AtkinsRéalis

Final Report

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A REVIEW OF THE ENVIRONMENTAL STANDARDS FOR AQUATIC ECOLOGICAL IMPACTS FROM NUTRIENT POLLUTION IN ENGLAND AND NORTHERN IRELAND

Notice

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Contents

Glossary

BOD	Biological oxygen demand
BQE	Biological quality element
CBB	Cyanobacterial biomass
CEFAS	Centre for Environment, Fisheries and Aquaculture Science
CIS	Common implementation strategy (EU guidance on eutrophication assessment)
DOC	Dissolved organic carbon
DIN	Dissolved inorganic nitrogen
DIP	Dissolved inorganic phosphorus
eDNA	Environmental DNA
EQR	Environmental Quality Ratio
EQS	Environmental Quality Standard
EU	European Union
G/M	The Good-Moderate boundary
GES	Good ecological status
H/G	The High- Good boundary
JRC	Joint Research Centre
LDTI	Lake trophic diatom index
MS	Member State
N	Nitrogen
NI	Northern Ireland
NVZs	Nitrate vulnerable zones
OLP	Ordinary least squared
OSPAR	The Convention for the Protection of the Marine Environment of the North-East Atlantic
P	Phosphorus
PICO	Problem; Intervention; Comparison; Outcome (Evidence review methodological approach)
PNEC	Predicted no effect concentration
r	The correlation between the predictor variable, x, and the response variable, y (correlation coefficient)
r^2	The proportion of the variance in the response variable that can be explained by the predictor variable in the regression model
rs	Spearman Rank correlation coefficients
RQ	Research question
RoI	Republic of Ireland
RP	Reactive phosphorus

SRP	Soluble reactive phosphorus
TDI	UK Trophic Diatom Index
TMDL	Total maximum daily loads
TN	Total nitrogen
TP	Total phosphorus
TraC	Transitional and coastal waters
Type R-C1	Low alkalinity lowland rivers
UKTAG	United Kingdom Technical Advisory Group
WFD	Water Framework Directive

Executive Summary

Background

Nutrient pollution has been identified as one of the main reasons why surface water bodies are failing to achieve Good Ecological Status (GES)/Potential in England and Northern Ireland (NI). The 2017 Water Environment Regulations aims to restore 77% of water bodies to GES (England) and 70% to Good Potential (NI), however, to achieve this, the underpinning legislation must be effective. The Water Framework Directive (WFD) Regulations set the legal limits for the acceptable concentration of pollutants in water bodies using Environmental Quality Standards (EQSs). Compared to other EU countries, the nutrient EQSs used in England and NI tend to be stringent, however they do not always cover the full range of nutrient fractions employed across the EU.

Since nutrient EQSs were last revised, pressures on the water environment have increased due to population growth, climate change and increased agricultural productivity. At the same time there has been ongoing monitoring and research that could provide further evidence to inform future EQS development. In light of this, this review aims to provide an initial assessment of whether the existing nutrient EQSs underpinning the WFD Regulations are fit-for-purpose, i.e., are they sufficient to protect ecological health and function, and do they reflect the latest available evidence.

Methodology

A systematic review of relevant literature was undertaken to establish an evidence base, which was used to explore two main Research Questions (RQs). The first RQ aimed to identify the current state of play in England and NI regarding nutrient EQSs for surface water bodies and the second RQ aimed to determine if the current nutrient EQSs are protective of ecological health and function in the surface water bodies of England and NI. Literature included academic and grey literature (including the documents which were produced by the UK Targets Advisory Group (UK TAG) when each of the current nutrient standards were derived), as well as documents provided by relevant organisations and public authorities.

Interviews with personnel who have knowledge/experience pertinent to this study were undertaken. The choice of interviewees was informed by the literature review process and input from the OEP. Interviewees were asked a series of questions relating to whether they felt the current nutrient EQSs are fit-for-purpose, cover the correct nutrient fractions, and were derived using appropriate methodologies. Following the completion of all nine interviews a virtual workshop was arranged; this facilitated the sharing of views and discussion on topics pertinent to the project. The workshop was attended by representatives from regulatory and non-departmental bodies in Northern Ireland, Wales, Scotland and England as well as members of UK and international research institutions. The workshop focussed on three key themes: (1) statistics used to set EQSs, (2) nutrients of focus, and (3) methodological advancements.

