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Short communication

What has happened to river macroinvertebrate biodiversity in England and Wales over the past 30 years?

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ABSTRACT

Freshwater systems have experienced 200 years of escalating pressure from growing human populations, industrialisation and agricultural intensification leading to pollution and degraded biodiversity. In Europe and North America, the last 30 years have seen large-scale efforts to reduce river pollution via legislation such as the EU Urban Wastewater Treatment Directive and the US Clean Water Act. Concern over water quality in rivers has received increasing public interest over the past five years, and there are questions about the true impacts of pollution on wildlife. Recent evidence suggests that the richness and abundance of freshwater macroinvertebrates (a group commonly used as indicators of water quality) has improved, especially in formerly polluted urban rivers in developed Western countries, although more recently this recovery has slowed or plateaued.

Here, we examine available evidence on changing macroinvertebrate biodiversity in UK rivers, focusing on trends in England and Wales over the past 30 years. By summarising current research on the status and trends of freshwater invertebrates using multiple sources of evidence, we confirm an aggregate increase in several measures of biodiversity, including family richness, species occupancy and the abundance of pollution-sensitive groups, although this stops short of the diversity expected in completely unpolluted locations. There are also local or regional departures from this aggregate trend, in some cases linked to nutrient pollution and climate change. We highlight a significant decline in government agency monitoring effort over the last 15 years. The continuation of monitoring across England and Wales is critical to detecting and diagnosing changes in invertebrate diversity and identifying solutions. Furthermore, we argue that current approaches for assessing the ecological status of rivers should more prominently report measures of biodiversity to aid our understanding of, and to better communicate to the public, the impacts of pollution and other related pressures at the national scale.

1. Introduction

An increasing number of studies have highlighted declines in invertebrate biodiversity (Biesmeijer et al., 2006; Hallmann et al., 2017, 2020; van Strien et al., 2019). Although these studies tend to be restricted to well-studied regions (e.g., Europe) and taxonomic groups (e.g., butterflies and moths), most available evidence suggests that the world's invertebrates are under considerable threat (Reid et al., 2019). The declines are linked with increasing anthropogenic pressures such as habitat loss or impairment, invasive non-native species, pollution and climate change (Wagner et al., 2021). In river ecosystems, however, recent studies have shown different patterns in freshwater invertebrate biodiversity to the declines evident in the terrestrial realm. For areas of Europe and North America, for example, available data indicate that formerly polluted urban rivers began to recover from the effects of insanitary and industrial discharges in the early 1990s (Haase et al., 2023; Haubrock et al., 2023; Outhwaite et al., 2020; Pharaoh et al., 2023; Qu et al., 2023; Rumschlag et al., 2023; Sinclair et al., 2024; van Klink et al., 2020a). Recent evidence aggregated across Europe suggests that this recovery may now be stalled or slowing, at least when measured as taxonomic and functional diversity (Haase et al., 2023).

In the UK, there are conflicting perceptions about what is currently happening to river ecosystem quality. Ecological status, measured under the EU Water Framework Directive (2000/60 EC), indicates marked

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variability across England, Wales, Scotland and Northern Ireland, but overall, only modest improvement since 2009 (Voulvoulis et al., 2017). Other studies of raw chemical data produce a more nuanced picture (Whelan et al., 2022). There are fears that, as elsewhere in Europe, any improvements to water quality in UK rivers beginning from ~1990 may now be stalling. In particular, public concern has focussed on discharges of untreated sewage from combined sewer overflows and on the effects of agricultural nutrients into rivers designated for their importance to conservation.

One means to appraise the overall trajectory of freshwater ecosystems in the UK is from biological data which have a range of advantages over chemical assessment – for example in integrating signals from different abiotic drivers of change or capturing the effects of both long and short-term variations in water quality (Johnson et al., 2025b). Although biota are monitored by government agencies, official reporting involves condensing and amalgamating the data into a statutory format (Murray-Bligh and Griffiths, 2022). These aggregated analyses mask underlying biodiversity trends, although a range of studies have used the data to make more detailed assessments that are available for review (Nunn et al., 2025; Qu et al., 2023).

