Multi-decadal improvements in the ecological quality of European rivers are not consistently reflected in biodiversity metrics

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Multi-decadal improvements in the ecological quality of European rivers are not consistently reflected in biodiversity metrics

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Humans impact terrestrial, marine and freshwater ecosystems, yet many broad-scale studies have found no systematic, negative biodiversity changes (for example, decreasing abundance or taxon richness). Here we show that mixed biodiversity responses may arise because community metrics show variable responses to anthropogenic impacts across broad spatial scales. We first quantified temporal trends in anthropogenic impacts for 1,365 riverine invertebrate communities from 23 European countries, based on similarity to least-impacted reference communities. Reference comparisons provide necessary, but often missing, baselines for evaluating whether communities are negatively impacted or have improved (less or more similar, respectively). We then determined whether changing impacts were consistently reflected in metrics of community abundance, taxon richness, evenness and composition. Invertebrate communities improved, that is, became more similar to reference conditions, from 1992 until the 2010s, after which improvements plateaued. Improvements were generally reflected by higher taxon richness, providing evidence that certain community metrics can broadly indicate anthropogenic impacts. However, richness responses were highly variable among sites, and we found no consistent responses in community abundance, evenness or composition. These findings suggest that, without sufficient data and careful metric selection, many common community metrics cannot reliably reflect anthropogenic impacts, helping explain the prevalence of mixed biodiversity trends.

Reports of human-driven species extinctions[1,2] and environmental change[3–5] indicate widespread degradation of Earth's ecosystems, particularly freshwaters[6]. However, a growing number of continental- and global-scale temporal studies of freshwater, terrestrial and marine communities have found no evidence of systematic, negative biodiversity changes[7–15], instead reporting a mixture of negative, positive and neutral changes. Such studies typically infer that negative biodiversity changes (often defined as declining abundance or taxon richness) indicate anthropogenic degradation[8,16,17], whereas positive changes indicate improving environmental quality[13,18,19]. Studies finding mixed biodiversity changes therefore suggest a balance of degradation, improvement and no change[7,9,11,12]. These studies have spurred debate about whether anthropogenic impacts are truly mixed[20] and the role methodological issues play in producing mixed biodiversity trends, including issues of poor data quality, quantity and representativeness[21–23].

One unaddressed explanation for the prevalence of mixed biodiversity trends is that common metrics used to summarize community change, such as abundance or taxon richness, cannot reliably indicate

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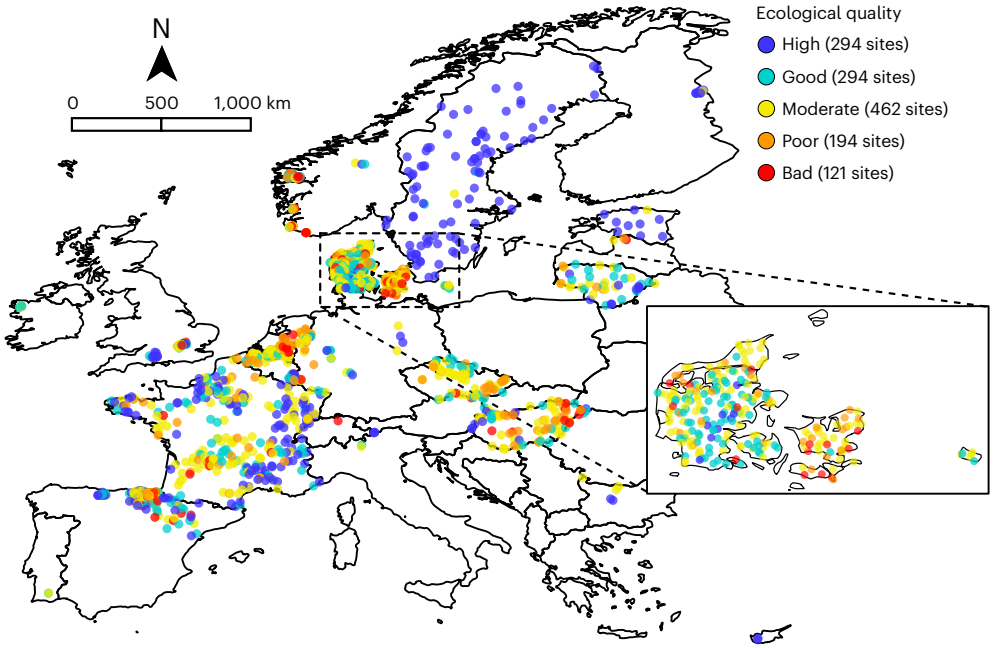

Fig. 1 | Locations and ecological quality of 1,365 river sites across Europe. Site colours indicate biomonitoring assessments of the ecological quality of the invertebrate community in the first year of sampling (calculated as the EQC; Methods). The EQCs of some densely clustered sites are hidden, as illustrated for Denmark. © EuroGeographics for the administrative boundaries[68].

anthropogenic impacts. This unreliability may occur because different stressors can have contrasting effects, species have varying tolerances to different stressors, and communities have different historical and environmental contexts that influence their response[24]. For example, anthropogenic impacts may drive declines in community abundance or richness in some localities[4,17,25], whereas others may show no overall change if gains match losses[26] or may even show increases when tolerant species proliferate[25,27]. Consequently, the mixed biodiversity trends found by many studies may be a result of variable community responses to anthropogenic impacts. An additional complication is that most biodiversity studies lack the pre-disturbance baseline data needed to contextualize observed trends. Without baselines, it is difficult to determine whether biodiversity changes result from anthropogenic impacts or from natural fluctuations[21,22], changing baselines or as a statistical artefact of the period chosen for analysis[28].

To better understand what broad-scale studies of local biodiversity trends can and cannot tell us, we must determine whether any aspect of biodiversity can consistently reflect anthropogenic impacts across broad spatial scales. Doing so involves first quantifying the degree of impact across numerous communities, which requires comparisons to unimpacted conditions. Next, variability in impact should be related to variability in common biodiversity metrics that summarize communities, such as abundance, taxon richness, evenness and temporal turnover, to determine whether any relationships are consistent across finer (for example, within sites and regions) to broader (for example, continental) spatial scales. Ideally, such analyses would use high-quality time-series data collected from similar taxa and habitats using similar sampling methodologies to ensure comparability[21,22,29].

While no dataset can perfectly fulfil these requirements, a feasible solution is to use organisms commonly collected by biomonitoring programs, such as riverine invertebrates[30]. Biomonitoring programs can compensate for missing historical baselines by replacing them with modelled or collected data from other minimally or least-impacted 'reference' communities. The degree of impact is then quantified using an index of the similarity between sampled versus reference communities, which we hereafter refer to as 'ecological quality'. Ecological quality indices provide what many biodiversity studies are typically missing, specifically a consistent measure of how communities have changed compared to reference conditions, with greater deviation indicating more severe anthropogenic impacts regardless of differences in stressors or community contexts. This index can then be related to common community metrics to identify any consistent associations. Moreover, established biomonitoring indices summarizing components of community composition, such as the occurrence of sensitive taxa[30,31], provide support for environmental changes inferred from changes in ecological quality. Last, riverine invertebrates have been sampled for decades worldwide following standardized methodologies[30], enabling robust time-series analyses and ensuring that the same taxa from the same habitats are compared.

In this Article, we used riverine invertebrate biomonitoring data from 1,365 sites across 23 European countries (Fig. 1) to fill the knowledge gaps outlined above. First, to characterize changes in anthropogenic impacts, we quantified temporal trends (1992–2019) in ecological quality at continent, country and site spatial scales. These analyses determine how communities have changed relative to baseline conditions and provide a European-scale assessment of long-term trends in ecological quality. Second, to identify community metrics that consistently reflect anthropogenic impacts, we related ecological quality to common metrics summarizing community abundance, biodiversity (for example, richness) and composition, and to common biomonitoring indices that reflect the occurrence of sensitive taxa.

Results

Continental-scale trends in ecological quality

Ecological quality was measured using ecological quality ratios (EQRs) and ecological quality classes (EQCs), as defined by the European Union Water Framework Directive (WFD[32]). EQRs are a continuous ratio of the similarity between sampled and least-impacted reference invertebrate communities. EQRs range from 0 (0% similarity) to 1 (100% similarity), and the values within this range are allocated into one of five numeric EQCs of 1 (high), 2 (good), 3 (moderate), 4 (poor) or 5 (bad) based on country-specific classification systems (detailed in Supplementary Table 1). EQCs are used to determine whether a given invertebrate community has satisfied the WFD target of achieving a 'good' or 'high'

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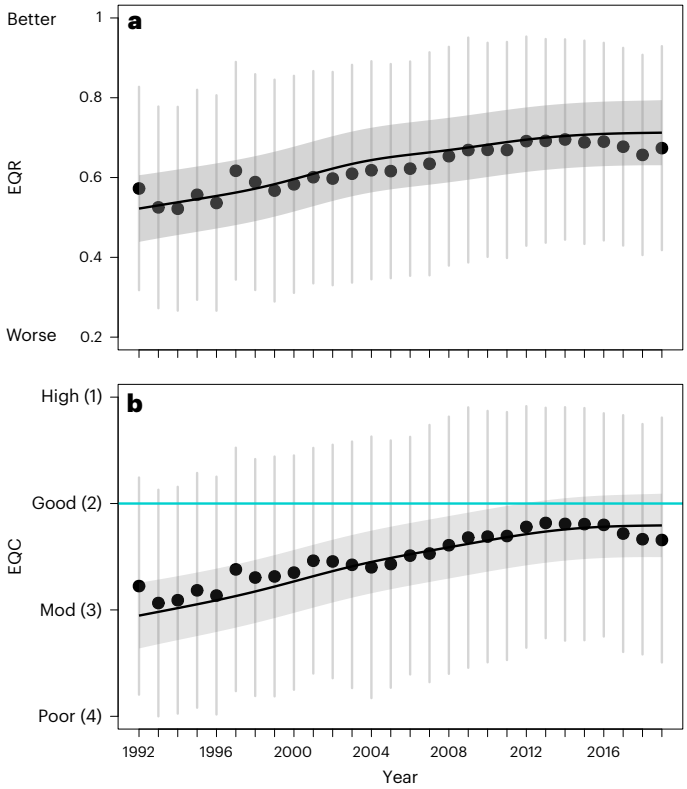

Fig. 2 | Continental-scale trends in ecological quality. Trends in EQRs (n = 19,660) (**a**) and EQCs (Mod, moderate; n = 19,697) (**b**) across 1,365 European riverine invertebrate communities during 1992–2019. Black points and grey vertical lines respectively indicate the annual means and standard deviations. Fitted relationships (black line) and 95% confidence intervals (grey background) were based on GAMM output. The European Union WFD target of a 'good' EQC is indicated by a light blue line in **b**. The 'bad' EQC (class 5) is not plotted.

