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Catchment controls on nutrients at coastal outlets: a national typology for England and Wales

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A R T I C L E I N F O

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A B S T R A C T

Riverine nutrient concentrations at coastal outlets reflect upstream catchment controls and underpin nutrient flux estimates. Existing catchment classifications are not tailored to this scale, limiting their application for interpreting land-to-sea patterns and informing management of marine pressures. This study develops a data-driven typology for 178 coast-draining catchments across England and Wales, grouping them into eight types through geospatial characterisation and clustering. Nutrient concentrations (2000–2024) were retrieved from water quality archives at 271 near-tidal sites.

The resulting typology is structured along a topographic gradient, further differentiated by dominant bedrock permeability. A two-way analysis of variance (ANOVA) confirmed significant differences between types for NH_4^+, NO_3^- and PO_4^{3-} ($p < 0.001$), with a significant seasonal effect for NO_3^- and PO_4^{3-} only. NO_3^- exhibited strong seasonality in arable lowland low- and high-permeability catchments where winter medians (8.8 and 8.4 mg N/L) exceeded summer values (5.1 and 6.4 mg N/L), reflecting enhanced leaching. NH_4^+ showed limited seasonality, with variability governed by point-source inputs. PO_4^{3-} displayed an inverse seasonal pattern, with higher summer concentrations from reduced dilution of point-source inputs. Despite representing only 1% of the total catchment area, the urban type dominated NH_4^+ and PO_4^{3-} concentrations, peaking in winter for NH_4^+ (0.176 mg N/L) and summer for PO_4^{3-} (0.51 mg P/L), while NO_3^- was controlled by arable types, with increasing urban influence in recent years. These results provide a framework for targeting nutrient management to the dominant controls of each type, supporting more effective nutrient reduction measures under the UK Marine Strategy and OSPAR commitments.

1. Introduction

Rivers across the UK form a pivotal connection between land and sea, transferring water, nutrients, and other constituents to coastal zones. Consequently, estuarine and marine environments are subject to the direct and indirect impacts of expanding inland human activities. Nutrients such as nitrogen (N) and phosphorus (P) are essential for supporting phytoplankton communities and maintaining ecosystem productivity (Delwiche, 1970; Hecky and Kilham, 1988; Redfield, 1958). Anthropogenic nutrient enrichment has disrupted this balance – driven by urban expansion, and industrial and agricultural intensification – leading to degraded water quality in rivers and widespread eutrophication in estuaries and shelf seas (Anderson et al., 2002; Brockmann et al., 1988; Jarvie et al., 2025; Nedwell et al., 2002; Seitzinger et al., 2010; Vermaat et al., 2008).

The introduction of synthetic P-fertilisers in the mid-19th century (Föllmi, 1996), followed by N-fertilisers (the Haber-Bosch process) in the early 20th century (Smil, 2004), caused a significant inland increase in reactive N and P inputs, with profound impacts on agricultural productivity and downstream nutrient loads to coastal areas (Beusen et al., 2016; Galloway et al., 2004; Seitzinger et al., 2010). Nitrogen inputs to shelf seas are primarily delivered through riverine discharge, the major source of reactive N in rivers is agricultural leaching from fertiliser application (Galloway et al., 2004). Atmospheric deposition of reactive N also contributes, although its impact is generally smaller compared to riverine inputs (Gruber and Galloway, 2008; Jickells et al., 2017).

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Geological weathering is the natural source of reactive P loads to seawater which is traversed mainly via riverine pathways (Benitez-Nelson, 2000; Paytan and McLaughlin, 2007), although this source has been largely masked by anthropogenic inputs from sewage effluent and agricultural fertilisers. This cascade makes riverine inputs a crucial focus for environmental monitoring and management on both ends of inland freshwaters and coastal waters.

The UK Marine Strategy Regulations (UKMS), introduced into UK law in 2010, require comprehensive assessments and management by UK countries to achieve "good environmental status" (GES) in their marine waters (Brennan et al., 2014). GES is defined through 11 'descriptors', reflecting various aspects of marine ecosystem health and associated pressures (DEFRA, 2025). Central to the eutrophication descriptor (D5), is the monitoring and reduction of nutrient inputs to coastal waters, in line with UK's commitments under the OSPAR Convention (the Oslo Paris Convention for the protection of the marine environment of the North-East Atlantic) (OSPAR, 1992). OSPAR adopted the recommendation 88/2 of 1988 of the Paris Commission (PARCOM) – its forerunner – with the aim of reducing inputs of nitrogen and phosphorus to the convention maritime area by 50% by 1995, relative to 1985 levels (Devlin et al., 2025). By 2005, six of the nine reporting countries had met the target for phosphorus, but only three for nitrogen (OSPAR, 2008). However, reporting under this recommendation was ceased in 2008, and it remains in force pending the development of country-specific targets within OSPAR's recent North-East Atlantic Environment Strategy (NEAES 2030) (OSPAR, 2021). The OSPAR Riverine Inputs and Direct Discharges (RID) programme is particularly important in this regard, providing a framework for tracking nutrient inputs from UK catchments since 1990 (OSPAR, 2015). The RID agreement mandates that contracting parties are required to report annually the inputs of 12 determinands, including 5 nutrients: ammonium (NH_4^+), nitrate (NO_3^-) and orthophosphate (PO_4^{3-}) total nitrogen (TN) and total phosphorus (TP). Across these policy frameworks, it is essential to improve our understanding of catchment attributes and nutrient sources – and how they shape the downstream nutrient inputs. This knowledge underpins accurate assessments, targeted monitoring, and practical measures to meet national and international obligations.

Previous studies have investigated the linkages between nutrient concentrations and underlying catchment characteristics; Davies and Neal (2007) applied multi-regression models to estimate nitrate and orthophosphate concentrations at 168 Harmonised Monitoring Scheme (HMS) sites across Great Britain (1990–1995). The models were fitted using baseflow index (BFI), rainfall data and land use information from the catchments. Rothwell et al. (2010a) adopted a similar modelling approach in Northeast England, predicting nutrient concentrations across around 800 river water quality sites (1995–2001). However, both studies relied on catchment-wide water quality data rather than only coastal discharge points. This introduces an element of uncertainty, as nutrient transport can vary within catchments, especially where point-sources are predominant (Crowson et al., 2024). Nonetheless, this set of catchment characteristics has proven robust across other determinands; Tye et al. (2022) and Williamson et al. (2023) used topography, land cover, rainfall and baseflow index to estimate dissolved inorganic and organic carbon fluxes respectively for major Great Britain rivers. These studies although having different objectives, converge on the importance of catchment's topography, hydrogeology, meteorology, and land use for understanding riverine nutrient fluxes to coastal areas.

The high complexity and heterogeneity of catchments in their attributes is a recurring challenge for researchers and policymakers, which is often addressed through clustering catchments into certain groups. Jarvie et al. (2002b) used Principal Component Analysis (PCA) to cluster the Humber catchments based on their attributes and assessed the variability in determinand concentrations across these clusters (1986–1996). Heasley et al. (2020) used self-organising maps to classify waterbodies (reach-scale) in England and Wales into 7 types based on their characteristics, to examine the variability in their physical habitat indices. Their reach-scale typology aligned with the spatial scales of the River Habitat Survey (RHS) and the Water Framework Directive (WFD) assessment (UKTAG, 2003). Another example of this classification was developed within the Demonstration Test Catchments (DTC) project (2009–2019) (Lovett et al., 2018; McGonigle et al., 2014; Old et al., 2020). The project clustered operational catchments through PCA into 6 types to inform agricultural diffuse pollution management (DEFRA, 2020), reflecting the catchment management hierarchy in England and Wales. Across Europe, catchment classification is well established at the WFD waterbody-scale (Kuentz et al., 2017; Lyche Solheim et al., 2019; Poikane et al., 2019). Beyond Europe, New Zealand offers another precedent: the River Environment Classification (REC) (Snelder and Biggs, 2002) and Physiographic Environment Classification (PEC) (Rissmann et al., 2024) classify catchments through a hierarchy of successive system levels including climate, geomorphology and lithology. Rissmann et al. (2024) used variance partitioning analysis to assess the spatial variation in water quality parameters across PEC classes. Although these studies differ in method and in spatial scale, they draw on a broadly common set of geospatial attributes: topography, geology, climate and land use. However, the shared rationale is that a classification built on the underlying controls is more transferrable than one based on the observed outcomes (Snelder and Biggs, 2002).

To the best of our knowledge, a coherent classification at the coast-draining scale, tied to coastal riverine inputs, has not been carried out across the OSPAR parties. This may be attributed to the UK's dense riverine drainage network and its high coastline-to-area ratio, which results in a higher number of independent coast-draining catchments and estuaries compared to other European countries (Davidson et al., 1991; Nedwell et al., 2002). No single UK river contributes more than 4% of national runoff, unlike other OSPAR parties such as France, Germany and Netherlands, where a few major rivers dominate total runoff (Littlewood and Marsh, 2005). Building on the PCA-based approaches, this study extends attribute clustering from regional and operational scales to the national scale of catchments draining directly to the coast. It is evident that the utility of clustering depends on its intended purpose – whether for scientific analysis, regulatory assessment, or practical management (Kondolf et al., 2016; Tarasova et al., 2024). UK riverine nutrient inputs reported under OSPAR's RID programme have been examined in a limited number of studies, during the 1990s, focusing on reported loads and regional trends (Jarvie et al., 1997; Littlewood et al., 1998). However, these assessments did not explicitly link reported loads to national-scale catchment typology. This paper provides the first typology of coast-draining catchments for England and Wales, linking catchment type to nutrient concentrations at coastal outlets. This study addresses two research questions:

(1) Can the coast-draining catchments of England and Wales be classified into distinct types using land use, topographical, hydrogeological, and meteorological attributes linked to nutrient sources?
(2) Do these types and their defining attributes explain seasonal and long-term patterns in riverine ammonium, nitrate and orthophosphate concentrations at coastal outlets?

Beyond regulatory reporting, developing coast-draining catchment typology could inform water quality data gap-filling in unmonitored and data-sparse regions, enhance load estimation and modelling approaches. Also, by resolving where nutrient pressures arise and how they vary by catchment type and determinand, it could support management targeted to the dominant control in each setting – rather than uniform national measures – to set and achieve nutrient reduction targets under OSPAR and the UKMS.

2. Data and methods

2.1. Catchment aggregation

The UK's dense rivers network poses challenges for using the current operational catchment delineated by the Environment Agency (EA) and Natural Resources Wales (NRW), as these are highly nested and many of them do not directly drain to the sea. This was addressed by using cumulative drainage areas, surface type (fresh or saline water) and flow direction data from the UK Centre for Ecology and Hydrology's Integrated Hydrological Digital Terrain Model (IHDTM) (Morris and Flavin, 1994; Morris et al., 1990), allowing 542 operational catchments to be spatially aggregated into 178 larger units representing distinct drainage areas, based on their downstream connectivity to tidal outlets (Fig. 1). Only aggregated catchments having at least one active and regularly sampled sites were retained for analysis, ensuring alignment between hydrological units and available water quality data.

