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Shifting sands of marine eutrophication assessments: building a future approach for UK marine waters

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The assessment of water quality, and in particular, of eutrophication, has been a core activity to establish, disseminate, and communicate the impact of anthropogenic influences on coastal and marine waters in the United Kingdom (UK) and globally. To date, the UK assessments of eutrophication have focused heavily on indicators, either singularly or in combination, associated with a numerical threshold, with supporting science concentrating on defining relevant thresholds and relating exceedances to management actions. However, as our understanding of the complexity of estuarine and coastal zone processes in terms of variability, time lags, ecological interactions and climate resilience has evolved, so too must the structure of our water quality assessments. This paper presents a review of existing UK eutrophication assessments, identifying what has worked and where gaps still exist, particularly as our ecosystems face rapid changes. From the gap analysis, we present a series of recommendations for future eutrophication assessments, assessing the feasibility of implementing those recommendations through consideration of effort, complexity and costs. This work presents a set of headline activities offering a renewed and revised approach to the structure of UK eutrophication assessments that will progress complex data flows, achieve enhanced alignment between directives, embed new indicators, greater understanding of ecosystem impacts and consideration of the shifting climate baseline.

KEYWORDS
eutrophication, nutrients, pelagic community, OSPAR, UK Marine Strategy

1 Introduction

Excess nutrients from fertilizer application, pollution discharge, and sewage are transported from lands to oceans, impacting on coastal water quality and ecosystem health (Devlin and Brodie, 2023; Devlin et al., 2023; OSPAR, 2023; Paerl and Piehler, 2008; Painting et al., 2007; Carstensen et al., 2011). Terrestrial runoff of waters polluted with excess nutrients (primarily nitrogen and phosphorus compounds) from point sources, such as sewage treatment works (STW) discharges and aquaculture, and diffuse sources such as fertilizer losses via river discharges, have had devastating adverse effects in coastal and marine ecosystems globally (Ngatia et al., 2019; Ryther and Dunstan, 1971; Smith, 2003). Biomass production of plant matter in coastal waters is often naturally limited by the availability of nitrogen and/or phosphorus and increased anthropogenic inputs of these substances can lead to increased biomass that disturbs the natural ecological balance in marine ecosystems (de Raús Maúre et al., 2021). This disturbance, the process of eutrophication, is seen globally as one of the biggest threats to marine ecosystem health. Eutrophication, like climate change, is a cross country, cross sectoral issue with coastal regions throughout the world being impacted through the input of national and transboundary elevated nutrients (Laurent et al., 2018; Meier et al., 2019).

Eutrophication has a substantial impact on our coastal and marine systems and can limit access to ecosystem services by acting as a pressure on biodiversity and wider ecosystem approaches and industries such as shellfish harvesting and fisheries (Rhodes et al., 2017; Kermagoret et al., 2019). Even at a low level, increased nutrient loads and changing proportions of nutrients can result in changes in phytoplankton biomass and communities which can affect higher trophic level species (Duarte, 2009; Duarte et al., 2009; Carstensen et al., 2011; Frenken et al., 2023; Ibáñez et al., 2023; Dory et al., 2024). Species shifts are frequently characterized by high biomass bloom events which can have significant economic impacts as they reduce attractiveness and amenity value of coastal waters resulting in societal upset (Willis et al., 2018; Andersen et al., 2019). Increased phytoplankton biomass reduces light penetration which in turn causes habitat loss by limiting areas where seaweeds and seagrasses can grow (Carolina, 2002; Foden et al., 2005). These habitats are important for maintaining fish nursery populations and biodiverse benthic organisms. More serious eutrophication impacts involve hypoxic events which harm many organisms but are particularly damaging to sessile benthic fauna, whose loss again affects the food web and biotic water quality regulation. Extreme hypoxia and anoxia lead to a loss of both biotic and abiotic water quality regulation, as previously sequestered nutrients re-enter the water column and bacterial denitrification processes change (Best et al., 2007; Devlin and Brodie, 2023). Well-documented adverse ecological responses of increased nutrient discharge to coastal and marine waters from across the world include harmful algal bloom events (HABs) (Paerl, 2008; Glibert and Burford, 2017), changed preponderance and dominance of certain types of fast growing plankton over other, more long-lived and structural benthic primary producers (seagrass, coral, macroalgae) (Lapointe et al., 2019, 2020), the creation of hypoxia and subsequent "dead zones" (Diaz and Rosenberg, 2008) habitat degradation, and adverse changes in aquatic food webs (Carpenter et al., 1998; Gross and Hagy, 2017).

Generally, the UK eutrophication assessment approaches focus on single metrics ("indicators") associated with a numerical threshold which are then integrated into one outcome of overall status (Best et al., 2007; Devlin et al., 2007a,b, 2009, 2012a; Greenwood et al., 2019). Historically, work has focused on the development of those thresholds, how they relate to ecosystem function and how to guide mitigative management actions from the outcomes of the assessment (Bricker et al., 2003; Bricker and Devlin, 2011; Ferreira et al., 2011). However, effective mitigation of eutrophication requires consideration of many layers of complexity, needing multiple, often cumulative actions over large spatio-temporal scales (Thornton et al., 2013). These challenges are well known with many studies recognizing the complexity of the problem due to large variations in hydrodynamics, water supply, inputs and susceptibility (Cloern, 2001; Cloern and Jassby, 2009; Duarte, 2009; Duarte et al., 2009).

As our understanding of the complexity of the coastal zone, in terms of variability, time lags, ecological interactions and resilience has evolved, so should the structure of our eutrophication monitoring and assessment. Pauly (2019) identifies the concept of shifting baselines, the phenomenon where each generation accepts the baseline as the earliest condition it experiences, with shift in baselines typically toward degradation. Our ecological systems are much different from decades ago facing multiple pressures whilst our technology to collect vast amounts of data continues to grow. Understanding this changing baseline in a complex pressure-response system requires multiple layers of information to inform and direct our understanding of what constitutes an acceptable and sustainable level of use for the marine environment. In addition, our shifting baseline needs to consider declining climate resilience, through cumulative impacts from multiple pressures, and the interactions of these pressures with increasing global temperatures (Atkins et al., 2011b; Patrício et al., 2016; Elliott et al., 2017; Laurent et al., 2018; Meier et al., 2019). Efforts to tackle eutrophication need to address the entire land-sea continuum from catchment to coast and be supported by monitoring a range of complex interactions and impacts (Thornton et al., 2013). Future approaches to eutrophication need a re-analysis of the issues, updating our frameworks and a rethinking of the complex solutions to achieve sustainable use of the marine environment.

This paper presents the outcomes of an evidence review and prioritization exercise, identifying gaps in current UK eutrophication assessment frameworks which have not fully considered how our system is changing and what is needed in future assessments to account for the complexity of the pressures that drive the impacts and the scale of potential solutions. It presents a review of historical and current eutrophication assessments that are being implemented in UK coastal and marine waters, identifying successes and challenges. The review and national consultations with eutrophication experts provided the baseline to identify five key challenges that need to be resolved for future eutrophication monitoring. We describe those challenges and outline a series of recommendations that can help achieve future success in eutrophication monitoring and assessment.

2 Methods

Part 1 of this work describes historical and current eutrophication assessment policies, identifying the aim of the assessments, the area in which it was applied, who was responsible for implementation, the structure of the assessment, and what was achieved though the implementation. This review, alongside national consultations with UK eutrophication experts provided the basis for part 2, a quantitative analysis of future eutrophication needs which were characterized into five broad thematic areas exploring future data needs, potential for further alignment, potential for new indicators, embedding greater understanding of ecosystem impacts and consideration of climate change. Part 3 presents a prioritization of the main recommendations under each theme, considering cost, feasibility and outcomes.

2.1 Review existing eutrophication assessment frameworks

We reviewed the existing environmental policies responsible for the assessment of eutrophication in the UK's coastal and marine waters, detailing their basis and implementation, overlapping spatial extents, and their component indicators and thresholds. We explored the evolution of eutrophication policy from coastal and marine areas and identify successes of both monitoring and policy implementation associated with UK eutrophication assessments. Finally, we summarized the indicators and reporting structures that form the assessment frameworks. The focus of this review was coastal to marine and freshwater eutrophication assessments were not included.

2.2. Identify key evidence gaps in current practices

Information from the review alongside national consultation with eutrophication experts across UK agencies were discussed in terms of what has worked and where gaps still exist. Topics considered essential to improve future eutrophication monitoring and assessment were characterized into five thematic areas exploring future data needs, potential for further alignment, potential for new indicators, embedding greater understanding of ecosystem impacts and consideration of climate change within eutrophication assessments. From these consultations, we identified evidence gaps within five broad themes ("data," "alignment," "indicators," "ecosystem," and "climate") to identify knowledge and evidence needed for better informed and effective management of eutrophication (Figure 1).

2.3 Prioritization of activities required to fill evidence gaps

Alongside the recommendations for future eutrophication assessments, an assessment of feasibility was carried out, developed from the review and expert knowledge of what is currently occurring in UK national eutrophication monitoring programs. Feasibility of developing solutions to the key evidence gaps is reported against the effort required to achieve recommended activity, the scale of complexity and potential costs of implementation. We finish with a discussion on those recommendations and how best to achieve successful implementation of a revised, updated eutrophication monitoring and assessment strategy for coastal and marine waters.

3 Results

3.1 Review existing eutrophication assessment frameworks

3.1.1 Developing environmental directives

The design, establishment and management of eutrophication assessments in the UK have been influenced by the experience gained through the implementation of EU directives now transposed into UK law and that of the devolved administrations, and the consecutive development of monitoring to comply with the requirements of the different directives addressing eutrophication. The UK also continues to be a signatory to international conventions, in particular that of OSPAR (the Oslo and Paris Convention for the Protection of the Marine Environment of the North Atlantic). We review the application of national laws and partnerships associated with the assessment in eutrophication in UK coastal and marine waters. Figure 2 gives a timeline of the implementation of key policies and associated assessments.

Eutrophication emerged as a major issue in the 1970s, with concern over the input of sewage outfalls into UK rivers. Rising awareness around sewage outfalls and direct nutrient inputs meant that eutrophication from the 1970s and into the 1990s was driven by the need to mitigate point source pollution with programmes of management measures focused primarily on the reduction of phosphorus and nitrogen from sewage treatment. Much of this initial work was developed from frameworks such as the EU Urban Wastewater Treatment Directive (91/271/EEC) (European Commission, 1991a) and the EU Bathing Water Directive (2006/7/EC) (Figueras et al., 1997; European Commission, 2006). The EU Urban Wastewater Treatment Directive (91/271/ECC; hereafter UWWTD) required Member States to ensure that urban areas collect and treat wastewater which would otherwise pollute rivers, lakes and seas and succeeded at altering compliance choices and improving related national policies (Kemp, 2001). It required compliance by 1998 for wastewater treatment for all settlements of >2,000 population equivalents to have (at least) secondary treatment with more advanced treatment required for towns leading to a step change in sewage treatment from the late 1990s. Whilst the EU Bathing Water Directive was aimed at bacterial reduction, there was a positive effect of catalyzing greater nutrient reductions from STWs.

However, issues around diffuse runoff from farming and agricultural practices through the 1980s and the 1990s that were not mitigated by the UWWTD resulted in the parallel development of polices aimed at the reduction of nutrient runoff from agriculture. The implementation of European directives continued with the onset of the Nitrates Directive (91/676/ECC), which aimed to protect water quality by preventing nitrates from agricultural sources that pollute ground and surface waters and by promoting the use of good farming practices (European Commission, 1991b; Massarelli et al., 2021). The Nitrates Directive was a major step toward acknowledging and protecting water against pollution from agriculture (Tunney, 1994). The implementation of Nitrogen

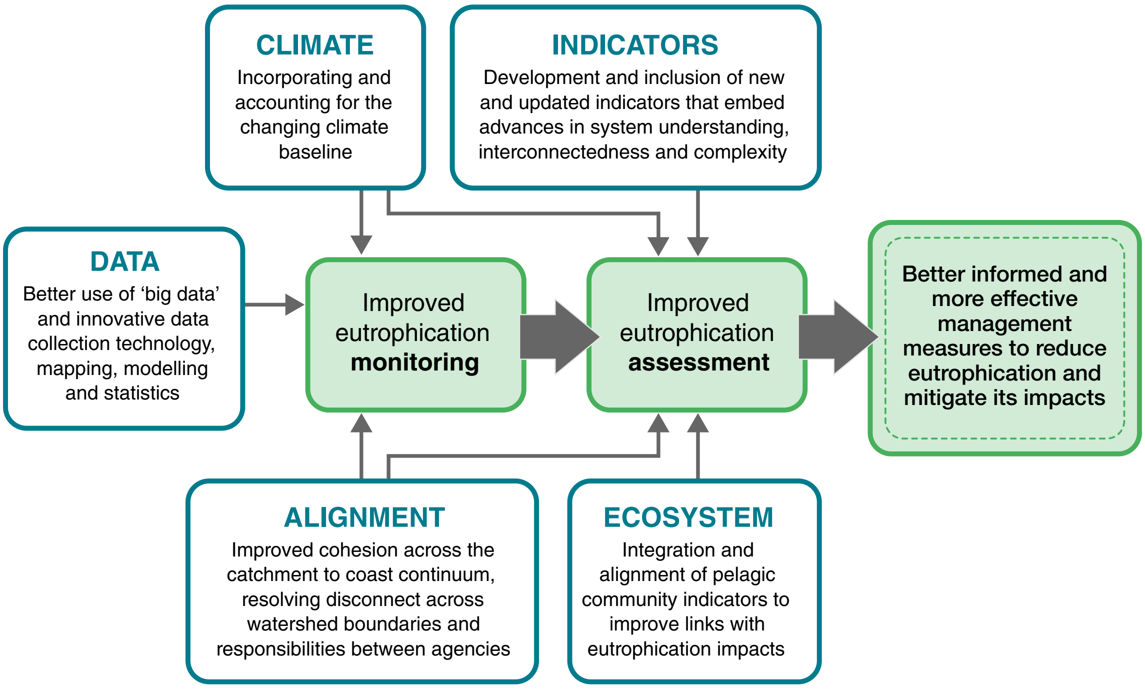

FIGURE 1
Five broad themes considered for the improvements of future marine eutrophication assessments.

Vulnerable Zones (NVZs) provides an example of a catchment-based program to reduce excess groundwater nitrogen (Smith, 2000; Johnson et al., 2007; Worrall et al., 2009) where areas of elevated nitrate sensitivity were identified and, in these areas, limits on nitrate additions were imposed.