Here we bring together evidence from a range of sources on trends in freshwater invertebrates from English and Welsh rivers in relation to their current state. We ask three questions.

1. What trends have been reported for freshwater invertebrate biodiversity in rivers over the past 30 years?
2. Are these trends plausible given our knowledge of local and international trends?
3. To what extent do current monitoring strategies assist or hinder national and local assessment of biodiversity change and its drivers?

In this study we will largely be discussing macroinvertebrate biodiversity with respect to family and species richness.

2. UK invertebrate monitoring data

Two data sources have been used to estimate trends in English and Welsh river macroinvertebrates (Table 1): (i) biological community data collected by UK environment protection agencies for the statutory purposes of managing river water quality and (ii) species occurrence records collected by volunteers and collated by multiple national recording schemes (typically used by scientists at a 1 km resolution). Both types of data source benefit from being long time series records which cover much of the country (Murray-Bligh and Griffiths, 2022; Pocock et al., 2015).

For scientists interested in the topic, biological community data with a high sampling density over a 30-year period are available from the Environment Agency (England) and Natural Resources Wales (Table 1). Similar data for Scotland, collected by the Scottish Environment Protection Agency, are also available, with digitized records beginning in 2006. Species occurrence records have been collated via recording schemes including the Riverfly Recording Schemes, the British Dragonfly Recording Society recording scheme, the Aquatic Coleoptera Recording Scheme, and the Conchological Society of Great Britain and Northern Ireland Non-Marine Mollusc recording scheme and supported by the Biological Records Centre (BRC) for the past 50 years but are spatially and temporally variable (Table 1). The national databases are heterogeneous, containing data that are collected for a range of purposes including long-term monitoring and surveillance, and short-term investigations, rather than the result of a single, systematic sampling scheme.

Apart from these large datasets with broad national coverage, there are focused long-term monitoring networks such as the Upland Waters Monitoring Network (Monteith et al., 2022) and the Llyn Brianne Stream Observatory (Larsen et al., 2018). Here we comment only on the data used to identify the aggregate national trends.

3. Trends in river invertebrate data

Several analyses of river macroinvertebrate data from the environmental regulators (Pharaoh et al., 2023; Qu et al., 2023; Vaughan and Ormerod, 2012) and the BRC (Outhwaite et al., 2020) (Table 1) have revealed underlying trends and enabled us to draw conclusions about average changes across England/Wales.

- Following an initial decline between the 1970s and 80s, overall river invertebrate biodiversity has increased at a national scale since 1990 (Fig. 1) (Outhwaite et al., 2020; Pharaoh et al., 2023; Qu et al., 2023).
- There has been only a modest increase in the arrival of alien species detected since 1989, the presence of non-native families increasing from 1.0 to 1.3 families per site over 30 years (Qu et al., 2023). However, the population sizes of several of these non-native species have increased exponentially between 2002 and 2019 (Wilkes et al., 2025). These include invasive crustaceans, molluscs, and annelids.
- The average increase in overall riverine macroinvertebrate diversity has slowed since the early to mid-2000s and may have levelled off (Pharaoh et al., 2023; Qu et al., 2023), similar to trends seen in continental Europe (Haase et al., 2023). In some English and Welsh regions, there is also evidence of local to regional decline (Pharaoh et al., 2024).

Table 1
Key details for the data sources available in England and Wales for long-term freshwater invertebrate monitoring. This includes the timescale over which monitoring data are available, the taxonomic coverage of the available data in terms of key invertebrate orders, the taxonomic level of the organism identification, the unit of the observations made, the spatial extent of the available records, the sampling methods used to collect the data, and the associated survey design.