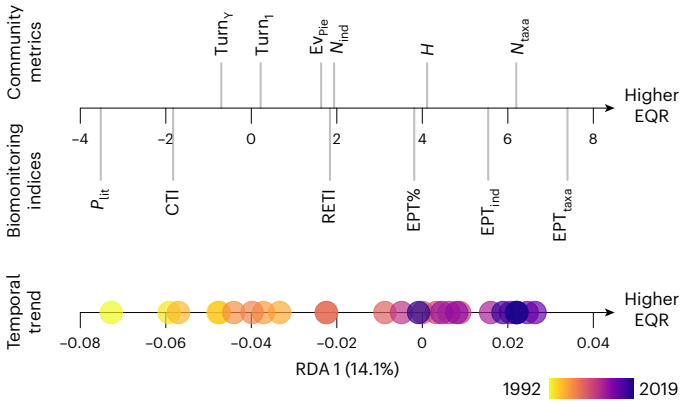

Fig. 3 | Continental-scale links between ecological quality, community metrics and biomonitoring indices. Redundancy analysis (RDA) of the continental-scale relationship between EQRs (black arrows) and the community metrics and biomonitoring indices (upper arrow); temporal trends in metrics/indices are also shown during 1992–2019 (lower arrow). Metrics/indices with higher or lower scores on RDA axis 1 indicate stronger relationships to ecological quality, with 0 indicating no relationship. The community metrics comprise abundance (N_{ind}), richness (N_{taxa}), evenness (Ev_{Pie}), Shannon diversity (H) and temporal turnover between consecutive years ($Turn_Y$) and compared to the first year ($Turn_1$). The biomonitoring indices comprise the total abundance (EPT_{ind}), and proportion ($EPT_{\%}$) and richness (EPT_{taxa}) of EPT, in addition to the Community Temperature Index (CTI), the proportion of littoral taxa (P_{lit}) and the Rhithron feeding type index (RETI; Extended Data Table 1). Temporal trends are visualized as the centroid position of all sites in each year and are coloured from earlier (yellow) to later (purple) years.

ecological quality status, whereas EQRs are better suited for statistical analyses because they provide a more precise representation of community similarity to the references. Some uncertainties exist in the degree to which EQRs/EQCs represent all anthropogenic environmental changes[33]. However, they are well-established measures of general impact[34] that are assumed to be comparable across countries[35].

Based on generalized additive mixed models (GAMMs), ecological quality (represented as both EQRs and EQCs) improved across our sites from 1992 until around the 2010s, evidenced by significant smoothed year terms in models for both EQRs (Wald test, n = 19,660, effective degrees of freedom (e.d.f.) = 5.06, F = 69.00, P < 0.001) and EQCs (Wald test, n = 19,697, e.d.f. = 3.98, F = 86.80, P < 0.001; Fig. 2). EQRs initially improved by around 0.006–0.013 yr^{-1}, with EQCs improving by about 0.035–0.05 classes per year (Extended Data Fig. 1). However, little to no change occurred after the early 2010s when EQRs plateaued around 0.7 and EQCs plateaued around 2.2, which is just below the target of a 'good' EQC value of 2 set by the WFD[32] (Fig. 2). These trends were robust to the inclusion or exclusion of individual countries, despite differences in time-series length among countries (Supplementary Table 2 and Supplementary Fig. 1).

Continental-scale metrics and indices

Ecological quality was moderately related to the community metrics and biomonitoring indices (based on a significant global permutation test; n = 19,654, $F_{1,19653}$ = 3,214.7, R^2 = 0.14, P = 0.001). Specifically, improvements in ecological quality from 1992 to the 2000s were most associated with increases in taxon richness, Shannon diversity and the Ephemeroptera, Plecoptera and Trichoptera (EPT) indices (Fig. 3), with increased EPT indicating improved water quality and habitat conditions (Extended Data Table 1). Other community metrics and biomonitoring indices showed weaker or no relationships (Fig. 3), excluding the Average Score Per Taxon (ASPT) index and the Saprobic Index, which were not included because not all countries use them.

A caveat to these results is that certain countries calculate ecological quality using multiple metrics and indices, which can partly incorporate those we analysed, particularly taxon richness, Shannon diversity, EPT richness, and the ASPT index (used in around 20–40% of sites; detailed in Supplementary Table 1). This results in a potential problem of circularity, although ecological quality can change even if some of its composite metrics do not (or vice versa) because multiple metrics/indices are typically used. To test for the influence of this circularity, we removed sites that use potentially circular metrics/indices and repeated our analyses. The removal did not substantially influence our results (Supplementary Figs. 2 and 3).

Country-scale trends in ecological quality

We quantified country-scale temporal changes in ecological quality and its relationships to the community metrics and biomonitoring indices for 15 countries with adequate data to parameterize individual models, which represented 99% of the sites. The continental-scale temporal improvements in ecological quality were driven by improvements in communities from Belgium, Denmark, France, Hungary, Norway and Spain (Fig. 4 and Supplementary Tables 3 and 4). Between 40% and 85% of the sampled communities from these countries were at good or high EQCs in their most recent year of sampling. Modelled temporal relationships for the EQC values indicated improvements from 3 to 2.2 in Belgium, from 2.9 to 2.2 in Denmark, from 2.4 to 1.5 in France, from 3.1 to 2.7 in Hungary, from 3.3 to 2.6 in Norway and from 3.8 to 2.2 in Spain (Fig. 4c). Conversely, we found no statistical evidence of improvements in the other countries, such as Czechia (EQCs remained stable around 3.2), Ireland (2.8), Lithuania (2.3), the Netherlands (3.5) and Sweden (1.0; Fig. 4d). Based solely on trendlines, ecological quality

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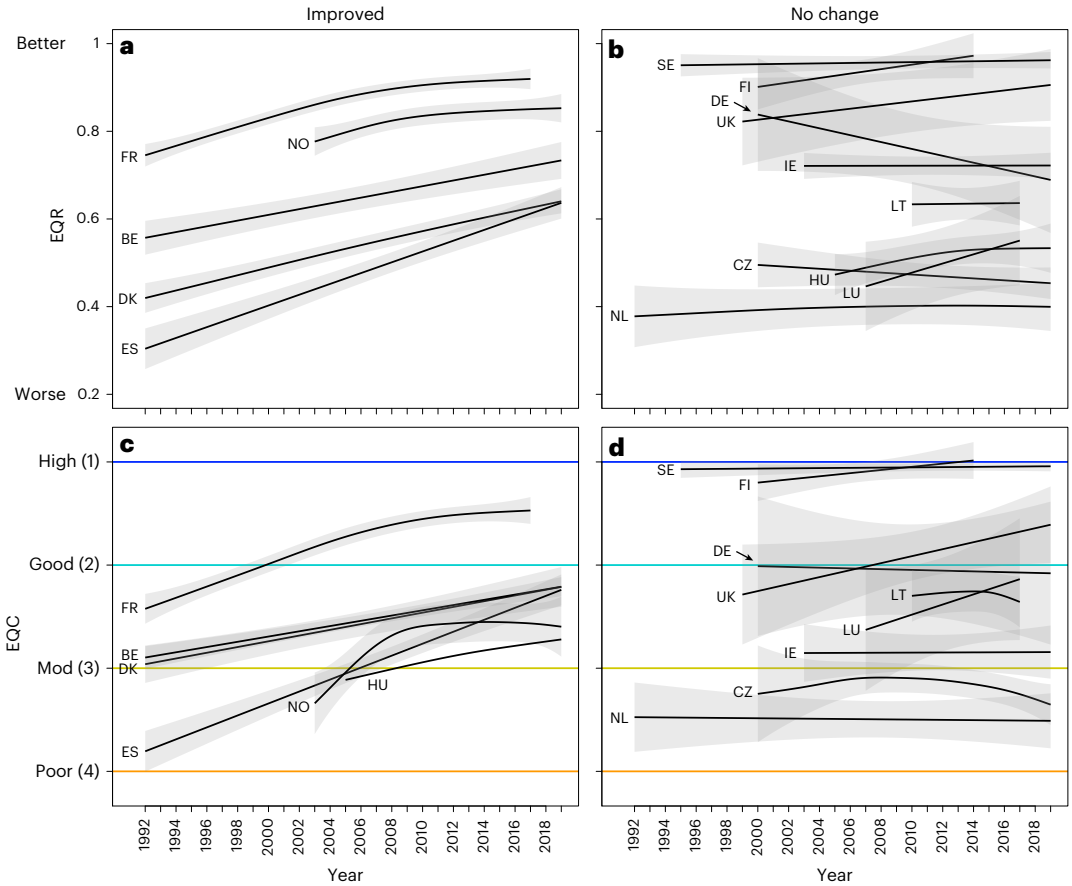

Fig. 4 | Country-scale trends in ecological quality. Temporal changes in EQRs (**a**,**b**) and EQCs (**c**,**d**) across European riverine invertebrate communities from 15 countries during 1992–2019. Fitted relationships (solid lines) and 95% confidence intervals (grey backgrounds) are based on GAMM outputs for Belgium (BE), Czechia (CZ), Denmark (DK), Finland (FI), France (FR), Germany (DE), Hungary (HU), Ireland (IE), Lithuania (LT), Luxembourg (LU), the Netherlands, (NL), Norway (NO), Spain (ES), Sweden (SE) and the UK. Countries in **a** and **c** have statistical evidence for improvements over time, whereas those in **b** and **d** have no evidence for change. EQC categories (**c**,**d**) are illustrated using coloured lines for 'high' (dark blue), 'good' (light blue), 'moderate' (yellow) and 'poor' (orange) classes. The 'bad' EQC (class 5) is not plotted.

may be improving in Luxembourg (modelled EQCs changed from 2.6 to 2.1 during 1992 through 2019), Finland (1.2 to 1.0) and the UK (2.3 to 1.6), versus degrading in Germany (2.0 to 2.1; Fig. 4d). However, these patterns were non-significant (Supplementary Tables 3 and 4) with large confidence intervals.