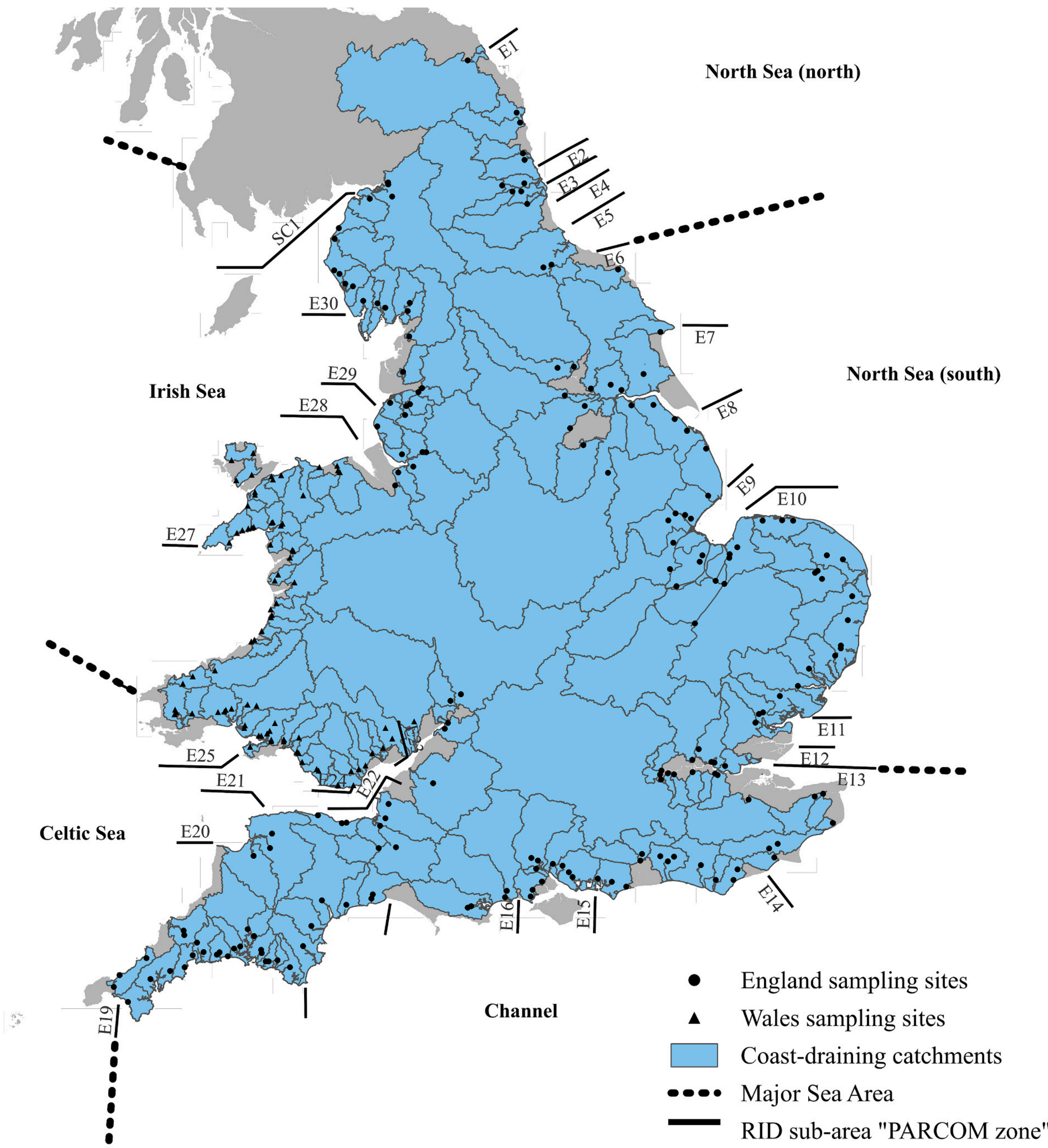

Fig. 1. Map of England and Wales's coast-draining catchments (178 catchments), aggregated from EA and NRW's operational catchments. Catchments downstream connectivity aligns with UKCEH's IHDTM, with cumulative drainage areas ranging between 48 and 10,388 km^2. The Tweed catchment, which is shared between England and Scotland, was supplemented with boundaries data from the National River Flow Archive (NRFA) due to lack of Scottish operational catchment data. Water quality sites for England (circles) and Wales (tringles) are shown, noting that some catchments are covered by multiple sites. Major OSPAR sea areas (dotted lines) and discharge zones (PARCOM zones; solid lines) assigned to catchments and their water quality sites are shown. E1-E30 denote the standardised OSPAR RID reporting zones for England and Wales.

2.2. Nutrients data

Freshwater monitoring sites were retrieved for England from the Environment Agency (EA) Water Quality Archive API (environment.data.gov.uk/water-quality-beta) and filtered to retain only riverine sites ('FRESHWATER–RIVERS' and 'FRESHWATER–UNSPECIFIED'). For Wales, riverine sites were extracted from Natural Resources Wales (NRW)'s OGC GeoPackage (Open Geospatial Consortium, datamap.gov.wales/layers/geonode:nrw_water_quality_archive_stations) with the types 'FRESHWATER–UNSPECIFIED' and 'FRESHWATER–NONCLASSIFIED RIVER POINTS' being retained. Riverine sites from both countries were then consolidated into one GeoPackage (24,595 sites) for geographical information system (GIS) processing and catchment attribution. In some cases, the most downstream monitoring sites fell just outside the catchment boundaries and were therefore not automatically attributed to the represented catchment. These sites were manually reassigned to the appropriate catchments based on their downstream connectivity.

Ammonium (NH_4^+), nitrate (NO_3^-) and orthophosphate (PO_4^{3-}) were consistently measured (2000–2024) across study catchments and were considered for analysis. Although total nitrogen and total phosphorus are reported as part of OSPAR's RID, they were sparsely measured – total nitrogen was measured from 2015 onwards at a limited number of sites – and were therefore excluded from the subsequent analysis. For Wales, nitrate was calculated as the difference between total oxidised nitrogen (NO_x) and nitrite (NO_2^-) where available. Concentrations below the limit of detection were imputed using Helsel (2011)'s Regression on Order Statistics (ROS), applied to each catchment data through the NADA R library (Python implementation of the ROS function is available at https://github.com/Hassanabubaker/censored_ros_impute). For subsequent analyses, a network of 271 riverine sites discharging at tidal limits was used (Fig. 1). This was based on historical RID monitoring sites in England and Wales (obtained from the Centre for Environment, Fisheries and Aquaculture Science (CEFAS)), with supplementary sites (active and regularly monitored) added in some historically unmonitored areas to achieve better spatial coverage. The regularity and annual sampling frequency at these sites were assessed and are presented in the results section. For each catchment, seasonal medians were calculated using two hydrological seasons: a wet winter season from October to March and a dry summer season from April to September. These descriptors were then evaluated as proxies for selecting catchment attributes to use in the clustering analysis (as detailed in the following section on catchment attributes) and post-clustering investigation of nutrient characteristics.

2.3. Catchment clustering analysis

Catchment clusters were obtained through integrating the K-means unsupervised clustering algorithm (Likas et al., 2003) and Principal Component Analysis (PCA). Catchment attribution was performed by spatially aggregating each group of attributes (Table 1) – topographical, hydrogeological, meteorological and land use – over the major coast-draining units in England and Wales. Each catchment attribute was estimated by clipping the catchment boundaries to the relevant national geospatial datasets. For each attribute group, qualitative or quantitative descriptors were estimated using spatial statistics in Python. Specific libraries were chosen according to the underlying raw data format; raster layers (land use, topography and rainfall) were processed with rasterio (Gillies et al., 2013), and vector layers (hydrogeology) with geopandas (Jordahl et al., 2020). Other datasets on known nutrient sources were explored based on the literature, but not all were used for clustering – e.g. Hydrology of Soil Types (HOST) (Boorman et al., 1995), loads entering sewage treatment works within catchments (Hoffmann et al., 2023), and fertiliser inputs (Jarvis et al., 2025). Each attribute group was separately assessed by PCA to ensure relevance for the clustering analysis. For example, when PCA was applied on the Low Flow

Table 1
Summary of catchment attribute groups, source datasets, derivation methods, and final variables used for catchment clustering derived via the k-means and PCA approach.

Catchment attribute group	Dataset	Method	Final clustering variables
Land use	Great Britain's land cover maps (LCM2021) at 25 m resolution (Marston et al., 2023)	UKCEH's 10 aggregated classes were generated from the 21 original classes (data documentation, table 1). From the correlation analysis, the aggregated classes "arable", "improved grassland", "semi-natural grassland" and "urban" were found significant to this analysis. Classes "improved grassland" and "semi-natural grassland" were combined into a "grassland" class; consistent with Davies and Neal (2007), both showed negative correlations with N and P.	*arable* *grassland* *urban*
Topography	UKCEH's IHDTM (2016 version) at 50 m resolution (Morris and Flavin, 1994; Morris et al., 1990)	Mean elevation was estimated from using the height layer of the IHDTM.	*alt*
Hydrogeology	Great Britain's bedrock permeability indices (vector data) (Lewis et al., 2020)	Catchments were characterised using the 'minimum permeability' class of the dataset, reflecting the most prevailing rate of vertical water movement. The original five classes (very high to very low) were aggregated into three classes (high, moderate, low).	*high permeability* *moderate permeability* *low permeability*
Meteorology	UK's daily rainfall, HadUK (2000–2024) at 5 km resolution (Hollis et al., 2019)	Average monthly wet days (using a 1 mm daily rainfall threshold) and average daily rainfall (adr) were calculated across the study years. Both descriptors were included in clustering to capture rainfall frequency and intensity, which improved group-wise cluster differentiation.	*Average monthly wet days* adr

HOST Groups (LFHG) as defined by Boorman et al. (1995), it did not lead to distinct catchments clustering, suggesting that these variables did not effectively capture the hydrogeological variability at the coast-draining catchment scale. Consequently, they were excluded from further analysis and substituted with the bedrock permeability indices (Lewis et al., 2020).

