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Significant improvement in freshwater invertebrate biodiversity in all types of English rivers over the past 30 years

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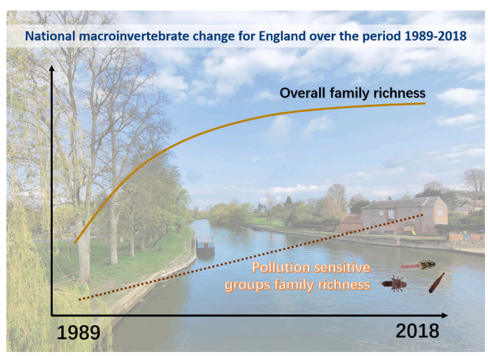
HIGHLIGHTS

- River macroinvertebrate richness has increased throughout England over the past 30 years.
- The recovery of pollution sensitive invertebrates has now reached the reference condition.
- Improvement seen across all regions, river types, land covers and wastewater exposures.

GRAPHICAL ABSTRACT



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ABSTRACT

There remains a persistent concern that freshwater biodiversity is in decline and being threatened by pollution. As the UK, and particularly England, is a densely populated nation with rivers of modest dilution capacity, this location is very suitable to examine how freshwater biodiversity has responded to human pressures over the past 30 years. A long-term dataset of 223,325 freshwater macroinvertebrate records from 1989 to 2018 for England was retrieved and examined. A sub-set of approximately 200 sites per English Region (1515 sites in total with 62,514 samples), with the longest and most consistent records were matched with predicted wastewater exposure, upstream land cover and terrain characteristics (latitude, altitude, slope gradient and flow discharge). To understand changes in macroinvertebrate diversity and sensitivity with respect to these parameters, the biotic indices of (i) overall family richness, (ii) Ephemeroptera, Plecoptera, Trichoptera (EPT) family richness, and (iii) the Biological Monitoring Working Party (BMWP) scores of NTAXA (number of scoring taxa) and (iv) ASPT (average score per taxon) were selected. A review of how close the BMWP scores come to those expected at minimally impacted reference sites was included. For all latitudes, altitudes, channel slope, river size, wastewater exposure levels, and differing proportions of upstream woodland, seminatural, arable and urban land cover, all

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diversity or sensitivity indices examined improved over this period, although this improvement has slowed in some cases post 2003. Mean overall family richness has increased from 15 to 25 family groups, a 66 % improvement. The improvement in mean EPT family richness (3 to 10 families, >300 % improvement), which are considered to be particularly sensitive to pollution, implies macroinvertebrate diversity has benefited from a national improvement in critical components of water quality.

1. Introduction

There has been a presumption that most environments around the world are suffering from continual biodiversity decline (Butchart et al., 2010). This is considered to be particularly so for freshwater environments (Dudgeon et al., 2006) which continues to be a concern (Reid et al., 2019), and for which we can find evidence in such places as China (Ma et al., 2023), with an urgent need for a recovery plan being requested (Tickner et al., 2020).

Modern societies use and dispose of vast numbers of different chemical substances (Muir et al., 2023) and concerns exist over risks from their discharge to water following wastewater treatment (Boxall et al., 2022; Daughton and Ternes, 1999). Various forms of pollution might be adversely affecting biodiversity (Groh et al., 2022; Malaj et al., 2014; Watson et al., 2019). Increasing numbers of chemicals are reported to be potentially harming freshwater organisms because their effects can be seen in the laboratory at 'environmentally relevant concentrations' (Weltje and Sumpter, 2017). Freshwater macroinvertebrates are a valuable measure of human impact on the environment (Wright et al., 1993), due to their natural biological diversity, multiple positions in the food web, range of tolerances to environmental changes, relative ease to monitor, and that their lives remain closely tied to a specific location. Hence, it is important to ascertain the status of freshwater macroinvertebrates in a developed and densely populated country like England, and assess whether or not their status has changed over time.

By international standards, English rivers would be considered highly exposed to human wastewater (Keller et al., 2014). In 2021, the population of England was estimated as 56,489,800 (The Office for National Statistics, 2022) which would translate to approximately 438 capita/km^2. It has previously been assessed that about 56 % of this population discharge their waste via wastewater treatment plants to inland rivers (rather than areas within the tidal limit/coastal), giving only 3.8 m^3 freshwater/capita/day to dilute this waste (Johnson et al., 2011). It is possible to demonstrate that macroinvertebrate biodiversity has been lower in areas of higher exposure to hazardous chemicals (Posthuma et al., 2019a), with lower ecological status related to higher wastewater exposure (Buttner et al., 2022). However, a vital question for the scientific community and policy makers is: are things improving? The assumption, based on the way results from the European Union Water Framework Directive (2000/60/EC) (WFD) are reported, might suggest they are not (Giakoumis and Voulvoulis, 2019). In contrast, an analysis of 40 plus years of monitoring data suggests that freshwater macroinvertebrate richness, abundance and biomass have improved in Europe (Haase et al., 2023) and perhaps in many other areas of the world (van Klink et al., 2020), although this opinion has been challenged by some (Desquilbet et al., 2020). In the UK, an analysis of different aquatic insect groups since 1970 showed a recovery had occurred which started in the 1990s (Outhwaite et al., 2020). Reviewing records from the last 10 to 20 years has also shown distinct improvements in aquatic insect and macroinvertebrate richness in a 9 year review period for 438 sites in Switzerland (Gebert et al., 2022), an 8 year review period of 1709 sites in the Netherlands (Hallmann and Jongejans, 2021), in France with a 25 year review period of 91 sites (Van Looy et al., 2016), and, albeit more modestly, in the USA (Rumschlag et al., 2023). An improvement in freshwater macroinvertebrate diversity in the UK, at least in urban areas, appears to have occurred based on a review of a 5 to 8 year period since the 1990s (Vaughan and Ormerod, 2012; Vaughan and Ormerod, 2014) and more recently re-analysed in a 28 year survey (Pharaoh et al., 2023). This improvement has been linked at least in part to improving water quality associated with more effective wastewater treatment (Johnson et al., 2019), driven by the European Urban Waste Water Directive.

There is good evidence that many of the basic constituents of water quality, such as metal concentrations, have improved in the UK (Whelan et al., 2022) over the past 30 years. This is important, as in a review of major hazardous chemicals in English rivers, the metals were found to have the highest relative risk (exposure × hazard) compared to a selection of other types of contaminants (Johnson et al., 2017). However, it has been demonstrated that insecticides have harmed macroinvertebrate diversity in European rural water bodies (Beketov et al., 2013; Liess et al., 2021). Lower pesticide tonnages may not necessarily reduce environmental risks if the new products are more potent (Schulz et al., 2021). The use of synthetic fertilizer has declined over the past 20 years, albeit from a high base (Muhammed et al., 2018). So, it is not clear whether macroinvertebrate biodiversity in rivers within catchments of intense agricultural activity may be following an improving trend; the suggestion is that in high arable areas at least, it might not (Ho et al., 2022; Pearson et al., 2016).

Chemical exposure is only one component of the environment that influences biodiversity. There are also physical and ecological characteristics such as temperature, flow regime, and morphology (Birk et al., 2020). Climate change, with its direct and indirect influences on temperature, flow, sediments, fish, plants and algae, may compensate for or add to other stresses on freshwater macroinvertebrates (Johnson et al., 2009; Mantyka-Pringle et al., 2014). Land cover may generate different types of human pressures on a water body (Utz et al., 2009). Ultimately, if important, the result of all of these factors will become apparent in changes to wildlife populations such as macroinvertebrate diversity.

England has a centralized database of freshwater macroinvertebrate data created and maintained by the Environment Agency (EA) and its predecessors, spanning >50 years. This was compared in this study with physical data for the same locations and human activity-related factors of upstream catchment land cover and predicted wastewater exposure. Data on individual macroinvertebrate taxa can be amalgamated into biotic indices of diversity or sensitivity. We aimed to test the hypothesis that national macroinvertebrate richness has increased over the period 1989 to 2018. We wanted to distinguish any changes firstly with respect to physical geographic factors (latitude, altitude, slope and river flow); secondly with respect to different proportions of land cover types (seminatural, arable/horticulture, woodland and urban land cover); thirdly with respect to different percentages of wastewater exposure; and finally, to discover whether differences could be discerned in trends for macroinvertebrate families considered to be more pollution sensitive?