The assessment of the impact of eutrophication in the UK for estuaries and coastal waters commenced with the EU Water Framework Directive (2000/60/EC; WFD) from 2001. This covers transitional and coastal water extending for one nautical mile (1 nm) beyond the low water mark for England, Wales and Northern Ireland and 3 nm from the "coastal baseline" in Scotland (European Commission, 2000). Between 2006 and 2008, all agencies collaborated formally to develop the WFD tools and methods for the initial assessments. The EU WFD was developed as a river basin programme, connecting upstream activities with downstream ecological health. Eutrophication assessments in coastal and offshore waters (beyond the WFD waters) were also developed from the requirements of the EU Marine Strategy Framework Directive (2008/56/EC; MSFD) (European Commission, 2008). The overarching aim of the MSFD was to implement measures to achieve Good Environmental Status (GES) by 2020. First, Member States developed marine strategies consisting of an initial assessment of environmental status, along with articulating the defining characteristics, targets and indicators representing Good Environmental Status (GES). Monitoring programmes to measure progress toward GES were then established, and, where necessary, programmes of measures to achieve or maintain GES were implemented. Implementation of the WFD and MSFD resulted in shared knowledge between participating countries, development of national indicators, and a robust monitoring framework that allowed tracking of the reduction of eutrophication pressures and impacts (Borja, 2005; Devlin et al., 2007a).

Additional marine eutrophication assessments are carried out under OSPAR, the mechanism by which 15 Governments and the EU cooperate to protect the marine environment of the North-East Atlantic. OSPAR started in 1972 with the Oslo Convention against dumping and was broadened to cover land-based sources of marine pollution and the offshore industry by the Paris Convention of 1974. These two conventions were unified, updated and extended by the 1992 OSPAR Convention. Eutrophication assessments are carried out under the OSPAR "common procedure" (OSPAR COMP) (Devlin et al., 2023). The OSPAR eutrophication assessment was implemented for OSPAR COMP1 (1996–2000) and OSPAR COMP2 (2006–2014) with OSPAR COMP3 (2013–2018) providing an updated Common Assessment Criteria for the Eutrophication status of the OSPAR Marine Area. The most recent "OSPAR COMP4" has recently been concluded and provides an updated eutrophication thematic assessment (2019–2023) for 15 countries across the North-East Atlantic (Devlin et al., 2023).

Brexit and departure from EU legislation has meant several changes to the UK environmental policies which were responsible for the assessment of eutrophication in coastal and marine waters. The WFD was transposed into separate environmental legislation for England, Wales, Northern Ireland and Scotland so that supporting pieces of water legislation would continue to operate after EU exit (1 January 2021). These are the Water Environment (Water Framework Directive) (England and Wales) Regulations, the Water Environment and Water Services (Scotland) Act 2003 (WEWS Act 2017) and The Water Environment (Water

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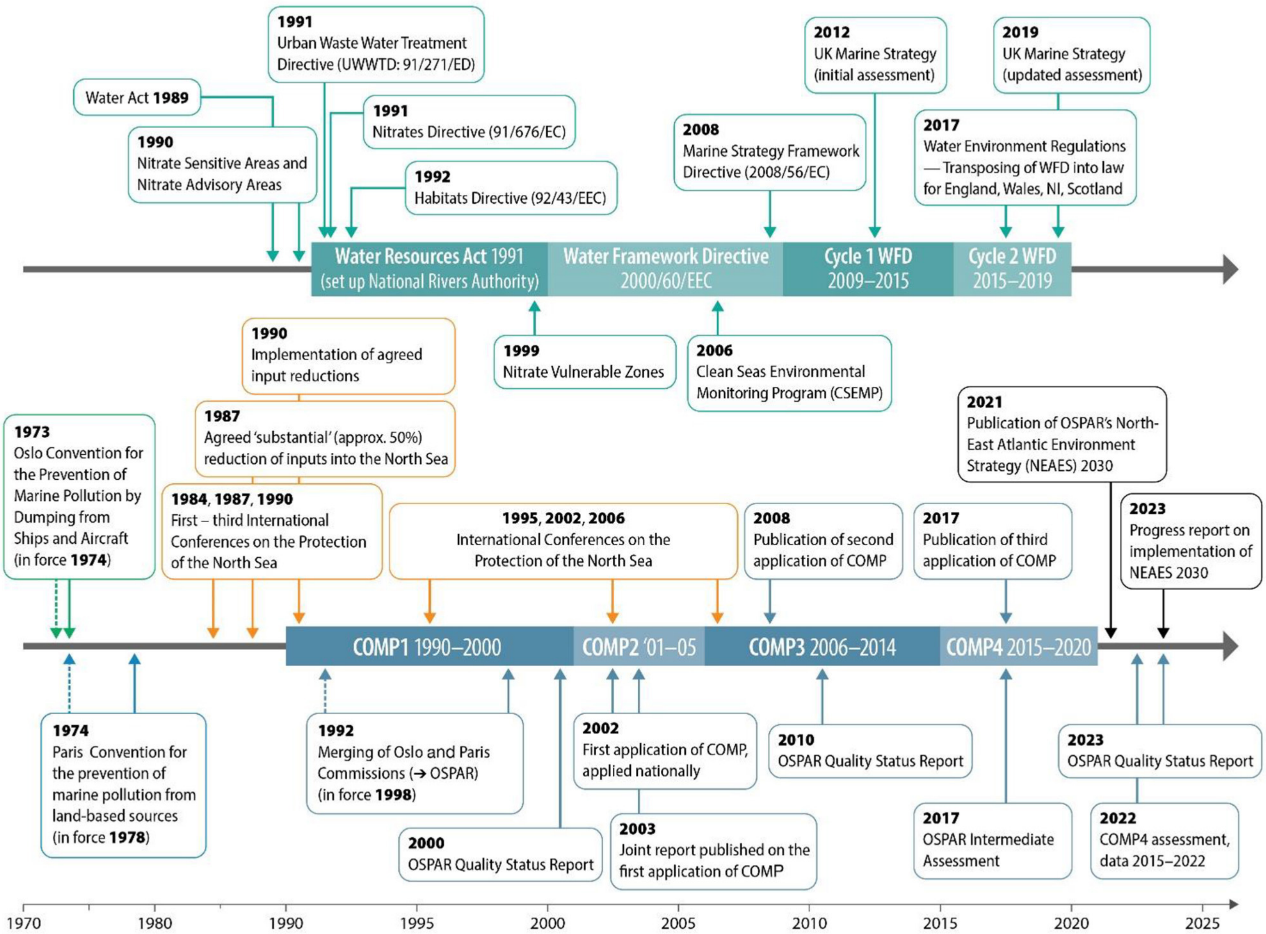

FIGURE 2
The evolution of eutrophication assessments for coastal and marine waters in the UK. The upper timeline focuses on the implementation of EU and national legislation with the bottom timeline showing the development of international agreements and frameworks, specifically the implementation of OSPAR.

Framework Directive) Regulations (Northern Ireland) 2017. For the purposes of this paper, the different regional approaches will be called WFD/WER—representing the three regional approaches for the Water Environment Regulation and Water Framework Directive across the UK, which is still the primary environmental directive that assesses transitional and coastal waters in the UK For coastal and offshore waters, the EU MSFD was transposed into national legislation through the Marine Strategy Regulations 2010 (covering England, Scotland, Wales and Northern Ireland) (DEFRA, 2012, 2019) and now sits within the UK Marine Strategy (UKMS). The DEFRA (2012) required action to achieve or maintain GES in our seas by 2020. The Regulations require the production of a "Marine Strategy" for all UK waters and that the approach is coordinated across all four UK Administrations and cooperate with other countries sharing our seas. This has resulted in the UK Marine Strategy Part One: an assessment of marine waters, objectives for GES and targets and indicators to measure progress toward GES (DEFRA, 2012, 2019) with the third report recently published (DEFRA, 2025). Eutrophication assessments under the recent UK Marine Strategy Part 1 include the coastal assessments carried out under WFD/WER and the coastal and offshore assessments carried out under the OSPAR thematic assessment (COMP4) with the two source assessments merging outcomes together but not fully integrating the source data.

3.1.2 Eutrophication indicators for UK monitoring and assessment

The type of indicators that make up a eutrophication assessment framework vary depending on the specific assessment, though many commonalities exist between eutrophication directives. The assessment of eutrophication usually includes (at least) three indicators: dissolved inorganic nutrients (nitrogen and phosphorus), phytoplankton biomass (typically measured as chlorophyll-a) and dissolved oxygen (Devlin et al., 2011, 2007a; Greenwood et al., 2019; Scavia and Bricker, 2006; Smith and Schindler, 2009; Van Beusekom et al., 2019).

Eutrophication has been defined as "the progressive enrichment of nutrients, leading to excessive plant growth," and developed tools to classify nutrient and plant status (COM, 2017). Eutrophication may be present if one of the sensitive biological elements (particularly phytoplankton or opportunistic green algae) are moderate or worse and the supporting element of nutrients (DIN in marine waters) is also moderate or worse. These conditions should be persistent over a period of assessment cycles and investigations suggest that the causes of failure are strongly linked to nutrients. The overall eutrophication assessment is based on this evidence and other supporting parameters.

The assessment of eutrophication in the recent OSPAR eutrophication thematic assessment is based on the degree of nutrient enrichment (Category I), the direct effects of nutrient enrichment (Category II) and the indirect effects of nutrient enrichment (Category III). For Category I, the nutrients common indicator is derived from winter mean concentrations of dissolved inorganic phosphorus (DIP) and dissolved inorganic nitrogen (DIN) (Heyden and Leujak, 2022). For Category II, the Chlorophyll common indicator is derived from growing-season mean concentrations of chlorophyll (Prins and Enserink, 2022) where chlorophyll EO and water sample data are combined, weighted as a function of *in-situ* confidence. For Category III, the Dissolved Oxygen common indicator assesses the concentrations of dissolved oxygen near the seafloor (Devlin et al., 2022). Assessment criteria and their corresponding area-specific assessment levels as set and agreed for COMP4 are applied for each given area. The results obtained are integrated to give the classification for the given area. The overall result of the assessment depends on the outcome of the direct and indirect effects (Categories II, III), following the one-out-all-out principle (OOAO) (Devlin et al., 2011; Ferreira et al., 2011). The UK Marine Strategy is a combination of WFD/WER and OSPAR methodology (Devlin et al., 2023).

We present a comparison of the indicators across the three main UK eutrophication directives within each category in Table 1, summarizing key eutrophication indicators, assessment criteria, and geographical coverage across these frameworks. Indicators are grouped into "Category I" which relate to causative factors including nutrient loads and nutrient enrichment, "Category II" which include the direct effects of nutrient enrichment (impacts on phytoplankton biomass and community), and "Category III" which include the indirect effects of nutrient enrichment (reduction in light penetration and dissolved oxygen concentration). The WFD/WER nutrient assessment is an aggregation of two indicators (Winter DIN and Winter DIP), whilst the OSPAR assessment is based on Winter DIN and Winter DIP measured separately. The WFD/WER uses the outcome of the nutrient indicators to formally identify high risk areas for eutrophication while the nutrient indicators are not considered in the final OSPAR assessment outcome, and assessment areas that have failed nutrient indicators under the UKMS have been designated as only partially meeting GES. There are also differences between the chlorophyll and dissolved oxygen indicators and how they are used in the final assessments within WFD/WER and OSPAR. Again, these are small but significant, and result in multiple differences between the two approaches for the final assessment of eutrophication (Greenwood et al., 2019).

3.1.3 Designating eutrophication status

Indicators are assessed by comparison to their respective thresholds; however, outcomes are classified differently under each separate eutrophication frameworks. The WFD/WER eutrophication assessment includes five assessment outcome categories with "high" the most positive and "poor and bad" indicating the most perturbed, and "moderate" and "good" designating somewhat negative and somewhat positive intermediate states, respectively. Geographically specific "reference conditions" (and class boundary thresholds) for each indicator under WFD/WER were constructed based on a combination of scientific review (Borja et al., 2004), thresholds accepted under previous directives and international agreements (e.g. OSPAR, Foden et al., 2011; Devlin et al., 2007a; Painting et al., 2007), expert knowledge and investigations of outputs between water bodies at low and high risk of eutrophication (Devlin et al., 2007b). Thresholds were also validated and modified by the consensus process of the first phase of the NE Atlantic Intercalibration process (Heiskanen and Carletti, 2009).

OSPAR designates spatial assessment areas as "problem" or "non-problem" areas with respect to eutrophication. In the first three OSPAR COMPs, OSPAR Contracting Parties evaluated the eutrophication status of their national marine waters using national assessment levels (thresholds) to assess nutrients and chlorophyll a (OSPAR, 2017). For OSPAR, assessment levels that were indicative of good ecological status (non-problem area) were based on reference conditions reflecting non-eutrophic conditions with boundary between "good" and "moderate" status derived by adding a 50% deviation to the reference condition. However, OSPAR Contracting Parties used different approaches in establishing reference conditions to derive these values, leading to variable assessment levels and different outcomes of the eutrophication assessment across national borders (Malcolm et al., 2002; OSPAR, 2003, 2008a, 2017; Foden et al., 2011). OSPAR COMP4 improved this through the development of an ensemble model approach used to derive pre-eutrophic conditions (Lenhart et al., 2010; van Leeuwen et al., 2023). Nutrient loads into the NE Atlantic were estimated from rivers under reference conditions, using the European model E-HYPE and observations (Lenhart et al., 2010; Stegert et al., 2021). The chlorophyll concentrations were modeled corresponding to the estimated nutrient concentrations under reference conditions. To allow for natural variability with a "slight disturbance," and in the absence of more specific information, the assessment level was defined as the concentration 50% above the area-specific background concentration derived from the ensemble model approach (Borja, 2005; OSPAR, 2003, 2008b). The UK Marine Strategy uses a combination of WFD/WER and OSPAR indicators and thresholds for coastal and offshore areas and applies the terminology "in/achieving GES" and "not in GES" for each indicator.

While the different directives for the assessment of eutrophication are somewhat compatible regarding the type of indicators and how they are assessed relative to thresholds, the integration frameworks, how individual indicator outcomes are brought together to provide a classification of eutrophication, status depends on what elements/indicators are used for the assessment. WFD/WER implements a "one out all out" (OOAO)

TABLE 1 Summary of current UK eutrophication indicators used to inform each indicator within the three UK frameworks [Water Framework, Water Framework Directive Regulations (WFD/WER for England, Wales), or Water Framework Directive (Northern Ireland, Scotland)], UK Marine Strategy and the OSPAR Common Procedure (COMP).