The selected variables were standardised using z-score (zero mean normalisation). No missing values were present in the dataset. Dimensionality reduction was applied via an initial PCA on these variables to identify principal components preserving significant variance. Four components were retained which cumulatively explained > 90% of total variance. K-means clustering was performed by iterating the number of clusters from 2 to 20. To ensure solution stability, K-means was initiated multiple times (n_init = 20) and a fixed random seed (random_state = 42), yielding consistent cluster results across runs (see python script for details). The optimal clusters number was determined through estimating Davies-Bouldin Index (DBI) (Davies and Bouldin, 1979) for clusters and Average Silhouette Index (ASI) (Rousseeuw, 1987) for catchments and the subsequent comparison of both metrics across all iterations, while considering the resulting cluster sizes. ASI is a quantitative metric that evaluates the proximity of each catchment relative to its assigned cluster, in comparison to other clusters (Shahapure and Nicholas, 2020), while DBI pertains to the separation between clusters (Heasley et al., 2020).

Following the clustering, post-clustering analyses were conducted to examine how the identified types relate to nutrient behaviour. Median seasonal concentrations calculated at the catchment scale (as described above) were used to derive cluster-level medians for ammonium, nitrate and orthophosphate, representing type-specific nutrient profiles for the period 2000–2024. Differences between clusters and seasons were assessed using Two-Way ANOVA, and the distribution of concentrations was explored using boxplots and spatial mapping. Additionally, annual mean concentrations were aggregated at the cluster-level to examine temporal trends and long-term differences between clusters. In this context, means were used to better integrate temporal variability within catchments, whereas medians were preferred for the above distributional analyses in order to reduce bias arising from skewed data, variability between catchments, and differences in cluster size.

3. Results

3.1. Catchment types

The selected catchment attributes resulted in four principal components (PC) explaining 90% of the cumulative variance; PC1 = 52%, PC2 = 16%, PC3 = 13%, and PC4 = 9%). The loadings of each attribute on these components are presented in (Fig. A.1). PC1 is characterised by strong positive loadings on rainfall variables (average daily rainfall and average monthly wet days), and strong negative loadings on *arable* and *high permeability*, while PC2 is dominated by a pronounced positive loading on *moderate permeability* and a negative loading on *low permeability* (Fig. A.3). PC3 has high positive loadings on both the *urban* and *low permeability* variables, with a strong negative loading on *high*

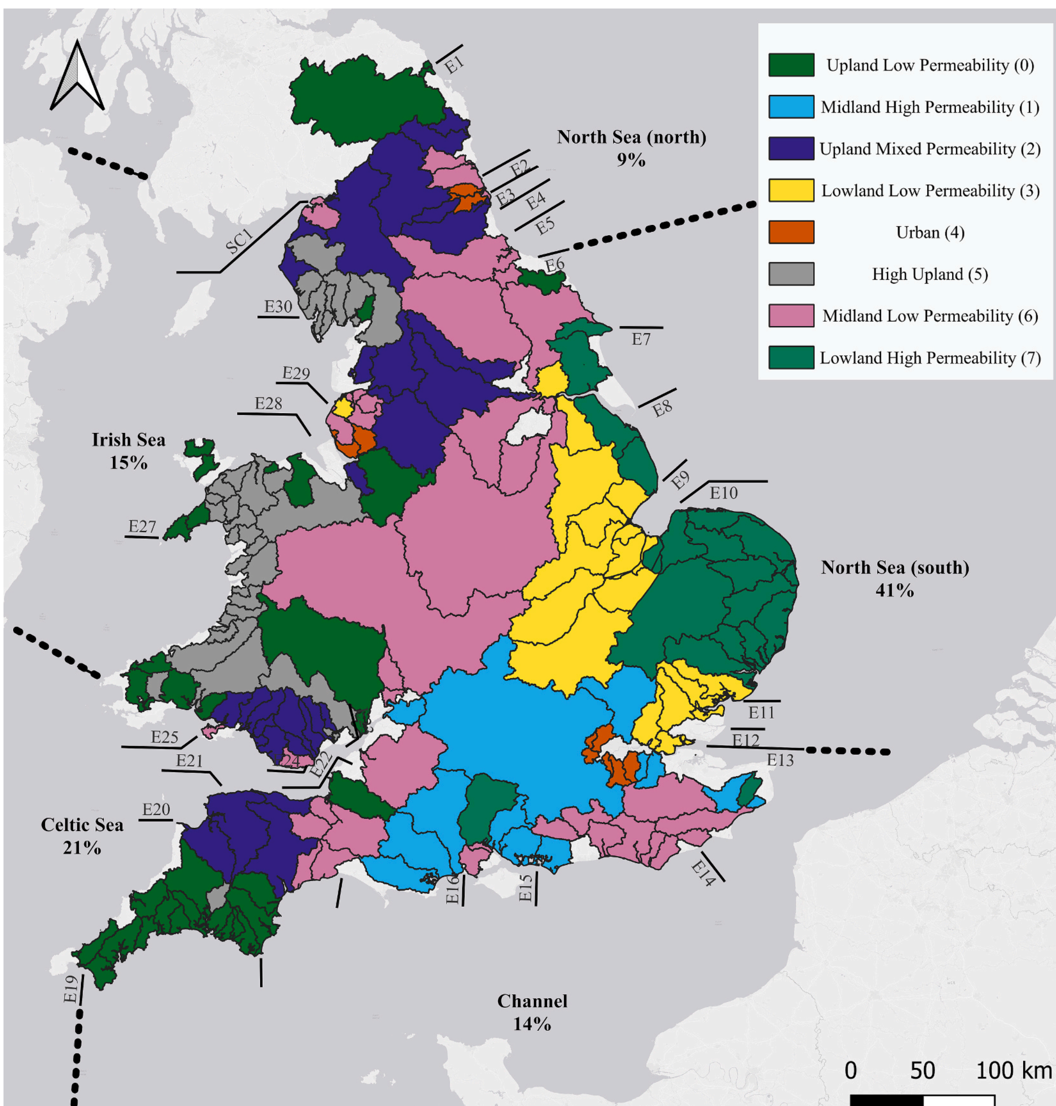

Fig. 2. Map showing the eight types of 178 coast-draining catchment across England and Wales. Major sea areas (dotted lines) and discharge zones (PARCOM zones; solid lines) are shown as per UK's historical reporting under OSPAR's RID agreement. The percentage of England and Wales's mainland drained to each major sea area is annotated below each area.

permeability. PC4 is characterised by positive loadings on *arable* and *moderate permeability*, and negative loadings on *urban* and *high permeability*.

Clustering catchment attributes using K-means and optimising the number of clusters based on Average Silhouette Index (ASI), Davies Bouldin Index (DBI) and practical cluster sizes resulted in 8 distinct catchment types across England and Wales (Fig. 2). After applying the initial PCA, the ASI indicated that the number of cluster ranges between 5 and 8 (ASI > 0.37; Fig. A.3). The ASI ranges from −1 to 1, where the values closer to 1 indicates clearer distinction between clusters. The observed ASI values (close to 0), suggest moderate separation and overlapping boundaries between catchment clusters in the 5 to 8 range. However, to ascertain the optimal number of clusters within this range, the Davies-Bouldin Index (DBI) was calculated, indicating that eight clusters had the lowest value (0.9), and thus the best separation (Fig. A.4).

The eight catchment types are summarised in Table 2, which shows the area proportion in each type and their mean values of catchment attributes. Fig. 3 provides more details on the distribution of catchment characteristics, highlighting the considerable complexity and variability among the clusters. Detailed attributes of each catchment are provided in "Supplementary material 2". Three of the identified types were significantly higher in elevation and rainfall parameters compared to the other types (Fig. 3*; d, h and i*). They are defined as High Upland (HU), Upland Low Permeability (ULP) and Upland Mixed Permeability (UMP). HU is characterised by high *mean elevation* (234 m), high *average daily rainfall* (5 mm) and high *average monthly wet days* (16 days), and its predominantly impermeable bedrock (92%). It also shows the lowest percentage of *arable* and *urban* land use, with *grassland* comprising on average 92% of the land use. HU catchments are located in the north-west of England and western Wales, draining to the Irish Sea, and their bedrock geology is composed of sandstone and slate (Dearman et al., 2011). ULP and UMP types have lower elevations (128 m and 188 m, respectively) and lower average daily rainfall (3.5 and 3.7 mm). However, both clusters show comparable geographical attributes and land use patterns, with *grassland* constituting the dominant land cover (ULP = 63% and UMP = 49%), noting that UMP is relatively more urbanised, with most catchments exceeding 10% *urban* land use (Fig. 3***.c***), such as rivers Mersey (32%), Darwen (31%) and Rhymney (28%). Another significant difference is observed in hydrogeology; ULP bedrock geology is underlain by mudstone and sandstone, resulting in consistently *low permeability* of 90%, similar to the HU type. In contrast, UMP has a more heterogeneous lithology including limestone, sandstone and mudstone, which is reflected in a mixed permeability with *moderate* (41%) and *low* (49%) indices. Spatially, UMP catchments are clustered in the north of England and around the Bristol Channel, while ULP catchments are concentrated in the southwest peninsula, with some catchments across Wales and a few scattered in England including the rivers Wye and Tweed, the largest catchments in this group.

Another two types, Lowland Low Permeability (LLP) and Lowland High Permeability (LHP), are distinguished by a predominance of *arable* land use (64% and 63%, respectively). The key attributes of these agricultural clusters include low elevations and low average daily rainfall (Table 2, Fig. 3). Most of LLP and LHP catchments are situated within the Anglian region and drain to the southern North Sea. Their main distinction lies in bedrock geology; LHP typically over Chalk aquifers (Allen et al., 1997; Dearman et al., 2011), has a high proportion of *high permeability* bedrock (89%), while LLP are dominated by *low permeability* (82%), reflecting its mudstone bedrock. The Urban cluster (U) is comprised of 8 urbanised catchments constituting only 1% of the total catchments area. These urbanised areas are concentrated at the tidal limits of the Thames, Tyne and Mersey estuaries. The final two types are primarily spanning the English Midlands and extend into the south near the Channel: Midland High Permeability (MHP) and Midland Low Permeability (MLP). MHP and MLP include England and Wales's three largest catchments, the Severn, Trent and Thames. They have mixed land use patterns (*arable*; 31 and 25%, *grassland*; 35 and 43%, and *urban*; 18 and 13%, respectively). Their fundamental difference lies in bedrock hydrogeology; MHP catchments, underlain by the southern Chalk aquifers, have an average *high permeability* of 69%, while MLP is dominated by *low permeability* bedrock (62%), with lesser contributions from *moderate* (18%) and *high permeability* (20%).