An examination of macroinvertebrate richness trends with respect to such a broad range of landscape features and wastewater exposures has not been attempted before on this scale. In particular, it could reveal if some parts of the landscape were experiencing increasing or declining richness trends different from the others. By discriminating different wastewater exposures at each river site, this could shed light on the current concern over the impacts of domestic wastewater with its associated chemical pollutants. The inclusion of metrics such as Ephemeroptera, Plecoptera, Trichoptera (EPT) family richness, and the Biological Monitoring Working Party (BMWP) score of ASPT (average score per taxon) were selected to provide insight on the fate of potentially sensitive families. By comparing the BMWP metrics of NTAXA and ASPT with the expected values at reference sites, this would enable an assessment of how far we have yet to go to reach biodiversity expected at unpolluted sites. It is accepted that family richness will not tell the whole story of the river ecosystems, such as changes in functional diversity, however, it is a start and could be sufficient to reveal major change.

2. Methods

2.1. Introduction to the data used

The basis of the assessment was the EA's Freshwater river macroinvertebrate surveys datasets (BIOSYS), which are publicly available from the Ecology and Fish data explorer (https://environment.data.gov.uk/ecology/explorer/). This dataset contains metric and taxa information from freshwater river macroinvertebrate surveys carried out across England from mid-1960s. For latitude, altitude, slope gradient and discharge category, the entire English macroinvertebrate dataset could potentially be utilized, as this type of information was usually recorded with the biological data in the same dataset. So, for these geographic parameters it was possible to utilise 239,943 observations (back to 1965).

We would have preferred to have gone back further in time beyond 1989, but the macroinvertebrate sampling was much less frequent before that year and the data quality less certain. The EA standardized the method of survey and introduced quality assurance procedures in 1990 (John Murray-Bligh, personal comment). The total number of sites that have been visited at least once across England is 24,484. Some of the sites surveyed may have been visited only a few times to record macroinvertebrates, whilst others have a consistent set of records going back over 30 years, with up to four samples per year. As well as basic geographic factors, we wanted to examine relationships with land cover and wastewater exposure. This information is not part of the public dataset. Therefore, we looked for sites with the longest continuous records, which generated a sub-set of 1515 from across England. The aim was to identify 200 in each English Region, to link to the upstream land cover and the modelled wastewater exposure for that reach. The coverage of these sites across the EA regions of England are shown in Figs. S1, S2. The selected 1515 sites offer 62,514 discrete sets of records of macroinvertebrates for consideration (Fig. 1).

2.2. Aquatic macroinvertebrate biomonitoring data

To describe the biodiversity of macroinvertebrates across these sites over time, we considered four biotic indices. Whilst the sampling methodology used by the EA has not changed over 30 years, the level to which the taxa have been recorded has (Murray-Bligh and Griffiths, 2022). Thus, whilst species level recording is now routine across England, this was patchy before 2014. Similarly, abundance was recorded in a semi-quantitative manner before 2000 and so could not be used in assessing the whole 30-year time period. The analysis, therefore, focused on presence-absence recording at the family level, which although not as detailed as species level, was at least consistent over the recording period. This included:

Family richness (FR): this is the number of families present at a site. One family may include several species. It is an indicator of the overall taxonomic community status. Oligochaeta were not differentiated further, despite different families having different tolerances to environmental quality.

Ephemeroptera, Plecoptera, Trichoptera family richness (EPT_FR): this is the total number of families within three major orders of stream insects that are considered to be particularly sensitive to water pollution. This index is a variant of the EPT ratio, which is often used internationally as a useful measure of water quality (Garcia-Criado et al., 1999; Kietzka et al., 2019).

BMWP_NTAXA: this is one of the indices used historically by the EA to record river ecological quality for the UK government to assess compliance to the WFD. BMWP stands for Biological Monitoring Working Party and NTAXA is the number of scoring taxa (Hawkes,

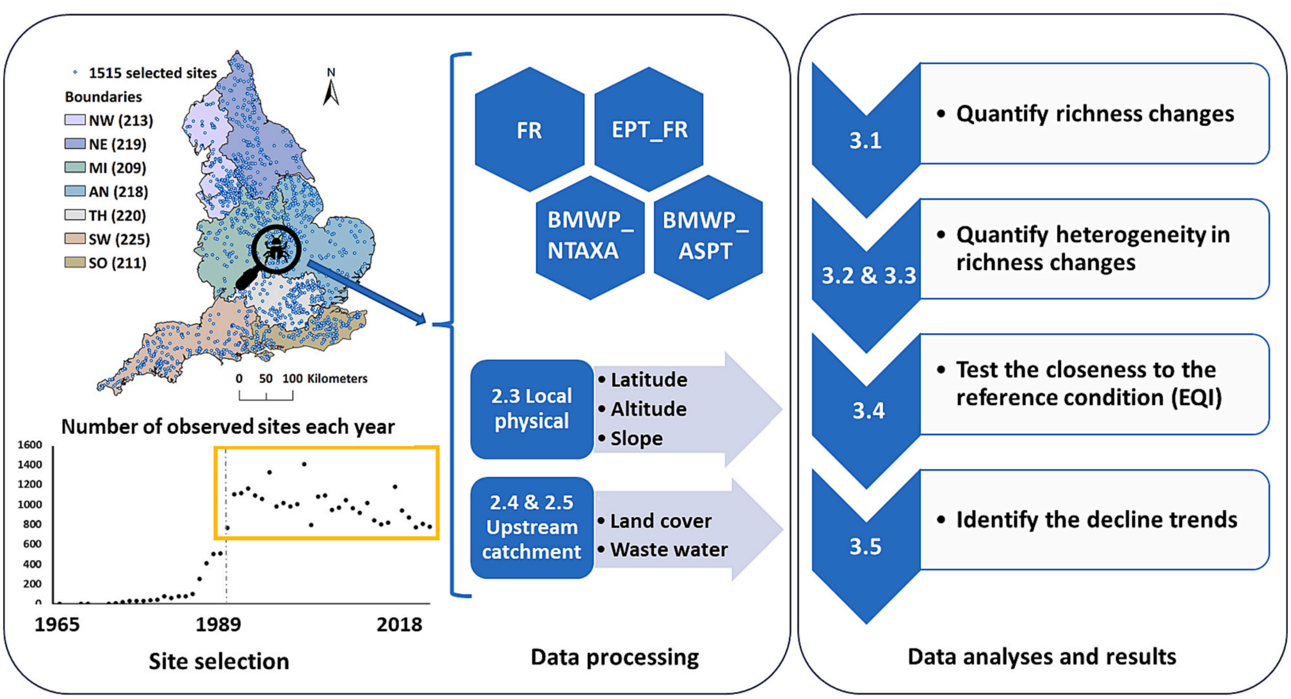

Fig. 1. Schematic diagram showing the site selection and project work-flow. NW: Northwest, NE: Northeast, MI: Midlands, AN: Anglian, TH: Thames, TH: Thames, SW: Southwest and SO: Southern region. FR: family richness, EPT_FR: Ephemeroptera, Plecoptera, Trichoptera family richness, BMWP_NTAXA and BMWP_ASPT: the number of taxa and average score per taxon in the Biological Monitoring Working Party, EQI: ecological quality index. The prefix numbers relate to sections within this paper.

1998). This is similar to family richness but only includes taxa with an assigned BMWP score. Thus, this method was capturing a smaller set of families to the overall family richness metric above.

BMWP_ASPT: this is a companion index to NTAXA. In this case, ASPT stands for average score per taxon present which have been given a form of pollution sensitive score, from 1 to 10. Thus, the higher the ASPT score, the greater the preponderance of pollution sensitive-taxa compared to pollution tolerant taxa (Walley and Hawkes, 1997).

In the UK, the BMWP scoring system has now been superseded by the WHPT (Walley, Hawkes, Paisley and Trigg), which includes both richness and abundance of the taxa to generate the scores. However, because true abundance quantification was only available post-2000, and the project wanted to include records pre-2000, we choose not to use WHPT. Where two values were available in the year at a site (nominally spring and autumn), we used the average. The potential uses and weaknesses of different biotic indices have been reviewed by Sandin and Johnson (2000). Although the focus of the study was on overall aggregated macroinvertebrate metrics, we also reviewed the national family richness changes in Bivalvia, Gastropoda, Annelida, Crustacea, Diptera, Odonata, Ephemeroptera, Plecoptera and Trichoptera, separately. To compare the biodiversity scores with an ideal or natural state, the RIVPACs model (River Invertebrate Prediction and Classification System) was used. We calculated the ecological quality index (EQI) based on the BMWP_NTAXA and BMWP_ASPT biological indices. The EQI is the ratio of the observed value to the reference or expected condition.

$$\text{EQI} = \frac{\text{value of biotic index observed in sample}}{\text{reference value of biotic index}}$$

The reference value was predicted by the RIVPACS model (Wright et al., 1998). The set of indices from the Biological Monitoring Working Party (BMWP_NTAXA and BMWP_NTAXA) was included in the RIVPACS predictive model, which allows an assessment of how different a sample is from an appropriate environmental quality level at any moment in time (Wright et al., 1993). This approach is pragmatic, acknowledging that pristine reference sites do not exist, but the diversity of minimally impacted sites of a suitable geology and hydrology can be used as suitable reference conditions. The EQI score allows an assessment of the degree of degradation in the biological status regardless of the river type (Murray-Bligh and Griffiths, 2022). A value close to 1 represents high ecological status, while the further below 1, the worse the status. We used the latest RIVPACS IV model for our database (https://www.fba.org.uk/rivpacs-and-rict/river-invertebrate-classification-tool).