Key Findings

In general, the nutrient EQSs employed in lakes are considered largely fit-for-purpose, however the general consensus was that certain riverine standards require improvement. Information obtained from this review relating specifically to transitional and coastal (TraC) waters was limited and further research is required to determine if the current standards are fit-for-purpose for these environments.

When the river P standards were being developed (2012) it was acknowledged that the relationships used to determine the thresholds were fairly weak and evidence suggests that the current values may not be stringent enough to protect ecological health. Since the current thresholds were developed, an updated reference model for diatoms has been developed and the Best Practice Guidance (EU) for determining thresholds has been updated, both of which could be employed if the threshold values were to be recalculated.

England and NI currently have nitrogen (N) and phosphorus (P) standards for lakes but only use one nutrient EQS in rivers and transitional and coastal (TraC) water bodies which use P and N respectively. However, evidence strongly suggests that having standards for both N and P is important for all surface water bodies and would likely promote more significant ecological improvements than tweaking current threshold values. However, developing new (additional) standards may also result in greater political and financial ramifications compared to updating the current standards.

Recommendations

Whilst the current standards adopted good practice at the time of development, there is now more monitoring data available and updated approaches to developing standards that better relate to ecological response. Recommendations include updating the current thresholds using the most recent data and following the methods described in the updated Best Practice Guide (e.g., using appropriate statistical approaches to utilise pressure-response relationships and quantifying uncertainty), as well as using the updated diatom reference model for river P thresholds.

Furthermore, it is recommended that EQSs should be developed for additional nutrient fractions, including Total P and Total N for rivers and soluble reactive P or Total P for TraC. Further approaches that also warrant consideration include exploring the statistical viability of including additional site/ type-specific parameters when calculating threshold values, as well as exploring the potential of using the seasonal mean (if sampling is increased) and/or alternative biological metrics. Lastly, it is recommended that additional data (existing or new) should be obtained to fill gaps in current data sets, and additional monitoring techniques should be explored which may provide the opportunity to develop more effective and ecologically relevant thresholds in the future.

1.1 Background

Freshwater and marine ecosystems are increasingly at risk from multiple anthropogenic stressors (Albini et al., 2023). Nutrient pollution has been identified in the River Basin Management Plans (RBMPs) as one of the main reasons why surface water bodies are failing to achieve Good Ecological Status (GES)/ Potential in England and Northern Ireland (NI). Specific nutrient targets have therefore been developed to drive improvement in water quality. The Water Environment (Water Framework Directive) (England and Wales) Regulations 2017 and the Water Environment (Water Framework Directive) Regulations (NI) 2017 (henceforth referred to collectively as the 'WFD Regulations') aim to restore 77% of water bodies to GES (England) (The OEP 2024a), which align with the Government's plan for clean and plentiful water (GOV.UK 2023). Similarly in NI, the NI Executive has set a working target to bring 70% of water bodies to 'Good Status' (The OEP, 2024b).

To improve water quality and meet these ambitious targets, the underpinning legislation must be effective. The WFD Regulations set the legal Environmental Quality Standards (EQSs) for pollutant concentrations within water bodies in England and Northern Ireland. Since nutrient EQSs were last revised, pressures on the water environment have increased due to population growth, climate change and increased agricultural productivity. At the same time, there is an increased public engagement (and expectation) relating to this issue. However, there has also been ongoing monitoring and research since the latest nutrient EQSs were determined/revised that could provide further evidence to inform future EQS development. In light of this, the Office for Environmental Protection (OEP) commissioned this review to provide an initial assessment of whether the existing nutrient EQSs underpinning the WFD Regulations are fit-for-purpose i.e., are they sufficient to protect ecological health and function, and do they reflect the latest available evidence.

1.2 Aim

The principal aim of this review was to improve the OEP's understanding of existing nutrient EQSs underpinning the WFD Regulations and assess whether they are fit-for-purpose. A systematic review of relevant literature was undertaken to establish an evidence base, which was used to explore two main Research Questions (RQs).

The first research question was to identify the current state of play in England and NI regarding nutrient EQSs for surface water bodies, addressing several specific points:

- What are the current EQSs for nutrients in England and NI?
- How were the current EQSs derived? (i.e., what methods were used to determine EQSs?)
- How does this differ from other countries within the same climate zone?