3.2. Riverine monitoring regime

Water quality sampling in the UK is typically performed monthly at best, due to the expensive and labour-intensive nature of monitoring, which restricts the ability to fully capture the temporal variability of nutrients in dynamic riverine environments. In this study, the data is used solely to assess the applicability of the identified catchment clusters and to examine their nutrient profiles. Potential biases and gaps in sampling frequency are not addressed in detail here, as they are outside the scope of this study and do not affect the main objective. Several studies have examined the impacts of sampling frequency and regularity on riverine loads estimation (Littlewood, 1995; Littlewood et al., 1998; Webb et al., 2000; Worrall et al., 2013). However, it is important to

Table 2
Summary of catchment attributes for the eight identified catchment clusters. Cluster IDs were randomly assigned by the K-means algorithm, while the labelling was informed by the differentiation between catchments on the PCA biplot (Fig. A.2). The key attributes presented are the means calculated across each cluster, corresponding to the distribution illustrated in the boxplots (Fig. 3).

Cluster id Cluster label	0 Upland Low Permeability (ULP)	1 Midland High Permeability (MHP)	2 Upland Mixed Permeability (UMP)	3 Lowland Low Permeability (LLP)	4 Urban (U)	5 High Upland (HU)	6 Midland low Permeability (MLP)	7 Lowland High Permeability (LHP)
Catchments count	31	13	30	16	8	30	33	17
Monitoring Sites Count	55	19	45	25	9	44	45	30
Proportionate area	13%	12%	15%	10%	1%	9%	29%	11%
arable (%)	*12*	*31*	*9*	*64*	*10*	*2*	*25*	*63*
grassland (%)	*63*	*35*	*49*	*18*	*21*	*68*	*43*	*19*
urban (%)	*7*	*18*	*13*	*11*	*60*	*3*	*13*	*9*
alt (m)	*128*	*92*	*188*	*36*	*57*	*234*	*85*	*47*
high permeability (%)	*5*	*69*	*10*	*14*	*25*	*3*	*20*	*89*
moderate permeability (%)	*5*	*0*	*41*	*4*	*17*	*4*	*18*	*0*
low permeability (%)	*90*	*30*	*49*	*82*	*57*	*92*	*62*	*11*
average wet days (d/month)	*14*	*11*	*15*	*10*	*11*	*16*	*12*	*11*
adr (mm)	*3.5*	*2.3*	*3.7*	*1.8*	*2.1*	*5*	*2.5*	*1.9*

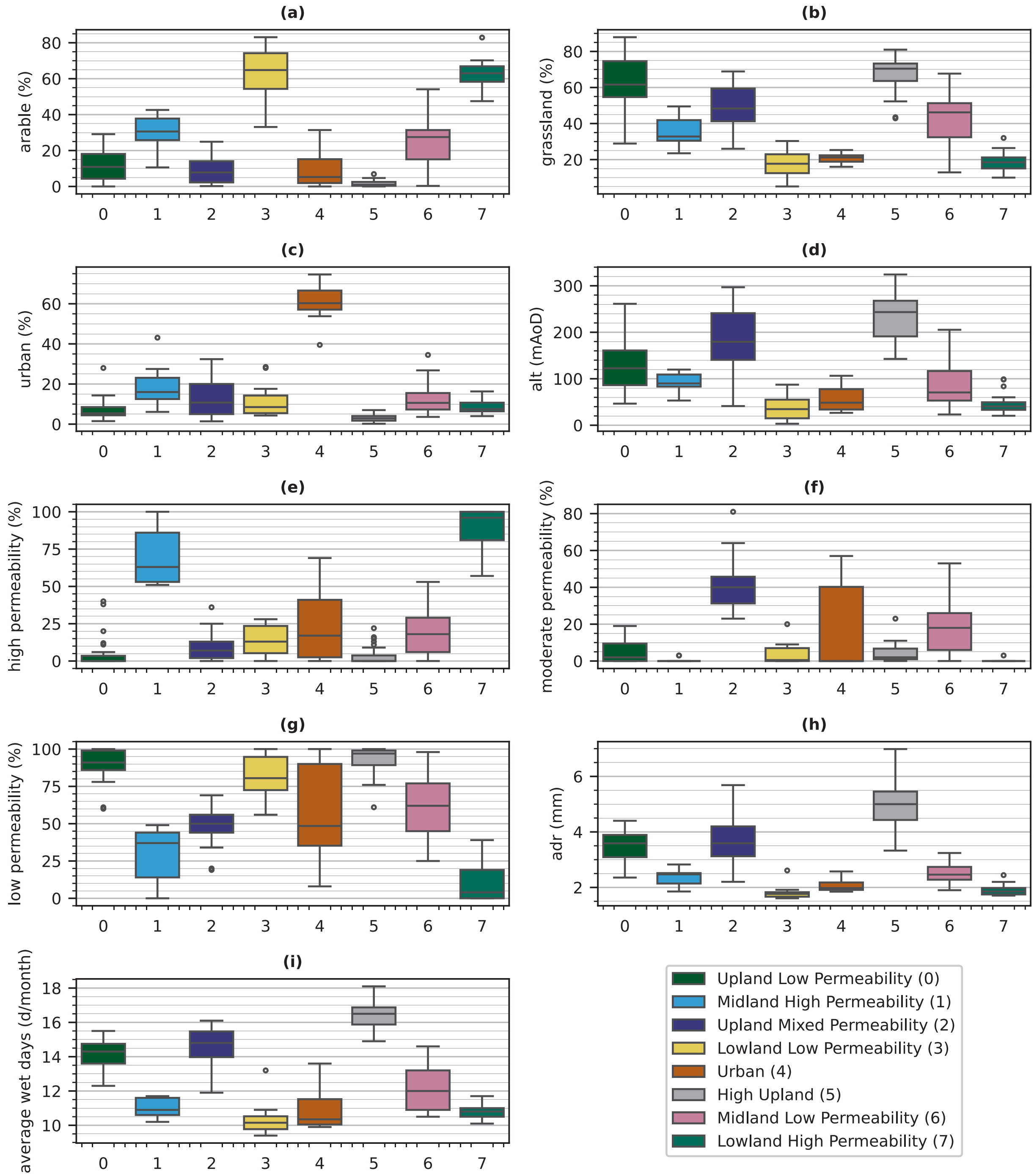

Fig. 3. Boxplots summarising the distribution of catchment attribute groups across the eight defined catchment types; [1] Land use: arable (a), grassland (b) and urban (c). [2] Topography: elevation (d). [3] Hydrogeology: high (e), moderate (f) and low (g) bedrock permeability. [4] Meteorology: average daily rainfall (h) and average monthly wet days (i). The central line represents the median, boxes indicate the interquartile range (25th and 75th percentiles), whiskers extend to 1.5 the interquartile range, and individual points represent outliers.

consider sampling limitations when interpreting the patterns and trends derived from the dataset. Variation in the number of sites and sampling regimes within clusters could bias annual averages towards areas or periods with higher sampling frequency.

Overall, 77% of sites were sampled for 20 years or more between 2000 and 2024 for nitrate, 86% for ammonium and 82% for orthophosphate. The remaining sites had shorter and more intermittent records with a minimum of 9 years for nitrate and 11 years for ammonium and orthophosphate. Fig. 4 illustrates, as an example, the variability of nitrate's sampling regime across the study sites. In 2000, 70% of sites were sampled monthly or more ($\geq$ 12 samples annually), but this proportion dropped to 30% in 2001 and 43% in 2002. During these first years, a small fraction of sites (2–7%) achieved weekly or higher sampling. From 2003 to 2013, the proportion of sites with monthly or greater monitoring steadily increased to reach 93% in 2007 – coinciding with the full implementation of the WFD's operational monitoring requirement – before declining to 69% by 2013 and 50% in 2014. Noting that between 2007 and 2019, a number of sites had sub-weekly or near-daily sampling, shown by the spread of points at higher frequencies (Fig. 4.***a***) but there was also an increase in sites with little or no sampling

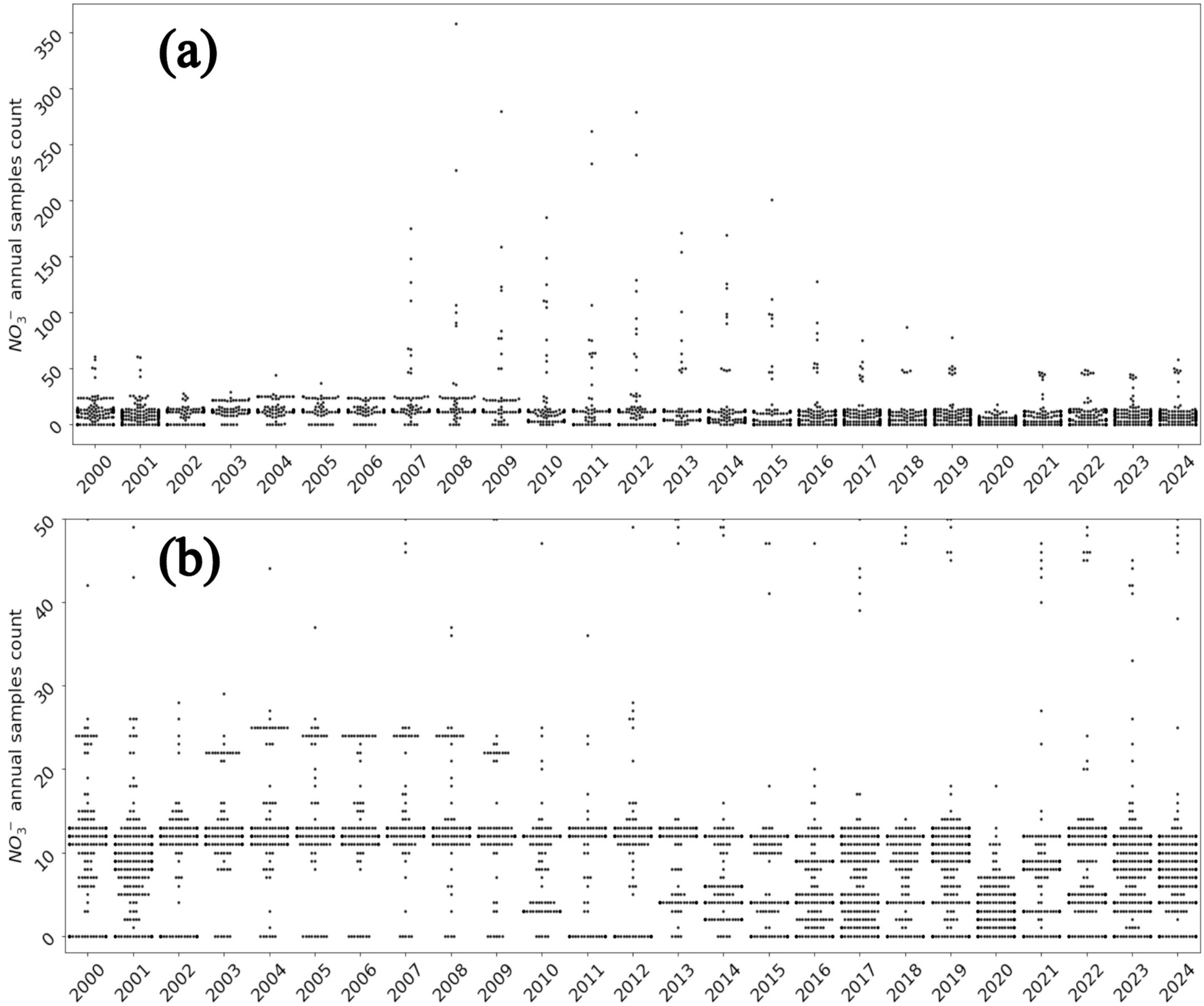

Fig. 4. (a) Scatter plot showing the annual number of samples of nitrate (NO_3^-) collected across the 271 study sites between 2000 and 2024. (b) Truncated version of "a" showing sites with up to 50 annual samples for clarity. The plot depicts the variability in annual sampling frequency to highlight the shifts in monitoring regime over time.