2.3. Linking sites to physical characteristics

All the EA macroinvertebrate sampling sites information in BIOSYS includes location (as geographic coordinates), so for latitude we could use all the 24,484 sites present in the database. Around 60 % (15,620 to 17,373) of the sites included information on altitude, slope gradient and discharge category (Table 1). The discharge, as used by RIVPACS, represents the naturalised annual mean discharge/flow in cubic meters per second (m^3/s) estimated for the period 1961–1990 from the LF2000 model (Young et al., 2003), which is converted to a discharge category. In this case, discharge can be viewed as a surrogate to the size of the river at the monitoring point. The observations available for each variable from year to year are illustrated in Fig. S3.

Table 1
A summary of the range of the tested environmental parameters.

Environmental parameters	Minimum	Maximum	Mean	Median
Latitude (°)	49.97	55.76	52.26	51.99
Altitude (m.a.s.l.)[a]	0	1000	240	230
Slope gradient (m/km)	0	500	7.64	3.3
Wastewater exposure(%)	0	94.68	0.34	0
Mean annual discharge (m^3/s)	<0.31	>80.00	6.64	2.36
Arable land cover (%)	0	99.33	30.33	23.45
Urban land cover (%)	0	89.2	14.41	7.37
Forest land cover (%)	0	81.17	11.42	7.95
Semi-natural land cover (%)	0	98.97	43.85	39.42

[a] Metres above sea level.

2.4. Estimating land cover upstream of macroinvertebrate sites

The land cover values upstream of each macroinvertebrate sample point were extracted using a National River Flow Archive bespoke Python script and the Integrated Hydrological Digital Terrain Model (IHDTM) (Morris and Flavin, 1990) (https://www.ceh.ac.uk/data/integrated-hydrological-digital-terrain-model). The IHDTM is used to identify the upstream area draining through a macroinvertebrate monitoring point. Although the UK Centre for Ecology and Hydrology Land Cover Map series contains data for several time points, the methods used are not consistent over time. We decided to base land cover estimates only on the 2015 assessment. To simplify the IHDTM output, the 21 land cover types mapped in the Land Cover Map 2015 (25 m raster) (Rowland et al., 2017) were grouped into four large categories: woodlands, arable/horticulture, seminatural and urban land cover types and their percentage covers in the upstream catchment were calculated (Table 1). The degree of coverage from year to year can also be found in Fig. S3.

2.5. Predicting the wastewater exposure at each macroinvertebrate site

As an indication of exposure to the range of contaminants discharged from municipal wastewater, we used a GIS-based water quality model, the LF2000-WQX (LowFlows2000 Water Quality eXtension) model (Williams et al., 2009) to predict the quantity of wastewater present in the river. This is a spatially explicit model that was designed to produce distributions of chemical concentrations across England and Wales. It predicts flow throughout catchments on the basis of a 40-year climate record and combines this with location and flow of Wastewater Treatment Works (WwTWs) provided by the UK Water Industry in 2008 when the model was established. When tested against a known human contaminant, such as steroid estrogens, its predictions for concentrations within rivers have been acceptable and well within an order of magnitude of actual measurements (Balaam et al., 2010; Jobling et al., 2006; Williams et al., 2003), but in this instance we used the given WwTW dry weather flow of each plant (the designed discharge of wastewater as a daily volume) as the driver. The predicted wastewater percentage only considers the contribution of those WwTWs that contribute 90 % of the dry weather flow discharged in a catchment. Thus, very small WwTWs and septic tanks are not considered. Each biota site was assigned the wastewater value predicted for that exact river reach (Fig. 1). Of the selected 1515 sites, there were 814 sites where the predicted wastewater exposure was zero, which in many cases were sites located upstream of the modelled network boundaries, such as small streams. The degree of predicted wastewater data availability can be seen in Fig. S3.

2.6. Identifying sites where a prolonged period of macroinvertebrate richness decline had occurred

This study focused on the national average macroinvertebrate trends. However, we considered it necessary to seek out sites where significant richness declines could have occurred. There are different ways of doing this with a large dataset, in this study we searched for sites where sufficient long-term data existed for family and EPT family richness and a consistent decline for a period of at least 5 years could be identified. In this case, a decline meant that for a period of 5 years or more, the average annual biotic value was lower than the previous year. The choice of a 5-year period of decline was pragmatic. It could be imagined that declines of 1 or 2 years might be due to natural causes such as from severe droughts or floods, whereas longer, more sustained declines might point to some chronic anthropogenic problem. It is also possible that some sites have 'see-saw' periods of repeated decline and then recovery, but these would be difficult to define and discover. So, the pragmatic method we used here could be said to be most appropriate for sites that were consistently bad.

2.7. Dividing up the data to assist interpretation and statistical analysis

To understand the variance pattern of macroinvertebrate communities under different environmental gradients, each of our chosen parameters were divided into four categories to assist comparing their individual trends across England for a 30-year period spanning from 1989 to 2018. The selection of the different categories was a compromise between similar sample size and a meaningful division between them. In most cases, they were divided according to the quantile values (25th, 50th, and 75th percentile). Latitude was divided into four equal intervals of 1.5 degrees. Discharge was only available as one of 10 categories (based on naturalised mean annual discharge described above), so in this case 4 classes were selected (categories 1, 2–3, 4–6, and 7–10, corresponding to flow discharge values of <0.31, 0.31–1.25, 1.25–10, $\geq$ 10 m^3/s). For predicted wastewater exposure, over half of the sites fell into the zero category and the remaining sites were divided by three quantile division. The patterns of 30-year trends have been smoothed by local polynomial regression fitting (using function 'loess' in package 'stats' (Cleveland et al., 2017). The grey error bands display the 0.95 confidence intervals around the smoothed curves. The difference among the four groups were tested by Analysis of Variance (ANOVA, by the function of 'avo') and then further examined with the "Tukey Honestly Significant Difference" test (by function 'tukeyHSD'), after ANOVA, to identify whether any specific pairs were significantly different from one another. Both methods used the R package 'stats'. To test the significance of any trends in overall family and EPT family richness, generalized linear mixed models (GLMM, through the function 'glmmTMB' from the package 'glmmTMB') were used. We modelled family and EPT richness indicators as response variables with negative binomial distribution (via the method 'nbinom1') for each individual site. In addition, we used all observations in regression models with family and EPT richness as well as NTAXA and ASPT as responses, which were explained by time and the nine abiotic variables included in this study. Predictors were scaled to unit variance and mean of zero before conducting the GLMM regression (using function 'scale' in R package 'base'). All data analysis and visualization were conducted in R-4.3.1.

3. Results

3.1. General national picture and characteristics of the available data

Using all the available data, for all sites across England, there was an unambiguous and continuing increase in EPT family richness and BMWP_ASPT (Fig. 2). For overall family richness the average number of families have increased from 15 to 25 over 30 years. With family richness and BMWP_NTAXA an increase can be seen, but unlike EPT and BMWP_ASPT, this has slowed post-2008 (Fig. 2). It will be noted that the trends for the whole EA dataset and for the subset of 1515 sites were the same, which provides reassurance that the long-term records from the smaller set of sites can be considered representative of the national picture. The entire dataset of observations for all sites for family and EPT richness (not just the means) are shown in Fig. S4. There are 227 families recorded in total. The highest number of families at any site within the selected sub-set of sites was 56. Statistical analysis showed a clear majority of sites enjoyed a statistically significant increase in richness (Fig. 3 and Table S1). Although there was an increase in overall family and EPT richness, this degree of change was not equally spread across invertebrate groups. For Crustacea, Diptera, Ephemeroptera, Plecoptera and Trichoptera, positive to very positive trends can be seen over the 30 year time period, but this was less clear for Gastropoda and Odonata with a decline for Annelida families being apparent (Fig. S5).