Category	Indicator	Current UK eutrophication assessment frameworks		
		WFD/WER	UK Marine Strategy—Part 1	OSPAR Common Procedure 4 (COMP4)
Areas assessed		Transitional (estuarine) waters (TW) to coastal areas up to 1 nm of coast	Coastal waters (CW) up to 3 nm, plumes and offshore waters	Plumes and coastal waters (seaward of 1 nm) and all UK marine waters
I. Drivers of eutrophication: physico-chemical	**TN and TP—**total nitrogen and phosphorus loads (inputs from land to sea)	Not used.	Common indicators outcome from OSPAR applied; no UK specific analysis.	Common indicator for trends in loads (diffusive sources, riverine inputs and direct discharges from sewage and industry) as well as atmospheric TN since 1990 at the scale of OSPAR sea areas. Not assessed against thresholds
	Nutrient concentrations—elevated level(s) of winter DIN and/or DIP	Winter mean DIN, DIP over 6-year period compared to "reference" level + 50%	WFD/WER methodology used for coastal waters and OSPAR COMP4 methodology used for plumes and offshore waters	Winter mean DIN over 5-year period compared to "reference" level + 50%
	N/P ratio—elevated winter N/P ratio	Used as part of "weight of evidence" to designate future risk	Common indicators outcome from OSPAR applied; no UK specific analysis	Trends in N:P ratios presented as part of INPUT reporting
II. Direct impacts from Elevated nutrients: Direct biological impacts	**Chlorophyll-*a* concentration—**90th percentile or mean	Mean of annual 90th percentile value of chlorophyll (March to Sept) calculated over 6-year period compared to "reference" level +50%	WFD/WER methodology used for coastal waters and OSPAR COMP4 methodology used for plumes and offshore waters	Mean of annual 90th percentile value of chlorophyll (March to Sept) calculated over 6-year period compared to "reference" level +50%
	Phytoplankton indicator species (area-specific)—Elevated levels of nuisance/toxic indicator species increased duration of blooms	Single species and total taxa count of all phytoplankton species (identified through microscopy) compared against "reference" taxa counts	In coastal waters only: single species and total taxa count of all phytoplankton species (identified through microscopy) compared against "reference" taxa counts	Not implemented for UK waters—though narrative of phytoplankton changes in biodiversity assessments included in OSPAR reporting
	Macrophytes including macroalgae (area-specific) Elevated levels (biomass or area covered) of opportunistic green macroalgae Area and size of saltmarsh and seagrass systems	Opportunistic macroalgae (used only for coastal waters) Extent and density and biomass, of nuisance green macroalgae on sediment, together with overwintering and entrainment. Measure during the period of maximum growth once every 3–6 years	Not included in eutrophication assessment in UKMS	Not implemented in OSPAR assessment
		Saltmarsh—metrics of area extent, dominance of saltmarsh zones, saltmarsh taxa diversity		
		Intertidal seagrass—change in species composition, change in areal extent of beds, change in percentage cover Subtidal Seagrass—bed extent, bed density, maximum bed depth		
III. Secondary impacts from elevated nutrients: indirect biological impacts	**Oxygen deficiency**	Dissolved Oxygen (DO) calculated from surface samples. 5th percentile calculated for each assessment area	Dissolved Oxygen (DO) calculated from surface samples. 5th percentile calculated for each assessment area	Bottom DO (July to October)—calculated from deepest sample within 10 m of seabed. 5th percentile calculated for each assessment area

(Continued)

TABLE 1 (Continued)

Category	Indicator	Current UK eutrophication assessment frameworks		
		WFD/WER	UK Marine Strategy—Part 1	OSPAR Common Procedure 4 (COMP4)
	Photic limit (water transparency of the water column)	Turbidity used in nutrient assessment for transitional waters	Not implemented	Not implemented
	Phaeocystis	Not implemented	Not implemented	Considered by OSPAR but not implemented in UK waters

DIN, dissolved inorganic nitrogen; DIP, dissolved inorganic phosphorus; GS, growing season, typically the period between March to September; Winter, November–February.
UK Marine Strategy Part One refers to a nationally based assessment of UK marine waters that measures progress toward GES.

process which takes the lowest value of biological elements (phytoplankton, macroalgae), while the most recent OSPAR common procedure (COMP4) used the lowest value of the chlorophyll and dissolved oxygen indicators, and the UK Marine Strategy (Part 1) combines outputs from both. Nutrients in all three directives cannot, on their own, designate a "failure" if the biological elements do not also "fail." A failure is when a waterbody is designated as less than moderate status under the WFD/WER or not meeting good environmental status (GES) under UK Marine Strategy (Part 1) and OSPAR. However, failing nutrients can cause potential risk, and is designated as a "potential problem area" under OSPAR or high risk of future failures under the recent UK Marine Strategy (Part 1) and is identified in the WFD/WER as failing physico-chemical standards.

3.1.4 From "increased growth" to "changes to the balance of organisms"

In two cases (ECJ, 2004, 2009) the European Court of Justice ruled that the definition of eutrophication in the UWWTD must take account of "significant harmful effects of the accelerated growth of algae and higher forms of plant life resulting from discharges of urban wastewater," that include "an undesirable disturbance of the balance of organisms present in the water." However, because of lack of agreement amongst OSPAR signatories about adding indicators of phytoplankton community composition, the OSPAR Common Procedure has remained focused on the bulk variables, chlorophyll concentration and oxygen deficiency, as the key common indicators of the direct and indirect biological impacts associated with eutrophication of the water column.

This was also the case for the eutrophication Qualitative Descriptor (QD) of the EU Marine Strategy Framework Directive (EU MSFD), as interpreted by the European Commission (COM, 2017). Instead, what we understand as the health, good functioning, and balance of organisms of the plankton in the pelagic habitats, and which COM (2017) refers to as "the essential features and characteristics and current environmental status of marine waters" in relation to the EU MSFD's aim to achieve Good Environmental Status, was assigned (Borja et al., 2010) to the Biodiversity QD and the Food-Webs QD. In a separate process from OSPAR COMP4, Biodiversity and Food-Web indicators have also been assessed by OSPAR in a recent Quality Status Report (QSR) on the pelagic habitats (PH) of the North-East Atlantic (McQuatters-Gollop et al., 2022; Holland et al., 2023a,b). The relevant "PH" indicators PH1/FW5, based on the abundances of plankton lifeforms and Food-Webs, PH2 (also used for Food-Webs) based on biomass at each trophic level, and PH3, based on numerical biodiversity. There is continuing work to disentangle the effects of nutrient enrichment, fisheries and climate change on the PH1 indicators (McQuatters-Gollop et al., 2019; Holland et al., 2024).

There would, thus, seem to be two options for conceptualizing the changes in pelagic habitats/ecosystem state associated with eutrophication. One option is to understand eutrophication as being the increased production of organic matter and changes in the balance of organisms in pelagic habitats are seen as a consequence rather than a part of eutrophication: the view currently taken by the EU in the MSFD (Borja et al., 2010; COM, 2017) and by OSPAR, the latter having separate strategies for eutrophication and biodiversity. The other option is to see nutrient enrichment as one of several anthropogenic pressures on pelagic habitats, leading to perturbations of their normal functioning which may lead to regime shift (Tett et al., 2007, 2013; Gowen et al., 2012) as the most significant of the "undesirable disturbances" that according to the ECJ make up the final component of the eutrophication process.

3.2 Identify key evidence gaps in current practices

3.2.1 Recognizing what has worked is the first step

There have been many successes associated with the implementation of the UK eutrophication assessments, leading to positive program of measures and the reduction of nutrient inputs from both direct and diffuse sources. Starting as early as 1988, the OSPAR Contracting Parties agreed to reduce nutrient emissions to the Greater North Sea by 50%. Since then, several OSPAR Recommendations, land management initiatives and controls from regulations such as the Urban Wastewater Treatment Directive (UWWTD) (Council Directive 91/271/EEC) and WFD/WER have been taken to combat eutrophication. These include measures targeting diffuse run-off from land, atmospheric nitrogen emissions, wastewater, and other point sources. These responses have led to significant improvements in nutrient loadings to the UK marine waters since the start of monitoring in 1990. On a European scale, many measures to reduce inputs have been implemented by European Union directives covering wastewater treatment, nitrates in agriculture, industrial emissions and water and marine management with dramatic improvements in the form of atmospheric nitrogen input reductions and a reduction in fertilizer use, since 1990 (OSPAR, 2023). The four applications of the Common Procedure have revealed a steadily improving trend in the eutrophication status of OSPAR Regions II, III, and IV. Success can also be measured by the application of the eutrophication assessments, with detailed assessments completed by regulatory agencies increasing our knowledge of high-risk areas, identifying where and how programmes of measures can be implemented.

However, during the last decade, improvement trends have slowed down, with heightened concerns regarding increasing nutrient inputs from diffuse agricultural loads, combined sewage overflows and the growing marine aquaculture industry.

The application and outcomes of these assessments, whilst identifying areas that are at risk from eutrophication, can and should be improved. The original assessments were set up to define eutrophication through a set of indicators, with thresholds that were developed on historical understanding and expert knowledge. This is an evolving area, and our understanding of what is required into the future to ensure our understanding of eutrophication, to fully utilize the many emerging data sources, to look beyond simple indicators of state, and to account for climate change, requires adaptation and updating of our current methods. We present a series of recommendations around the improvement of data, alignment between different UK assessment frameworks, updating and improving eutrophication indicators, embedding ecosystem indicators within the eutrophication assessment and consideration of the interactions between climate and eutrophication (Figure 1). These recommendations set out priority areas for future assessments which are then assessed considering cost, feasibility and outcomes.

3.2.2 Improving data flows

3.2.2.1 Autonomous and remotely collected data

Our long-term monitoring datasets are the critical baselines from which we have developed and continue to develop our understanding of the changes in our coastal and marine systems. However, it is not feasible or affordable to collect all data all the time. The way we collect eutrophication assessment data is evolving, from traditional vessel-based sampling data (*in-situ* sampling and *ex-situ* analysis) to include autonomous data from *in-situ* sensors, satellites, and biogeochemical models (Bean et al., 2017). Furthermore, machine learning and artificial intelligence are changing the way we collect, analyse and report data within eutrophication assessments (Borja et al., 2024). There will always be a need to continue long-term, traditional *in-situ* sampling alongside the collection of new data types to ensure continuity of data collections for analysis and identification of trends and validation of the high frequency data sets (Addison et al., 2018; Mack et al., 2020; Holland et al., 2025), but as our technology changes, so must our approach to data streams, data repositories and assessments. We must consider data processes that can fully integrate novel and high frequency data into our statutory monitoring programs to improve understanding of complex coastal and marine processes (Dafforn et al., 2016; Addison et al., 2018) whilst ensuring robust validation of these new technologies (García-García et al., 2019; Holland et al., 2025).

High frequency and autonomous data collected and analyzed *in-situ* and instruments deployed from ships, continuously running on ships, and deployed on buoys is still not readily used in current assessments, despite increasing the data frequency substantively for some water quality parameters. Nutrient data rely predominantly on *in-situ* samples, with buoy-deployed sensors providing additional high-temporal resolution data in limited areas (Figure 3). Bottom-water oxygen concentrations also rely predominantly on *in-situ* data but have more recently been supplemented with ship-deployed profiler sensor data significantly increasing the spatial and temporal data coverage (Greenwood et al., 2010; Hull et al., 2020, 2021). Chlorophyll is measured from water samples as well as sensors, with increased spatial and temporal resolution resulting from "FerryBox" data: an automated flow through system which continuously measures surface water concentrations aboard the ship, and from remote sensing data (Petersen et al., 2003, 2008; Harvey et al., 2015; Bean et al., 2017; El Serafy et al., 2023).

Remote sensing data can provide a valuable source of monitoring data, that includes chlorophyll, measures of turbidity and primary productivity and ecological health (Devlin et al., 2013, 2015; Petus et al., 2014, 2016, 2019; Capuzzo et al., 2018; Patricio-Valerio et al., 2022; Mohseni et al., 2022). Satellite earth observation data for chlorophyll were used for the first time in the recent OSPAR eutrophication thematic assessment (Devlin et al., 2023; van Leeuwen et al., 2023) combining data from *in-situ* sampling with higher frequency and higher spatial resolution data from remote sensing chlorophyll data (Lavigne et al., 2021) with outcomes transferred into the upcoming 2025 UK Marine Strategy assessments for marine waters (Figure 3). However, remote sensing (satellite) data has not yet been included in WFD/WER assessments due to higher uncertainties in nearshore waters, which are particularly important for eutrophication, and have observation gaps related to cloud cover (Klemas, 2011; Wei et al., 2020). Biogeochemical modeling data can also provide a potential means of supplementing limited observation data including for nutrients and oxygen, but uncertainties and model biases have prevented modeling data from being included in assessments, though some models have been invaluable for setting historical thresholds (Stegert et al., 2021; van Leeuwen et al., 2023). Full integration of high frequency data within relevant assessment areas will improve detection of when and where changes in eutrophication status are occurring.

Whilst frequency of data collection is increasing in some areas due to improved technology, some aspects of the UK's eutrophication monitoring programme are suffering significant reductions related to decreasing sampling-based monitoring in response to budget reductions. For example, the number of offshore samples taken for nutrient concentrations during the winter months which make up the eutrophication assessment period has been decreasing over the last 10 years, given the costs associated with large field programs and offshore sampling. *In situ* vessel sampling is resource intensive and requires ship-based sampling

	Type of data	Start date	Data characteristics	Spatio-temporal frequency	Use in eutrophication assessments
	Riverine Load	1975	Water quality (ammonium, nitrate, phosphates, total dissolved nitrogen and phosphorus, suspended particulate matter) sampling of rivers at their tidal limit. 'Direct discharges' from sewage, aquaculture and other industries downstream of riverine monitoring stations monitored separately.	Sampling is ideally monthly, though some rivers are monitored weekly and some only quarterly. Direct monitoring is also ideally monthly but now quarterly in many cases. Concentration data are combined with flow measurements to estimate discharges; flow data is available at higher temporal resolution. Data are reported for 39 segments of coast, with more than 150 rivers monitored.	Provides input into national and OSPAR assessments of nutrient discharges, presented as long-term trends of annual loads.
	Ship-based water quality sampling	1975	In-situ samples taken via Niskin bottles from research vessels, analysis carried out ex-situ (boat or laboratory) for salinity, dissolved inorganic nutrients, chlorophyll, phytoplankton community composition and abundance, dissolved oxygen and suspended particulate matter.	Collected on sporadic research cruises, plus targeted monitoring surveys throughout the year. Potential spatial coverage of UK Exclusive Economic Zone (EEZ) offshore.	Main data source for all UK assessments, both inshore and offshore.
	Ship-based sensor data via single or multiple conductivity-temperature-depth (CTD) profilers	1985	Depth profiling of conductivity, temperature, depth, fluorescence, turbidity, dissolved CO_2 (pCO_2), photosynthetically active radiation (PAR) and dissolved oxygen when taking water samples. Data requires different levels of calibration.	Collected on sporadic research cruises, plus targeted monitoring surveys in winter and summer. Potential spatial coverage of UK EEZ.	CTD data has been submitted to the ICES database and can now potentially be used in future assessments, particularly dissolved oxygen data. Agreement on depth profiling data submission still in progress for UK.
	SmartBuoys	2000	Conductivity, temperature, depth, fluorescence, turbidity, PAR, dissolved oxygen. Autonomous water sample data measured by sensors and dissolved inorganic nutrients, phytoplankton community composition and abundance determined on autonomously collected water samples.	Autonomous, high frequency (30 min) measurements at discrete locations.	Autonomous water sample data including water and plankton samples has been submitted to the UK MERMAN database and is available for OSPAR and UKMS assessments.
	High Resolution models	2007	Salinity, temperature, chlorophyll, dissolved inorganic nutrients, dissolved oxygen, suspended particulate matter, phytoplankton groups.	High temporal frequency (seconds–days). High spatial frequency (metres–kilometres).	Not yet used in assessments. However, models were used to develop current thresholds and assessment areas for UK marine areas.
	FerryBox	2009	Conductivity, temperature, fluorescence, turbidity, dissolved oxygen measured by sensors.	Autonomous, high frequency (1–5 min).	Potential spatial coverage of UK EEZ. Data products have been developed based on daily averages. Not yet used in assessments.
	Satellites	2001	Ocean colour measured by sensors. Data requires different levels of calibration.	High spatial frequency (0.3–4 kilometres).	Chlorophyll products included in last OSPAR eutrophication thematic assessment. Satellite data is also included for variability.