Country-scale metrics and indices

Taxon richness and Shannon diversity were the community metrics that showed the strongest relationships to changes in ecological quality in most countries (Fig. 5 and Extended Data Fig. 2), aligning with the continental-scale patterns, but trends varied spatially. For example, richness showed less change in relation to ecological quality in Germany (Fig. 5c), Denmark (Fig. 5d) and particularly the Netherlands (Fig. 5i) compared to the other countries, with similarly weaker relationships for Shannon diversity in Denmark (Fig. 5d), France (Fig. 5f) and the Netherlands (Fig. 5i). In addition, the degree to which the community metrics were related to ecological quality varied widely among countries, from ecological quality explaining almost 30% of the total variation in metrics/indices in some countries (for example, Lithuania; Fig. 5h) down to less than 10% in others (for example, Sweden; Fig. 5j). These spatial differences indicate that the community metrics varied more in relation to ecological quality in some regions (those with more explained variation) versus less in others.

Ecological quality was always positively related to biomonitoring indices of water/habitat quality in all countries, specifically EPT richness and the ASPT index in countries that use this index. Relationships to other indices were country specific (see Supplementary Figs. 4–7 for statistics). For example, in Czechia (Fig. 5b), Denmark (Fig. 5d), Spain (Fig. 5e) and the Netherlands (Fig. 5i), years with better ecological quality were more strongly associated with a lower proportion of taxa with preferences for littoral habitats compared to other countries, which could indicate a stronger influence of flow alteration in these regions (Extended Data Table 1). Similarly, ecological quality was more strongly associated with the Community Temperature Index and the Saprobic Index in Germany (Fig. 5c) and the Netherlands (Fig. 5i), which may indicate a stronger influence of warming and organic pollution.

Site-scale ecological quality, metrics and indices

Site-scale ecological quality trends were often positive (Fig. 6; 37% with positive slopes and confidence intervals that did not overlap 0), aligning with the general improvements shown in our other analyses. However, 57% of sites showed no strong evidence for change, indicating substantial site-scale variability in whether ecological quality was changing. The ecological quality of 6% of all sites also tended to decrease over time, which encompassed sites in 17 of the total 23 countries.

Site-scale temporal changes in ecological quality showed the strongest relationships to changes in taxon richness (Fig. 6a) followed by Shannon diversity (Fig. 6b), with weaker to no relationships to all other community metrics (Fig. 6). However, even the more consistent relationships varied widely among sites, as evidenced by generally low R^2 values. For example, 24% of sites showed the same direction of

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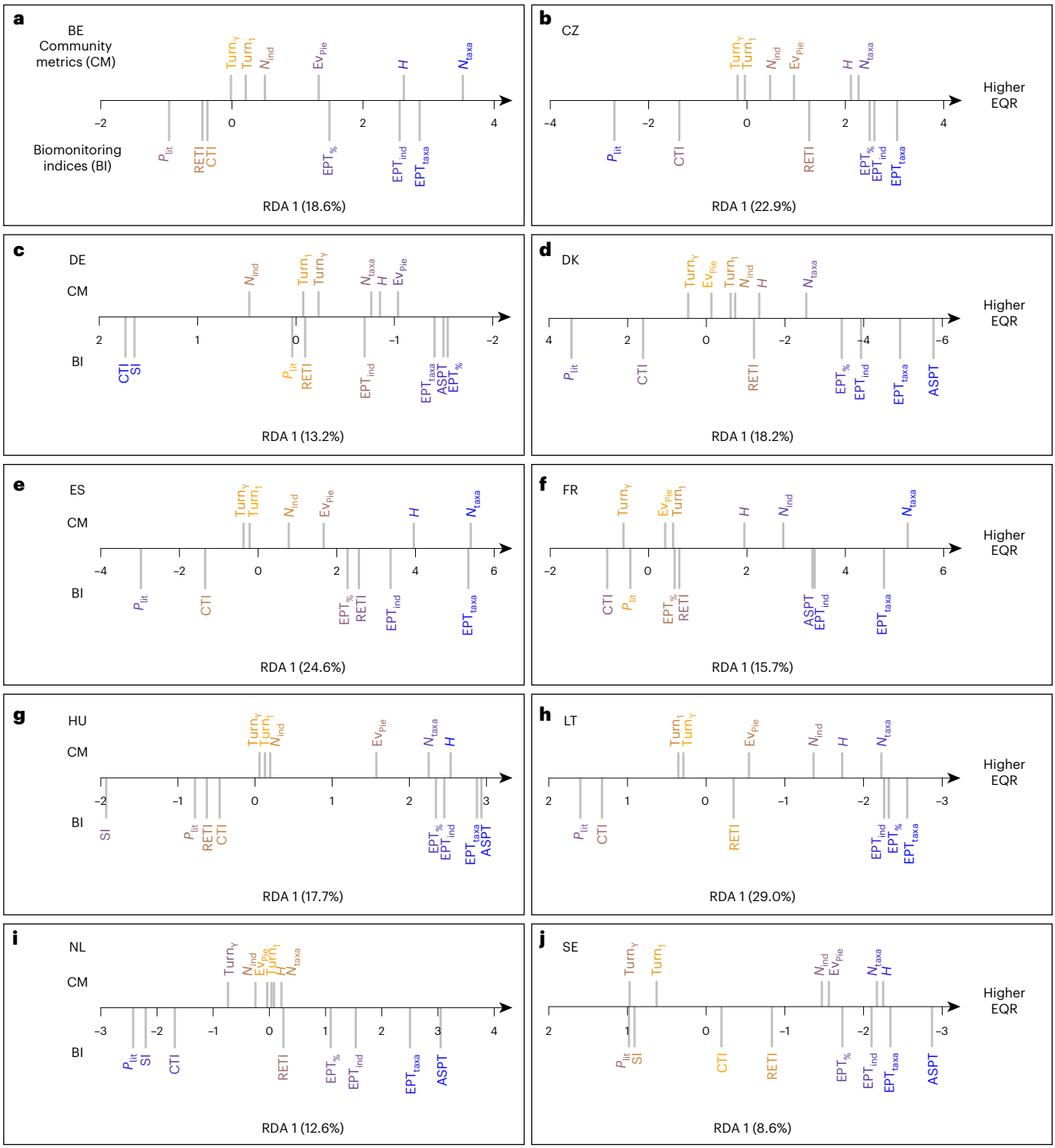

Fig. 5 | Country-scale links between ecological quality, community metrics and biomonitoring indices. RDA of the relationship between EQRs (black arrows) and community metrics (CM) and biomonitoring indices (BI) for BE (**a**), CZ (**b**), DE (**c**), DK (**d**), ES (**e**), FR (**f**), HU (**g**), LT (**h**), NL (**i**) and SE (**j**). Only the ten countries with the most comprehensive datasets are plotted (see Extended Data Fig. 2 for the other countries). The community metrics comprise N_{ind}, N_{taxa}, EV_{Pie}, H, $Turn_Y$ and $Turn_1$. The biomonitoring indices comprise the EPT_{ind}, $EPT_{\%}$ and EPT_{taxa}, in addition to the ASPT index, the CTI, the P_{lit}, the Saprobic Index (SI), and the RETI (Extended Data Table 1). Metrics and indices are coloured from orange to blue based on their respective weaker to stronger relationships to ecological quality, quantified based on their loadings on RDA axis 1.

change in both richness and ecological quality (here 'change' means a slope value whose confidence intervals do not overlap 0), but 31% showed no change in richness when ecological quality changed or vice versa, and 2% showed opposing changes (Fig. 6a and Extended Data Table 2). This variability was more pronounced in community metrics with weaker relationships to ecological quality and lower R^2 values, such as Shannon diversity for which only 11% of sites showed matching relationships and 43% showed no match; that is, either Shannon

