

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

Jonathan F. Jupke¹, Sebastian Birk^{2,3}, Apostolos Apostolou⁴, Jukka Aroviita⁵, Annette Baattrup-Pedersen⁶, Peter Baláži⁷, Libuše Barešová⁸, Saúl Blanco⁹, María Borrego⁹, Herman van Dam¹⁰, Elias Dimitriou¹¹, Christian K. Feld^{2,3}, Maria Teresa Ferreira¹², Gana Gecheva¹³, Joan Gomà¹⁴, Nikola Hanžek¹⁵, Ida Marie Haslev⁶, Tsvetelina Isheva⁴, Aurelien Jamoneau¹⁶, Jenny Jyrkänkallio-Mikkola¹⁷, Maria Kahlert¹⁸, Ioannis Karaouzas¹¹, Satu Maaria Karjalainen⁵, Adriana Olenici⁹, Piotr Panek¹⁹, Petr Paril²⁰, Edwin T.H.M. Peeters²¹, Marek Polášek²⁰, Didier Pont²², Audrone Pumputyte²³, Leonard Sandin²⁴, Lucia Sochuliaková²⁵, Janne Soininen¹⁷, Igor Stanković¹⁵, Michal Straka^{20,26}, Mirela Šušnjara²⁷, Tapio Sutela²⁸, Juliette Tison-Rosebery¹⁶, Marija Gligora Udovič²⁷, Michiel Verhofstad²⁹, Petar Žutinić²⁷, Ralf B. Schäfer¹

1 Institute for Environmental Sciences iES, RPTU Kaiserslautern-Landau, Campus Landau, Fortstraße 7, 76829 Landau, Germany

2 Faculty of Biology-Department of Aquatic Ecology, University of Duisburg-Essen, 45117 Essen, Germany

3 Centre for Water and Environmental Research (ZWU), University of Duisburg-Essen, 45117 Essen, Germany

4 Department of Aquatic Ecosystems, Institute for Biodiversity and Ecosystem Research at the Bulgarian Academy of Sciences, Gagarin 2, Sofia 1113, Bulgaria

5 Finnish Environment Institute, Paavo Havaksen tie 3, 90570 Oulu, Finland

6 Aarhus University, Department of Ecoscience, Aarhus, Denmark

7 Water Research Institute, Nabr. arm. gen. L. Svobodu 7, 81249 Bratislava, Slovakia

8 Czech Hydrometeorological Institute, Na Šabatce 17, 143 06 Praha, Czech Republic

9 Diatom Lab. Universidad de León. La Serna 58 - E24007 Leon, Spain

10 Consultancy for Water and Nature, Spyridon Louisweg 141, 1034 WR Amsterdam, The Netherlands.

11 Institute of Marine Biological Resources & Inland Waters, Hellenic Centre for Marine Research, 46.7 km Athens-Sounio Av., 19013 Anavyssos, Attica, Greece

12 Forest Research Centre and Associate Laboratory TERRA, Instituto Superior de Agronomia, Tapada da Ajuda, 1349-017 Lisboa, Portugal

13 Faculty of Biology, Plovdiv University, Tsar Asen 24, 4000 Plovdiv, Bulgaria

14 Faculty of Biology, Universitat de Barcelona, Av. Diagonal 643, 08028 Barcelona, Spain

15 Josip Juraj Strossmayer Water Institute, Ulica grada Vukovara 220, HR-10000 Zagreb, Croatia

16 INRAE, UR EABX, F-33612 Cestas, France

17 Department of Geosciences and Geography, PO Box 64, FI-00014 University of Helsinki, Finland

18 Swedish University of Agricultural Sciences, Department of Aquatic Sciences and Assessment, PO Box 7050, 750 07 Uppsala, Sweden