FIGURE 3
Different types of data collected in the UK Eutrophication monitoring programs describing the resolution, data characteristics and if the data has been used in UK Eutrophication assessments.

during logistically challenging bad weather months in winter. Sensor data for nutrients tends to be spatially limited to inshore sites and (for UK) at three autonomous sites (Mills et al., 2003, 2004). However, satellite and modeling data offers a new source of data for offshore waters and are already becoming an invaluable source of data for coastal and marine waters.

3.2.2.2 Monitoring inputs from land to sea

Accurate and timely information on nutrient concentrations and nutrient loads is integral to strategies designed to improve human wellbeing and manage the underlying drivers of water quality impairment and inform program of management measures (Joo et al., 2012; Pellerin et al., 2016). UK input monitoring for nitrogen and phosphorus has decreased in recent years, particularly in England and Scotland. For OSPAR, this monitoring consists of riverine inputs as well as "direct discharges" from sewage, industry and marine aquaculture (Axe et al., 2022) and aims to capture 90 % of inputs (Joo et al., 2012; Pellerin et al., 2016). Reductions in data frequency can impact our collective ability to understand if river systems are changing.

Recent studies have identified a common problem for many coastal waters, where abatement of phosphorus loads has occurred at a much faster rate than nitrogen abatement and mitigation (Lu and Tian, 2017; Ngatia et al., 2019; Devlin and Brodie, 2023). This has led to imbalanced nutrient ratios, where rivers and coastal systems are experiencing reductions associated with phosphorus but stabilization and/or or increases in nitrogen most likely due to increases in diffuse N from agriculture and direct sewage inputs. These nutrient imbalances can impact on plankton communities in coastal waters (Romero et al., 2012) but limited monitoring of nutrient loads and concentrations leads to a lack of understanding of the extent of this issue and coupled with changing climate and shifts in seasonality means that we may not be tracking these changes with sufficient confidence in our data.

The recent OSPAR eutrophication thematic assessment also identified issues around the monitoring of aquaculture loads, as the extent of the growth in aquaculture within the OSPAR Maritime Areas has been substantial (Axe et al., 2022; OSPAR, 2023). A gap exists concerning the agreement of minimum environmental standards for aquaculture across the OSPAR Maritime Area which is relevant to UK waters given the rise of aquaculture in many coastal areas. A more strategic approach to monitoring nutrient inputs could rationalize the collection of high-density data to focus on a few rivers, sub-catchments, and transitional and coastal waterbodies. This would require detailed analysis of existing monitoring data to identify candidate waterbodies that could best represent the types of coastal systems across the UK as well as incorporating land-use data and modeling which are not currently integrated with marine eutrophication assessment frameworks.

3.2.3 Improving the alignment between directives

3.2.3.1 Connecting the catchment to coast

Whilst the WFD/WER, UK Marine Strategy and OSPAR COMP advocate for a river basin approach, the directives are not always aligned. There is a disconnect between geographical boundaries and indicator thresholds which hinders understanding of eutrophication status across the continuum from transitional and coastal to marine waters (Foden et al., 2011). Alignment between decision making on programmes of measures and downstream impacts is not possible when there is disaggregated policy implementation across agencies, reporting to different government areas and stakeholders (Figure 4). Historically, UK terrestrial and marine environmental policies have been largely delivered in isolation despite the marine system being explicitly connected to the land with most of the marine pollution originating from terrestrial sources (Howarth, 2008). This has resulted in a disconnect between inshore and offshore assessments across the arbitrary policy line at 1 nm (3 nm in Scotland) that separates coastal water bodies under WFD/WER from the full extent of riverine plumes and the OSPAR and UK Marine Strategy assessment areas further offshore. This disjointed approach hinders our understanding of how land-based management measures are impacting our coastal systems. There is also a disconnect between eutrophication indicators and thresholds developed for OSPAR and those developed and applied under the WFD/WER, reflected in the UK Marine Strategy where the outcomes from different assessments were combined without full harmonization of data or assessment structures. More work is required to progress beyond the current method of simply combining the coastal and offshore assessment outcomes derived by different methodologies (Devlin et al., 2007a, 2023; Foden et al., 2011).

Future assessments should consider a fully integrated catchment to coast approach (Waterhouse et al., 2011; Brodie et al., 2012; Creighton et al., 2021) which has greater potential to change the input of terrestrial contaminants into our marine environment with subsequent positive effects on the coastal ecosystem. Managing catchments to control diffuse pollution into downstream coastal systems will also provide benefits for freshwater systems. Future eutrophication assessments should be managed more holistically, bringing diverse stakeholders together to ensure common and opposing interests are included, and to assess impacts over the ecological and hydrological boundaries.

3.2.3.2 The importance of riverine influenced areas

Until recently, eutrophication assessments carried out for the UK Marine Strategy and the OSPAR COMP have used assessment areas defined by geographical or political boundaries rather than those which are ecologically coherent and fully represent the extent of terrestrial influence in marine waters (Foden et al., 2011; van Leeuwen et al., 2015; Devlin et al., 2023). The UK coastal zone covers up to 10% of the Exclusive Economic Zone (EEZ) though that number varies dependent on the definition of coastal zone. The coastal zone is dynamic with high spatial and temporal fluctuations influenced by tides, stratification, wind and river discharges, which all influence variation in the water quality measurements. This was recognized within the initial implementation of the EU WFD (Vincent et al., 2022) which developed an approach to defining transitional and coastal typologies characterized by tidal range, mixing, salinity and depth. However, the seaward edge of the coastal assessment areas was defined by a 1 nm offshore limit for England, Wales and NI and by a 3 nm limit for Scotland, leading to abrupt delineations between the nearshore coastal areas assessed under the WFD/WER and coastal to offshore areas assessed under UK Marine Strategy and OSPAR.

Riverine freshwater plumes are the major transport mechanism for nutrients, sediments and pollutants and connect the land with the receiving coastal and marine waters. Knowledge of the variability in the extent of freshwater influence into UK marine waters is relevant for environment managers to develop strategies for improving ecosystem health and risk assessments (Schroeder et al., 2012; Devlin et al., 2012b, 2015). An approach using satellite derived suspended particulate matter (SPM) and *in situ* salinity provided a first estimate of the physical, chemical and biological processes (Greenwood et al., 2019) with the area of riverine influence mapped using salinity and satellite derived suspended particulate matter (Ivanov et al., 2020; Fettweis et al., 2023; Desmit et al., 2024). Sea surface salinity is the most traditional conservative tracer of freshwater discharge; however, it can be difficult to extract direct satellite-based salinity measurements with sufficient spatial resolution for coastal applications (Schroeder et al., 2012).

Although these mapping methods can improve assessments across ecologically homogeneous areas, defining river plumes seasonally and with a relatively high resolution in coastal areas is still required to assess water quality conditions across estuarine and intertidal habitats (Fronkova et al., 2022; Heal et al., 2023). Plume mapping of UK waters has now progressed to deriving and mapping the Forel Ule color scale, as determined from high resolution Sentinel-3 satellite imagery data at 1 km^2 resolution for England and Wales and using the 4 km^2 resolution for Scottish coastal waters (Fronkova et al., 2022). Using the relationship between ocean color and water quality parameters, recent work has defined geographically resolved assessment areas through the mapping of ocean color (Greenwood et al., 2019; Fronkova et al., 2022; Heal et al., 2023) (Figure 5). The most recent OSPAR QSR implemented a new set of ecologically relevant assessment areas (Figure 6) developed specifically for eutrophication assessment with common indicators

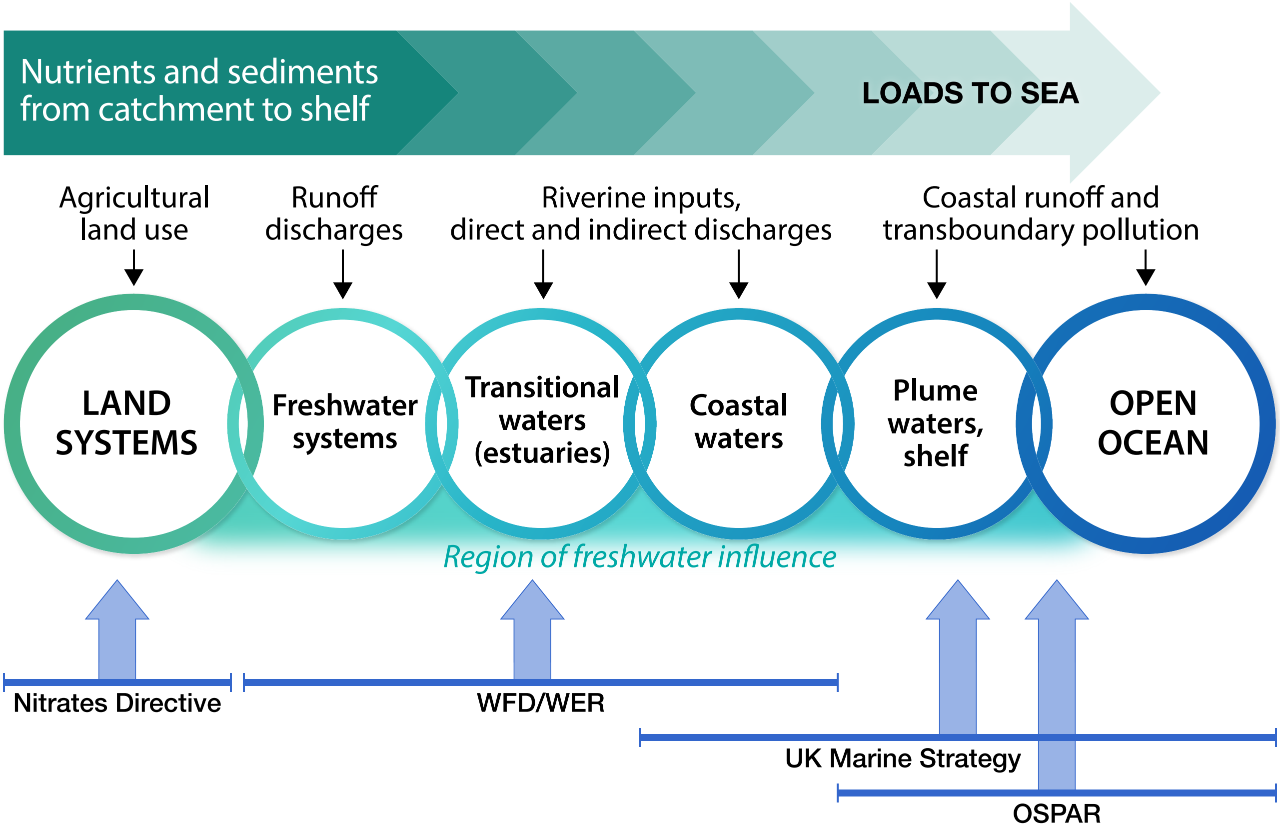

FIGURE 4
Eutrophication assessments span across the land to sea continuum, working across different regulations, upstream to downstream systems and a wide range of stakeholders. Different monitoring programs from across UK environment and regulatory agencies feed into the UK's national eutrophication assessment.

and harmonized thresholds across national boundaries (Devlin et al., 2023; Lenhart et al., 2010; van Leeuwen et al., 2023). The new OSPAR assessment areas and thresholds, carried across for the UK Marine Strategy eutrophication assessment provides a much stronger alignment between nearshore and coastal marine waters than previous assessments, with the inclusion of plume areas and more localized coastal assessment units (Figure 5) which can be expanded into future assessments (Devlin et al., 2023).

3.2.4 Improving our indicators of eutrophication

The eutrophication frameworks, outcomes and subsequent program of measures were based on best available information around key eutrophication indicators including nutrients, phytoplankton biomass (measured as chlorophyll) and dissolved oxygen (OSPAR, 2003, 2008a). WFD/WER assessments also use phytoplankton counts in coastal waters. These indicators are, and continue to be, highly relevant to measuring the extent and impact of eutrophication (Bricker and Devlin, 2011; Devlin et al., 2011). However, using these indicators only can limit our understanding of ecosystem impacts, and there is an urgent need to expand, both in the improvement of our current indicators and through the development of new indicators.