Dataset	Timescale	Taxonomic coverage	Taxonomic level (e.g. species)	Observation unit	Spatial extent	Sampling methodology	Survey design
BIOSYS Environment Agency KiECO Natural Resources Wales	Begins with few sites in 1970s and 1980s. Main record keeping 1989 onwards.	All river macroinvertebrates.	Mixed (group, family, species). Species recorded more frequently after 2009/2011.	Presence/absence. Log abundance categories (before 2002). Abundance (after 2002).	England and Wales. England:36,000 sites, 291,000 samples Wales: 2500 sites, 14,100 samples	3-min kick sample and 1 min hand search. A small number of other methods.	Sampling focus in rivers. Systematic sampling with long-term monitoring of sites.
National Recording Schemes *via* Biological Records Centre[a]	Most data from 1980 onward, with some records from earlier years.	Largely the adult flying stage of insects (Odonata, Plecoptera, Trichoptera, Ephemeroptera, Aquatic beetles) plus molluscs	Mixed but mostly to species (group, family, species).	Presence-only (pseudo-absence)	UK, spatial coverage will vary by scheme	Ad hoc observations of species	Un-structured sampling by volunteers[b].

[a] Note that the BRC freshwater data up to 2004 incorporated information from the Environment Agency BIOSYS dataset.
[b] Note that some recording schemes periodically incorporate data from government datasets, including BIOSYS.