(increased density of points below 10). However, the period 2003–2007 shows a more consistent and inclusive monitoring regime, with fewer sites falling below monthly frequency. Sampling frequency dropped sharply in 2020, coinciding with the COVID pandemic, with only 3% of sites maintaining monthly sampling. After 2020, no sites returned to high-frequency monitoring, and at best, only 28% of sites maintained a monthly regime. A table of annual number of samples for each nutrient across sites is provided in "Supplementary material 3".

3.3. Nutrients across catchment types

The distribution of seasonal nutrient concentration across catchment types is presented in (Fig. 5). Two-Way ANOVA indicated a significant statistical difference ($p < 0.001$) between the identified catchment types for ammonium (NH_4^+), nitrate (NO_3^-) and orthophosphate (PO_4^{3-}). For NH_4^+, the test showed no significant effect of season ($p = 0.1$) nor significant interaction between season and cluster ($p = 0.9$). In contrast, both factors were significant for NO_3^- (season; $p = 0.003$ and season-cluster interaction; $p < 0.001$). For PO_4^{3-}, there was a significant impact of season ($p = 0.02$) and no significant interaction between the two factors ($p = 0.5$). Detailed median seasonal nutrient concentrations for each catchment are provided in "Supplementary material 2".

3.3.1. Ammonium

The urban type had the highest median ammonium (NH_4^+) concentration in summer (0.10 mg N/L) and winter (0.176 mg N/L) (Fig. 5.***a***), with the highest values recorded in Sankey Brook and Ditton Brook catchments near Liverpool. The elevated NH_4^+ concentrations reflect the intense urbanisation in type 4 catchments. A similar pattern was identified in the urban type's annual mean concentrations (Fig. 6.***a***), particularly before 2005, after which NH_4^+ showed a decline that reflected improvements in sewage treatment in compliance with the Urban Wastewater Treatment Directive (UWWTD, 91/271/EEC). However, two spikes appeared in 2002 and 2018, where the annual means were biased towards the urban catchments near Liverpool; the four London catchments have not been sampled in those years.

Very high ammonium concentrations were observed in several types, such as the River Weaver (ULP type) and the River Lee (MHP type). More prevalent outliers appeared within the MLP type in the adjacent Yarrow and Lostock, Douglas, and Alt catchments, and within the UMP type, in

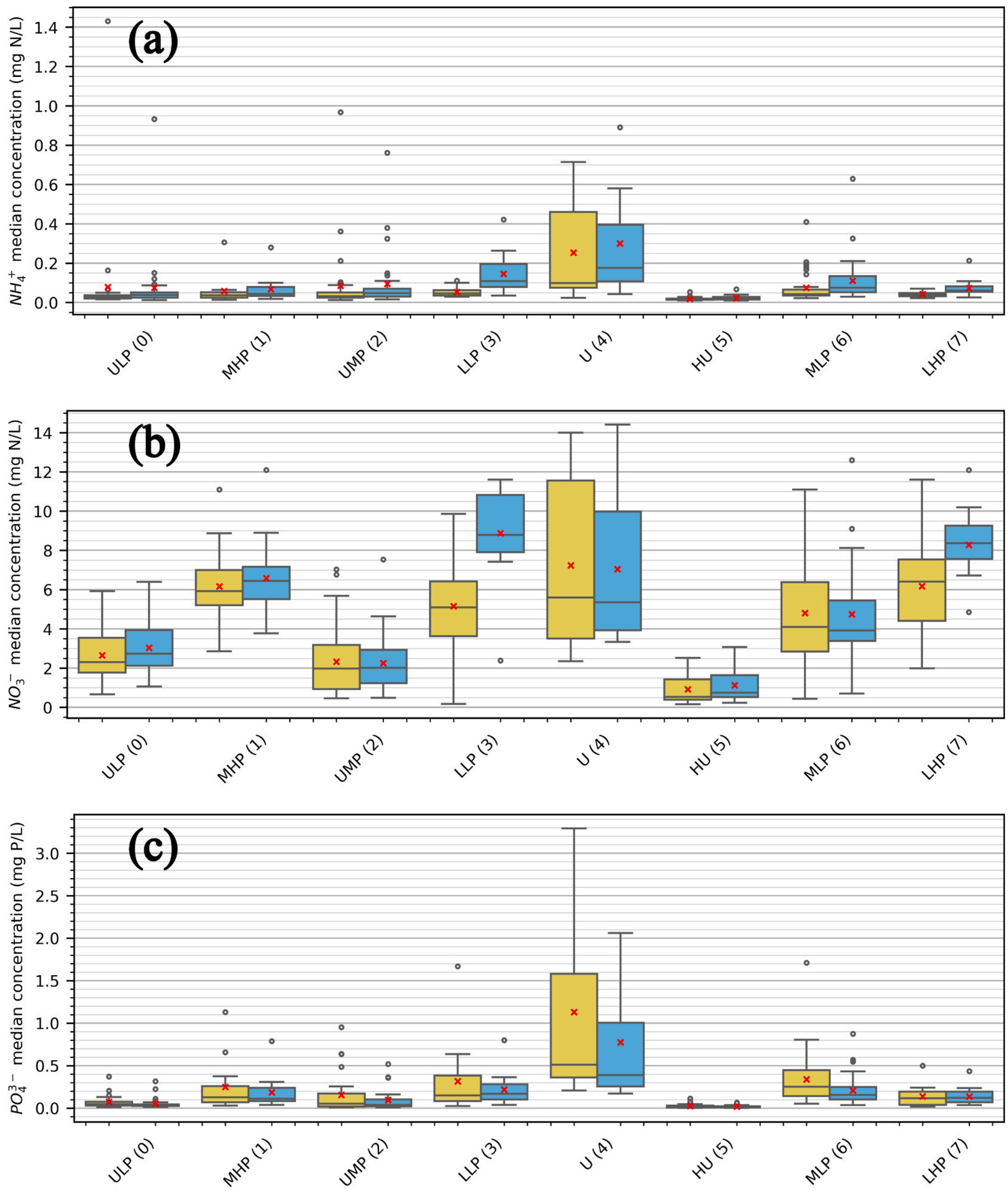

Fig. 5. Boxplots summarising the distribution of catchment median concentrations for (a) ammonium, (b) nitrate and (c) orthophosphate during summer (yellow) and winter (blue) across catchment types between 2000 and 2024. The central line represents the median, boxes indicate the interquartile range (25th and 75th percentiles), whiskers extend to 1.5 the interquartile range, and individual points represent outliers. The red 'x' denotes the cluster mean value. Due to the skewness of distributions and the influence of extreme values that are evident in the significant disparities between mean and median estimates; the focus of interpretation is on median values. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

the Eden, Mersey, Aire and Darwen catchments. Most of these extreme values are associated with highly urbanised and industrialised areas around the Mersey (Rothwell et al., 2010b) and Humber (Jarvie et al., 2002b) regions (Fig. 7). Most notable is the Eden catchment in Cumbria which has less than 3% urban land use, yet it recorded elevated NH_4^+ values, reaching more than 0.7 mg N/L (Fig. 5), driven by a monitoring site in Carlisle that showed consistent spikes between 2008 and 2015, however, Carlisle's major STW is located downstream of this site. These localised point sources produce concentrations comparable to those observed in urban catchments despite the predominantly rural land use of the wider basin. Consequently, while type medians reflect broader catchment characteristics, outliers represent site-specific anthropogenic inputs. The HU cluster had the lowest NH_4^+ concentrations, summer and winter medians were 0.016 and 0.018 mg N/L, respectively. This indicates the minimal urban and agricultural activities in these catchments, where both urban and arable land cover averaging less than 5%.

Although the Two-Way ANOVA showed no significant interaction between cluster and season for NH_4^+; the median ratio of winter to summer concentration for each type (s-ratio) suggests a broadly consistent winter enrichment across types. Individually, most catchments (145 of 178) showed elevated ratios in winter (s-ratio > 1, Fig. 7), indicating that higher winter concentrations are a general feature of NH_4^+ dynamics. This likely reflects increased hydrological connectivity and mobilisation during wetter months, combined with reduced in-stream processing in winter. The arable types had the highest ratios (LLP $=$ 2.3 and LHP $=$ 1.4) which reflects the increased contribution from nitrogen fertilisers leaching in the wet season (Cooper and Hiscock, 2023), noting that there is a relative difference between the two types.

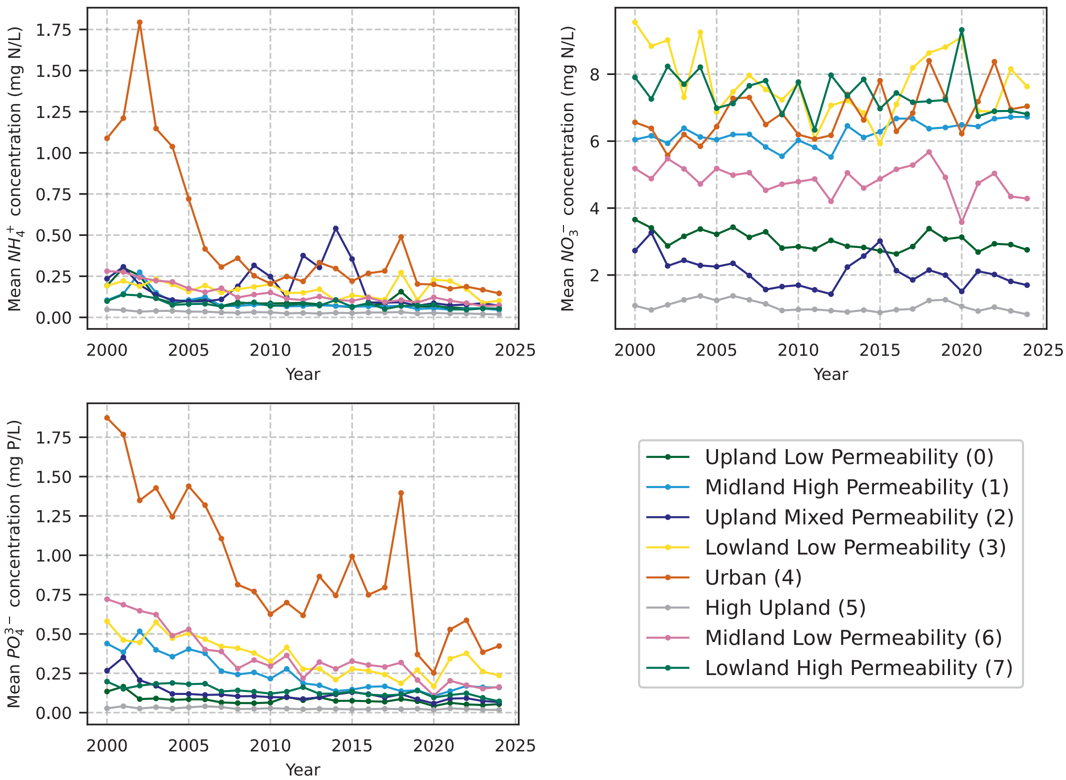

Fig. 6. Temporal variability of annual mean concentrations across catchment types; (a) ammonium, (b) nitrate and (c) orthophosphate, during the period 2000–2024.