3.2.4.1 Indicators should consider trends and trajectory of change

Measuring nutrient loads and concentrations are important parts of understanding the trajectory of risk and potential impact of eutrophication. Reduction of diffuse nitrogen and other agricultural pollutants after the implementation of EU and UK directives, whilst initially successful, has leveled off over time. The abatement of nitrogen has not kept pace with the scale required across all catchments and diffuse nitrogen losses are now the main source of nitrogen loading into coastal and marine waters (Worrall et al., 2016; Zhang et al., 2022). At the same time, there has been an increase in the farmed fish industry in some coastal regions resulting in increasing point source introduction of nutrients to the environment that are not considered in our current assessments (Olsen et al., 2008; Holmer, 2010; Edwards, 2015). Our current assessments, using a multimetric approach incorporating several indicators, are based on status assessment (a single value aggregated over 6-year cycle), but associated trend assessments are not currently part of the quantitative assessment of eutrophication indicators. The incorporation of trend data into assessments can demonstrate the trajectory of change and help in predictions of future state. However, in the recent UK Marine Strategy and OSPAR COMP, long term changes in nutrient inputs were assessed using Mann-Kendall analysis to detect trends,

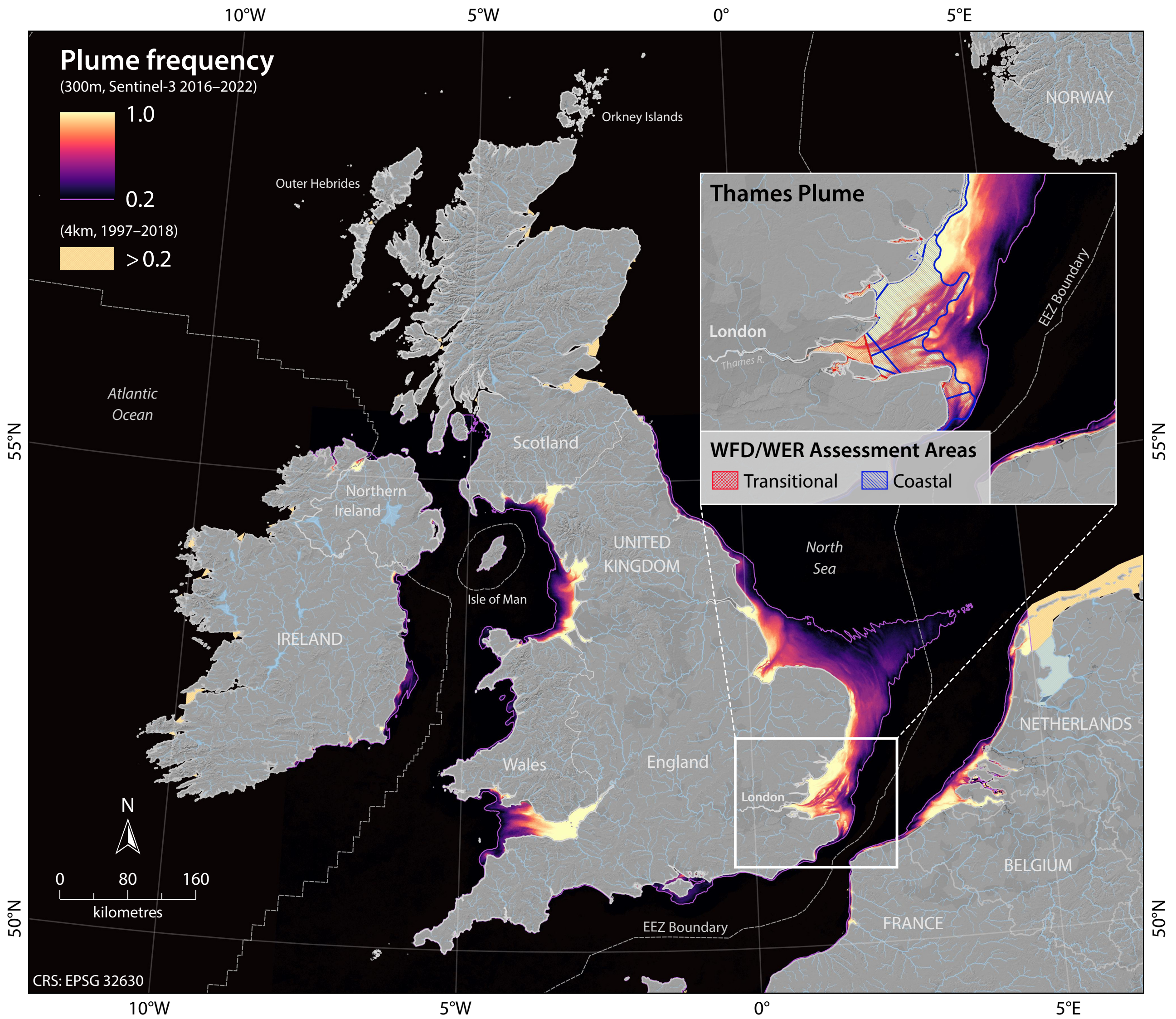

FIGURE 5
Riverine influenced areas for the UK are based on two different types of plume methodologies, with a higher 300 m resolution plume extent derived from the Forel Ule Index (FUI) from Sentinel-3 and lower resolution plume extent derived from Pitarch et al. (2019) 4 km FUI data. The FUI derived from Sentinel-3 has been used to map the extent of flood plumes, with FUI values of 10 or above being representative of riverine-influenced areas. The purple line designates the maximum plume extent, where the area is exposed to riverine plume at least 20% of the time between 2017–2022. Both approaches have been applied in England, Wales and Northern Ireland waters, with only the 4 km resolution imagery applied in Scotland. Development of plume imagery detailed in Fronkova et al. (2022).

quantifying (where present) monotonic trends in timeseries data, based on comparing each year's value to all preceding years (Devlin et al., 2023). This non-parametric method is useful for long term environmental data as it is not affected by any transformation of the annual data values, and it is flexible for time-series with missing data points (Bedford et al., 2020; Desmit et al., 2020). Non-parametric trend tests require only that the data be independent and can tolerate outliers (e.g., resulting from a change in analytical detection limit) and missing values in the data. Expanding the use of trend analysis such as Mann-Kendall will allow greater scrutiny of the direction of travel alongside assessment of state.

3.2.4.2 Indicators that consider susceptibility to eutrophication

Phytoplankton, through their chlorophyll cells, absorb light to provide energy for photosynthesis. Light is rarely limiting in offshore waters, but the more turbid, variable inshore waters can make it difficult for plankton to absorb enough light to grow (Devlin et al., 2008). The clarity and composition of highly dynamic estuarine and coastal waters can help inform susceptibility of the coastal and marine waters to eutrophication (Cloern, 1987, 1999, 2001). Vertical attenuation of light through the water column is attributable to the optically active components of phytoplankton, suspended particulate material (SPM) and chromophoric dissolved

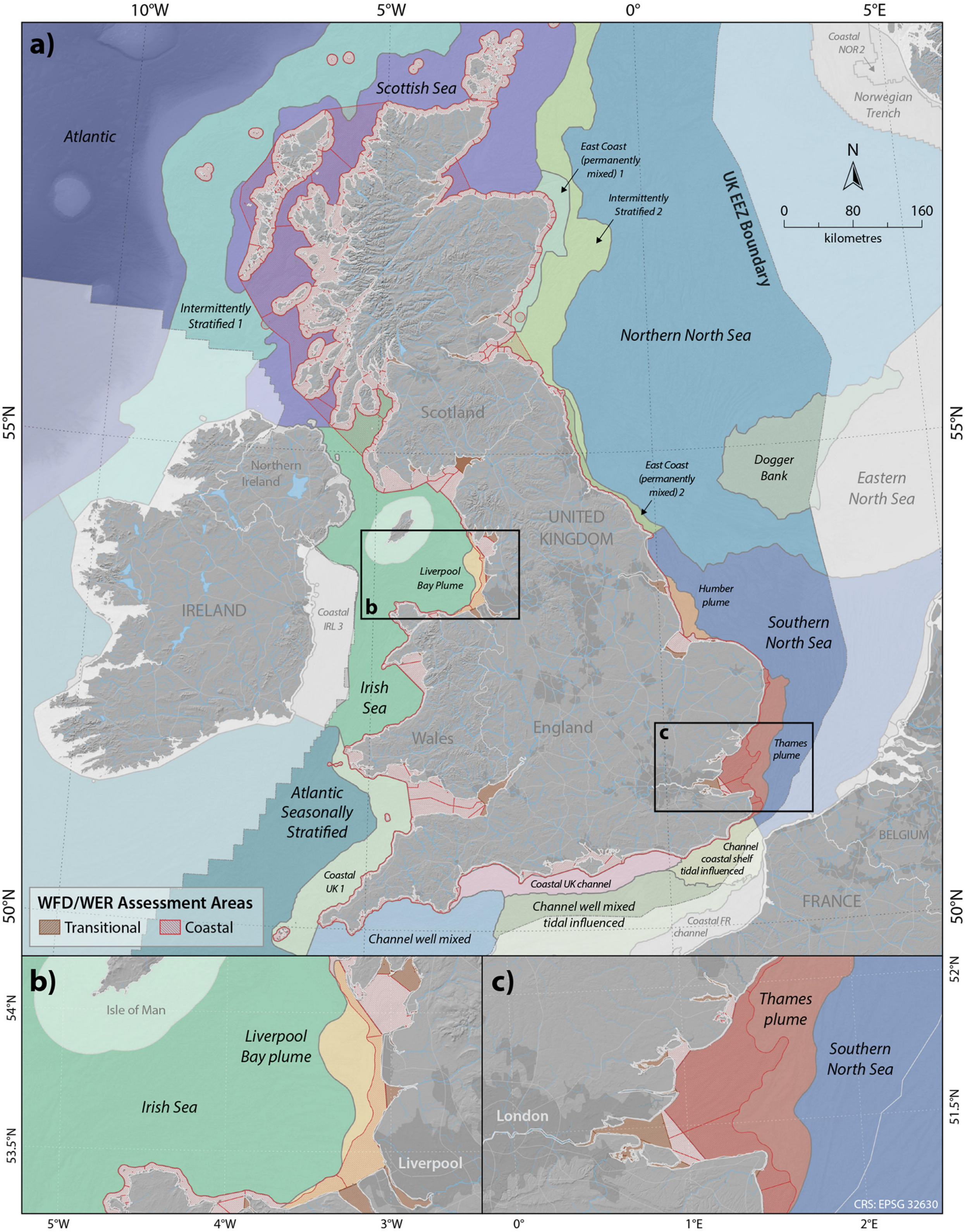

FIGURE 6
UK assessment areas reported under OSPAR Quality Status reporting. Note that several of these areas are shared with other OSPAR member states including Atlantic, Irish Sea, Northern North Sea and Southern North Sea, areas outside of the UK EEZ are lighter in color. WFD/WER coastal assessment units are highlighted in red. **(a)** Full extent of the UK assessment areas for coastal and offshore waters zoomed in to estuarine and coastal areas of **(b)** Liverpool Bay and **(c)** Thames embayment.

organic matter (CDOM). SPM is routinely measured in all UK monitoring programs but is not used as a standalone indicator. However, SPM is used in WFD/WER eutrophication assessments when nutrient concentrations exceed thresholds, and an additional (higher) threshold is applied if SPM is high (Greenwood et al., 2019). In offshore waters, CDOM originates predominantly from bacterial decomposition of phytoplankton cells, whereas in coastal waters, CDOM is dominated by humic and fulvic acids of terrestrial origin and transported to the seas through freshwater runoff from the land as well as autochthonous CDOM from salt marshes, mangroves, inter- and sub- tidal benthic microalgae, seagrasses, macroalgae and corals (Carder et al., 1989). CDOM is not routinely measured for eutrophication monitoring and assessment (Foden et al., 2008) despite being a key measurement for understanding riverine influenced plumes and light dynamics.

CDOM, ocean color, light attenuation, turbidity and SPM are all important elements in understanding the extent of the estuarine and coastal systems and the dynamics of the light conditions. Foden et al. (2008) discuss how applying a simple dose-response model of nutrient enrichment to risk of eutrophication does not consider the important role light plays in marine waters, and limits understanding of the complex interactions at play. Cloern (1999, 2001) recognizes system attributes that "filter" responses to changes in nutrient loading, including the underwater light climate, horizontal exchange, tidal mixing, grazing and biogeochemical processes. This complex response determines susceptibility, which influences the assessment of eutrophication status. The light climate is highly variable in UK waters and therefore of particular significance regarding the risk of eutrophication (Devlin et al., 2008; Foden et al., 2008, 2011).

Over the last century, the world oceans and coastal regions have experienced changes to marine lightscapes in two fundamental ways. Firstly, regions such as the Norwegian Fjords have experienced a long-term reduction in water clarity, referred to as Coastal Darkening (Aksnes et al., 2009) with large-scale drivers that are connected to effects of climate change such as more frequent and intense rainfall, increased temperatures, and other human activities that increase erosion (Dupont and Aksnes, 2013; Organelli et al., 2017; Frigstad et al., 2023). A reduction in the light availability will impact key eutrophication indicators such as phytoplankton (Capuzzo et al., 2018; Opdal et al., 2019; Wollschläger et al., 2021). Secondly, some coastal regions are experiencing a brightening of the night-time light environment linked to urbanization, on- and offshore infrastructures, fisheries, and shipping (Smyth et al., 2022; Davies et al., 2023). Knowledge of changes in natural light conditions needs to be a key part of eutrophication (and climate change) assessments into the future. The latest round of assessments show that the UK has many high nutrient transitional and coastal waters, but most of these were not deemed eutrophic due to not being able to show an "undesirable disturbance." For example, conditions in UK coastal waters with high turbidity and large tidal systems are assumed to not support the proliferation of high biomass (Cloern, 1987; Thornton et al., 2002; Tweedley et al., 2016). However, recent work has shown that high growth can still occur under turbid conditions, with productivity remaining high despite light limitation (Gonçalves Leles et al., 2018). More work on the role of mixoplankton is required as it is now recognized that most phytoplankton and as much as half the protist-zooplankton combine both plant-like photosynthesis and animal-like consumer activity synergistically within the same single-cell (Ward and Follows, 2016; Stoecker et al., 2017; Gonçalves Leles et al., 2018; Mitra et al., 2024).

3.2.4.3 Indicators require improved understanding of natural variability

Our understanding of natural variability needs to evolve alongside our understanding of the complexity of our coastal and marine systems. The recent OSPAR eutrophication thematic assessment used an ensemble model approach to derived pre-eutrophic conditions (Lenhart et al., 2010; Stegert et al., 2021; van Leeuwen et al., 2023). Nutrient loads into the north-east Atlantic were estimated from rivers under historic conditions prior to 1900, using the European model E-HYPE and observations with historic (pre-eutrophic) conditions (Lenhart et al., 2010; Stegert et al., 2021). Reference chlorophyll concentrations were derived corresponding to the estimated nutrient concentrations under reference conditions. To account for natural variability allowing for a "slight disturbance" in the absence of more specific information, the assessment levels (thresholds) were then defined as a concentration of 50% more than the area-specific background concentration derived from the ensemble model approach (Malcolm et al., 2002; Borja, 2005; OSPAR, 2008b, 2022) which set the threshold between Non-Problem and Problem Areas. This is equivalent to the boundary setting good/moderate for the EU Water Framework Directive (WFD) and boundary setting between GES and non-GES for UK Marine Strategy and the EU MSFD (Claussen et al., 2009; Maas-Hebner et al., 2015; Topcu and Brockmann, 2021). However, when using long term monitoring data from the recent OSPAR eutrophication thematic assessment the natural variability for the common indicators (from 2015 to 2020) in UK waters varies between 24% and 56% (OSPAR, 2023). Whilst a deviation from baseline is important for all assessment processes, future assessments need to analyse "true" natural variability and look for commonalities and differences between assessment areas and different indicators.