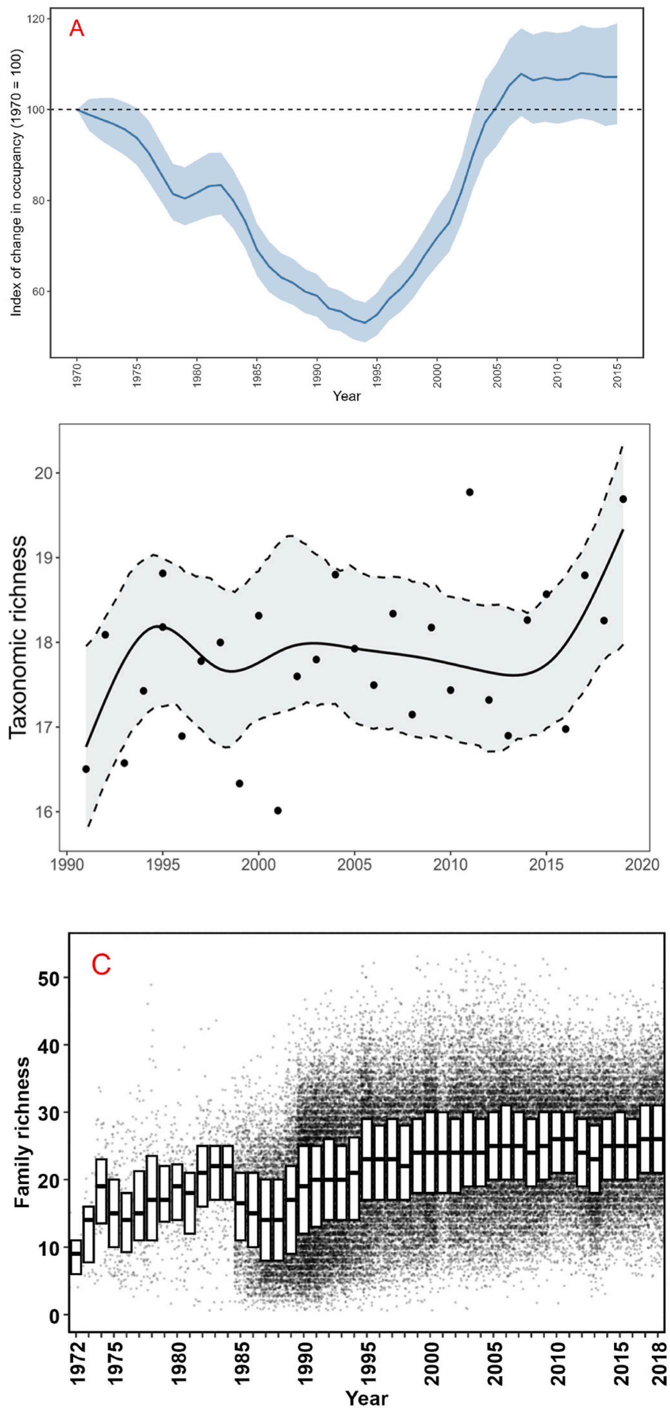

Fig. 1. Examples of studies showing (A) change in annual occupancy from 1970 to 2015 for a selection of freshwater invertebrates across the UK (Outhwaite et al., 2020), (B) change in mixed taxonomic richness across England and Wales from 1991 to 2019 (Pharaoh et al., 2023) and (C) macroinvertebrate family richness over time for 1519 observation sites and all observations across England (with median, 25th and 75th percentiles) (Qu et al., 2023).

- This slowing or levelling off observed since the mid-2000s for overall riverine family richness in England is less apparent for mayfly, stonefly and caddisfly taxa (Qu et al., 2023). The abundance picture can be mixed, with some dragonfly, beetle and true fly families declining between 2002 and 2019 (Wilkes et al., 2025) but still the broad picture is of stability or increase for most guilds (Sadykova et al., 2026).
- The Average Score per Taxon (ASPT) - a measure of the tolerance of the invertebrate community to oxygen stress and organic pollution (Armitage et al., 1983) - has increased continuously since 1990 in ways that are consistent with progressive recovery (Pharaoh et al., 2023; Qu et al., 2023). However, there is evidence that trends have been relatively static for many of the other groups since ~2010 (see Fig. 2 (Pharaoh et al., 2023),).
- Over the last 20 years (the period when the BIOSYS data included quantitative abundance data) many groups of macroinvertebrates that are sensitive to poor water quality have become more abundant in English and Welsh samples (Qu et al., 2023; Sadykova et al., 2026), including sensitive families of Ephemeroptera, Plecoptera and Trichoptera (EPT). Changes in abundance vary across trophic levels, with herbivorous and carnivorous invertebrates generally increasing and decomposers declining, but patterns are complex (Powell et al., 2022). Overall, of those species where we can be confident of change, more are increasing in abundance than decreasing (Defra, 2024; Sadykova et al., 2026).
- Overall, average family richness is still lower than that expected in the absence of human impacts (Qu et al., 2023) particularly in recovering urban systems (Pharaoh et al., 2023).

Whilst there is general agreement about the qualitative changes in macroinvertebrate communities, obtaining unbiased, quantitative estimates is hampered by the heterogeneous nature of the databases, lacking overall, statistically rigorous sampling schemes. Sampling site selection is generally non-random, and subsets of the data will have been collected according to different – and often unknown – site selection rules. As a consequence, there is a non-trivial task to reduce bias in the estimates of change through a combination of modelling and/or weighting samples to obtain a more accurate national picture (Outhwaite et al., 2020; Pharaoh et al., 2023). Recent analysis has indicated that care is needed in the choice of data filtering procedures, taxonomic aggregation, model specification as these can influence the outcome of on trend estimates (Wilkes et al., 2025).

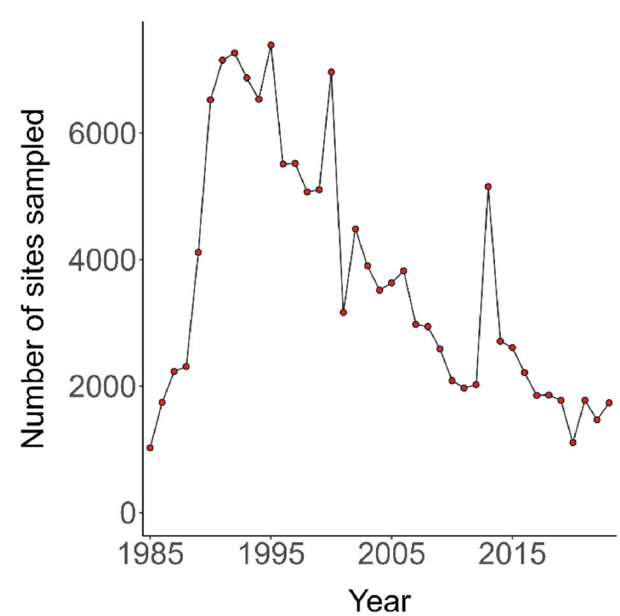

Fig. 2. The total number of sites sampled for freshwater macroinvertebrates by the Environment Agency in England each year, from 1985 to 2023 from data recorded in the BIOSYS database.