This corresponds to their contrasting hydrogeology; LHP is dominated by aquifers which incline to buffer and delay nutrient transport, while LLP is dominated by impermeable bedrock, leading to event-driven response and rapid mobilisation of nutrients in winter.

3.3.2. Nitrate

The arable types (LLP and LHP) had the highest median annual nitrate (NO_3^-) measurements prior to 2005 but were then matched or surpassed by the urban cluster up to 2024 (Fig. 6.***b***). This shift is simultaneous with major sewage improvements and the expansion to secondary treatment. The Urban Wastewater Treatment Directive (UWWTD) mandates secondary treatment (biological) for agglomerations ranging between 2,000–15,000 population equivalents (PE), and tertiary treatment for those exceeding 15,000 PE (UWWTD, 91/271/EEC). The secondary treatment provides enhanced nitrification that reduces ammonium inputs but increases NO_3^- concentrations (Earl et al., 2014). LLP and LHP had the highest median NO_3^- concentrations in winter (8.8 and 8.4 mg N/L, respectively), followed by the MHP (6.4 mg N/L) and urban (5.4 mg N/L) types (Fig. 5.***b***). Both LLP and LHP of eastern England (Fig. 8), exhibited pronounced seasonality (s-ratio; LLP = 1.6 and LHP = 1.3), shown also by the sharp rise in winter boxplot's lower whiskers (Fig. 5.***b***) – All catchments within these two types exhibited s-ratio greater than one, indicative of increased inputs from fertiliser leaching during wetter months. In contrast, the urban type displayed more complex NO_3^- dynamics; where some catchments were summer dominated while others peaked in winter. This variability likely indicates differences in the relative contribution of point sources and hydrologically driven diffuse sources. In effluent-dominated systems, low summer flows increase the proportion of treated wastewater discharged into rivers, resulting in higher nitrate concentrations. By contrast, catchments where NO_3^- inputs are controlled by runoff and groundwater recharge, tend to exhibit maximum concentrations in winter, associated with rainfall-driven leaching of fertiliser (Crowson et al., 2024; Davies and Neal, 2007; Neal et al., 2000).

The midland types follow the urban and arable types in NO_3^- concentrations (Fig. 5.***b*** and Fig. 6.***b***). MLP displayed little seasonal change (summer = 4.1 and winter = 3.9 mg N/L), while MHP showed more comparable medians to the arable clusters (summer = 5.9 and winter = 6.4 mg N/L), suggesting increased mobilisation likely associated with the greater proportion of arable land (31%) and Chalk aquifers (high permeability bedrock = 69%), particularly in the southern catchments (Jarvie et al., 2002a; Jarvie et al., 2005b). As previously noted for ammonium, the River Lee catchment in London is a persistent outlier in the MHP type for all nutrients (Fig. 5); this is likely linked to the influence of Deephams Sewage Treatment Works (STW) in London, which discharges to Pymmes Brook before its confluence with the river upstream of the monitoring site. Deephams serves nearly one million in population equivalent (PE), besides the upper catchment's substantial arable land (40%), further contributing to elevated nitrate and ammonium concentrations. Overall, mapping the distribution of seasonal median concentrations demonstrate that heavily urbanised catchments, and those with a mix of urban and arable land, consistently exhibited high NO_3^- concentrations in both seasons, while predominantly arable catchments escalated during the winter season (Fig. 8).

3.3.3. Orthophosphate

The eight urban catchments dominated orthophosphate (PO_4^{3-}) concentrations, with median concentrations of 0.51 mg P/L in summer and 0.39 mg P/L in winter, with the highest values recorded in Ditton Brook and River Wandle. The seasonal pattern is evident across all catchments within this type, where the highest PO_4^{3-} concentrations were recorded in summer (Fig. 5.***c***). A marked decline in the annual average concentration has occurred since 2000 (Fig. 6.***c***), this drop reflects the previously mentioned improvements of the UWWTD, while further reductions after 2013 correspond to the European Union's regulations limiting phosphates in washing (2013) and laundry (2017) detergents, which were enforced during that period (Comber et al., 2013; Richards et al., 2015). However, a spike is present in 2018 where

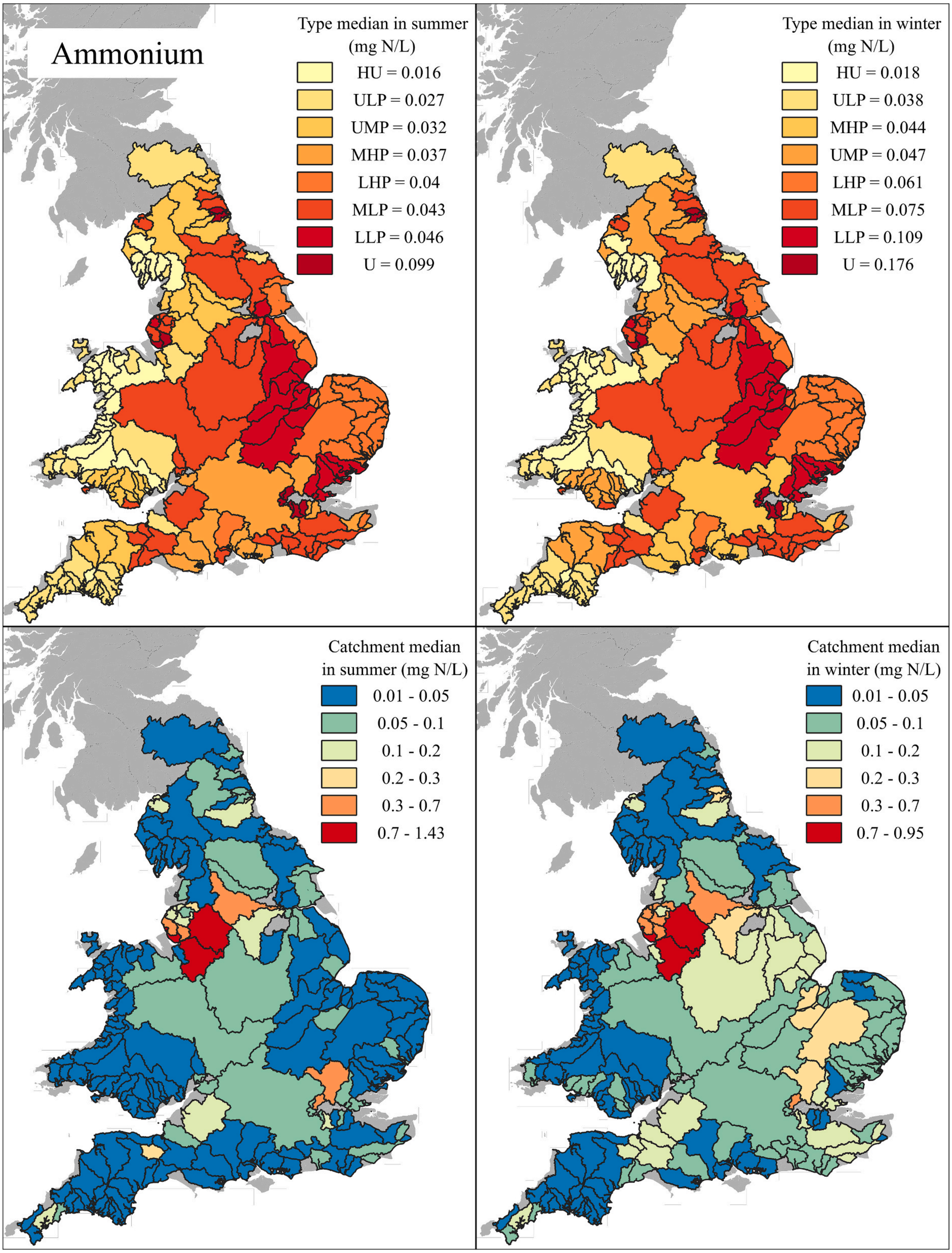

Fig. 7. Spatial distribution of median ammonium (NH_4^+) concentrations across catchment types (top panels) and individual catchments (bottom panels) from 2000 to 2024, shown for summer (Apr-Sep, left) and winter (Oct-Mar, right).

the mean annual is biased toward Ditton and Sankey Brooks near Liverpool, where phosphate removal (tertiary treatment) had not yet taken place according to the UK's latest report to UWWTD in 2018 (van Dijk et al., 2023). In 2018, the four London catchments were not sampled, and Newcastle catchments were under-sampled (4 samples) further impacting the average.