3.2.4.4 Indicators that consider societal impacts, including environmental, social and economic drivers

Embedding environmental and societal information alongside our more traditional monitoring and assessment processes is required to fully integrate ecosystem state, pressures, stakeholders, and policy (Kristensen, 2004; Borja et al., 2006; Patrício et al., 2016). This could include quantification of economic and environmental connections, greater integration of the reporting of complex interactions between social, economic, and ecological factors, multi-disciplinary frameworks and enhanced community engagements.

One such well known framework that has achieved many positive results is the DPSIR framework, which incorporates Drivers (D), Pressures (P), State (S), Indicators (I), and Response (R). The DPSIR framework is a widely used approach to understand interconnected layers and measure the driving forces of change (Elliott et al., 2017; Kristensen, 2004; Martin et al., 2018; Patrício et al., 2016). Simple messaging and clear linkages between human-induced drivers, pressures, state, impacts, and human welfare are crucial to drive outcomes into policy implementation

(Kristensen, 2004; Borja et al., 2006; Elliott et al., 2017; Martin et al., 2018). Drivers and pressures can be complex and difficult to measure and understanding the interactions between human drivers and ecological pressures is a key component to any monitoring and evaluation program, particularly one that spans from land to sea and multiple stakeholders (Oesterwind et al., 2016). The DPSIR framework has been adopted by the European Environment Agency and others (Borja et al., 2006; Atkins et al., 2011a; Patrício et al., 2016) and describes a framework for assessing the causes, consequences and responses to eutrophication in a holistic way including managing catchment to coast processes (Bowen and Riley, 2003; Langmead et al., 2007; Maccarrone et al., 2014; Le Gentil and Mongruel, 2015). DPSIR frameworks act as decision support systems which can enhance communication, knowledge transfer and interaction among scientists and policymakers, facilitating engagement among stakeholders and enhancing the legitimacy of the decision-making process (Newton and Weichselgartner, 2014).

Additionally, programs that include clear elucidation of cause and consequence, socio-economic pathways, and greater levels of engagement with communities are all attributes of successful monitoring and evaluation programs that could be applicable to UK coastal and marine systems (Figure 7). Embedding a greater understanding of natural capital into monitoring programs has been a successful way to incorporate the system flows between ecology, goods and services, and benefits to human wellbeing (Rhodes et al., 2017). Kermagoret et al. (2019) shows increasing eutrophication leads to a degradation of ecosystem service bundles, particularly for nutrient and pathogen regulation/ sequestration, or for the support of recreational and leisure activities. A cost benefit analysis on freshwater eutrophication in England and Wales, recognizing many data gaps, estimates the damage costs of freshwater eutrophication to be £75.0–114.3 million yr^{-1} (Pretty et al., 2002) highlighting the severe impacts of nutrient enrichment and eutrophication on many sectors of the economy. Estimates of local economic benefits for seaside recreation and waterfront property in Denmark through reduction of agricultural N losses were estimated at €35 million, with co-benefits of up to €57 million (Andersen et al., 2019). Conversely, the value of functioning coastal systems and nature based solutions to remove anthropogenic nitrogen can have wide ranging positive impacts and reduce costs of waste remediation (Watson and Beaumont, 2024). Cost accounting and reporting of natural capital benefits can be a valuable tool for positive policy implementation. Estimates of damage costs against policy response should be a critical part of any future eutrophication assessment.

Integrated approaches like the cross-sectoral and transdisciplinary One Health monitoring and evaluation framework (Stentiford et al., 2020), that emphasizes the interconnections between the health of humans and ecosystems, are highly applicable to an integrated eutrophication approach. These holistic approaches recognize the benefits of programs that are relevant to a range of end-users and contribute to positive changes through management actions that engage and represent the values of a diverse range of stakeholders impacted by the decision making. This is particularly true for local and regional stakeholders but can also extend to international partnerships and frameworks.

Greater engagement of the community, not only through data collection (i.e., "citizen science") but as an important part of the evaluation side, and by becoming embedded in decision making around policy and governance, should be a key requirement of greater success in monitoring and evaluation programs (Bischof, 2010; Cigliano et al., 2015; Darling et al., 2019). Local citizen scientists can cover important spatial gaps and when combined with long-term monitoring data from regulatory agencies, can add benefit to eutrophication monitoring programs (Loiselle et al., 2024).

3.2.4.5 Integration of indicators to multidirectional value of state

Current UK eutrophication indicators are assessed in terms of unidirectional exceedances. This approach does not fully encompass the range of conditions that support positive ecological functioning. For example, a chlorophyll threshold which, when exceeded, indicates only that increasing biomass can potentially result in undesirable disturbance to the ecosystem (Tett et al., 2007). However, recent work has also identified declining primary productivity (Capuzzo et al., 2018) due to warming waters and changing nutrient imbalances (Ryther and Dunstan, 1971; Lu and Tian, 2017; Nohe et al., 2020; Xu et al., 2020). Our eutrophication approach needs to consider the optimal range of values for each indicator, providing benchmarks for upper and lower limits. This approach would recognize that for some indicators, that there is a seasonal and temporal shape to the optimum range that needs to be considered and not just a single upper threshold (Tett et al., 2007, 2008; Bedford et al., 2020). Additionally, many of the indicators are intrinsically linked, and need to be operating in the optimum range across a wide range of interconnected ecological measurements.

Safe Space functioning is an emerging concept that is becoming more visible in our understanding of ecosystem state. Rockström et al. (2009) defined the boundaries for a "safe operating space for humanity, which outlines nine planetary goals that need to be supported together." The underlying concept of ecosystems being dependent on various conditions, all existing in the same conceptual space, should become a tangible part of our future thinking. Eutrophication assessments could develop a range of conditions that need to be met to ensure ecological functioning and resilience (Figure 8). This could be adapted from existing indicators of nutrients, biomass and dissolved oxygen, measures of important physico-chemicals such as turbidity and expand to look at corresponding plankton community metrics focusing on changes in plankton lifeforms impacted by eutrophication and primary productivity.

3.2.5 Improving our understanding of ecosystem impacts

3.2.5.1 Eutrophication as a plankton pressure

Eutrophication is a complex process and often associated with not only changes in overall algal biomass but also with changes in plankton community structure. Common eutrophication indicators (e.g., chlorophyll a, nutrients, dissolved oxygen) are not adequate for understanding biodiversity changes, especially those associated with the proliferations of HABs (Glibert, 2017; Anderson et al., 2002).

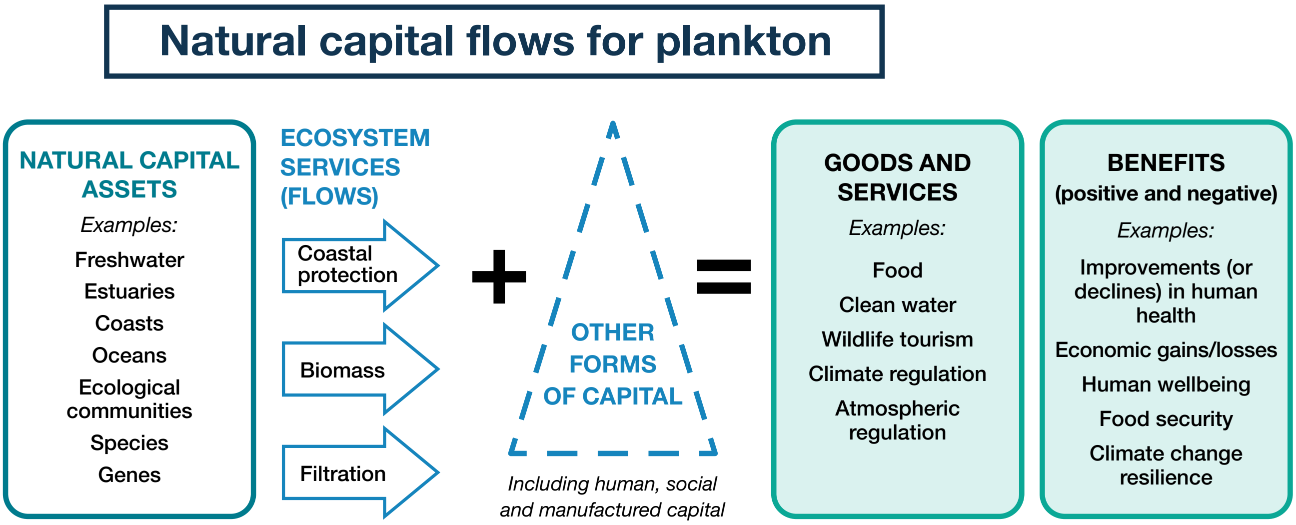

FIGURE 7
Schematic illustration of how natural capital represents the flow between the asset and ecosystem service into goods and services and benefits for humans. Diagram adapted from Devlin and Wenger (2024).

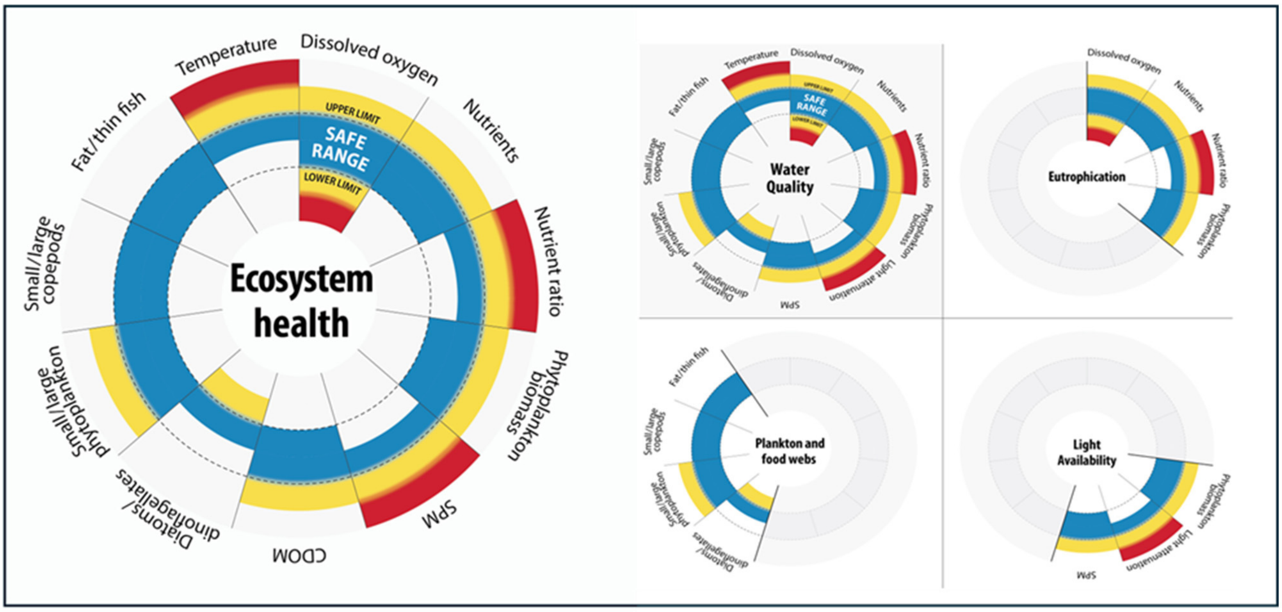

FIGURE 8
Visualization of the concept of "safe space" functioning, which can be adapted to represent a range of conditions that need to be supported together, identifying upper and lower limits around that condition. CDOM, chromophoric dissolved organic matter; SPM, suspended particulate matter.

Many coastal waters are experiencing nutrient imbalances due to the more successful mitigation of phosphorus over nitrogen and the continuing excess of nitrogen over silicon compounds. High N:Si favors non-silicified micro-algae, leading, for example, to blooms of Phaeocystis following silica depletion in the southern North Sea (Davidson et al., 2012). High N:P might favor mixotrophic nanoflagellates, which can obtain P from other micro-organisms, although this seems most relevant under oligotrophic conditions (Duhamel et al., 2019). High N:P ratios also correlate with Phaeocystis colony dominance in the southern North Sea (Lancelot et al., 2009). Many international marine waters have reportedly undergone regime shifts in phytoplankton community composition; the proportion of diatoms has decreased, (predominately in inshore, nutrient enriched coastal waters) whereas that of non-diatoms such as dinoflagellates and cyanobacteria has increased (Xiao et al., 2018; Lu et al., 2023; Chen et al., 2024). A shift in the phytoplankton community composition from diatoms, which have traditionally been dominant, to non-diatoms, can impact on the plankton community and foodwebs (Granéli and Turner, 2002; Verity et al., 2002; Lu and Tian, 2017; Chen et al., 2024). This change has several consequences, including reduced energy transfer to higher trophic levels, higher respiration rates (higher oxygen consumption and CO_2 production), and accumulation of fewer economically valuable organisms damaging the regulatory, provisioning, cultural, and supporting service functions within the marine ecosystems (Yunev et al., 2017; Chen et al., 2024). The consequences of changes in diatom and dinoflagellate dynamics in the phytoplankton community is a major concern because these taxa play key roles in ecosystem processes and form the basis of many aquatic food webs (Menden-Deuer and Lessard, 2000). Dinoflagellates account for 75% of all harmful phytoplankton species (Smayda and Reynolds, 2003) which can adversely affect human health as well as marine fisheries and aquaculture (Anderson et al., 2002).

Different species of phytoplankton have different traits (Bedford et al., 2018, 2020; Graves et al., 2023; Holland et al., 2023a,b; McQuatters-Gollop et al., 2017), most notably size and shape, growth rate, life history, and behavior such as motility, that together determine their ecological niche and preferred environmental conditions, and phytoplankton are a major driver for global carbon fixation and biogeochemical cycles. Shifts in plankton species composition from diatoms to dinoflagellates may indicate a shift in the balance of organisms due to eutrophication. The composition of the phytoplankton community could be compared with area-specific reference conditions and be expressed by the ratio of diatoms to dinoflagellates. To maximize the utility of the plankton lifeform approach for informing the management of marine ecosystems, changes in the abundance of lifeforms need to be attributed to drivers of change. These drivers can include "directly manageable" anthropogenic pressures such as eutrophication caused by nutrient loading (Bedford et al., 2020; Ostle et al., 2021; Graves et al., 2023; Holland et al., 2023b).

Whilst nutrients and phytoplankton biomass are key indicators for eutrophication assessments, there has been an increasing focus on marine protection and, more generally, biodiversity protection which has led to efforts to connect eutrophication with its impacts on impact on the balance of plankton lifeforms in UK waters (Graves et al., 2023; Holland et al., 2023a; McQuatters-Gollop et al., 2024). Reanalysing historic phytoplankton data collected under the various assessments found an increase in small phytoplankton cells, relative to the decreasing large cells and an increase in the dinoflagellate group, which are generally much less nutritious and more toxic than the decreasing diatoms (Graves et al., 2023). The UK Marine Strategy provided a framework within which to include plankton in the biodiversity assessment and developed plankton tools that applied a lifeforms approach grouping plankton by their common key functional traits (McQuatters-Gollop et al., 2019; Bedford et al., 2020; Graves et al., 2023; Holland et al., 2023a). Improving our ability to track nutrient pressures against biological impact should expand on this by considering pelagic communities explicitly within the eutrophication assessments.

