. 2000). If the purpose of the typology  
322 system is to define types with similar reference communities though, biotic type descriptors  
323 should not be used to avoid circularity (Bailey et al. 2004, Stoddard et al. 2006). A way to  
324 circumvent this would be to use Generalized Dissimilarity Models (Ferrier et al. 2007, Latombe  
325 et al. 2017) to weight abiotic variables according to the intensity of turnover along their  
326 gradients.

## 327 **5. Conclusion**

328 Our study is the most comprehensive evaluation of European river typology systems to  
329 date. Despite variation across evaluation metrics and taxonomic groups, we showed that

330 current broad-scale typology systems fail to capture the community composition of different  
331 taxonomic groups beyond their spatial autocorrelation. We propose several avenues for  
332 advancing the field. Most notably by combining existing segment- and region-based systems.  
333 In an age of increasing data availability, the context becomes scarce and context is what  
334 ecosystem typologies can provide. Therein lies their great value for ecological research and  
335 environmental policy. When science and policy are restricted to national, fine-scale tools they  
336 remain blind toward issues that transcend political borders. Therefore we encourage future  
337 work toward broad-scale river typologies specifically catered toward delimiting distinct biotic  
338 communities.

### 339 **Declaration of competing interest**

340 The authors declare that they have no known competing financial interests or personal  
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373

## 374 **Data availability**

375 We cannot make the complete database available. We published supplementary data on  
376 Zeondo (currently [here](#)). R code is available in the associated Github repository  
377 <https://github.com/JonJup/European-river-typologies-and-community-compositi>

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