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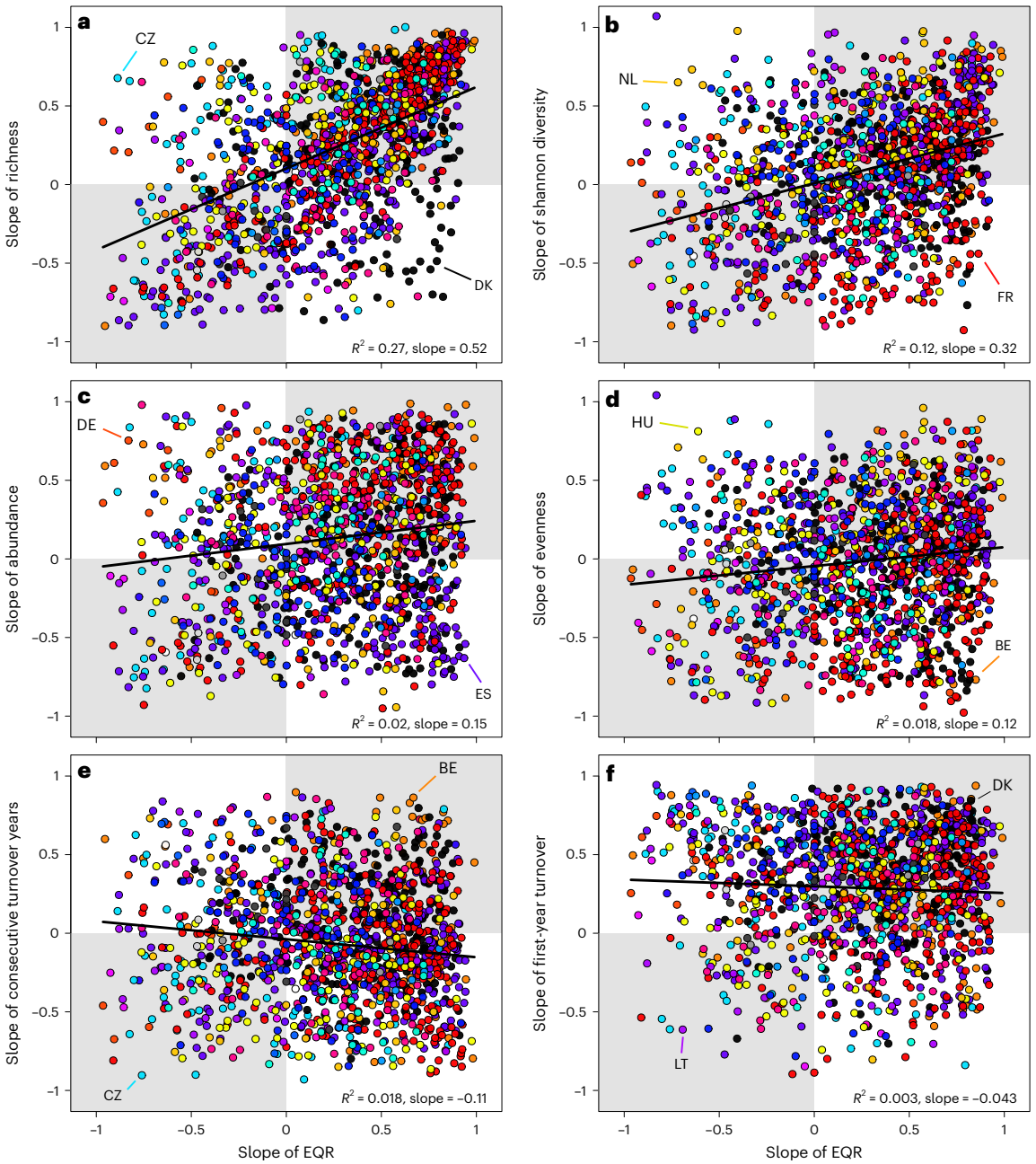

Fig. 6 | Site-scale links between ecological quality and community metrics. Relationships between the slopes of the EQR at each site and the slopes of taxon richness (**a**), Shannon diversity (**b**), abundance (**c**), evenness (**d**), temporal turnover between consecutive years (**e**) and temporal turnover between each year and the first year (**f**). Sites with matching ecological quality and metric trends are in the grey shaded areas, whereas opposing relationships are in the white areas. Sites are coloured by country, and some example countries with sites that show strong opposing relationships to the overall trend are indicated with coloured lines. Best-fit lines (black), R^2 values and estimated slopes are based on the associated linear mixed models.

diversity did not change when ecological quality did, or vice versa (Fig. 6b). Of the biomonitoring indices, ecological quality primarily showed positive relationships to the EPT and ASPT indices and particularly to EPT richness (Extended Data Fig. 3).

Discussion

Our results have important implications for upscaling local biodiversity trends into broader inferences about anthropogenic impacts and for monitoring and analysing biodiversity change. Many studies report various changes in animal and plant biodiversity and use these changes to infer likely drivers. For example, several studies report positive changes in European freshwater[11–13,18,31] and marine biodiversity[11,19] and suggest these trends reflect improvements in water and habitat quality. However, questions remain about potential issues with making such linkages, including problems in analysing sporadic biodiversity time series with missing baselines[21,22], variable community responses and the quality and representativeness of the underlying datasets (for example, ref. 35). Our findings help to resolve these questions using European riverine invertebrates as a case study. We found that ecological quality generally increased from the 1990s to 2010s, as did the number of sensitive taxa, indicating reduced anthropogenic impacts, albeit the required 'good' ecological status has not yet been achieved on average. Better ecological quality likely occurred owing to European policies introduced in the 1990s and 2000s to reduce pollution, such

as through improved wastewater treatment[36]. Increases in ecological quality then plateaued after the 2010s. Further research is needed to determine why improvements stalled[18], but likely candidates include unaddressed stressors, such as diffuse pollution and physical habitat modification[37,38], intensifying stressors such as climate change[39] and emerging stressors such as new pesticides, pharmaceuticals and other substances[40,41]. Regardless of the specific drivers, the general improvements we found in ecological quality match positive changes in European freshwater biodiversity reported by other studies during the same period[11,12]. This match illustrates that, despite different stressors and stressor effects, biodiversity can show consistent responses to anthropogenic impacts across broader spatial scales.

Improvements in ecological quality showed the strongest relationships to increases in taxon richness across all spatial scales, suggesting that richness could be a reliable broad-scale indicator of anthropogenic impacts. Richness is commonly used in biodiversity and biomonitoring assessments for a range of taxonomic groups partly owing to the comparative ease of data collection and metric calculation[42,43]. However, its usefulness is debated because it requires a harmonized taxa list across regions, it does not reflect compositional changes, its response depends on the spatial scale of study, and the baseline data to contextualize how and why richness has changed is generally lacking[21–23,27,44]. Despite these limitations, our results indicate that richness can provide meaningful insights into general patterns of anthropogenic impacts (other studies have found similar results[4]). This relationship likely occurred in our dataset because better river conditions can increase richness by increasing habitat quality, quantity and heterogeneity[45,46] and by increasing the presence of pollution-sensitive species[30,31]. This association may apply beyond riverine invertebrates given that taxon richness is often positively related to habitat quality for a variety of taxonomic groups[4,47,48]. Studies that identified mixed changes in local richness across large geographic areas, including in plants, fishes, birds, mammals and terrestrial insects[7–9,25,49,50] may therefore be correct when inferring a similar mixture of negative impacts, improvement and no change. These studies still, however, suffer from issues of representativeness. For example, sampling an equal mixture of degrading and improving sites will undoubtedly produce mixed responses, but this does not mean the sampled sites represent the state of anthropogenic impacts across the globe. Similarly, data are often missing for certain continental regions, such as the Mediterranean in our dataset, and from outside North America, Europe or Oceania[21,22]. These limitations mean that further work is required to evaluate the degree of anthropogenic impact, and the usefulness of richness as a broad-scale indicator, across different major biogeographic regions.

While richness was broadly positively related to ecological quality, this relationship was highly variable among countries and at the site scale, with most sites showing no response or even negative relationships. No change in richness as ecological quality changes could occur at sites where taxa losses are balanced by gains[26]. Alternatively, richness may change even when ecological quality does not due to natural population declines and colonization processes[20,26] or human-driven species introductions and range expansions[25]. We also observed some opposing relationships. For example, sometimes worsening impacts were associated with higher richness, which can occur when tolerant species and non-natives establish[51], or sometimes improvement was associated with lower richness, which may occur if declines in tolerant taxa outweigh gains in sensitive species[27]. This response variability highlights that richness may be a reliable indicator of impact across broader spatial scales, but this requires a large quantity of data to control for high spatial heterogeneity in responses among sampling sites[23]. Smaller-scale studies or those with less data may therefore find richness to be an inconsistent indicator of anthropogenic impacts[8,26,52]. Furthermore, the high spatial variability we found in richness responses suggests that studies may not be able to decompose broader-scale richness trends into finer-scale categories, such as by different regions, taxonomic groups or habitat types[7,11,49] and assume that richness responds similarly to anthropogenic impacts across categories[53,54]. Accounting for variability in richness responses may be best accomplished through multimetric approaches that combine changes in two or more metrics, such as richness and a composition metric. This approach better captures changes in different aspects of each community, which may more reliably reflect anthropogenic impacts and provide more consistent information for management and conservation[55].

Most community metrics, specifically Shannon diversity, abundance, evenness and temporal turnover, showed little to no general relationship to ecological quality. This result shows how changes in anthropogenic impacts can fail to translate to consistent changes in many common community metrics across broader spatial scales, which may partly explain why broad-scale biodiversity studies often find mixed trends. Such inconsistency may be more pronounced for metrics compared across communities from different taxonomic groups or habitat types[7–9,11,12], given the high variability we found even within approximately the same system, that is, invertebrates sampled from the river bottom following similar methodologies. Community metrics other than those we examined may provide more consistent insight into anthropogenic change, such as observed:expected richness[56], genetic diversity, functional diversity or trait composition[57–59]. However, responses in these types of metrics can be similarly variable across communities[10,18]. Alternatively, measuring the 'quality' of a community in a different way, for example, using ecosystem functionality, could produce more consistent responses in community metrics that best reflect relevant functions, such as abundance/biomass[17] or evenness[60]. Using biodiversity to infer anthropogenic impacts therefore requires careful consideration of which community metrics are the best indicators for the habitat types and taxa in question and what the most suitable way to measure impact is. The answers to these questions will also undoubtedly change depending on whether the study is broad in scale and so requires general indicators versus focusing on finer-scale changes in particular regions or ecosystems[61].

Our analyses have two principal limitations that we cannot address. Although our results are supported by a robust dataset and match other reported conclusions about improvements in European freshwater communities[13,31,37], they are limited first by the spatial coverage of our sites and second by the temporal duration of monitoring. Spatially, our analyses are restricted to only river sites for which we could obtain data that met our criteria. Consequently, ecological quality trends informed by more spatially extensive datasets may reliably reflect country-scale changes (for example, Denmark or France), but trends informed by less extensive datasets (for example, Ireland or Norway) may not reflect the overall status of rivers in the region. Temporally, our analyses were restricted to starting in the early 1990s because reliable monitoring data across different countries were only available during this period. Our results therefore reflect how communities have changed during the past 30 years but cannot reflect the full extent of change compared to historical, pre-disturbance baselines.

With the above caveats in mind, our findings show that some community metrics, specifically richness, can consistently indicate anthropogenic impacts across broad spatial scales. However, variability in community responses means that such inferences must be made carefully, ensuring comparison of similar taxa and habitats and with an appropriate amount of data. In addition, we found many commonly used community metrics cannot consistently indicate anthropogenic impacts. Acknowledging and incorporating this variability into biodiversity analyses and monitoring programs is essential for identifying impacted communities and for better protecting biodiversity in an era of global change.