Compared to other parts of the country, the higher altitude sites, steeper slope gradient sites and those with the most seminatural land cover tend to be in the west and north of England (Figs. 4 and 5). The sites with higher discharge are linked to the main stem of the major rivers. Sites with higher percentage of upstream woodland are mostly in

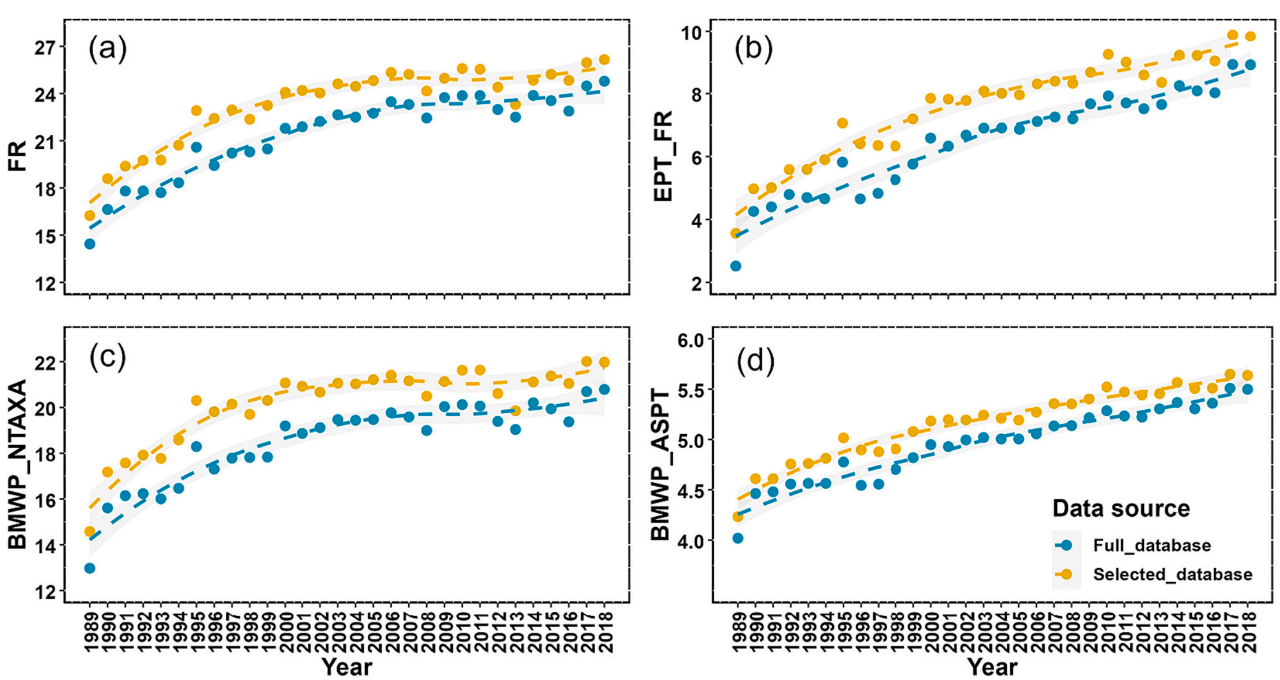

Fig. 2. Macroinvertebrate diversity trends for the full dataset of 24,484 sites across England with their 223,325 observations together with the selected 1515 sites with their 62,514 observations over time assessed as (a) overall family richness (FR), (b) the richness of Ephemeroptera, Plecoptera, and Trichoptera at the family level (EPT_FR), and two measures used by the Environment Agency to score macroinvertebrates on their susceptibility to pollution (c, BMWP_NTAXA and d, BMWP_ASPT) for rivers between 1989 and 2018.

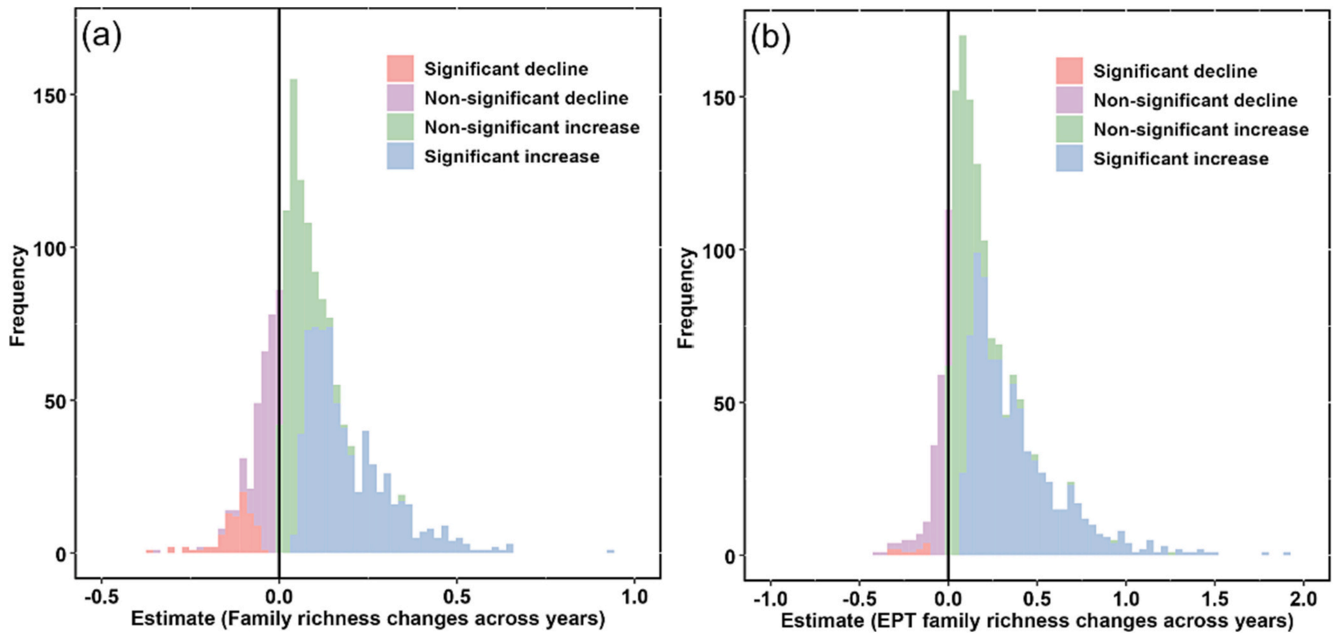

Fig. 3. Statistical analysis by generalized linear mixed model to assess the proportion of sites with significant change in (a) total family richness and (b) EPT family richness over time. The trends were assessed for each of the 1515 sites individually. The presence of an increasing or declining trend are shown by the positive and negative estimates. In this case, the significance of the trend was based on a *p* value <0.05. Further information on the proportions in these different categories are shown in Table S2.

the south east. Sites with greater arable/horticultural land cover are more common in the central and eastern region. Sites with more urban land cover and wastewater exposure correspond to the location of the major population centres in England. The graphs showing the overall number of samples (Num_sample in Fig. 4), reveal a decline of almost 80 % in sampling effort from the 1990s' to 2018. However, this decline in effort was broadly proportional across England, be it high or low latitude, altitude, slope or discharge, so we do not consider that a bias is introduced. For the smaller subset of 1515 sites matched with land cover and wastewater exposure (Num_sample in Fig. 5), the sampling effort was consistent over time, as indeed this was why they were selected for particular focus in this study.

3.2. Physical influences on biodiversity over time

Increases in richness occured at all latitudes (Figs. 4 and 6), although for the most northern latitude the improving trend appears to reverse after 2003. Absolute scores for EPT family richness and ASPT were lower in the central region (Midlands) compared to the north and south. Increases in richness occured at all altitudes, but there is a greater richness or proportion of sensitive macroinvertebrates (represented by BMWP_ASPT and EPT_FR) found at the higher altitude class (> 85 m) (Fig. 4). The influence of latitude was recorded as statistically significant at *P* <0.001 (Table S3) for all the macroinvertebrate indices. Increases in richness occured at all slopes from gentle to steep, but there is a consistent preference for the sensitive macroinvertebrates (BMWP_ASPT and EPT_FR) for the steeper slope class (slope > 8.5 m/km, Figs. 4 and S5). Increases in richness occured across sites at all river sizes, here represented by discharge, from small to large rivers, but there is a consistent preference for sensitive macroinvertebrates (BMWP_ASPT and EPT_FR) for the bigger river class (> 10 m^3/s, Fig. 4). The significance and relationships, as assessed by GLMM models, for the different macroinvertebrate indicators and geographic, land cover and wastewater factors are shown in Table S2. These analyses emphasise the positive relationships of the EPT richness and ASPT with altitude and strongly negative relationships of all macroinvertebrate indicators with upstream urban land cover (Table S2).