The MLP type follow the urban type in PO_4^{3-} summer concentrations (0.25 mg P/L, Fig. 5.*c*), followed by LLP (0.15 mg P/L), MHP (0.13 mg P/L) and LHP (0.12 mg P/L). Consistent with the above seasonal pattern in the urban type, winter medians across most types were equal to or

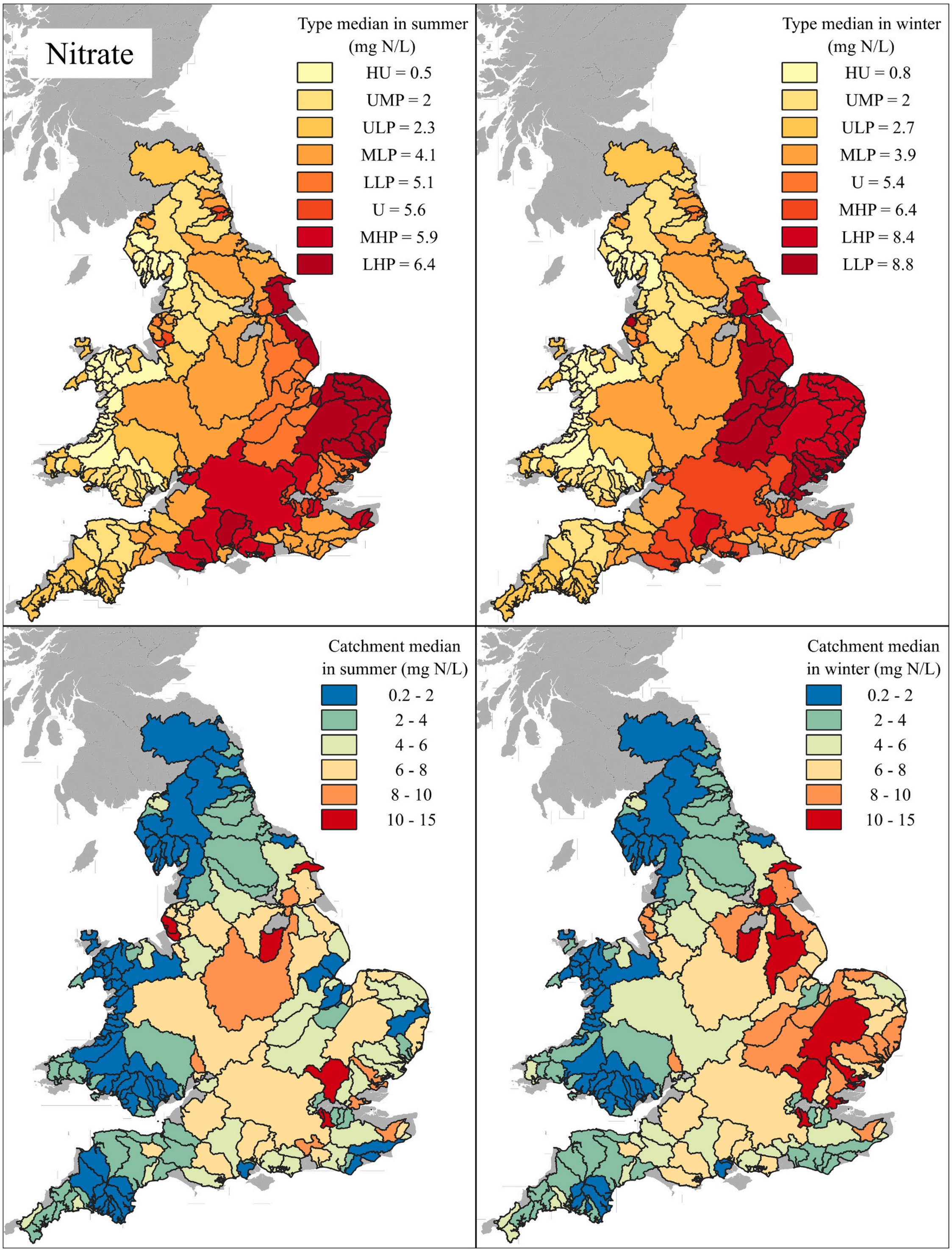

Fig. 8. Spatial distribution of median nitrate (NO_3^-) concentrations across catchment types (top panels) and individual catchments (bottom panels) from 2000 to 2024, shown for summer (Apr-Sep, left) and winter (Oct-Mar, right).

lower than summer medians (MLP = 0.16, MHP = 0.11 and LHP = 0.12 mg P/L), indicating the dominance of point sources. Most catchments (125 of 178) exhibited higher PO_4^{3-} concentrations in summer (s-ratio < 1, Fig. 9). However, LLP deviates from this pattern (winter median = 0.17 mg P/L); this deviation could be attributed to a sediment-derived contribution to orthophosphates. The arable types, LLP and LHP, have undergone decades of fertilisers application (Withers et al., 2001), accumulating legacy phosphorus in their soils and channel sediments; higher flows in winter dilute the water column below the equilibrium phosphorus concentration of the bed sediments, promoting desorption of legacy and freshly eroded sediment-bound phosphorus (Bowes et al., 2015; Jarvie et al., 2005a; Jarvie et al., 1998). The contrast between LLP and LHP reflects permeability; greater surface runoff in LLP delivers more sediment, and hence more dissolved phosphorus, than the infiltration-dominated LHP.

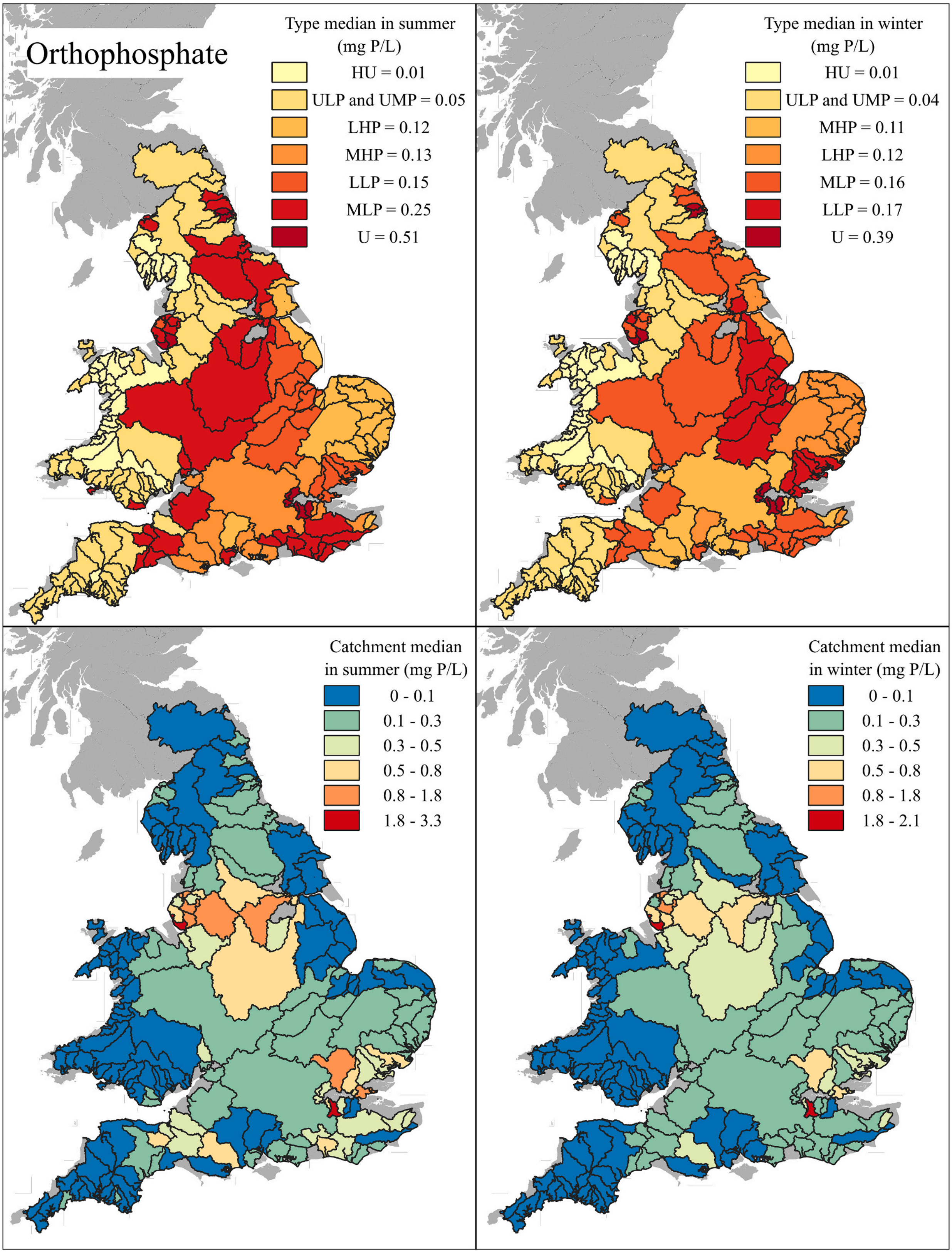

Fig. 9. Spatial distribution of median orthophosphate (PO_4^{3-}) concentrations across catchment types (top panels) and individual catchments (bottom panels) from 2000 to 2024, shown for summer (Apr-Sept, left) and winter (Oct-Mar, right).

4. Discussion

4.1. Catchment typology

The proposed clustering framework has been developed to capture the complex interplay between land use, topographical, hydrogeological, and meteorological controls that regulate nutrient mobilisation and transport in UK catchments. The framework is designed to be scientifically robust and reproducible, with potential applicability beyond the UK where data availability may be limited. This is achieved by reliance on widely available and transferable descriptors. The resulting typologies of coast-draining catchments across England and Wales reflect differences in catchment response and nutrient dynamics. The identified typology is broadly comparable to the previously proposed typologies of Lovett et al. (2018) at the operational catchment scale and Heasley et al. (2020) at the reach scale, particularly where spatial units are similar (e.g. operational catchments draining directly to the sea). The fundamental difference, however, lies in their intended application and scale. The present typology is explicitly structured around coast-draining units to interpret long-term riverine nutrient delivery to the marine environment. The aggregation to coast-draining units integrates upstream heterogeneity and captures a cumulative representativeness of catchment variables at the point of delivery to tidal limits. This approach differs from operational- or reach-scale classifications, which are more suited for local assessments or interventions but do not necessarily reflect integrated nutrient export to marine environments.

The separation of catchment types along gradients of topography, land use and bedrock permeability highlights the dominant controls on nutrient dynamics. The identified gradients are consistent with the well-established physical geography of UK catchments. Upland regions, characterised by higher rainfall, steeper topography and predominantly grassland cover, are located in the north and west of the country. Lowland catchments in the south and east are distinguished by a flatter topography and more intensive arable and horticulture land use, a consequence of the underlying geology and the long-standing patterns of agricultural development. These climatic and topographical variations are closely linked to hydrogeological contrasts, with impermeable upland terrains promoting rapid and event-driven runoff, while permeable lowland aquifers favour subsurface flow and longer residence times. Urban land use does not conform to this gradient as distinctly, given its concentration around major population centres and estuaries.