Methods

Riverine invertebrate data

We collated annual data on invertebrate community composition that was consistently collected from 1,365 river sampling locations across

23 European countries. These data primarily come from Haase et al.[18], although additional data for Czechia and Lithuania were provided via requests to ecologists and environmental managers. An advantage of this European-scale analysis is that all invertebrate biomonitoring and index calculation is performed in compliance with the European Union WFD, ensuring comparability among regions. Across all countries, the included rivers encompass a wide range of river sizes (Strahler order mean ± s.d. of 4.5 ± 1.9, range 1–10), catchment sizes and severity in anthropogenic impacts, from more pristine to heavily impacted ecosystems (Fig. 1). The time series ranged between 1992 and 2019, and each consisted of at least 7 years of data. Sampling was always conducted at the same river sites, during the same seasons (any three consecutive months) and using the same methods across years. Invertebrates were generally collected following WFD-compliant methods across countries, that is, primarily multi-habitat kick-net samples collected from the river bottom. Taxa were identified to family, genus or species level, although some were classified to intermediate (for example, Chironominae at subfamily) or higher levels (for example, Oligochaeta at subclass), with Chironomidae and Gammaridae typically the most abundant taxa across countries. The mean starting year for the time series was 1999, the mean end year was 2017, with a mean of 15 sampling years per site and a mean total time-series length of 18 years (see Supplementary Table 2 for further time-series details).

Ecological quality

The WFD is the principal piece of European protective water legislation that aims for all freshwaters to reach a 'good' or 'high' ecological status[32]. The ecological status of a river is quantified using multiple environmental parameters and taxonomic groups, but here we focused specifically on status measured using the ecological quality of the invertebrate community. We used WFD-compliant methods to calculate EQRs and EQCs. EQRs and EQCs were calculated by our co-authors using country-specific metrics/indices for the invertebrate data they provided (country-specific methods are detailed in Supplementary Table 1). We used the EQCs as a policy-relevant indication of the status of a community, whereas we used the EQRs in most statistical analyses because they are continuous rather than discrete and thus represent ecological quality more precisely.

Common community metrics

We calculated six community metrics for each river site and year: (1) abundance (number of individuals); (2) taxon richness (number of taxa); (3) evenness measured using Pielou's index[62]; (4) diversity measured using the Shannon index[63]; and (5 and 6) temporal turnover measured as the percentage difference in the proportional abundance of each taxon between consecutive years[8] and between each year and the first year[26] based on the Sørensen index. We chose these metrics because all are commonly used (or advocated for use) in biodiversity analyses and biomonitoring. Using multiple metrics also allowed us to examine the link between ecological quality and different aspects of the invertebrate community.

Biomonitoring indices of water and habitat conditions

We calculated eight invertebrate biomonitoring indices that can indicate changes in water quality and habitat conditions (detailed in Extended Data Table 1). Three indices respectively reflect the (1–3) abundance, richness and proportion (percentage of the community) of EPT, which encompass species that are often the most sensitive to anthropogenic impacts. Higher EPT values indicate the community contains more sensitive taxa. Two additional indices, (4) the ASPT index and (5) the Saprobic Index, reflect expert assessments of taxon-specific tolerances to anthropogenic impacts, usually chemical or organic pollution. Higher values of the former and lower values of the latter indicate the community contains more pollution-sensitive taxa. Last, we included (6) the Community Temperature Index, which reflects preferences for wider versus narrower temperature ranges[64]; (7) the proportion (%) of littoral taxa, which can reflect community responses to flow alteration; and (8) the Rhithron feeding type index, which reflects changes in the proportional abundance of different feeding guilds based on the assumption that certain guilds dominate in more impacted rivers. All indices are commonly used in European river biomonitoring[65], except for the Community Temperature Index which we included as an indicator of climate warming despite such indicators not yet being commonly used.

Statistical analyses

We split our analyses into three parts: (1) a continental-scale analysis that examined overall temporal ecological quality trends and their relationships to the metrics/indices across countries, and (2) a country-scale and (3) a site-scale analysis that examined variability in these trends and relationships at finer spatial scales. All analyses were performed in R version 4.2.0[66].

To quantify continental-scale changes in ecological quality, we modelled temporal trends in EQCs and EQRs across countries using GAMMs, which enable modelling nonlinear trends through time. The response variable for these models was the EQC or EQR for each site and year. The predictor variables included a smoothed term for year modelled using thin-plate regression splines and a basis dimension of $k = 10$, which we confirmed via comparisons to the e.d.f. and based on whether the relationship changed when the basis dimension was increased. We also included a random slope and intercept term for country to help control for differences among countries in sampling methods and effort, and random intercept terms for sampling year and sampling month to control for non-independence among samples collected from the same years and months. In addition, we included a first-order autoregressive structure to control for temporal autocorrelation in samples collected from the same site in consecutive years. We found no strong evidence for spatial autocorrelation (Supplementary Figs. 8 and 9). Significance ($P < 0.05$) of the smoothed year term in the finalized models was assessed with Wald tests.

To delineate continental-scale relationships between ecological quality and the community metrics and biomonitoring indices, we combined redundancy analysis (RDA) with GAMMs. We used the RDA to identify which metrics/indices were most related to changes in ecological quality and then used GAMMs to quantify the shape and strength of these relationships. The RDA modelled similarities (based on Euclidean distance) in the community metrics and biomonitoring indices across all sites and years in relation to the EQRs (excluding the ASPT and Saprobic Index, which are not calculated in all countries). Abundance was $\log_{10}$-transformed, and all metrics were converted to *z*-scores before analysis (that is, centred to their country-specific means and standard deviations) to enable comparison of metrics with different country-specific units or ranges, such as abundance. We identified the variables most related to ecological quality based on their loadings onto RDA axis 1, that is, the dimension representing changes in the EQRs. Relationships between EQRs and metrics with the highest loadings were then confirmed using GAMMs that included the same random effects and other control variables as the continental-scale models.

To quantify country-scale temporal change in ecological quality and its relationships to the community metrics and biomonitoring indices, we analysed the data for 15 separate countries that had samples from at least 10 sites (comprising 99% of our dataset), thus providing enough information to parameterize models for each country. We modelled temporal trends in EQCs and EQRs within each country using GAMMs following the methods used in the continental-scale analysis. We also conducted 15 RDAs that related all applicable metrics/indices for each country to their respective EQRs and used GAMMs to further examine these relationships.

To quantify the site-scale relationships between ecological quality and the community metrics and biomonitoring indices, we calculated

the slopes of temporal change in the EQRs and metrics/indices for each site. Slopes were calculated using robust regressions[67] to downweight the importance of data from the first and last years, which can strongly influence slope estimates in time-series analyses[21,28]. We then related the EQR slopes to the associated slopes for each community metric and biomonitoring index at each site using linear mixed models. These models included a random slope and intercept term for each country, and the contribution of each site was weighted by the $\log_{10}$-transformed inverse of the summed squared standard errors of its slope estimates to ensure that slopes with more error contributed less to modelled relationships.

Reporting summary

Further information on research design is available in the Nature Portfolio Reporting Summary linked to this article.

Data availability

All community metrics, biomonitoring indices and ecological quality data needed to reproduce our analyses are publicly available from Figshare at https://doi.org/10.6084/m9.figshare.24486769.

Code availability

All code used for our analyses is available upon request.