3.3. Land cover and wastewater influences on biodiversity over time

The results of the influences of land cover type and wastewater exposure on biodiversity are shown in Figs. 5, 7 and Table S2. Increases in richness occured across sites with a low to high percentage of woodland cover in the upstream catchment. The amount of upstream woodland in the catchment did not strongly influence macroinvertebrate richness, probably due to the generally very low proportion of woodland in England (the quartiles show that ¾ of the chosen sites have <14 % woodland in the upstream catchment). Increases in richness occured from low to high seminatural land cover in the upstream catchment. The more sensitive macroinvertebrates (EPT_FR and BMWP_ASPT) have a preference for river sites with upstream land cover with higher seminatural land in the catchment (Table S4). We observed a strong positive relationship beween EPT_FR and seminatural land cover (Table S2 & S4). Increases in overall family richness occured across sites from low to high arable/horticulture land cover in the upstream catchment. In contrast, sites with a higher proportion of upstream arable/horticulture land cover had a lower EPT family richness, nevertheless, richness increased over time. The proportion of arable/horticulture in the upstream catchment had a significantly negative effect on the sensitive macroinvertebrate indices (EPT_FR and BMWP_ASPT, Table S4). In this case, the highest category we chose was land with >49 % of the upstream catchment being laid down to arable/horticulture land cover type.

Increases in richness occured across sites whatever the degree of urban land cover. However, there is a marked preference of the more sensitive macroinvertebrates (BMWP_ASPT and EPT_FR) for sites with lower upstream urban land cover (Figs. 5 and 7). Upstream urban land cover had a significantly negative relationship to all macroinvertebrate indices, particularly for EPT_FR (with *p* value <0.001, Table S2). Increases in richness occured across sites from low to high predicted wastewater exposure. There were more sensitive macroinvertebrates families or proportions of them (BMWP_ASPT and EPT_FR) in the sites with the least wastewater exposure (Figs. 5 and 7). Nevertheless, the steady increase in the presence of sensitive macroinvertebrate groups, even in rivers with the highest wastewater exposure class (>13 %

Latitude (°) 24,484 sites		Altitude (m.a.s.l.) 17,373 sites		Slope gradient (m/km) 15,768 sites		Discharge (m^3/s) 15,620 sites	
Range	Num_site	Range	Num_site	Range	Num_site	Range	Num_site
[50, 51.5)	10050	[0, 15)	4289	[0, 1.5)	4166	< 0.31	9387
[51.5, 53)	6375	[15, 45)	4231	[1.5, 3.5)	3843	[0.31, 1.25)	3764
[53, 54.5)	6550	[45, 85)	4398	[3.5, 8.5)	3854	[1.25, 10)	1868
[54.5, 56]	1509	[85, 1000]	4455	[8.5, 500]	3905	≥ 10	601

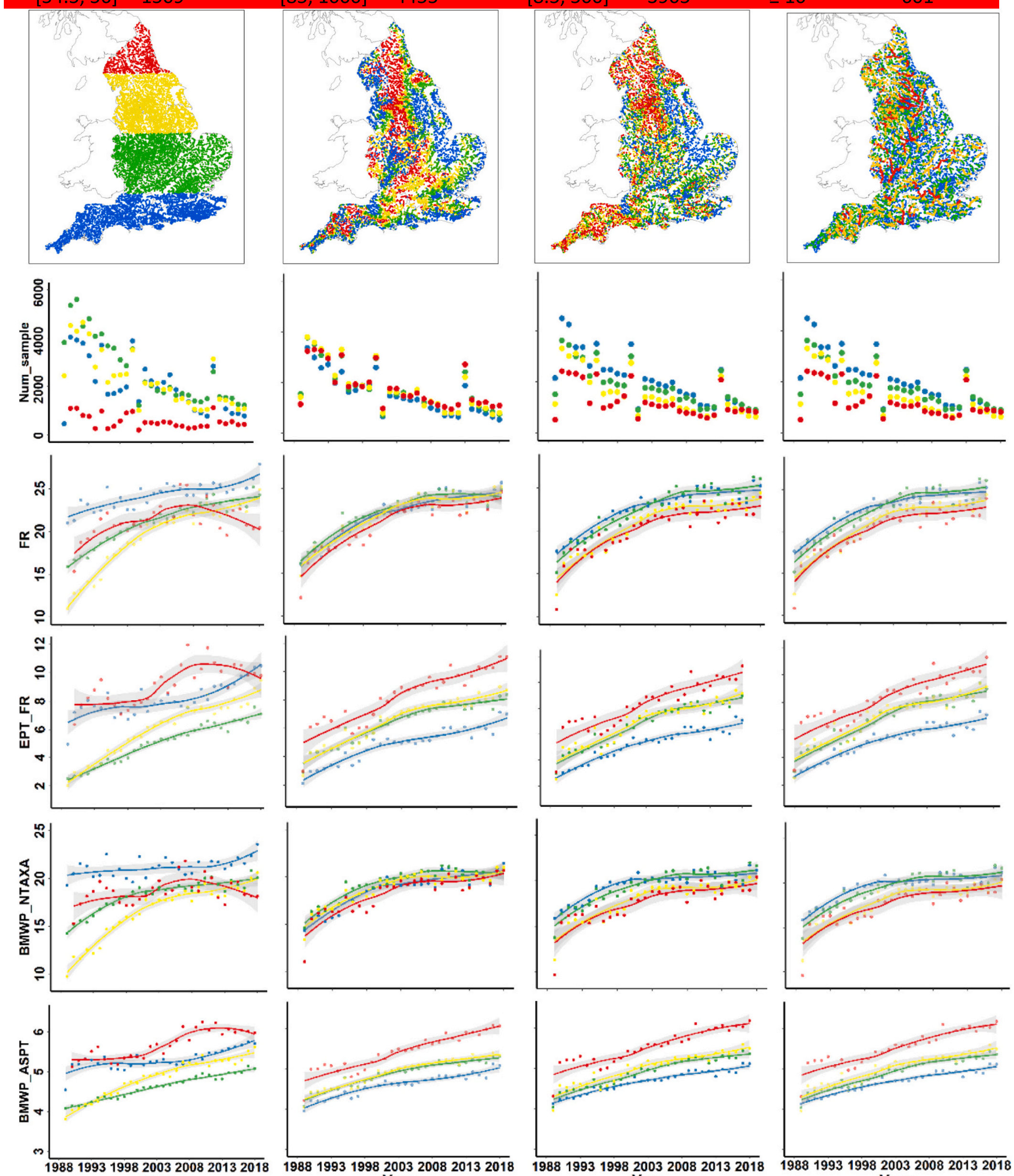

Fig. 4. Macroinvertebrate diversity trends over time for 15,620 to 24,484 sites assessed as overall family richness (FR), the richness of Ephemeroptera, Plecoptera, and Trichoptera at the family level (EPT_FR), and two measures used by the Environment Agency to score macroinvertebrates on their susceptibility to pollution (BMWP_NTAXA and BMWP_ASPT) for rivers in England between 1989 and 2018 as a function of latitude, altitude, slope, or flow discharge. The dots show the annual average value, the line is fitted by Local Polynomial Regression and the grey shaded areas are the 95 % confidence interval of that line fitting. The heading table gives the number of records in each division, the map shows the location of the sites (Num_site) used and the number of samples (Num_sample) taken each year is also shown. The colours in the figures represent a progression from lowest (blue) to low (green) to high (yellow) and highest (red). m.a.s.l. is the short name of the units of altitude, which is meters above sea level.

wastewater present in mean annual flow), is notable.