19 Chief Inspectorate for Environmental Protection, ul. Bitwy Warszawskiej 1920 r. 3, 02-362 Warszawa, Poland

20 Department of Botany and Zoology, Faculty of Science, Masaryk University, 61137, Brno, Czech Republic

50 **21** Aquatic Ecology and Water Quality Management Group, Wageningen University,
51 P.O. Box 47, 6700 AA Wageningen, The Netherlands
52 **22** Institute of Hydrobiology and Aquatic Ecosystem Management, University of
53 Natural Resources and Life Sciences, Gregor-Mendel-Strasse 33, Vienna, Austria
54 **23** Aplinkos apsaugos agentūra, A Juozapavičiaus g.9, 09311 Vilnius, Lithuania
55 **24** Norwegian institute for nature research (NINA), Lillehammer, Norway
56 **25** Department of Biology and Ecology, Faculty of Natural Sciences, Matej Bel
57 University, Tajovského 40, 97401, Banská Bystrica, Slovakia
58 **26** T.G. Masaryk Water Research Institute, 612 00, Brno, Czech Republic
59 **27** Faculty of Science, University of Zagreb, Horvatovac 102a, HR-10000 Zagreb,
60 Croatia
61 **28** Natural Resources Institute Finland, Paavo Havaksen tie 3, 90570 Oulu, Finland
62 **29** FLORON: Plant Conservation Netherlands, Toernooiveld 1, 6525ED, the
63 Netherlands

64
65 Corresponding Author: Jonathan F. Jupke, jonjup@protonmail.com
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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. Introduction

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

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

Ecologists commonly use ecosystem typologies to delineate ecosystems with similar biological communities. These typology systems usually defined contiguous areas (regions) as mapping units and focused on terrestrial ecosystems. Region-based typologies are appropriate for terrestrial (Olson et al. 2001) and marine ecosystems (Spalding et al. 2007), 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 (Aguilar 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 (Illies 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 et 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 Globevnik
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 km², medium-large 100–10.000 km² and very large >10.000 km². 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.

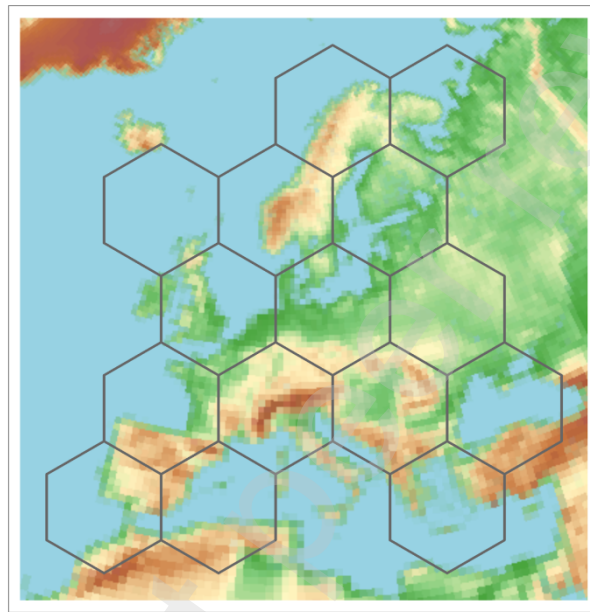
ID	Name
RT1	Very large rivers
RT2	Lowland, calcareous or mixed, medium-large
RT3	Lowland, calcareous or mixed, very small-small
RT4	Lowland, siliceous incl. organic, medium-large
RT5	Lowland, siliceous incl. organic, very small-small
RT6	Mid-altitude, calcareous incl. organic, medium-large
RT7	Mid-altitude, 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