References

1. Barnosky, A. D. et al. Has the Earth's sixth mass extinction already arrived? *Nature* **471**, 51–57 (2011).
2. Ceballos, G. et al. Accelerated modern human-induced species losses: entering the sixth mass extinction. *Sci. Adv.* **1**, e1400253 (2015).
3. Yasuhara, M., Hunt, G., Breitburg, D., Tsujimoto, A. & Katsuki, K. Human-induced marine ecological degradation: micropaleontological perspectives. *Ecol. Evol.* **2**, 3242–3268 (2012).
4. Newbold, T. et al. Global effects of land use on local terrestrial biodiversity. *Nature* **520**, 45–50 (2015).
5. Reid, A. J. et al. Emerging threats and persistent conservation challenges for freshwater biodiversity. *Biol. Rev.* **94**, 849–873 (2019).
6. Tickner, D. et al. Bending the curve of global freshwater biodiversity loss: an emergency recovery plan. *BioScience* **70**, 330–342 (2020).
7. Vellend, M. et al. Global meta-analysis reveals no net change in local-scale plant biodiversity over time. *Proc. Natl Acad. Sci. USA* **110**, 19456–19459 (2013).
8. Dornelas, M. et al. Assemblage time series reveal biodiversity change but not systematic loss. *Science* **344**, 296–299 (2014).
9. Blowes, S. A. et al. The geography of biodiversity change in marine and terrestrial assemblages. *Science* **366**, 339–345 (2019).
10. Millette, K. L. et al. No consistent effects of humans on animal genetic diversity worldwide. *Ecol. Lett.* **23**, 55–67 (2020).
11. Pilotto, F. et al. Meta-analysis of multidecadal biodiversity trends in Europe. *Nat. Commun.* **11**, 3486 (2020).
12. van Klink, R. et al. Meta-analysis reveals declines in terrestrial but increases in freshwater insect abundances. *Science* **368**, 417–420 (2020).
13. Outhwaite, C. L., Gregory, R. D., Chandler, R. E., Collen, B. & Isaac, N. J. B. Complex long-term biodiversity change among invertebrates, bryophytes and lichens. *Nat. Ecol. Evol.* **4**, 384–392 (2020).
14. Wagner, D. L., Fox, R., Salcido, D. M. & Dyer, L. A. A window to the world of global insect declines: moth biodiversity trends are complex and heterogeneous. *Proc. Natl Acad. Sci. USA* **118**, e2002549117 (2021).
15. Carvalheiro, L. G. et al. Species richness declines and biotic homogenisation have slowed down for NW-European pollinators and plants. *Ecol. Lett.* **16**, 870–878 (2013).
16. Schipper, A. M. et al. Contrasting changes in the abundance and diversity of North American bird assemblages from 1971 to 2010. *Glob. Change Biol.* **22**, 3948–3959 (2016).
17. Hallmann, C. A. et al. More than 75 percent decline over 27 years in total flying insect biomass in protected areas. *PLoS ONE* **12**, e0185809 (2017).
18. Haase, P. et al. The recovery of European freshwater biodiversity has come to a halt. *Nature* **620**, 582–588 (2023).
19. de los Santos, C. B. et al. Recent trend reversal for declining European seagrass meadows. *Nat. Commun.* **10**, 3356 (2019).
20. Kuczynski, L., Ontiveros, V. J. & Hillebrand, H. Biodiversity time series are biased towards increasing species richness in changing environments. *Nat. Ecol. Evol.* **7**, 994–1001 (2023).
21. Gonzalez, A. et al. Estimating local biodiversity change: a critique of papers claiming no net loss of local diversity. *Ecology* **97**, 1949–1960 (2016).
22. Cardinale, B. J., Gonzalez, A., Allington, G. R. H. & Loreau, M. Is local biodiversity declining or not? A summary of the debate over analysis of species richness time trends. *Biol. Conserv.* **219**, 175–183 (2018).
23. Valdez, J. W. et al. The undetectability of global biodiversity trends using local species richness. *Ecography* **2023**, e06604 (2023).
24. Catford, J. A., Wilson, J. R. U., Pyšek, P., Hulme, P. E. & Duncan, R. P. Addressing context dependence in ecology. *Trends Ecol. Evol.* **37**, 158–170 (2022).
25. Elahi, R. et al. Recent trends in local-scale marine biodiversity reflect community structure and human impacts. *Curr. Biol.* **25**, 1938–1943 (2015).
26. Hillebrand, H. et al. Biodiversity change is uncoupled from species richness trends: consequences for conservation and monitoring. *J. Appl. Ecol.* **55**, 169–184 (2018).
27. Ludsin, S. A., Kershner, M. W., Blocksom, K. A., Knight, R. L. & Stein, R. A. Life after death in Lake Erie: nutrient controls drive fish species richness, rehabilitation. *Ecol. Appl.* **11**, 731–746 (2001).
28. Fournier, A. M. V., White, E. R. & Heard, S. B. Site-selection bias and apparent population declines in long-term studies. *Conserv. Biol.* **33**, 1370–1379 (2019).
29. Magurran, A. E. et al. Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends Ecol. Evol.* **25**, 574–582 (2010).
30. Bonada, N., Prat, N., Resh, V. H. & Statzner, B. Developments in aquatic insect biomonitoring: a comparative analysis of recent approaches. *Annu. Rev. Entomol.* **51**, 495–523 (2006).
31. Pharaoh, E., Diamond, M., Ormerod, S. J., Rutt, G. & Vaughan, I. P. Evidence of biological recovery from gross pollution in English and Welsh rivers over three decades. *Sci. Total Environ.* **878**, 163107 (2023).
32. Water Framework Directive (WFD). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy. *Off. J. Eur. Comm.* **327**, 1–72 (2000).
33. Birk, S. et al. Three hundred ways to assess Europe's surface waters: an almost complete overview of biological methods to implement the Water Framework Directive. *Ecol. Indic.* **18**, 31–41 (2012).
34. Bennett, C. et al. Bringing European river quality into line: an exercise to intercalibrate macro-invertebrate classification methods. *Hydrobiologia* **667**, 31–48 (2011).
35. Desquilbet, M. et al. Comment on 'Meta-analysis reveals declines in terrestrial but increases in freshwater insect abundances. *Science* **370**, eabd8947 (2020).

36. Giakoumis, T. & Voulvoulis, N. The transition of EU water policy towards the Water Framework Directive's Integrated River Basin Management Paradigm. *Environ. Manage.* **62**, 819–831 (2018).
37. *The European Environment—State and Outlook 2020* (European Environment Agency, 2020); https://www.eea.europa.eu/soer/publications/soer-2020
38. Gozlan, R. E., Karimov, B. K., Zadereev, E., Kuznetsova, D. & Brucet, S. Status, trends, and future dynamics of freshwater ecosystems in Europe and Central Asia. *Inland Waters* **9**, 78–94 (2019).
39. O'Briain, R. Climate change and European rivers: an eco-hydromorphological perspective. *Ecohydrology* **12**, e2099 (2019).
40. Wolfram, J., Stehle, S., Bub, S., Petschick, L. L. & Schulz, R. Water quality and ecological risks in European surface waters—monitoring improves while water quality decreases. *Environ. Int.* **152**, 106479 (2021).
41. Bernhardt, E. S., Rosi, E. J. & Gessner, M. O. Synthetic chemicals as agents of global change. *Front. Ecol. Environ.* **15**, 84–90 (2017).
42. Pereira, H. M. et al. Essential biodiversity variables. *Science* **339**, 277–278 (2013).
43. Deiner, K. et al. Environmental DNA metabarcoding: transforming how we survey animal and plant communities. *Mol. Ecol.* **26**, 5872–5895 (2017).
44. Anderson, C. B. Biodiversity monitoring, earth observations and the ecology of scale. *Ecol. Lett.* **21**, 1572–1585 (2018).
45. Miller, S. W., Budy, P. & Schmidt, J. C. Quantifying macroinvertebrate responses to in-stream habitat restoration: applications of meta-analysis to river restoration. *Restor. Ecol.* **18**, 8–19 (2010).
46. Ferreira, W. R. et al. Importance of environmental factors for the richness and distribution of benthic macroinvertebrates in tropical headwater streams. *Freshw. Sci.* **33**, 860–871 (2014).
47. Hodgson, J. A., Thomas, C. D., Wintle, B. A. & Moilanen, A. Climate change, connectivity and conservation decision making: back to basics. *J. Appl. Ecol.* **46**, 964–969 (2009).
48. Mortelliti, A., Amori, G. & Boitani, L. The role of habitat quality in fragmented landscapes: a conceptual overview and prospectus for future research. *Oecologia* **163**, 535–547 (2010).
49. Crossley, M. S. et al. No net insect abundance and diversity declines across US Long Term Ecological Research sites. *Nat. Ecol. Evol.* **4**, 1368–1376 (2020).
50. Johnson, R. K. & Hering, D. Spatial congruency of benthic diatom, invertebrate, macrophyte, and fish assemblages in European streams. *Ecol. Appl.* **20**, 978–992 (2010).
51. Vellend, M. The biodiversity conservation paradox. *Am. Sci.* **105**, 94–101 (2017).
52. Dornelas, M. et al. A balance of winners and losers in the Anthropocene. *Ecol. Lett.* **22**, 847–854 (2019).
53. Heino, J. et al. Metacommunity organisation, spatial extent and dispersal in aquatic systems: patterns, processes and prospects. *Freshw. Biol.* **60**, 845–869 (2015).
54. Heino, J. The importance of metacommunity ecology for environmental assessment research in the freshwater realm. *Biol. Rev.* **88**, 166–178 (2013).
55. Ruaro, R., Gubiani, É. A., Hughes, R. M. & Mormul, R. P. Global trends and challenges in multimetric indices of biological condition. *Ecol. Indic.* **110**, 105862 (2020).
56. Hawkins, C. P. Quantifying biological integrity by taxonomic completeness: its utility in regional and global assessments. *Ecol. Appl.* **16**, 1277–1294 (2006).
57. Vandewalle, M. et al. Functional traits as indicators of biodiversity response to land use changes across ecosystems and organisms. *Biodivers. Conserv.* **19**, 2921–2947 (2010).
58. Miraldo, A. et al. An Anthropocene map of genetic diversity. *Science* **353**, 1532–1535 (2016).
59. Violle, C., Reich, P. B., Pacala, S. W., Enquist, B. J. & Kattge, J. The emergence and promise of functional biogeography. *Proc. Natl Acad. Sci. USA* **111**, 13690–13696 (2014).
60. Maureaud, A. et al. Biodiversity–ecosystem functioning relationships in fish communities: biomass is related to evenness and the environment, not to species richness. *Proc. R. Soc. B* **286**, 20191189 (2019).
61. Jarzyna, M. A. & Jetz, W. Taxonomic and functional diversity change is scale dependent. *Nat. Commun.* **9**, 2565 (2018).
62. Pielou, E. C. The measurement of diversity in different types of biological collections. *J. Theor. Biol.* **13**, 131–144 (1966).
63. Spellerberg, I. F. & Fedor, P. J. A tribute to Claude Shannon (1916–2001) and a plea for more rigorous use of species richness, species diversity and the 'Shannon–Wiener' Index. *Glob. Ecol. Biogeogr.* **12**, 177–179 (2003).
64. Haase, P. et al. Moderate warming over the past 25 years has already reorganized stream invertebrate communities.*Sci. Total Environ.* **658**, 1531–1538 (2019).
65. Vitecek, S., Johnson, R. K. & Poikane, S. Assessing the ecological status of European rivers and lakes using benthic invertebrate communities: a practical catalogue of metrics and methods. *Water* **13**, 346 (2021).
66. R Core Team. *R: A Language and Environment for Statistical Computing* (2022); https://www.r-project.org
67. Huber, P. J. *Robust Statistical Procedures 2*nd edn (SIAM, 1996).
68. EuroGeographics. Countries—Administrative Units (2020); https://ec.europa.eu/eurostat/web/gisco/geodata/reference-data/administrative-units-statistical-units/countries

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Author contributions

J.S.S. and P.H. conceived the study. E.A.R.W. cleaned the data, and J.S.S. performed the analyses. J.S.S. and P.H. wrote the majority of the manuscript. F.A., M.A.-C., J.A., N.J.B., L. Barešová, J.B., L. Bonacina, N.B., M.C.-A., Z.C., E.d.E., A.D., G.D., T.E.E., V.E., M.J.F., M. Ferréol, M. Floury, M.A.E.F., R.F., P.L.M.G., J.H., D.H., K.-L.H., S.C.J., R.K.J., L.K., B.K., L.L., A.L., P.L., A.W.L., B.G.M., T.M., D.O., R.P., V.P., P.P., F.P., M.P., J.J.R., R.B.S., A.S.-K., A. Scotti, A. Skuja, M.S., R.S., H.T., V.T., I.T., R.V., G. Várbíró, G. Velle, R.C.M.V. and S.V. provided invertebrate data or contributed to calculating ecological quality values for their respective countries, and all edited the manuscript.