3.4. The proximity to reference conditions

Using data available from the EA for the BMWP score methods of NTAXA and ASPT for our 1515 selected sites, we plotted the EQI scores, which revealed the proximity to the appropriate reference condition (Figs. 6 and 7). These show that, in general, average NTAXA EQI scores are close to, but still below the reference condition, whereas the average EQI ASPT scores have now largely reached the reference condition from 2008 onwards. The mid-northern latitude rivers have shown the greatest improvement in EQI scores over time. In terms of NTAXA EQI scores, the

Woodland land cover (%)		Semi-natural land cover (%)		Arable&horticultural land cover (%)		Urban land cover (%)		Wastewater exposure (%)	
These parameters have been determined for 1515 selected sites									
Range	Num_site	Range	Num_site	Range	Num_site	Range	Num_site	Range	Num_site
[0, 5)	375	[0, 27)	379	[0, 3)	379	[0, 2)	375	"zero"	840
[5, 8)	382	[27, 46)	379	[3, 21)	376	[2, 5)	380	(0, 3]	224
[8, 14)	379	[46, 75)	378	[21, 49)	380	[5, 13)	380	(3, 13]	224
[14, 89]	379	[75, 100]	379	[49, 99]	380	[13, 89]	380	(13, 95]	237

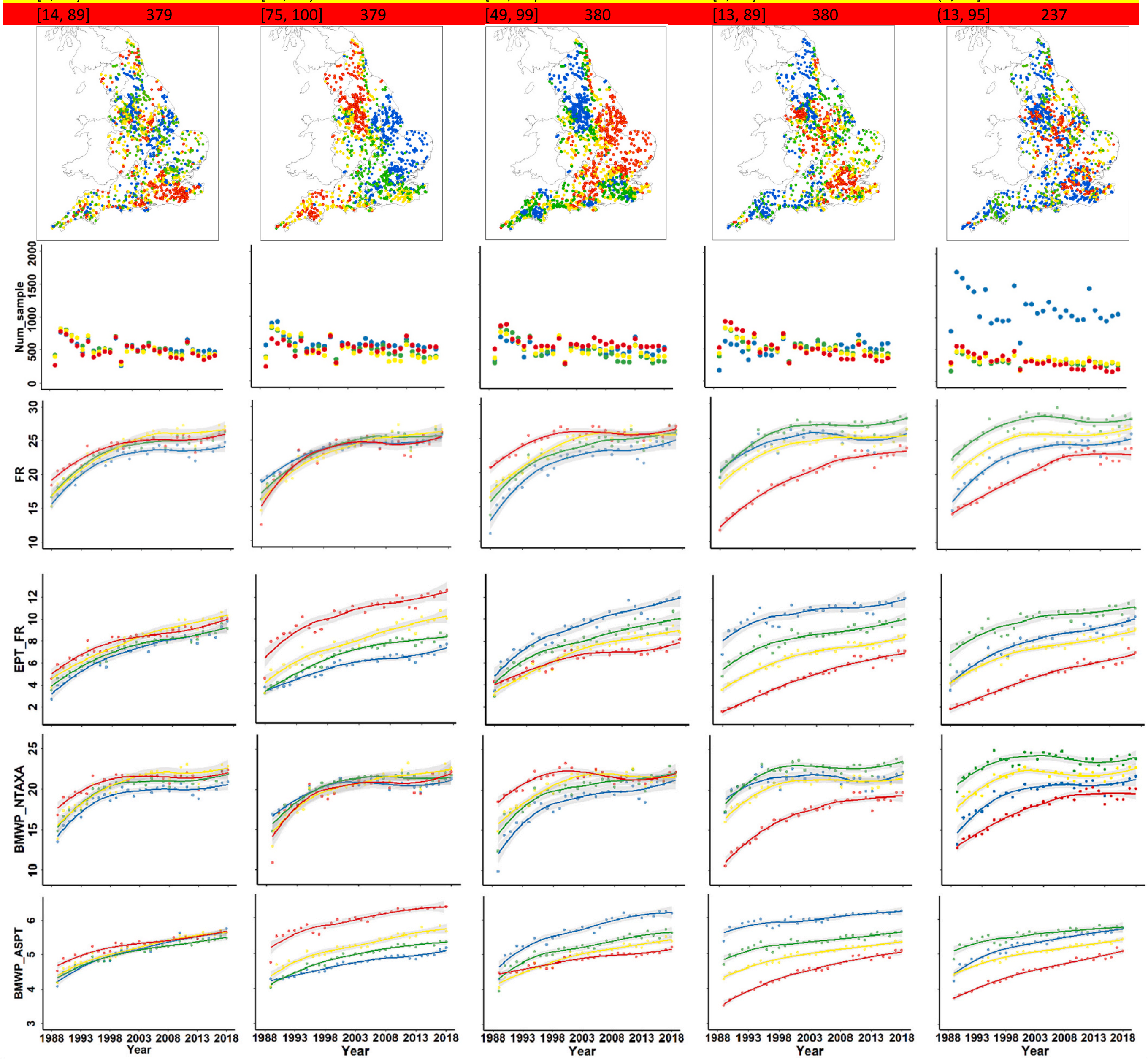

Fig. 5. Macroinvertebrate diversity trends over time for 1515 selected sites assessed as overall family richness (FR), the richness of Ephemeroptera, Plecoptera, and Trichoptera at the family level (EPT_FR), and two measures used by the Environment Agency to score macroinvertebrates on their susceptibility to pollution (BMWP_NTAXA and BMWP_ASPT) for rivers in England between 1989 and 2018 as a function of upstream land cover and wastewater exposure. The dots show the annual average value, the line is fitted by Local Polynomial Regression and the grey shaded areas are the 95 % confidence interval of that line fitting. The heading table gives the number of records in each division, the map shows the location of the sites (Num_site) used and the number of samples (Num_sample) taken each year is also shown in rivers in England. The colours in the figures represent a progression from lowest (blue) to low (green) to high (yellow) and highest (red).

sites with the highest seminatural land cover were now closest to the reference condition.

3.5. *Identifying sites where a decline of at least 5 years occured*

Only 69 sites (4.5 %) for overall family richness and 17 sites (1 %) for EPT family richness endured a sustained period of decline of at least 5 years over the 30 year study period (Fig. S6). Subsequently, most sites recovered from this decline period (data not shown). The site locations tended to be mostly rural (low urban and low wastewater). By this 5-year metric, declining sites are clearly in the minority.

Latitude (°)		Altitude (m.a.s.l.)		Slope gradient (m/km)		Discharge (m^3/s)	
These parameters have been determined for 1515 selected sites (1484 sites with EQI data available)							
Range	Num_site	Range	Num_site	Range	Num_site	Range	Num_site
[50, 51.5)	495	[0, 15)	280	[0, 1.5)	434	< 0.31	488
[51.5, 53)	413	[15, 45)	664	[1.5, 3.5)	494	[0.31, 1.25)	561
[53, 54.5)	476	[45, 85)	415	[3.5, 8.5)	314	[1.25, 10)	333
[54.5, 56]	100	[85, 1000]	395	[8.5, 500]	242	≥ 10	102