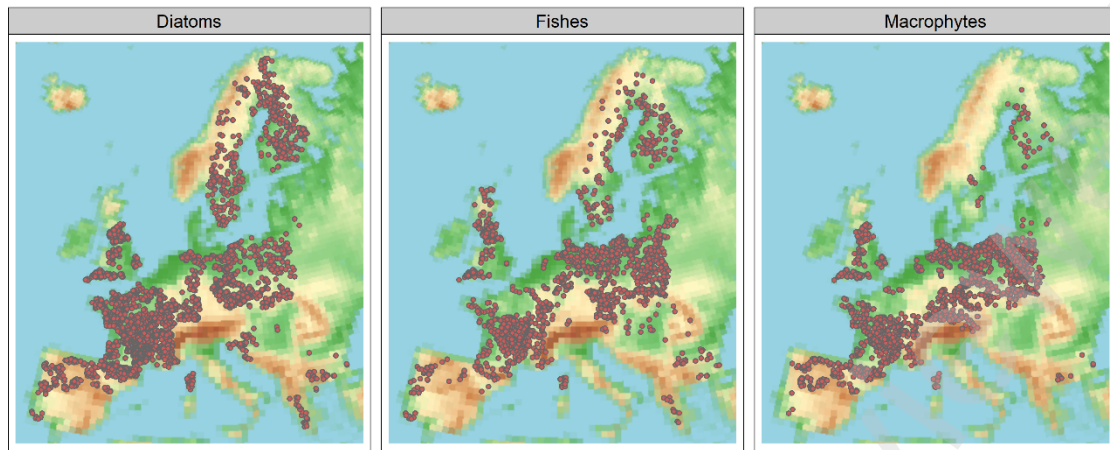
100 with 15 cells (types) over Europe (Figure 1). We chose 15 types as this approximately matches
101 the average number of types from the other typology systems. To evaluate whether size or
102 form impact the results, we created multiple SA classifications. Besides the 15-cell hexagonal
103 classification, 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 S13 for analysis). We included samples taken
118 between May and September and only included the most recent sample at each site (Figure
119 3).