3.2.5.2 Nutrient imbalances and plankton health

Human-induced inputs of phosphorus (P) and particularly nitrogen (N) into the biosphere continue to be problematic for inshore waters. The ratio of N to P, which is sensitive to the relative anthropogenic inputs of the two nutrients, has emerged as a significant driver of environmental change, impacting organisms, ecosystems, and global food security (Penuelas and Sardans, 2023). Historically, P has been the main nutrient controlling upstream freshwater productivity, whereas N limitation is more prevalent in most coastal waters (Brodie et al., 2011; Devlin et al., 2023). However, controls on production and nutrient cycling in estuarine and coastal systems are physically and chemically distinct from those in freshwater counterparts, and upstream nutrient management actions (predominately P controls) have exacerbated N-limited systems further downstream (Paerl, 2008). These changing anthropogenic activities have caused imbalances in N and P loading in UK waters (Figure 9) making it difficult to control eutrophication by reducing only one nutrient. Effective management of inputs of both nutrients are needed for long-term control of eutrophication in our coastal and marine waters. There has been concern about shifting nutrient ratios for some time with increased reporting of the impact of global nutrient loads and concerns on the export of nitrogen increasing faster than phosphorus (Glibert, 2017). Changes in nutrient loading, balances and concentrations can affect plankton, their biodiversity and toxicity with unintended consequences for HABs, mixoplankton, plankton communities and the food web (Gonçalves Leles et al., 2018; Bedford et al., 2020; Graves et al., 2023; Holland et al., 2023a; Mitra et al., 2024).

3.2.6 Consideration of climate resilience and shifting baselines

Climate change is impacting the environmental baseline, with changes in rainfall patterns affecting the delivery of freshwater and associated nutrients and sediment to the coastal marine environment with subsequent effects on ecosystem processes. Climate change has the potential to increase nutrient run off and algal growth and modify the interactions between planktonic and pelagic organisms. Summer rainfall is predicted to decrease alongside increases in the intensity of summer and autumn rainfall events (Cotterill et al., 2023). Shifts in seasonal rainfall patterns can change both the intensity and frequency of nutrient inputs where sudden, large events can cause excessive flooding and pulses of increased inputs and sediments into the marine environment. Predicted changes in the timing and intensity of winter rainfall may lead to increased riverine inputs of nitrogen and phosphorus (Ockenden et al., 2017; Bussi et al., 2016). It is expected that climate change will result in more hydrological extremes and higher river discharges, particularly in the northern parts of the North Sea (Willems and Lloyd-Hughes, 2016). These intense events may lead to large nutrient loads entering the coastal environment, particularly if they coincide with recent agricultural applications of fertilizer or slurry and/or intense rainfall following prolonged spells of very dry weather. Changes in the frequency of such intense events may also lead to changes in discharges from combined sewer overflows (CSOs), with the potential to alter nutrient inputs to the coastal environment. The magnitude of these changes is expected to vary widely between catchments, depending on land use, type of agriculture and agricultural practices and also on future land use and socio-economic developments (Arheimer et al., 2012).

Increased water temperatures have been shown to lead to phenological shifts, biogeographical changes and changes in abundance of plankton (Brander et al., 2016; Eker-Develi et al., 2022). Temperature change and eutrophication are known to affect phytoplankton communities (McQuatters-Gollop et al., 2022),

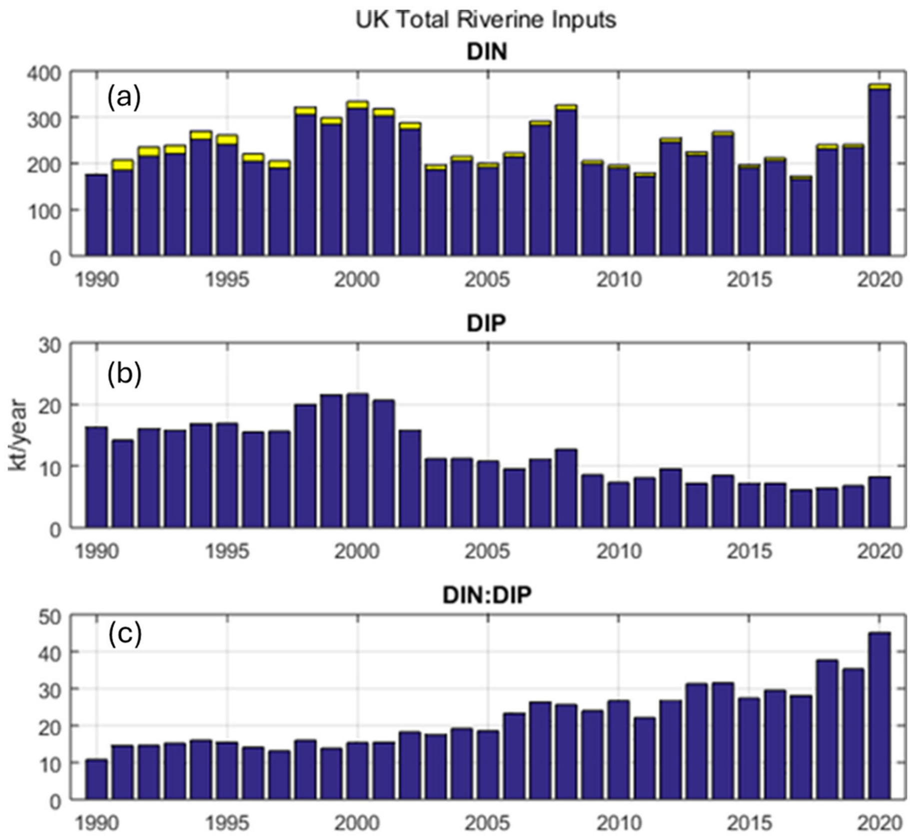

FIGURE 9
Long term changes in nutrient inputs from the UK including **(a)** Dissolved Inorganic Nitrogen (DIN), **(b)** Dissolved Inorganic Phosphorus (DIP) and **(c)** the ratio of DIN:DIP, with the increase or stabilization of DIN compared to decreasing DIP concentration leading to a five-fold increase in DIN:DIP ratio over the past 30 years.

but we have limited information on the effects of interactions between simultaneous changes in temperature and nutrient loading within coastal ecosystems. Ecological time-series are critical for understanding the drivers of change, especially of the interaction between eutrophication and climate factors (Edwards et al., 2010). Such interactions have been key in driving diatom-dinoflagellate dynamics in coastal systems such as the East China Sea where studies have shown that diatoms preferred lower temperature and higher nutrient concentrations, while dinoflagellates were less sensitive to temperature and nutrient concentrations but tended to prevail at low phosphorus and high N:P ratio conditions (Xiao et al., 2018). These different traits of diatoms and dinoflagellates resulted in the observation that increasing stratification and increasing terrestrial nutrient input could promote dinoflagellates over diatoms (Xiao et al., 2018).

Increases in recycling rates of organic matter in the water column with increasing temperature have the potential to increase the availability of inorganic nutrients for phytoplankton growth during the year and therefore increase chlorophyll. Increasing storminess and intensity of rainfall have the potential to increase the turbidity of UK shelf seas. Studies have shown that turbidity in the North Sea has increased over the past 100 years as a result of changes in storminess (Capuzzo et al., 2018; Wilson and Heath, 2019). Changes in the underwater light climate due to increases in turbidity have been linked to decreases in productivity (May et al., 2003). Any increases in storminess during the stratified period could act locally to increase dissolved oxygen concentrations in bottom waters by enhancing mixing from the surface. Increased storminess could also increase resuspension of sediments from the seabed which would increase the oxygen demand for remineralisation of the organic matter, leading to a decrease in oxygen concentration (Mahaffey et al., 2023). Increases in riverine nutrient loads from altered precipitation patterns may enhance algal biomass with the potential to decrease dissolved oxygen in riverine influenced coastal waters through increased oxygen demand when the organic matter is remineralised. Being able to predict the impact of climate change and understand potential mitigation measures is key to successfully managing the marine environment and will require adjustments to our current eutrophication indicators (Table 2).

3.3 Prioritization of solutions required to fill evidence gaps

From our review of eutrophication assessments, we have collated recommendations to improve future eutrophication monitoring programs through addressing the evidence gaps

TABLE 2 Requirements for future eutrophication assessments to fully consider climate interactions for the three common indicators most frequently used across all eutrophication assessments.

Eutrophication indicator	Requirements for and roadmap to achieving eutrophication assessments that account for climate change
Trends in nutrient inputs	• Review/reanalyse historic input data to improve understanding of baseline • Improve link between current monitoring and modeling of inputs in order to integrate available information and inform new monitoring programmes • Improve representation of relevant processes in marine biogeochemical models and related model uncertainty to allow model outputs to be used as a warning system, evaluating the likely marine impacts of potential future input scenarios
Chlorophyll biomass—phytoplankton changes	• Improve incorporation of plankton lifeforms into eutrophication assessment alongside developing links to food web indicators in order to develop a greater understanding of the relationship between eutrophication drivers and plankton community change (i.e., beyond chlorophyll/biomass) and thus consider system-level climate change impacts
Dissolved oxygen	• Determine the mechanisms driving spatial and temporal trends in dissolved oxygen and confidently identify when and where changes in dissolved oxygen are being driven by human-induced activity such as ocean warming or nutrient enrichment from background natural variability • Address the lack of long-term data in regions outside the North Sea and overall poor data resolution which hampers assessment of the occurrence, frequency and spatial extent of oxygen deficiency in UK coastal and shelf waters and the ability to confidently test coastal and shelf-sea models which are required to understand climate change impacts

Development of any new indicators such as pelagic lifeforms would also need to account for changing climate baselines.

within the five themes of data, alignment, indicators, ecosystem and climate.

Following the recommendations to improve future eutrophication assessments, we next provide an assessment of feasibility, developed from the review and expert knowledge of what is currently occurring in UK national eutrophication monitoring programs.

Feasibility of developing and implementing the key evidence gaps into future monitoring and assessment is reported against the effort required to achieve the recommended activity, the scale of complexity and potential costs of implementation (Figure 10). Effort is defined as the level of agency involvement and if the work can be delivered from one or more agencies, the level of stakeholder participation required and if any components of the work has already been agreed. Complexity is a measure of the type of data required to deliver the work, with the more complex systems being more difficult to deliver and includes a consideration of how easily the data changes could be made and if those changes are dependent on cross agency, cross stakeholder agreement. Costs are a simple classification of high costs related to ongoing staff, vessel and field time against lower costs that are not dependent on implementing aspects of a field-based monitoring program.

Key recommendations under each thematic area are described in Figure 11, summarizing the evidence gaps identified through the review and consultation, a list of activities that could address these evidence gaps, and then the outcomes of the feasibility assessment (effort, scale, cost). Additionally, a short narrative on impact of the improvement and progress that has already occurred is presented (Figure 11).

Recommendations for improved data flow require improved accessibility and usability of autonomous data, and clear processes for high level data products from autonomous and high frequency data (Figure 11). There is an urgency to restore nutrient load monitoring to enable clear pathways for a program of measures. Alignment improvements focus on the harmonization of all aspects of the different eutrophication monitoring and assessment frameworks, achieving true integration across the catchment to coast encompassing the full riverine influenced area that links coastal and offshore processes. A range of indicator improvements are presented, including more information on factors that influence susceptibility to eutrophication, refining understanding of natural variability in development of thresholds and consideration of social, economic and environmental benefits. All the indicator improvements could lead to a fully integrated assessment, aligned with the Safe Space theory of a range of conditions that support optimum benefits. Improving our understanding of the ecosystem benefits requires consideration of eutrophication as a plankton pressure, linking nutrient imbalances and impacted water quality with changes in plankton functionality. Finally, climate change and eutrophication are intrinsically linked, with the potential to exacerbate impacts, knowledge of these interactions is critical in a changing climate world.

4 Discussion

4.1 The shifting baseline of eutrophication

Etymologically, eutrophication refers to good nourishment and what was thought to be a natural process in postglacial lake evolution. However, the meaning has changed, as shown by European Court of Justice judgements (ECJ, 2004, 2009), relating to breaches of the UWWTD, that describe it as a process caused by human actions that lead to undesirable disturbance to the balance of organisms and the quality of water. Undesirable disturbance was defined as “a perturbation of a marine ecosystem that appreciably degrades the health” (Tett et al., 2013) or “threatens the sustainable human use of that ecosystem” (Tett et al., 2007, 2013; Ferreira et al., 2011). From the perspective of a natural capital approach, the term “undesirable” is a link (1) to specific concerns about the impact of anthropogenic nutrients on the ecosystem services provided by coastal seas, and (2) to the perturbations of planetary N and P fluxes that are amongst the pressures pushing Earth’s biosphere outside a “safe operating space” for humanity (Rockström et al., 2009; Richardson et al., 2023).

	H High	M Medium	L Low
Effort required to achieve recommendation	Requires multi-agency input and stakeholder support. Requires agreement across agencies/directive with high risk of stakeholder conflict.	Still reliant on multi-agency input but agreement on harmonisation. Requires agreement across agencies/directive with medium risk of stakeholder conflict and majority of stakeholders informed of work.	Agreed positioning from all agencies, work already started. Stakeholders informed and already participating.
Scale of complexity required to achieve recommendation	Complex, dynamic systems requiring multi-layered data analysis. High levels of data governance, data agreements.	Complex and variable ecological systems but analysis already occurring, experience with indicators and data governance agreed.	Less complex more stable system, with minimal data processing and analysis required.
Potential costs of implementation with achieving recommendation	High, ongoing costs and significant staff time, and will require substantial field time/vessel costs.	Medium, ongoing costs and significant staff time, and may require field time/ vessel costs.	Low ongoing costs and staff time focused on desk-based work with existing data. No field time/vessel costs.

FIGURE 10
Qualitative ranking of different factors for measuring difficulty, complexity and cost of each recommendation for future eutrophication monitoring.

Eutrophication is indeed a wicked problem (Thornton et al., 2013), impacting different ecosystem services and different sectors of society in different ways. For examples, changes in nutrient levels in the southern North Sea have probably contributed to changes in flatfish catches (a provisioning service) and waterbird numbers (a cultural service to birdwatchers) (Engelhard et al., 2011; Philippart et al., 2007; van Roomen et al., 2012). Use of the sea's regulating services for nutrient cycling reduces costs for householders and fish-farmers who might otherwise have to pay for removing nutrients from farm or domestic effluent (Martino et al., 2019). However, overuse of these regulating services runs the risk of harming provisioning and cultural services as a result of e.g. increasing frequency of HABs, shifts in marine food-webs that might be detrimental to commercial fisheries, and more decaying green algae on beaches.