4. Are these trends plausible?

4.1. Why improvements in macroinvertebrate biodiversity may have occurred from the 1990s

National trends represent conditions averaged across sites, and these indicate that positive trends in river freshwater biodiversity began in the 1990s, with more macroinvertebrate taxa increasing in abundance than declining. A powerful feature of the macroinvertebrate community data generated by UK environment agencies is that the quality of sample processing (detection and identification of taxa) by biologists was assessed consistently over the 1991-2010 period. These quality control data reveal that approximately 13% of the increase in family richness seen over these two decades could be attributed to improvements in the laboratory (pers. comm. JL Pretty, JF Murphy, JI Jones, J Murray-Bligh). So, while taxonomic skills have progressed measurably, this alone cannot account for the increased richness. Changes in environmental conditions are a plausible alternative for the observed trends.

Over the past 30 years there have been important reductions in concentrations of many chemicals that are critical to water quality (Whelan et al., 2022), including metals, ammonia, biological oxygen demand (BOD) and phosphate. Some of this change can be attributed to the implementation of the European Urban Wastewater Treatment Directive (UWWTD, 1991 start with full compliance in 1998) whose conditions the UK applied to all rivers. It is possible to speculate that industrial decline which has reduced water pollution (Johnson et al., 2019) and the steep decline in coal burning, reducing atmospheric pollution (Tipping et al., 2010) have also improved key aspects of water quality especially in upland, base-poor waters. A recent statistical analysis of 1457 macroinvertebrate monitoring sites across England found that of 41 different chemical, physical, habitat and geographic variables, a decline in the metals zinc and copper were most closely associated with the increase in invertebrate family richness (Johnson et al., 2025b). Zinc and copper are and have had industrial, agricultural and domestic applications but the widespread basis of their contamination has probably come from historic atmospheric pollution from coal burning. It is plausible that further releases of these elements in mine wastes in old mining areas in uplands are responsible for some of these local negative trends (Johnson et al., 2025b; Pharaoh et al., 2024; Qu et al., 2023).

Not all water quality trends have been positive, however, and for example there remain concerns over emerging contaminants such as human and veterinary pharmaceuticals as well as ongoing problems with nutrients of agricultural origin (Perkins et al., 2021; Whelan et al. 2022; Boxall et al., 2024). Additionally, there have been increases in water temperatures with global climate change, which may also be influencing diversity in both positive and negative ways (Baranov et al., 2020; Jourdan et al., 2018). Current evidence suggests that the improvements in sanitary water quality since 1990 may have offset climate warming across large parts of England and Wales (Vaughan and Gotelli, 2019).

Contrasting trajectories in different aspects of water quality, climate and local biodiversity may explain some the departures among invertebrate communities from the national pattern and there is considerable variability in the response of individual sites that span increases and declines. Many sites found to have endured periods of decline of at least five years duration were in rural areas (Qu et al., 2023). The same study found a decline in richness had been occurring over the last seven years in northern England. Johnson et al. (2024) found a consistent pattern of improvement in the pollution sensitivity of stream macroinvertebrate communities in the generally urban Great Manchester region between 2010 and 2019 but declines in rural Lincolnshire over the same period. Pharaoh et al. (2023) found that both urbanised and agricultural catchments improved over the 1991-2019 period, but that recovery was more pronounced in urban rivers; the latter continued to experience increases in taxon richness right up to 2019, while rural stream communities changed little after 2010. Further examination of these spatial variations has revealed declines between 1991-93 and 2017-19 in EPT richness in some northern and western rural and upland areas of England and Wales that were apparently best explained by nutrients and climate change (Pharaoh et al., 2024).