Competing interests

A. Scotti is affiliated with APEM, which is an environmental consultancy company, although they provided no funding for this study. The other authors declare no competing interests.

Additional information

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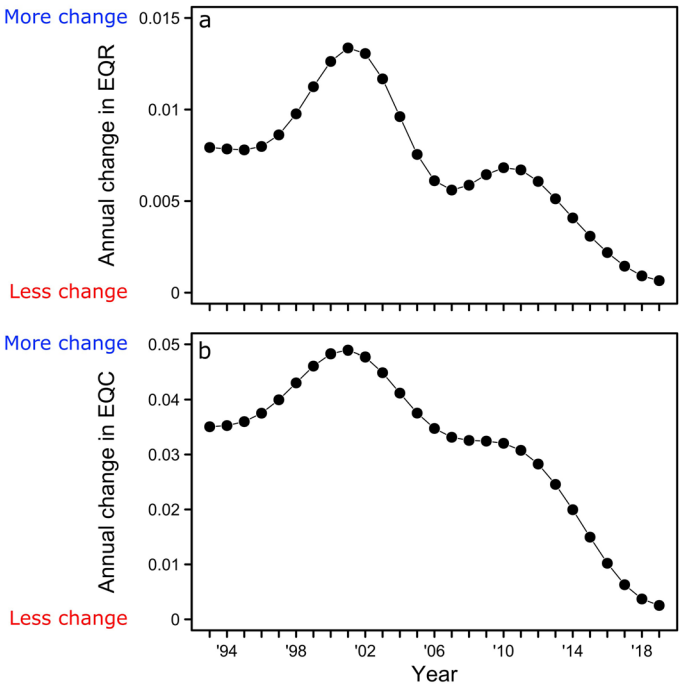

Extended Data Fig. 1 | Year-to-year changes in ecological quality. Differences in the predicted (**a**) EQRs and (**b**) EQCs between each year and the previous year during 1993–2019. For example, the 1993 values are the absolute differences in the predicted EQRs/EQCs between 1992 and 1993. Thus, values closer to 0 indicate less change between successive years. Predicted values for the EQRs and EQCs were obtained from their respective generalized additive mixed models (that is, the fitted relationships in Fig. 2).

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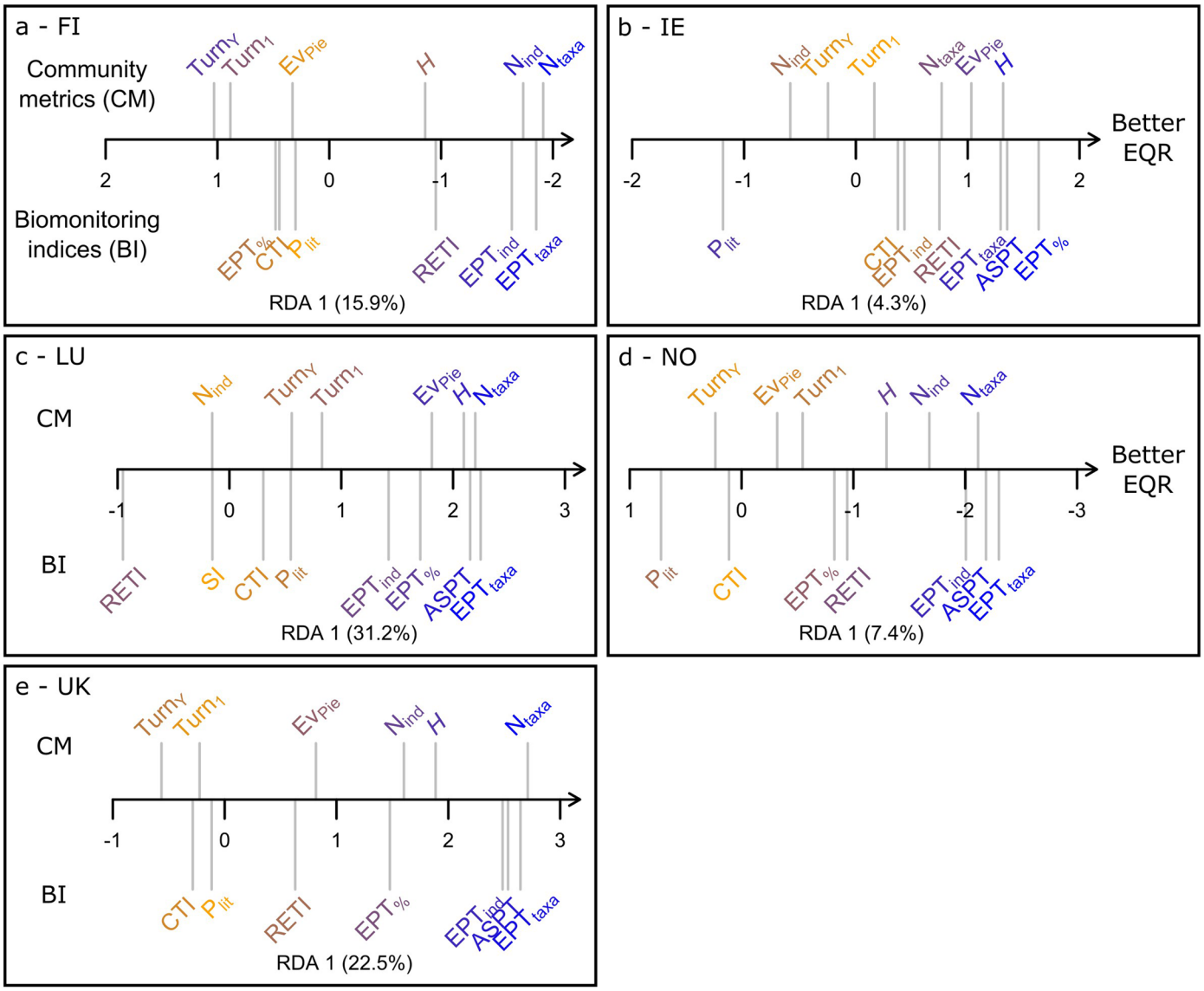
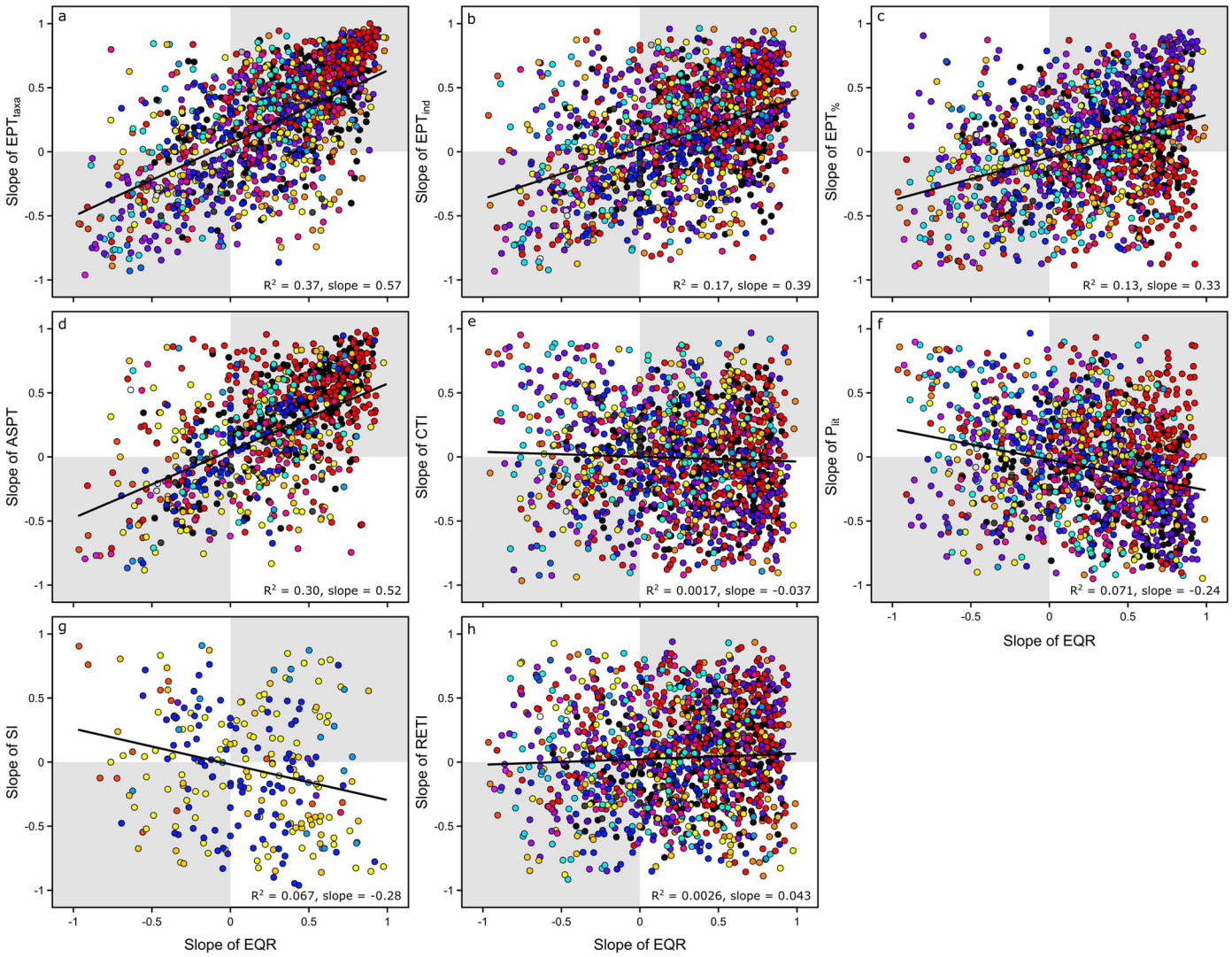