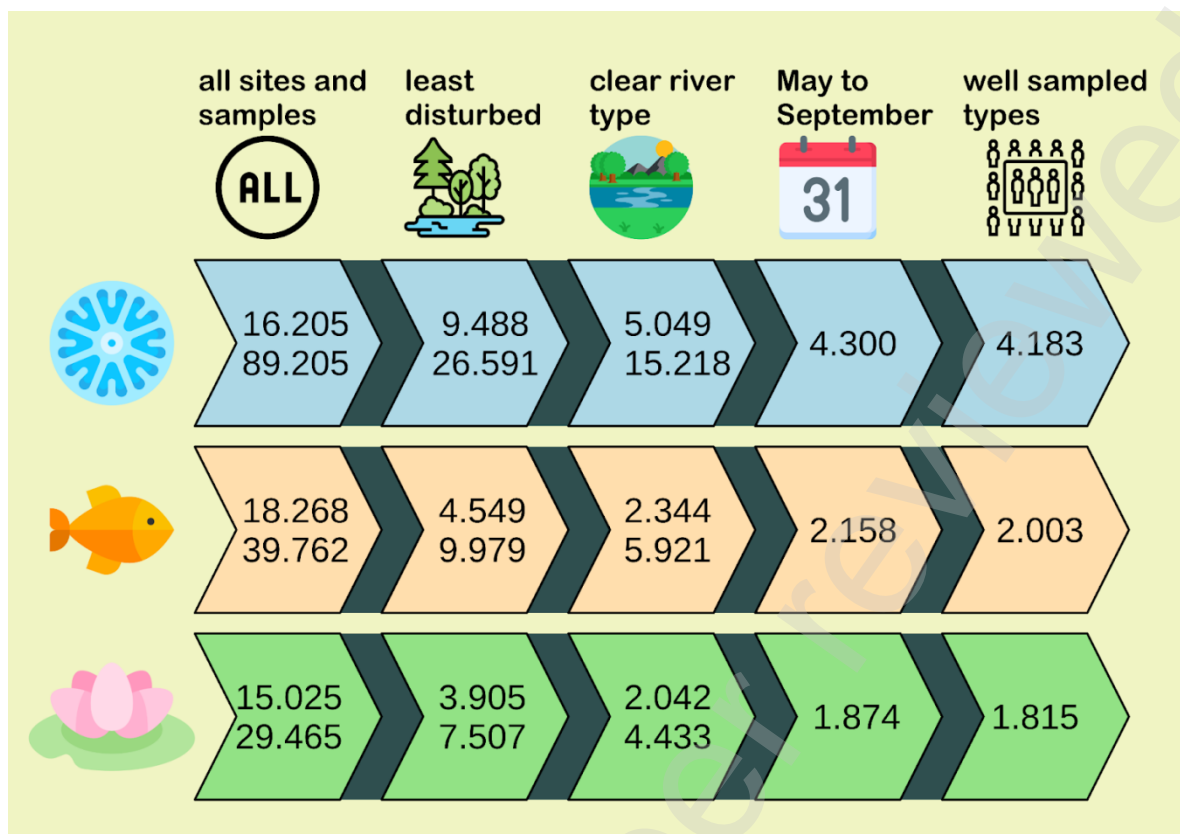


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121 *Figure 2: Spatial distribution of sampling sites for diatoms, fishes, and aquatic macrophytes. The map only shows sampling*
 122 *sites that we deemed least disturbed and could assign unambiguously to one stream in the digital representation of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



137
 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 assigned 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.*

146
 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 ≤ 0.24 as least disturbed (see
 154 SI4 for rationale). In addition, we removed samples with ≤ 10 diatom species, ≤ 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 CaronDB.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 ζ), 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 ≥ 0.5 (Clark & Gorley 2006) and to outperform the
169 SA classifications. CS is the difference between mean similarity within-types (\underline{W}) and mean
170 similarity among-types (\underline{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

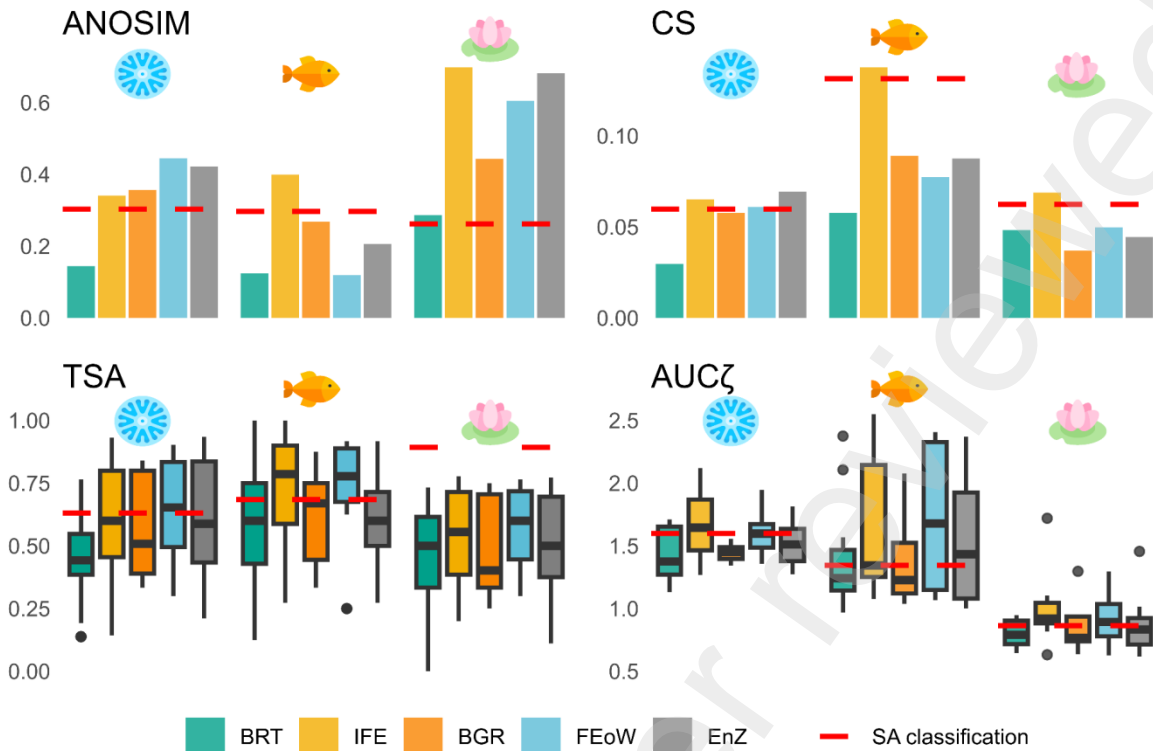
182 given number of sites (Hui and McGeoch 2014). Zeta diversity extends to multi-site
183 comparisons through the number of considered sites, the zeta degree. For example, ζ_3 is the
184 average number of species shared between three sites. Zeta diversity decreases
185 monotonically with increasing 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 ζ). 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 ζ 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).