Looking at the different macroinvertebrate groups recorded by the EA over the period 1989-2018 in more detail, Mollusca and snails increased in richness, Annelida (worms) decreased in richness, Crustacea increased in richness, Diptera increased in richness, no change in Odonata richness, Ephemeroptera, Plecoptera and Trichoptera all increased in richness (Qu et al., 2023). Examining abundance changes for some orders from the period 2003 to 2018, including 26 Ephemeroptera taxa, 14 Plecoptera taxa, 25 Diptera taxa, 65 Trichoptera taxa, 9 Odonata taxa, 34 Coleoptera taxa, 15 Hemiptera taxa and 23 Gastropoda taxa, it was found that the mean abundance trend for these orders was stable or positive except for Hemiptera which had some decline (Sadykova et al., 2026). Thus, the broad picture is that whilst there are some groups, orders or taxa which are losers, the overall trend is stable to positive for most recorded riverine macroinvertebrates.

While more work is required to assess spatial and temporal patterns, particularly the slowing of recovery in riverine macroinvertebrate richness over the past 10 years and some deterioration in rural rivers, it seems clear that river invertebrate communities across England and Wales reflect improved conditions by comparison with 30 years ago, when many factors contributed to make rivers less habitable (Harkness, 1982; Johnstone and Horan, 1996; Langford et al., 2009). Given the important position of invertebrates in the river food web, this is an encouraging indication of the health of our rivers at the national scale.

4.2. Comparison with international trends

If increased macroinvertebrate diversity can be accounted for by improvements in the river environment (Johnson et al., 2025b), we should expect to see similar patterns in other countries where environmental improvements have been made. Increasing richness in river macroinvertebrates has been seen in other European countries that have implemented the UWWTD, including the Netherlands from 1990 to 2015 (Hallmann and Jongejans, 2021), France from 1987 to 2009 (Van Looy et al., 2016), Germany from 1990 to 2020 (Haubrock et al., 2023) and Switzerland from 2010 to 2019 (Gebert et al., 2022). Recent pan-European scale analyses reported an increase in river macroinvertebrate richness over the 1968-2020 period – but both studies reported a qualitatively similar pattern in which these changes have slowed through time (Haase et al., 2023; Sinclair et al., 2024). Meta-analyses, globally (van Klink et al., 2020b) and at a European scale from 1980 to 2020 (Pilotto et al., 2020), have also shown general increases in either abundance or richness, respectively, in freshwater invertebrates, and this is hypothesised to be linked to clean water efforts. Although caution has been expressed that the former meta-analysis with only 20 sites representing Africa, South America, and large parts of Asia may not be robustly representative of true global trends (Jähnig et al., 2021).

In streams across the United States, the total density of macroinvertebrates decreased by 11% while taxon richness at sites increased by 12% over the 1993-2019 period (Rumschlag et al., 2023). In contrast to patterns seen in the UK, macroinvertebrate diversity in US urban streams decreased through time, while diversity of all other land uses (agriculture, grassland, forestry/wetland) increased. There has been a dearth of similar long-term, spatially extensive trend analyses from other regions of the world though there are some case studies published showing localised declines such as in China (Ma et al., 2023).

5. Are current monitoring strategies helpful for reporting biodiversity change?

5.1. Quantity of monitoring

Our ability to detect river macroinvertebrate diversity trends reliably depends a great deal on the consistent, and high-quality river monitoring undertaken over the long-term by regulatory agencies. However, monitoring by these agencies in England and Wales has declined (numbers of sites routinely visited) by around 75% from a peak in the mid-1990s (Fig. 2). Thus, the statistical power to detect national changes in the trajectory of freshwater invertebrates is likely to be much lower than it was, although the size of this effect will depend in part upon how the reduction in sampling effort is manifested e.g. whether sites have been lost from particular river types or regions, or spread across England and Wales, and whether sites have been lost altogether or are just sampled at a lower frequency. Nevertheless, the magnitude of the decline is such that it is likely to compromise our ability to detect trends and thus to feed back into policy development. We accept that there have been challenges to public spending since the 2008 global economic crisis, but perhaps given the benefits of high quality information on rivers are shared with sewerage undertakers and their customers, they could also share some of the burden of support.