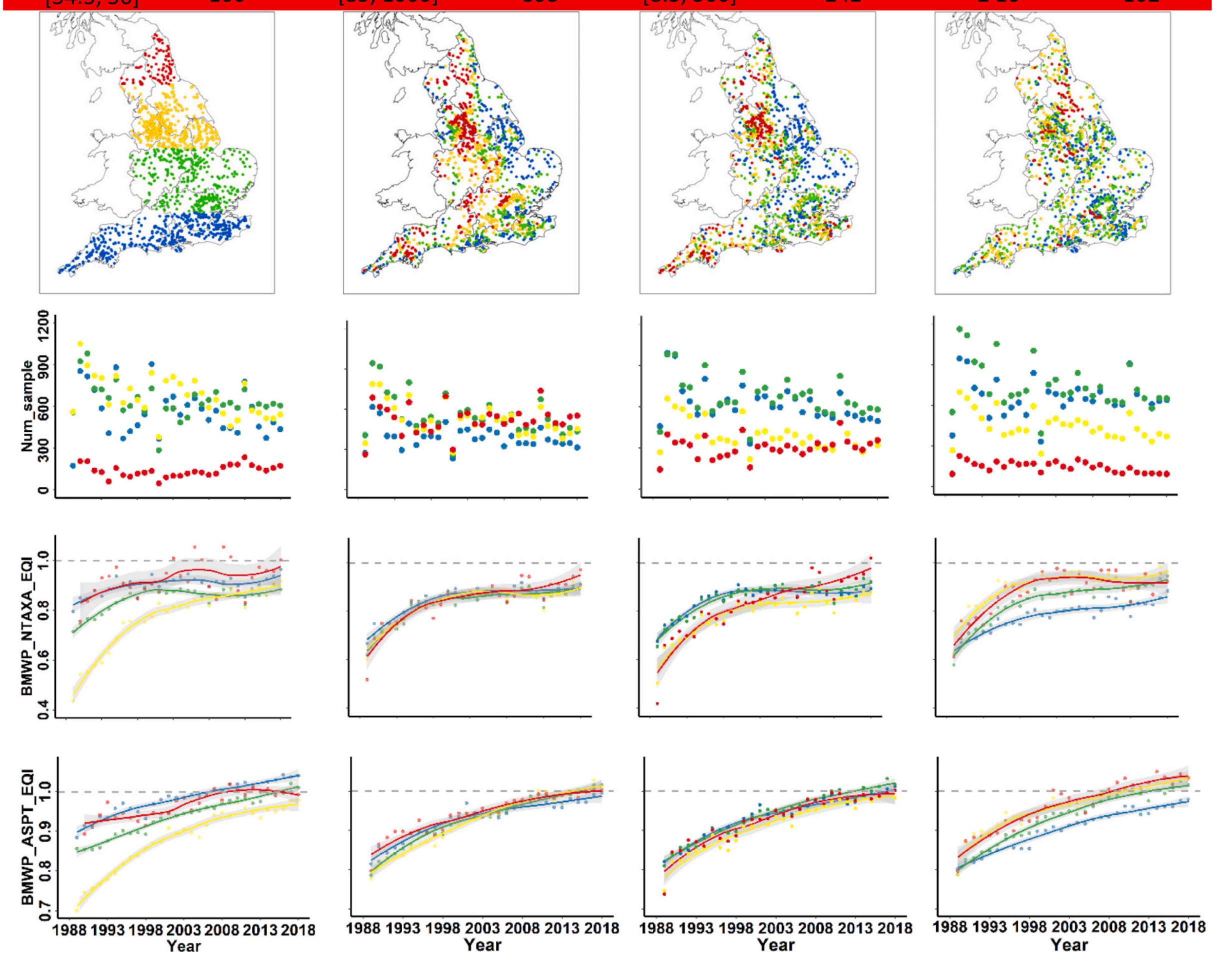

Fig. 6. Macro-invertebrate diversity trends for 1515 selected sites (1484 sites with EQI data available) over time assessed as environmental quality index (EQI) based on two of the biological indices used by the Environment Agency to score macroinvertebrates on their susceptibility to pollution (BMWP_NTAXA_EQI and BMWP_ASPT_EQI) for rivers in England between 1989 and 2018 as a function of latitude, altitude, slope, or flow discharge. The dashed line at 1.0 is the reference condition (ideal for that location). The dots show the annual average value, the line is fitted by Local Polynomial Regression and the grey shaded areas are the 95 % confidence interval of that line fitting. The heading table gives the number of records in each division, the map shows the location of the sites (Num_site) used and the samples (Num_sample) taken each year is also shown. The colours in the figures represent a progression from lowest (blue) to low (green) to high (yellow) and highest (red). (m.a.s.l. is the short name of the units of altitude, meters above sea level.

3.6. *Arrival of alien species*

There is an argument that both the improvement of family richness metrics and the ASPT scores are connected with an expansion in non-native species and not related to improvements in water quality or habitat. But the scale of the improvement (average 15 to 25 families) and consistency of improvements across all landscapes and in timing, would argue against this. We reviewed a recent list of taxa considered non-native in the UK (https://www.gbif.org/dataset/d03f444a-7192-4fa0-bf7a-63714585c611). The incidence of alien species had indeed increased, but this change for alien families was relatively modest, going from an average of only 1.0 to 1.3 family per site over the 30-year study period (Fig. S8).

4. Discussion

4.1. *Overall biodiversity changes over time*

Neither family richness or BMWP_NTAXA can reveal on their own the full potential complexity of the resident biodiversity. However, a consistent improvement in these measures of macroinvertebrate diversity due to the arrival of new families can be seen until around 2003 onwards, after which the trend slows (Figs. 2, 4 to 7). This is a change from around 15 to 25 families (66 % increase) on average. This is +1.73 % change/year from 1989 to 2018 for overall family richness. It may be that whilst the range of families have stabilised, there may still be an increase in species richness occurring (a finer level of taxanomic richness than family level); the coarse taxonomic resolution of the data

Woodland land cover (%)		Semi-natural land cover (%)		Arable&horticultural land cover (%)		Urban land cover (%)		Wastewater exposure (%)	
These parameters have been determined for 1515 selected sites (1484 sites with EQI data available)									
Range	Num_site	Range	Num_site	Range	Num_site	Range	Num_site	Range	Num_site
[0, 5)	371	[0, 27)	374	[0, 3)	372	[0, 2)	369	"zero"	814
[5, 8)	371	[27, 46)	367	[3, 21)	373	[2, 5)	375	(0, 3]	224
[8, 14)	377	[46, 75)	367	[21, 49)	366	[5, 13)	366	(3, 13]	224
[14, 89]	365	[75, 100]	376	[49, 99]	373	[13, 89]	374	(13, 95]	222

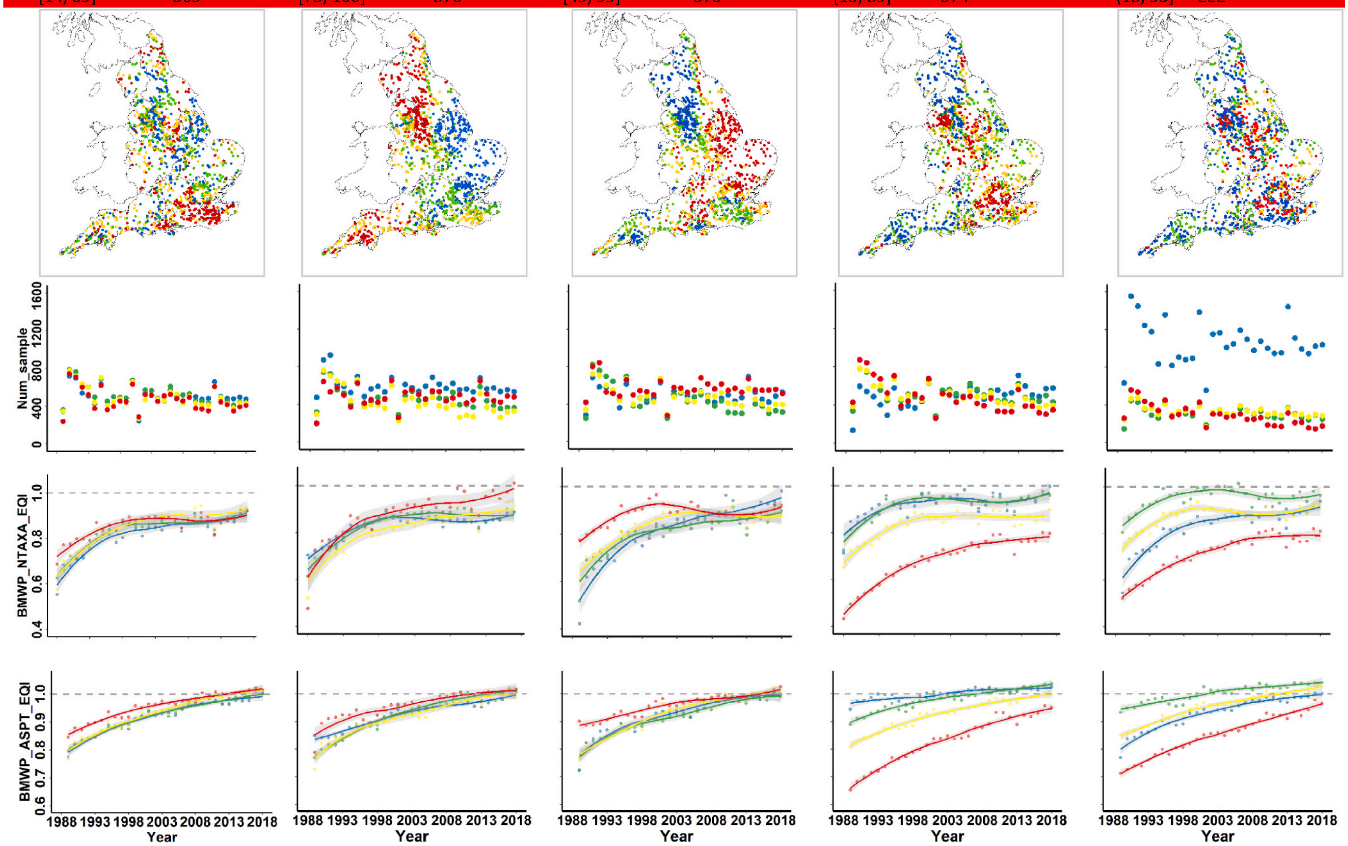

Fig. 7. Macro-invertebrate diversity trends for 1515 selected sites (1484 sites with EQI data available) over time assessed as environmental quality index (EQI) based on two of the biological indices used by the Environment Agency to score macroinvertebrates on their susceptibility to pollution (BMWP_NTAXA_EQI and BMWP_ASPT_EQI) for rivers in England between 1989 and 2018 as a function of upstream land cover and wastewater exposure. The dashed line is the reference condition (ideal for that location). The dots show the annual average value, the line is fitted by Local Polynomial Regression and the grey shaded areas are the 95 % confidence interval of that line fitting. The heading table gives the number of records in each division, the map shows the location of the sites (Num_site) used and the samples (Num_sample) taken each year is also shown in rivers in England. The colours in the figures represent a progression from lowest (blue) to low (green) to high (yellow) and highest (red).

selected in this study is a limitation here. The smaller sub-group of EPT family richness continued improving up to 2018, in most cases from around 3 to 10 families, which represents an increase >300 %. %. On average, this is +3.92 % changes/year for EPT family richness in the observed English rivers over the past 30 years, which is much higher than the +0.45 % changes/year for this class reported in a recent study based on all Europearn rivers (Haase et al., 2023). The change in different individual invertebrate groups is shown in Fig. S7.