Nutrient pollution is frequently related to diffuse sources, complicating identification and mitigation of the impacts of single sources. Le Moal et al. (2019) acknowledge these difficulties and urge a different approach to diffuse and large ranging nutrient sources where we need to address: (i) the long term cumulative impact of far reaching anthropogenic activities, (ii) the consequences of multiple, and often cumulative, actions which can be very distant both in space and time, (iii) the difficulty in disentangling past and present causes from past anthropogenic legacy (O'Higgins et al., 2014). Our future programs need to consider what Thornton et al. (2013) calls "the wicked problem of eutrophication" where multiple, often cumulative actions, remote both in space and time from the pressure, enact variable responses and trajectories along the land-sea continuum. This is compounded by complex decision-making required from multiple stakeholders with different end-user requirements, limited interactions, and government agencies reporting to different policy levers with increasing costs and reduced funding.

4.2 Contextualizing eutrophication assessments in the UK

Advances in eutrophication monitoring and successful implementation of programmes of measures has been driven by decades of environmental protection offered under policies such as the EU Urban Wastewater Treatment Directive, Nitrates Directive, Water Framework Directive, EU Marine Strategy Framework Directive, the UK Marine Strategy, and the WFD/WER and the requirements to report under the OSPAR COMP and "and for the OSPAR "Inputs of nutrients to the marine environment" indicator. This process recognizes the achievements to date under national and international environmental directives and set out a path toward improving our approach to ensure continued development of our understanding of eutrophication and all its complexities. The shift from assessments under EU environmental directives to a nationally based marine strategy has been challenging but has also acted as a catalyst to rethink how we manage our environment in respect to eutrophication. Ecosystem management requires adaptation and modification, allowing the adaptation of the environmental legislation and management to change as our understanding and the rate and scale of impact changes.

4.3 Refining our future assessments

4.3.1 Improved data flows

We require new and novel ways of collecting data to help us to understand the highly variable processes that influence the scale and extent of eutrophication. Improving our understanding of eutrophication within the national assessments will be achieved through the better use of high frequency and autonomous data and improved technology, ensuring the optimisation of the large datasets. However, the complex process of ensuring sufficient calibration of sensor data and integrating the higher resolution

Theme	Headline	Evidence Gaps	Activities that could resolve evidence gaps	Feasibility			Impact and Progress
				Effort required	Scale of Complexity	Costs	
DATA	**Autonomous and remotely collected data**	No clear consensus on integrating autonomous and high frequency data alongside traditional monitoring to improve assessments.	Improve accessibility and usability of autonomous data, including the integration of data collected over minutes to days. Agreement on data products extracted from high frequency data. Ensure robust validation to allow inclusion in assessment.	H	H	M	**High impact** through technological advances. **Progress** rapid, driven by a stepwise increase in autonomous and remote data collection. High frequency data improving understanding of variability and biological processes. **Future work** will focus on higher frequency data collection in data-poor areas.
	Monitoring inputs from land to sea	Limited understanding of the impact of reduced monitoring frequency on data confidence and assessments.	Improve confidence in monitoring and assessment of land-to-sea nutrient inputs. Ensure sufficient site location and frequency to track pollution sources from land into coastal and marine waters.	M	M	H	**High impact** when used to prioritise program of measures for reduction of nutrients and improve modeling of land-to-sea inputs. **Progress** underway, particularly if existing monitoring can be built upon and improved.
ALIGNMENT	**Catchment to coast**	Assessment areas, thresholds are not harmonised across different directives.	Harmonise assessment areas, indicators and thresholds from catchment to the coast across different policy frameworks.	M	M	L	**Medium impact** through integrated monitoring and assessment frameworks. **Progress** underway given harmonisation has occurred in coastal to marine waters under OSPAR. Similar techniques could be applied for estuarine to coastal waters under WER/WFD.
	Riverine influenced areas	Limited understanding of the role of the riverine plume, and improving connections between inshore and offshore environments.	Improve monitoring and assessment of riverine influenced areas (plumes) at the boundary between inshore and offshore eutrophication assessments. Improved monitoring at small scale including combined sewage overflows and fish farms.	M	M	L	**Medium impact** through better tracking of pathways from source to sea and improved assessment areas. **Progress** ongoing, with substantive work already made with riverine-plume influence mapped for UK. Differences in resolution need to be corrected, and agreement on the overlap between coastal WER/WRD and plume waters.
INDICATORS	**Trends and trajectory of change**	Trend assessment not included in current eutrophication assessments.	Incorporate analysis of trends and trajectory of changes of status or state into eutrophication assessments.	M	L	L	**Medium to high impact** when used alongside existing assessment frameworks. **Progress** ongoing, through the embedding of trends into assessments, enabling evaluation of change trajectories in final reporting of state.
	Light attenuation	Not enough consideration in eutrophication assessments of susceptibility factors influencing the complex interactions between nutrients and primary/ secondary responses.	Incorporate new indicators or update current indicators based on measurements of optical components of the system. Potential indicators include light attenuation (K_d) and Chromophoric Dissolved Organic Matter (CDOM).	M	M	L	**High impact** when used to understand waterbody resilience to eutrophication impacts. **Progress** ongoing through the increased collection of CDOM, and improvements in sensors allowing high resolution data collection of light, SPM, chl-*a* and CDOM. **Future work** will focus on improving data analysis and linking it to biological responses.
	Natural variability and background	Natural variability not incorporated at indicator or area level.	Improve incorporation of natural variability and background conditions in indicator thresholds to ensure that the distinction between natural and anthropogenic is based on real environmental data for each indicator.	M	L	L	**Medium impact** in improving understanding of natural variability (by parameter and waterbody). **Progress** ongoing through calculation of new measures of natural variability for each indicator. **Future work** relies on agreement to change existing approaches in all assessments.
	Holistic approaches	Social and economic frameworks are not an integral part of future eutrophication assessments.	Develop assessment metrics that consider social and economic drivers alongside environmental. Processes such as Natural Capital, DPSIR and OneHealth are all examples of considering eutrophication as part of the ecosystem approach.	H	H	M	**High impact** to consider human and environmental impacts within one assessment. **Progress** ongoing under recent UK marine Natural Capital and Ecosystem Assessment Programmme, alongside well established DPSIR and One Health frameworks. **Future work** needs to consider how to broaden current assessments to measure society and economic gains without increasing costs.
	Integration of indicators	Confusion in the integration of indicators in different assessments can impact on overall messaging.	Implement a Safe Space assessment approach. Improved integration of all indicators to identify a simple and communicable assessment that represents the complexity of the system.	M	H	L	**High impact** in an integrated understanding of cumulative pressures and impacts. **Progress** ongoing with the development of a Safe Space Approach for plankton, building new methodology based on current assessments.
ECOSYSTEM	**Eutrophication as a plankton pressure**	Disconnect between eutrophication and biodiversity impacts.	Improve understanding of links between eutrophication and plankton community health.	M	H	M	**Medium impact** through improved understanding of how nutrients are impacting plankton. **Progress** ongoing, with UKMS and OSPAR linking eutrophication and biodiversity indicators.
	Nutrient imbalances	Current assessments do not consider the role of limiting nutrients impacting plankton community.	Consider linkages between nutrient ratio imbalances and pelagic (plankton) community change indicators within eutrophication assessment. Report N:P ratios in nutrient loads, transitional and coastal waters and offshore alongside pelagic community and food web indicators.	M	H	M	**High impact** as one of key pressures impacting on coastal plankton and foodwebs. **Progress** ongoing across UK government agencies measuring pico-, nano-, phyto- and zooplankton. Link to data recommendations for collection of data in data poor areas (winter periods) to improve N and P data.
CLIMATE	**Climate resilience**	Current eutrophication assessments are limited in responding to changing climate baseline.	Implement recommendations from Greenwood et al., 2024 on the updating of current eutrophication indicators in respect of a changing climate baseline.	L	M	L	**High impact** through improved understanding of climate and eutrophication. **Progress** underway with work by the Clean Safe Sea Evidence Group, which identifies the criteria that will need future proofing for climate impacts.

FIGURE 11
Recommendations for future eutrophication monitoring and assessment in UK marine waters categorized into themes of data, alignment, indicators, ecosystem and climate.

data into data workflows and repositories means that all available data types have not always been available for use in assessment. Ongoing investment into high-quality calibration of autonomous data will be required to allow the transition to a next-generation assessment utilizing available data alongside high-resolution data (Bean et al., 2017; Daniel et al., 2020).

4.3.2 Improved alignment

Disaggregation between the natural gradient of coast to sea does not achieve harmonized assessments. A new integrated approach will help us reveal a fuller picture of the health of our seas and oceans bringing ecosystem, economics and social criteria together to encapsulate the full value of a functioning ecosystem. Future assessments should continue to assess the continuum between estuaries and coastal systems, highlighting the importance of the riverine influenced area that lies between the UK transitional and coastal waters and further offshore assessment areas (Greenwood et al., 2019; Fronkova et al., 2022; Heal et al., 2023). Developing a catchment to coast approach should always encompass clear understanding of anthropogenic change, the technical ability to link that change to an activity or sector, transparent pathways for positive management action, recognition and incorporation of divergent societal concerns and policy tools that support sustainable outcomes for the marine environment (Devlin and Wenger, 2024). Alignment of governance strategies, integrated monitoring programs, more regular cross-agency engagement and harmonized assessments would be a first step in achieving a co-ordinated catchment to coast program.

4.3.3 Improved indicators

4.3.3.1 Trend assessments

Using long term cross-sectoral monitoring data will improve applicability for assessing changes in landscape-level environmental conditions and improve our ability to support short- and long-term management decisions (Maas-Hebner et al., 2015). The presentation of long-term data for nutrient inputs can be valuable in understanding what is changing over time, linking program of measures to reducing nutrient inputs, or conversely, showing the lack of appropriate program of measures resulting in an increase in one or more of the nutrients being measured.

4.3.3.2 Understanding "true" natural variability

Whereas our current suite of environmental indicators (including OSPAR PH1) seems reasonably complete and widely applicable in UK coastal waters, reference conditions and threshold values may need to be set locally and reflect both the local environment and local preferences for desirable conditions.

4.3.3.3 Susceptibility to eutrophication

Understanding of susceptibility and resilience needs further work, and consideration of these complexities should be considered in the development of new indicators or supporting evidence alongside the more traditional indicators.

4.3.3.4 Indicators of social, economic and environmental wellbeing

Assessment of eutrophication needs a social as well as an ecological context. Cost-benefit analysis involving ecosystem services and natural capital could assist in this but will need better models for the response of services to changes in nutrient loading, and for the preferences of different communities (or stakeholder categories) for these services. More integrated assessments such as the One Health approach, the DPSIR approach and natural capital accounting (Ross and Carter, 2013; Cork et al., 2016; Stentiford et al., 2020) explore the multiple interconnections that exist between environmental, animal and human health as a technique to incorporate multiple benefits to multiple end-users. Processes or programmes that could deliver greater social cohesion in our understanding of eutrophication need to measure environmental success through alternative indicators, such as ones that measure the protection of assets within marine natural capital, and holistic approaches that consider health between the environment and humanity as intrinsically linked in terms of water quality improvements. This will enable a safe space approach that is essential to ensuring sustainable and resilient human and environmental health.

4.3.4 Improved understanding of ecosystem impacts

Improvements in our understanding of the complex connections between nutrients and the pelagic community allow us to see how increases and imbalances in nutrients impact trophic interactions and the food web. Incorporating information on pelagic habitats will expand our eutrophication assessments from the more commonly used indicators of nutrients, chlorophyll and dissolved oxygen toward better estimates of plankton community shifts resulting from nutrient imbalances, and related impacts on ecosystem functioning. Further harmonization of data sources and assessment methods between the eutrophication and pelagic and food web assessments is expected to increase comparability and improve understanding of how nutrient loads affect ecosystem functioning and at which nutrient concentrations the ecosystem can be considered healthy (OSPAR, 2023).

4.3.5 Improved understanding of climate change and eutrophication

Climate change and eutrophication are intrinsically linked. While much is known about each individually, it is crucial to understand how they are interconnected and how climate change exacerbates marine pollution. Climate change leads, among other impacts, to floods and droughts which cause stronger variations in nutrient inputs. Future assessments need to consider the ecological effects of this greater variability and how to adequately monitor and assess this shifting baseline for the purposes of future eutrophication assessments (OSPAR, 2023).

5 Conclusion

We can improve our eutrophication assessments, despite the complexity of the drivers and impacts, and the uncertainty related to mitigation and recovery processes. Although this account has focussed on eutrophication in UK estuaries and shelf waters, much of our argument is more widely applicable. Solutions are possible, though almost never simple, and rely on a combination of long-term strategies, improved sewage and groundwater infrastructure, best management practices around agriculture and aquaculture, detailed monitoring and assessment and close partnerships between all stakeholders, public users, and government. In the UK case, improving stakeholder engagement with river trusts, conservation groups, farmers, water operators and communities needs to continue to be an integral part of the eutrophication assessment.

To enable government to continue to implement these strategies, and successfully measure their progress, UK agencies must continue to work collaboratively to innovate and improve the efficiency of monitoring programmes. This review provides an opportunity to look across national obligations and disparate approaches, identifying key evidence gaps when coastal and marine waters are continually impacted by nutrients alongside other pressures. The historical context of UK approaches to the monitoring and management of eutrophication is important when considering both the science used to inform the design of the current assessments and what science may be needed in the future to ensure robust, evidence-based decision-making. This review provides that historical context, adds the outcomes of expert discussion sessions, and identifies knowledge gaps including those involving human wellbeing and natural capital and, in doing so, sets out a framework to evolve marine eutrophication assessment and management into the future. Bringing together potentially conflicting voices to work collaboratively is the best way to progress and evolve eutrophication assessments. On the larger scale, we need to assess UK marine discharges not only for their contribution to the much-disturbed planetary N and P cycles, but also for their impact on the functioning of marine ecosystems considered as part of the biosphere.

Data availability statement

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

Author contributions

MD: Writing – original draft, Writing – review & editing. CG: Writing – original draft, Writing – review & editing. NG: Writing – original draft, Writing – review & editing. MA: Writing – original draft, Writing – review & editing. EB: Writing – original draft, Writing – review & editing. MH: Writing – original draft, Writing – review & editing. AM-G: Writing – original draft, Writing – review & editing. PT: Writing – original draft, Writing – review & editing. DT: Writing – original draft, Writing – review & editing. MB: Writing – original draft, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

The author(s) declared that they were an editorial board member of Frontiers, at the time of submission. This had no impact on the peer review process and the final decision.

Generative AI statement

The author(s) declare that no Gen AI was used in the creation of this manuscript.

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