Complementary data from biological records provides information about a broader range of taxa and supports the overall narrative of recovery. That this information exists at all is down to the dedication and interest of volunteer recorders and the recording schemes. However, this unstructured data can only provide trends in species occupancy (the proportion of occupied sites) and not in abundance. As such, these trends may underestimate change over time as large reductions in abundance can occur before a corresponding change in occupancy is observed. 'Citizen science' approaches have repeatedly been shown to be a robust complement to statutory monitoring in river environments (e.g. (Bishop et al., 2020; Brooks et al., 2019). However, they very rarely provide the same taxonomic resolution of data or standardised monitoring design that can be achieved by regulatory monitoring, and care must be taken not to over-rely on volunteers to fill gaps in standardised monitoring schemes.

5.2. Biodiversity metrics and the Water Framework Directive ecological status assessments

'Good' or 'High' ecological status waterbodies as assessed under the WFD are in the minority in England, at only 16%, but around 40% in Wales and over 60% in Scotland. In all cases, however, improvements over recent years have been modest (Giakoumis and Voulvoulis, 2019), a result mirrored in many continental European countries with similar population densities. In the UK, the reported ecological status of a waterbody is derived from several different components (biological, physico-chemical, and hydromorphological) each of which contains a number of quantified elements (the surface waters classification hierarchy). Within the biological component, quantified elements include phytobenthos, macrophyte, macroinvertebrate and fish community data. The phytobenthos metric is designed to detect the impact of phosphorus enrichment. The four macrophyte metrics include measures of taxonomic and functional richness as well as a metric responsive to nutrient enrichment (Johnson et al., 2025a). Thus, these metrics are not intended to reflect overall plant diversity, but rather to focus on those sensitive to nutrient enrichment. Macroinvertebrate data are reported as ASPT and taxonomic richness (NTAXA) which, although broad, are intended to reflect organic pollution. Fish community data are summarised as a taxonomic richness metric and believed to reflect river physical/habitat suitability. Importantly, only the lowest scoring metric within each component determines the returned classification (High, Good, Moderate, Poor or Bad) and the lowest classified component determines the overall ecological status. If the physico-chemical or hydromorphological component is classified as less than 'Good' (which is becoming more likely over time as chemical standards, such as phosphate become more stringent) then the waterbody cannot be classed as 'Good' even if the biological component indicates a healthy river (the 'one out, all out' approach; (Hering et al., 2010).

Thus, whilst information on a wide range of biodiversity metrics is gathered in the WFD process, the subsequent processing and moderation of the score by physico-chemical and hydromorphological factors will not reflect positive developments or biodiversity in most cases. The shortcomings of this WFD ecological status approach and alternative metrics of river health focusing instead on unbiased biodiversity recording has been suggested in the UK (Johnson et al., 2025a).

5.3. Communicating data with policymakers and the public

The use of aggregated environmental indicators, such as the UK Biodiversity Indicators published by JNCC (Joint Nature Conservation Committee (JNCC), 2023), to assess general trends in biodiversity at large spatial scales is becoming increasingly common. These indicators are routinely applied to monitor progress towards international targets like the Global Biodiversity Framework and the Sustainable Development Goals. The aggregate indicators are designed to simplify complex ecological trends and so can be easily quoted (and mis-quoted) by the media. In this context, the choice of indicator is vital, since the high-level statistics provided by the chosen indicator have the potential to define both policy and public rhetoric (Burgass et al., 2017).