4.2. Overall change in the community over time

Whilst in a general sense, improvements in overall family richness have slowed post-2003, as is apparent across Europe (Haase et al., 2023), the BMWP_ASPT score and EPT_FR have continued to improve. The improving BMWP_ASPT score demonstrates an increasing preponderance of higher scoring (sensitive) macroinvertebrates and a decline in the low scoring (pollution tolerant, such as Oligochaeta) families present at a site, something implied in the analysis of Vaughan and Ormerod (2014), as well as in studies done in France (Van Looy et al., 2016) and the Netherlands (Hallmann and Jongejans, 2021). In the analysis of the 1515 sites here, increases in several groups are contrasted by a slight decline in the number of annelid families richness (Fig. S5). Analysis of national overall abundance (total numbers of specimens) over time showed no particular increase or decrease over the period 2003 to 2018 (Fig. S7). Quantitative abundance data were only available after 2000 and detailed analysis by others has shown abundance trends by functional group has differed over time, with decomposers showing a relative decline (Powell et al., 2022). A decline in decomposer abundance could reflect a decline in organic matter, measured as biological oxygen demand, over time (Whelan et al., 2022). Organic rich, polluted rivers may support many individuals but few families. The divergence of trends in richness from trends in total abundance as water quality improves has been observed by others (Hallmann and Jongejans, 2021).

4.3. The implications of the biodiversity and community change results for water quality issues

All measures of biodiversity and community composition improved steadily until the 2003–2008 period, after which overall community richness improvement slowed. However, some components of the community are maintaining an increasing trend in richness as shown with EPT family richness and BMWP_ASPT, (which can be seen for different groups in Fig. S5). Perhaps some aspects of water quality remain at a level that are inhibitory to some families. There are many non-chemical factors that will influence freshwater biodiversity, including habitat physical features, the presence of macrophytes, flow and of course the surrounding landscape (Lemm et al., 2021; Mielke et al., 2022; Roy et al., 2003). We do know that across England the concentrations of many of the water quality chemicals of greatest concern have declined (Whelan et al., 2022). It is interesting to note that sites with more arable/horticulture upstream or urban land cover or the greatest wastewater exposure are all continuing to show steady improvements in the presence of sensitive macroinvertebrate families. Of these, currently, the sites with the highest arable/horticulture upstream have better biodiversity and sensitive macroinvertebrate presence than those sites with the highest urban or wastewater exposure.

The 4.5 % of sites with a > 5-year decline period in family richness could be found in any region of England (Fig. S6). The family richness of those sites on average started from 30 families, and dropped to 21; while the average EPT family richness declined from 11 to 6. They tended to be in more rural catchments, often with a high seminatural (pasture) component. Without investigating further, these declines may have been connected with isolated poor farming practice connected with manures/slurry/silage liquor or habitat damage. This reveals that we should not assume high wastewater or high urban sites will always be the sites where declines are most likely.

4.4. How far have we got to go to achieve a satisfactory macroinvertebrate biodiversity?

In recent years, there has been considerable despondency about the low numbers of European rivers achieving good or high ecological status (Carvalho et al., 2019; Grizzetti et al., 2017), with toxic chemicals often being considered to be playing a significant role in suppression of biodiversity (Lemm et al., 2021; Malaj et al., 2014; Posthuma et al., 2019b). But when scientists have looked previously at temporal changes in macroinvertebrate biodiversity in the UK (Outhwaite et al., 2020; Vaughan and Ormerod, 2012), in a manner similar to that employed in this study, an improving trend with time has been reported, which can also be seen in some continental European countries (Gebert et al., 2022; Hallmann and Jongejans, 2021; Van Looy et al., 2016). To address the question "how far is this improvement from an ideal condition", using the RIVPACS based EQI score (Figs. 6 and 7), these show that average BMWP_ASPT scores have mostly reached the reference condition. The BMWP_NTAXA are in many cases close to, but not quite at, the same level as the reference condition. The recent slowing of the rise in BMWP_NTAXA scores (and overall family richness) may, perhaps, reflect a residual presence of key universal stressors whose levels need to be reduced further. There is something of a paradox, as whilst this research, and others, show an increase in freshwater invertebrate richness has taken place over the past 30 years, there has been a general decline in terrestrial invertebrate richness over the same period in the same country (Mancini et al., 2023).

4.5. Limitations of the data and this research

To investigate change with respect to the broad physical parameters of latitude, altitude, slope and discharge, we used practically the entire English macroinvertebrate dataset of up to 24,484 sites. From a peak in the early 1990s, the amount of sampling of these sites fell over time, leaving by 2018 only around 1700 sites being routinely sampled. By and large, this decline in sampling effort was proportional across the different categories we chose for these parameters (Figs. 4 and S3). For the 1515 matched sites selected on the basis of their long record, this relatively consistent sampling effort over time can be seen in Fig. 5, with most of these sites having records covering a period of at least 25 years (Fig. S2). The consistency in the national picture of diversity increase, shown both in the full macroinvertebrate site dataset and for the selected 1515 sites with individual long-term records, suggests this change is genuine and not due to any idiosyncrasies associated with a reduction in national sampling.

This study does not imply that every region, catchment or reach of every river in England is in an ideal state. Whilst there has been a significant improvement in the crude family richness metric, it does not mean that all species are recovering, nor can we say that functional diversity is ideal. We do not dwell on abundance of the different families. The slowing of improvements in family richness and NTAXA since the early 2000 period may be sending a message that inhibitory factors remain, or perhaps new ones have arrived. However, none of these caveats should overshadow the observation that national macroinvertebrate diversity is far better now than it was in 1989.

5. Conclusions

As England is within a densely populated island (GB), with modest sized rivers with limited wastewater dilution capacity, and subject to intensive agriculture, this positive change at the coarse level of macroinvertebrate family richness should be of international note. The national recovery, at least as far as the proportion of average sensitive families present, has now reached what might be considered a target reference condition.

Whilst sites with higher wastewater exposure or probable higher pesticide exposure (arable/horticulture land cover sites) are less rich in macroinvertebrates, these have improved significantly over the past 30 years. The issue is not whether rivers are chemically contaminated, but rather whether those contaminants at their current levels matter significantly to the local wildlife.

Evidence exists that similar improvements in macroinvertebrate richness have occurred in recent decades in many parts of continental Europe. Whilst freshwater macroinvertebrates could be influenced by many different habitat, physical and chemical factors, what is striking is the; (i) universality (improvements in all types of rivers), (ii) identical timing (improvements are occurring at precisely the same time). The implication is that this is not a coincidence and that something positive has been happening across Europe most likely connected to improving freshwater quality. It may be that some universal pollutant(s), present in every river type, rural and urban, upland and lowland which is/are critical to macroinvertebrates, have declined in concentration.

As currently set out, the European WFD does not give prominence to any one marker of ecological good status above another. It could be argued that, as biodiversity is the marker that effectively reflects ecological status, so improvements at different trophic levels (algae, macroinvertebrates and fish) are the best guide to success or otherwise in the management of our rivers (above chemicals) and should therefore be considered the indicator 'primus inter pares'.

CRediT authorship contribution statement

The study was designed by YQ and AJ. Data analyses were done by YQ with support from VK, NB, ME, FE, MJ, and AJ. The manuscript was written by YQ and AJ, revised by JS and all authors contributed to improve the manuscript. All authors gave final approval for publication.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data supporting the results in the paper can be archived. We are preparing to submit the data separately. It will be openly available. We are glad to accept any request to view these data if that is desired.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.167144.

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