Extended Data Fig. 2 | Country-scale links between ecological quality, community metrics, and biomonitoring indices. Redundancy Analyses (RDAs) of the relationship between the Ecological Quality Ratios (EQRs; black arrows) and the community metrics and biomonitoring indices for (**a**) Finland (FI), (**b**) Ireland (IE), (**c**) Luxembourg (LU), (**d**) Norway (NO), and (**e**) the United Kingdom (UK). The community metrics comprise abundance (N_{ind}), richness (N_{taxa}), evenness (Ev_{Pie}), Shannon diversity (*H*), and temporal turnover between consecutive years ($Turn_Y$) and compared to the first year ($Turn_1$). The biomonitoring indices comprise the total abundance (EPT_{ind}), proportion ($EPT_{\%}$), and richness (EPT_{taxa}) of Ephemeroptera, Plecoptera, and Trichoptera, in addition to the Community Temperature Index (CTI), the proportion of littoral taxa (P_{lit}), and the Rhithron feeding type index (RETI; all indices are described in the *Methods* and Extended Data Table 1). Metrics and indices are coloured from orange to blue based on their loadings on RDA axis 1, with blues indicating stronger relationships to ecological quality.

https://doi.org/10.1038/s41559-023-02305-4



Extended Data Fig. 3 | Site-scale links between ecological quality and biomonitoring indices. Relationship between the temporal slope of the Ecological Quality Ratio (EQR) at each site and the slope of (**a**) the richness of Ephemeroptera, Plecoptera, and Trichoptera (EPT_{taxa}), (**b**) EPT abundance (EPT_{ind}), (**c**) the proportion of EPT taxa ($EPT_{\%}$), (**d**) the Average Score Per Taxon (ASPT) index, (**e**) the Community Temperature Index (CTI), (**f**) the proportion of littoral taxa (P_{lit}), (**g**) the Saprobic Index (SI), and (**h**) the Rhithron feeding type index (RETI). Sites are coloured by country and sites with matching ecological quality and biodiversity trends are in the gray shaded areas, whereas opposing relationships are in the white areas.

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Extended Data Table 1 | List and description of invertebrate biomonitoring indices

Metrics	Abbreviation	Meaning	Specific stressors	Number of countries
Ephemeroptera, Plecoptera, and Trichoptera	EPT	The abundance, richness, and proportion of EPT taxa[65]. Higher values indicate more EPT taxa, which tend to occur in less-impacted rivers.	–	23
Average Score Per Taxon	ASPT	Average of pollution-tolerance scores for present taxa (also weighted by abundance in the UK). Higher values indicate communities comprising taxa associated with less-impacted rivers.	–	14
Saprobic Index	SI	Abundance-weighted index of taxon-specific saprobic values. Higher values indicate communities comprised of taxa that tend to occur in enriched rivers.	Organic pollution[65]	8
Community Temperature Index	CTI	Abundance-weighted average of taxon-specific preferences for temperature variability. Higher values indicate communities characterized by taxa with warmer and wider temperature preferences.	Warming[64]	23
Proportion of littoral taxa	P_{Lit}	Proportion of the invertebrate community comprising taxa with any affinity for littoral habitats (based on the stream zonation trait from www.freshwaterecology.info). Higher values indicate more littoral taxa, which tend to prefer slower currents.	Flow alteration[65]	23
Rhithron feeding type index	RETI	Proportion of feeding types associated with more upstream habitats, specifically grazers, shredders, or taxa that feed on woody debris[65]. Higher values indicate communities comprised of taxa that tend to prefer less-impacted rivers.	–	23

These indices were used to determine whether changes in ecological quality corresponded to shifts in sensitive versus tolerant invertebrates, which provides supporting evidence that ecological quality reflects general anthropogenic impacts on river water and habitat quality. Some indices can also indicate the effects of specific stressors. We also list the number of countries for which each index was calculated out of 23 total in our dataset. References for the ASPT indices are provided in Supplementary Table 1.

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Extended Data Table 2 | Site-scale variability in the relationship between ecological quality and community metrics

Metric	Matching	No match	Opposing
Abundance	13%	40%	6%
Evenness	6%	45%	6%
Richness	24%	31%	2%
Shannon diversity	11%	43%	3%
Turnover (consecutive)	7%	45%	4%
Turnover (first-year)	4%	43%	18%

Proportion of sites (out of 1,365) that match the overall relationship between the slope of a given community metric and the slope of the Ecological Quality Ratio (EQR), compared to those that exhibit no matching relationship (either the metric changes when ecological quality does not or vice versa), or opposing responses. For example, the overall relationship between the slopes of richness and the EQRs is positive (Fig. 6a) and 24% of sites match this trend. Similarly, the overall relationship between the slopes of consecutive turnover and the EQRs is negative, specifically turnover tends to decline as ecological quality improves (Fig. 6e), and 7% of sites match this relationship. Note that 'change' in a given metric or the EQRs is determined as a slope value whose confidence intervals do not overlap 0.

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For all statistical analyses, confirm that the following items are present in the figure legend, table legend, main text, or Methods section.

n/a	Confirmed	
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☐	☒	The statistical test(s) used AND whether they are one- or two-sided *Only common tests should be described solely by name; describe more complex techniques in the Methods section.*
☐	☒	A description of all covariates tested
☐	☒	A description of any assumptions or corrections, such as tests of normality and adjustment for multiple comparisons
☐	☒	A full description of the statistical parameters including central tendency (e.g. means) or other basic estimates (e.g. regression coefficient) AND variation (e.g. standard deviation) or associated estimates of uncertainty (e.g. confidence intervals)
☐	☒	For null hypothesis testing, the test statistic (e.g. *F*, *t*, *r*) with confidence intervals, effect sizes, degrees of freedom and *P* value noted *Give P values as exact values whenever suitable.*
☒	☐	For Bayesian analysis, information on the choice of priors and Markov chain Monte Carlo settings
☒	☐	For hierarchical and complex designs, identification of the appropriate level for tests and full reporting of outcomes
☒	☐	Estimates of effect sizes (e.g. Cohen's *d*, Pearson's *r*), indicating how they were calculated

Our web collection on statistics for biologists contains articles on many of the points above.

Software and code

Policy information about availability of computer code

Data collection	No software was used for data collection.
Data analysis	All data analyses were performed using R version 4.2.0.

For manuscripts utilizing custom algorithms or software that are central to the research but not yet described in published literature, software must be made available to editors and reviewers. We strongly encourage code deposition in a community repository (e.g. GitHub). See the Nature Portfolio guidelines for submitting code & software for further information.

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All manuscripts must include a data availability statement. This statement should provide the following information, where applicable:

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A data availability statement is provided in the manuscript.

Research involving human participants, their data, or biological material

Policy information about studies with human participants or human data. See also policy information about sex, gender (identity/presentation), and sexual orientation and race, ethnicity and racism.

Reporting on sex and gender	NA
Reporting on race, ethnicity, or other socially relevant groupings	NA
Population characteristics	NA
Recruitment	NA
Ethics oversight	NA

Note that full information on the approval of the study protocol must also be provided in the manuscript.

Field-specific reporting

Please select the one below that is the best fit for your research. If you are not sure, read the appropriate sections before making your selection.

☐ Life sciences ☐ Behavioural & social sciences ☒ Ecological, evolutionary & environmental sciences

For a reference copy of the document with all sections, see nature.com/documents/nr-reporting-summary-flat.pdf

Ecological, evolutionary & environmental sciences study design

All studies must disclose on these points even when the disclosure is negative.

Study description	We collated annual time series data on riverine invertebrate community composition at 1,365 sites across 23 European countries via data requests to ecologists and environmental managers.
Research sample	The time series each consisted of at least seven years of data, with sampling conducted at the same sites, during the same seasons (any three consecutive months), and using the same methods within sites through time.
Sampling strategy	Sites were selected based on which rivers were being monitored for the ecological quality of their invertebrate communities in their respective countries. The data collected from these sties also needed to satisfy our data requirements listed in the "Research sample" section above.
Data collection	Invertebrates were generally collected via multi-habitat kick-net samples collected from the stream bottom.
Timing and spatial scale	Sites encompassed 23 European countries spanning across northern, central, and southern Europe. The time series across sites ranged between 1992 and 2019. Data was often collected annually, although some sites can exhibit gaps of several years between samples.
Data exclusions	Data collected prior to 2000 in Germany was excluded because sampling protocols changed in this year to comply with the European Union Water Framework Directive.
Reproducibility	No experiments were performed as part of this study.
Randomization	We controlled for both spatial and temporal relationships among sites. We controlled for spatial similarities by grouping data by the country in which it was collected and by testing for spatial autocorrelation. We controlled for temporal relationships by grouping data by the year and month in which it was collected and by controlling for temporal autocorrelation among samples collected from the same site in successive years.
Blinding	Blinding was not relevant to this study because data acquisition required open contact among all of the ecologists and environmental managers that provided data.

Did the study involve field work? ☐ Yes ☒ No

Reporting for specific materials, systems and methods

We require information from authors about some types of materials, experimental systems and methods used in many studies. Here, indicate whether each material, system or method listed is relevant to your study. If you are not sure if a list item applies to your research, read the appropriate section before selecting a response.

Materials & experimental systems

n/a	Involved in the study
☒	☐ Antibodies
☒	☐ Eukaryotic cell lines
☒	☐ Palaeontology and archaeology
☐	☒ Animals and other organisms
☒	☐ Clinical data
☒	☐ Dual use research of concern
☒	☐ Plants

Methods

n/a	Involved in the study
☒	☐ ChIP-seq
☒	☐ Flow cytometry
☒	☐ MRI-based neuroimaging

Animals and other research organisms

Policy information about studies involving animals; ARRIVE guidelines recommended for reporting animal research, and Sex and Gender in Research

Laboratory animals	NA
Wild animals	No animals were collected for this specific study. Only data that had already been collected for other purposes in each region was used.
Reporting on sex	NA
Field-collected samples	No samples were collected or stored specifically for this study.
Ethics oversight	No ethical approval or guidance was required because no animals were collected for this study.

Note that full information on the approval of the study protocol must also be provided in the manuscript.

Plants

Seed stocks	NA
Novel plant genotypes	NA
Authentication	NA