The WFD ecological status metric was originally designed to indicate any issues to managers at the waterbody scale. Therefore, it inevitably produces a conservative narrative, and, as we have shown, aggregated data treatment can mask more informative biodiversity trends in individual quality elements. The high-level statistic 'percentage of waterbodies with good ecological status' now features in England's official indicator to assess progress towards the Environment Act 2021 (https://www.legislation.gov.uk/ukpga/2021/30/contents) (B3 State of the water environment). It is also increasingly widely quoted in public facing materials, for example The Rivers Trust 'State of our Rivers' reports. We suggest that revealing the actual biological data underpinning the aggregated biodiversity metrics and broader assessments of water quality or ecological status could help the public and policymakers take a more informed view (Johnson et al., 2025a). In particular, fuller use of the raw data could help to diagnose which of a wider range of stressors, such as climate change, are impacting wildlife, leading to better outcomes than might be possible now.

5.4. Limitations to assessing national macroinvertebrate trends

When discussing overall national trends, it must be acknowledged that most of the data discussed comes from larger rivers, and other freshwater habitats are relatively neglected. In particular, headwater streams, wetlands, small lakes and ponds within a catchment are not well served by monitoring so we know much less about the biodiversity in these ecosystems (Riley et al., 2018). This represents a significant weakness in our national biodiversity assessments. Further, due to local factors, not all catchments or even regions that are monitored will share in the macroinvertebrate richness increase or abundance changes (Pharaoh et al., 2024; Qu et al., 2023). Finally, as previously mentioned, monitoring effort by the Environment Agency in England and Wales has been generally declining in recent years (Fig. 2) and so this reduces to some degree the confidence in the current macroinvertebrate status.

5.5. Summary observations

- Riverine macroinvertebrate biodiversity recovery has occurred in association with some improvements in chemical water quality although there are departures from this overall pattern.

- The way WFD ecological status is calculated masks significant progress achieved in components of freshwater biodiversity.
- Riverine macroinvertebrate biodiversity has not fully recovered to that predicted at relatively unpolluted state.

5.6. Recommendations

- Public interest in the threats to water quality has never been greater and reliable evidence depends on consistent and widespread monitoring (wildlife, water quality and physical change). Sufficient field monitoring and further investigation will be critical to reveal the key pressures preventing further recovery to direct cost-effective remediation. The decline in support received by regulatory agencies to support their monitoring therefore raises significant public risks in detecting and understanding the quality of freshwater ecosystems. We recommend research to identify an ideal level of monitoring required to identify changes to freshwater biodiversity at the local and national levels, and that monitoring effort should be scaled to meet these requirements. Appropriate investigative research is also required to understand the processes responsible for the trends detected.
- We advocate monitoring systems and associated indicators explicitly designed for tracking changes in freshwater biodiversity at the UK, four country, and regional levels. Further to this, we recommend disentangling biodiversity data (an 'impact' variable) from bio-indicators designed to reflect changes in water quality (a 'state' variable). As such we welcome the inclusion of freshwater taxa in the official English indicator used to track change in biodiversity (Defra, 2024). We know very little about trends for wildlife in headwater streams, ponds, small lakes and wetlands and monitoring should not overlook these important components of national biodiversity.
- Given their influence on water quality and the impacts that perceptions of the public can have on the water industry, it would seem reasonable that they share perhaps a greater part of the burden of supporting monitoring efforts with public bodies. Better quality data on our rivers can bring light and take out the heat from the debate on their status, and therefore would be worth investing in.

CRediT authorship contribution statement

A.C. Johnson: Writing – original draft, Funding acquisition, Conceptualization. **Charlotte L. Outhwaite:** Writing – original draft, Methodology. **N.J.B. Isaac:** Writing – review & editing, Methodology. **K. Powell:** Writing – review & editing, Writing – original draft, Methodology. **I. Bishop:** Writing – review & editing, Methodology. **D.B. Roy:** Writing – review & editing. **J.I. Jones:** Writing – review & editing, Methodology. **J.F. Murphy:** Writing – review & editing, Methodology. **Y. Qu:** Writing – review & editing, Visualization, Methodology. **I.P. Vaughan:** Writing – review & editing. **S.J. Ormerod:** Writing – review & editing. **M.A. Wilkes:** Writing – review & editing.

Declaration of competing interest

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Data availability

Data will be made available on request.

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