4.2. Nutrients dynamics across catchment types

4.2.1. Nitrate

Nitrate shows the clearest expression of catchment controls, largely governed by arable land use and bedrock permeability. This is consistent with previous studies, which have shown that NO_3^- concentrations are more reliably explained by arable land use compared to nutrients more strongly influenced by point sources (Crowson et al., 2024; Davies and Neal, 2007; Jarvie et al., 2002a; Rothwell et al., 2010a). Agricultural types exhibit the highest concentrations, particularly in winter, reflecting the dominance of diffuse leaching from fertiliser application (Crowson et al., 2024; Jarvie et al., 2005b; Rothwell et al., 2010b). The seasonal contrast differs between the two types and evident in (Fig. 5.***b***); LLP shows wider summer distributions that shift upward and become more compressed in winter, indicating a strong seasonal increase, while summer and winter distributions largely overlap in the LHP type. This pattern reflects the hydrogeological setting of the two types; permeable aquifers in the LHP sustain buffered groundwater NO_3^- inputs, while the poorly drained soil of LLP promotes stronger winter flushing signals. These findings are consistent with Davies and Neal (2007) who reported higher nitrate concentrations in areas with low rainfall and high baseflow index (BFI), reflecting stronger groundwater influence. BFI was initially considered for this study, but it was not retained due to its less significant correlation with NO_3^- concentration, compared to the applied bedrock permeability classification. This finding indicates that permeability-based descriptors may be more suitable at the present scale, where integrated subsurface controls on NO_3^- transport may be more effectively captured by geological classes than by catchment-averaged BFI.

Nitrate concentrations in the urban type are predominantly point-source driven, producing variable and often muted seasonality. This is evident by the temporal shift in urban NO_3^- concentrations (Fig. 6.***b***) following wastewater treatment upgrades; reductions in NH_4^+ were accompanied by increased nitrification and subsequent NO_3^- export – secondary treatment redistributes nitrogen speciation rather than achieving net removal. While NO_3^- is generally more strongly associated with arable farming, a previous study of catchments in Northwest England suggested that the highest concentrations can also occur in highly urbanised and industrialised catchments, reflecting substantial point-source inputs (Rothwell et al., 2010a). Earlier studies of the Humber Basin (Davies and Neal, 2004; Jarvie et al., 2002b) reported stronger correlations between NO_3^- concentrations and urban and industrial areas than with arable cover. This contrast with the present findings likely reflects the scale of analysis. While those studies captured regional statistical associations, the present typology isolates source signals within clusters defined by dominant land use. For instance, the MLP type, representing approximately 72% of the Humber, exhibits limited seasonality, consistent with a mixed contribution of agricultural inputs and persistent point sources, particularly during low summer flows when effluent dilution is weakest.

4.2.2. Ammonium

Ammonium is a recognised indicator of sewage effluent and industrial discharge, reflecting point-source pressures more directly than nitrate or orthophosphate (Robson and Neal, 1997), though it is more susceptible to within-channel processing (e.g. nitrification and biological uptake) which can attenuate concentrations between source and monitoring point (Davies and Neal, 2004). The dominance of the urban type is therefore expected; however, the early 2000 s decline (Fig. 6.***a***) represents the more analytically significant signal, aligning with UWWTD-driven improvements in sewage treatment and consistent with national-scale assessments of wastewater compliance (Whelan et al., 2022). The more instructive observations lie outside the urban type. Several catchments in predominantly rural types recorded NH_4^+ concentrations comparable to urban systems – most notably the Eden in Cumbria, where less than 3% urban land cover masks consistent point-source spikes at a single monitoring site in city of Carlisle between 2008 and 2015. Similar signals are observed in catchments associated with historically industrialised areas around the Mersey and Humber estuaries. These extremes reinforce the interpretation above and highlight a key methodological consideration; while a catchment typology that is defined by land use effectively capture dominant source characteristics, localised point sources can generate concentrations disproportionate to the surrounding landscape and are therefore likely to be underrepresented in aggregated land use-based classifications. The arable types, LLP and LHP show the strongest winter enrichment, reflecting enhanced nutrient mobilisation during wetter months. The contrast between them mirrors the hydrogeological controls discussed above for nitrate. This consistent pattern across both NH_4^+ and NO_3^- strengthens the case that bedrock permeability is a primary control on seasonal nitrogen dynamics in agricultural catchments, with direct implications for the timing of nutrient delivery to coastal receiving waters.

4.2.3. Orthophosphate

Orthophosphate concentrations peak in summer across most catchment types, a pattern governed by flow dilution rather than increased inputs – unlike nitrate whose seasonality is largely source-driven. Low summer flows concentrate effluent-derived phosphate, while higher winter discharges dilute point-source contributions (Jarvie et al., 2008;

Jarvie et al., 2006; Jarvie et al., 2005b; Mellander et al., 2015; Rothwell et al., 2010b). This distinction is diagnostically useful, as summer-dominated PO_4^{3-} generally indicates point-source influence. This is consistent with the urban type recording the highest concentrations in both seasons, and with most types exhibiting median s-ratios below one. The exception is the arable types (LLP and LHP), where fertiliser inputs may partially offset winter dilution, resulting in slightly higher median winter PO_4^{3-} concentrations. However, the effect of this seasonal signal is stronger in the LHP cluster while magnitude is higher in the LLP catchments.

Differences between the low-permeability types (MLP and LLP) and their high permeability counterparts (MHP and LHP) – given their broadly comparable land use characteristics – likely point to the role of three hydrogeological controls regulating flow pathways: first, a substantial proportion of phosphate is transported to rivers bound to sediment particles via surface runoff; the increased infiltration capacity within MHP and LHP catchments, reduces surface runoff generation, limiting the delivery of sediment-bound phosphate (Cooper et al., 2020; Jarvie et al., 1998). Second, in comparison to the runoff-dominated MLP and LLP catchments, greater baseflow contributions in MHP and LHP catchments result in higher year-round discharge, further diluting point-source inputs (Jarvie et al., 2005a; Mellander et al., 2015). Third, Chalk aquifers in LHP and MHP – to a lesser extent – catchments, act as biogeochemical buffers, promoting phosphorus retention through $CaCO_3$ co-precipitation within the subsurface, thereby reducing the transfer of dissolved phosphorus to the river (Jarvie et al., 2005b; Neal et al., 2002). While the dissolved fraction of phosphorus, PO_4^{3-}, is the focus of this study, rather than the particulate or total phosphorus, it is important to note that the two fractions are coupled; river sediments act as both a sink and a source for phosphorus, releasing PO_4^{3-} through desorption and resuspension when sediment is mobilised (Bowes et al., 2015; Cooper et al., 2020; Jarvie et al., 2005a; Jarvie et al., 1998).

4.3. Conclusion and implications for nutrients management and policy

The presented typology provides clear differentiation in catchment characteristics and nutrient behaviour, but its limitations must be acknowledged. Within-type variability remains substantial despite statistically significant between-types differences, reflecting the inherent heterogeneity of large catchments where localised point sources – especially for NH_4^+ and PO_4^{3-} – can dominate individual site signals that aggregated descriptors do not resolve. Sampling irregularity introduces a further constraint; uneven monitoring frequency and spatial coverage can bias seasonal and annual summaries, as demonstrated by the recurrent mean annual spikes discussed above. The clustering is also sensitive to descriptor selection – while the applied variables were chosen for robustness and reproducibility, processes such as in-stream nutrient processing, land management practices, and fine-scale hydrological connectivity are not explicitly captured. The typology also reflects catchment characteristics over the study period (2000–2024) and does not explicitly encode legacy nutrient stores or shifts in land use, climate, or management. These are expected limitations at national scale and do not undermine the typology's utility, but they define its appropriate scope. For practitioners, the typology is a framework for interpreting dominant controls and broad spatial patterns, not for predicting site-specific conditions. The clustering was built on catchment controls and assessed against nutrient concentrations; river flow and nutrient loads were not incorporated, so the typology characterises concentration behaviour rather than contribution to coastal loads – the highest-concentration catchments are not necessarily the highest-load ones. The proposed typology provides a transferable framework that can be applied beyond nutrients to interpret the behaviour of other determinands, particularly in distinguishing between conservative and non-conservative transport dynamics.

The results showed that land use and hydrogeology are major factors that control nutrient dynamics across England and Wales, distinguishing eight types with contrasting magnitudes, seasonality, and transport pathways. The proposed framework provides a robust basis for improving the estimation and interpretation of riverine inputs for OSPAR's RID reporting and provides a foundation for further work by both researchers and practitioners. It could be applied in gap-filling sparse concentration data – a recurrent constraint in long-term load estimation studies (Civan et al., 2018; Earl et al., 2014; Littlewood and Marsh, 2005; Nedwell et al., 2002; Worrall et al., 2009), where water-quality records are typically sparser in space and time than flow data. By assigning unmonitored or under-sampled catchments to a type with a characteristic nutrient signature, the typology provides a means for inferring concentrations where direct observations are limited. The typology can also help to target future monitoring. By identifying where within-type variability is greatest and where aggregated descriptors least resolve local signals – notably ammonium and orthophosphate in point-source-dominated types such as the urban and some MLP catchments, where individual site signals diverge most from the median – it can direct high-frequency monitoring towards catchments and determinands where it would be most valuable.

The framework could also inform more effective, nutrient- and catchment-specific management strategies. By distinguishing the dominant transport pathways and source contributions for each type, the framework demonstrates why blanket regulatory approaches may be ineffective. This is particularly relevant as OSPAR aims to develop area-specific nutrient reduction targets under the NEAES 2030, succeeding the paused PARCOM commitments (OSPAR, 2008). A catchment typology provides a structure for setting and applying such targets based on the dominant controls in each type, rather than using a single national threshold. For nitrogen, the contrast between catchment types is particularly instructive; in arable LHP catchments underlain by permeable aquifers, legacy groundwater nitrate represents a long-term water quality commitment that will respond slowly to surface interventions – management measures implemented today may take decades to manifest in riverine concentrations (Ascott et al., 2021; Bell et al., 2021; Jarvie et al., 2005b). In urban catchments, improvements under the UWWTD and the shift towards secondary treatment reduced ammonium but shifted nitrogen towards oxidised forms – achieving a net nitrogen reduction will require tertiary denitrification at major STWs. For phosphorus, the results emphasise the importance of transport pathways rather than source magnitude alone. In low permeability catchments (e.g. LLP and MLP), where surface runoff drives rapid mobilisation of particulate phosphorus, mitigation strategies should prioritise pathway disruption (e.g. erosion control, buffer strips, and flow attenuation) alongside point-source reduction. Conversely, in permeable catchments (e.g. LHP and MHP), reduced surface runoff limits particulate transport, and focusing on point sources, particularly under low-flow conditions, becomes more relevant. The variability in seasonality and mixed source contributions observed in transitional clusters across nutrients further indicates that catchment-specific, source- and pathway-informed strategies are needed. Aligning management strategies with dominant catchment processes is essential for achieving sustained reductions in nutrient inputs from source to sea.

CRediT authorship contribution statement

Hassan Mohamed: Writing – original draft, Visualization, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Naomi Greenwood:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Richard J. Cooper:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Carolyn Graves:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Yi He:** Writing – review & editing, Supervision.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jhydrol.2026.135970.

Data availability

Full details and workflows of the catchment characterisation, nutrient data processing, and clustering analysis are provided in the project's GitHub repository (https://github.com/Hassanabubaker/Coast-Draining-Catchments-Typologies).

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