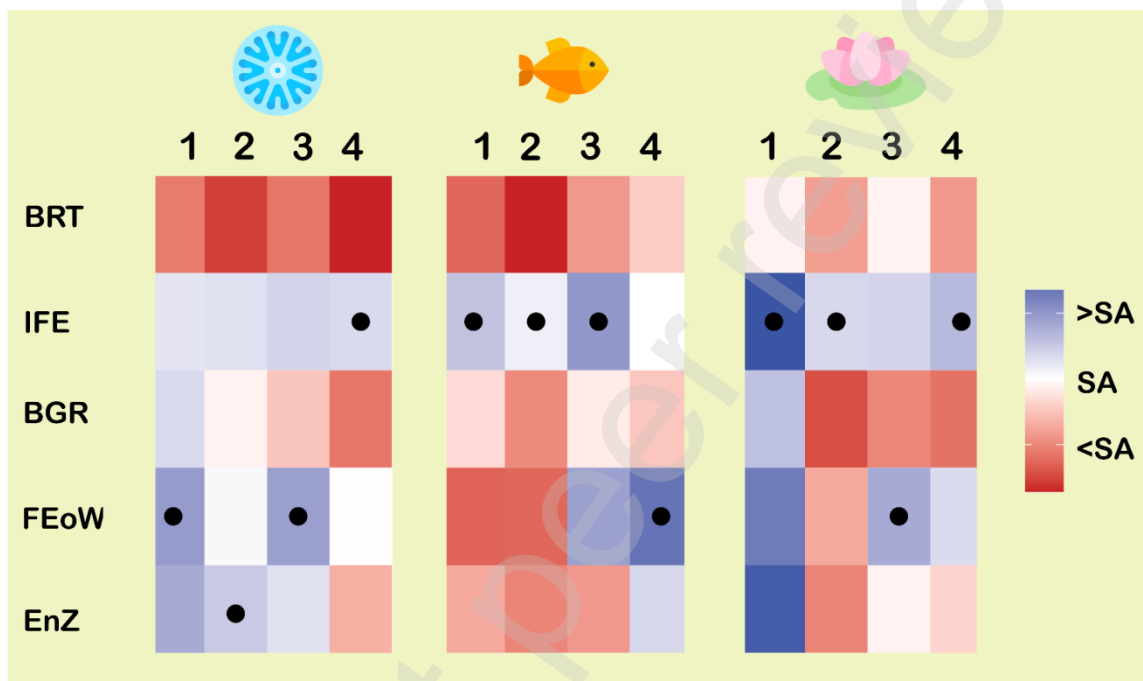


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Figure 4: Results of the four cluster analyses, Analysis of similarities (ANOSIM), Classification Strength (CS), Typical Species Analysis (TSA), and area under the zeta diversity decline curve (AUC ζ), for the Broad River Types (BRT), Illies Freshwater 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 ζ , 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.

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 (FEoW), 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, FEOw 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, Baatrup-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
341 relationships that could have appeared to influence the work reported in this paper.

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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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