The second research question was to address if the current nutrient EQSs are protective of ecological health and function in the surface water bodies of England and NI, specifically addressing the following:

- Does the evidence suggest that current standards (and inherent simplifications/assumptions), including the statistic adopted, are protective of ecological health and function?

- Does the evidence suggest that alternative approaches (for example, use of alternative statistics or the inclusion of alternative/ additional nutrient fractions) will provide a closer link to ecological response?

1.3 Structure of the report

This report is set out as follows:

- **Chapter 2 –** Methodology, covering the Rapid Evidence Assessment completed, and the interviews and the workshop held as part of evidence gathering from specialists.
- **Chapter 3 –** Sufficient background on the WFD regulations is provided for the reader to understand the role of nutrients in Ecological Status of a surface waterbody. Details associated with the existing EQSs for nutrients (in England and NI) pertinent to this study are summarised and 'current' status for nutrients in the surface water bodies of England and NI is presented.
- **Chapter 4 –** Findings and discussion.
- **Chapter 5 –** Conclusions.
- **Chapter 6 –** Recommendations.

2. Methodology

2.1 Rapid evidence assessment

A Rapid Evidence Assessment was undertaken following the approach described below.

2.1.1 Structured search

A structured search of the academic literature was conducted using the search engine Web of Science based on the methodology described by Collins et al. (2015). A search string was designed using an adapted Population, Intervention, Comparator, and Outcomes (PICO) framework (James et al., 2014). Terminology within each category, as presented in Table 2.1, were linked by the Boolean command 'OR' and each category was linked by the command 'AND' to form the search string. This ensured that at least one of the terms from each category was included in the title, abstract or key words of the articles identified when using the search string.

Further refinements were made to help ensure the most relevant literature was identified for review. This included:

- The addition of a 'NOT' category within the search string to remove articles that mention China, America, 'United States of America' and the USA (in the title, abstract or keywords).
- Language was limited to English.
- The date was limited to the last 21 years (2005 to 2025).
- Articles were also filtered by the 'Web of Science categories' (Environmental Sciences, Water resources, Marine freshwater biology, Geoscience multidisciplinary, Water resources, Ecology, Geography physical, Fisheries, Biodiversity conservation, Toxicology, Multidisciplinary sciences, Limnology, Geography, Chemistry multidisciplinary).
- Filter by countries within the same Koppen-Geiger climate zone as the UK: France, Belgium, Netherlands, Germany, Denmark, New Zealand, Switzerland (lowland), Austria (small parts), Czech Republic (small parts), Spain (North), Poland (west), Sweden (southern Coast), Norway (southern coast), Australia (southern regions) as defined by the interactive Koppen-Geiger map[1].

After running the search string in Web of Science, the top 150 articles (ordered by relevance) were filtered for relevance following a two-stage process. The relevance of each article was initially reviewed based on the title, and designated a category of 'relevant', 'somewhat relevant', or 'not relevant'. Following this, the second stage of the filtering process reviewed the abstracts of those articles deemed 'relevant' or 'somewhat relevant' at the title filtering stage into 'relevant' or 'not relevant'. The assessment of relevance was checked by an independent checker. The spreadsheets containing the full article list downloaded from the Web of Science, and the filtering is provided in Appendix A. Relevance was determined based on two key criteria:

- Geographical/ climatic relevance: studies in areas that are not geographically/ climatically comparable with the UK (i.e., do not have the same Köppen-Geiger climate classification - Cfb: warm temperate, fully humid, warm summer) were not taken forward for review.
- Relevance of content to the research questions addressed in this study.

[1] https://koppen.earth/ (Accessed 04/06/2025)

Table 2.1. Terminology categories used in the search string

Population: surface water bodies	Intervention/ exposure: nutrient pollution	Comparator	Outcome	Geographical context
Water quality	Nutrient*	Environmental quality standard*	Sufficient*	England
Aquatic ecology	Phos*	EQS*	Insufficient*	Ireland
Aquatic health	P	Water framework directive	Adequate*	NI
Ecosystem health	TP	WFD	Inadequate*	Wales
Ecological health	SRP	Limit*	Effective*	Scotland
Ecological function	Orthophos*	Standard*	Ineffective*	United Kingdom
Ecological status	Nitr*	Threshold*	Fit for purpose	UK
River*	TON	Natura	Limit*	
Freshwater*	TN	Site* of special scientific interest	Efficien*	Britain
Lake*	Ammoni*	SSSI*	Implement*	GB
Coast*	Sediment*	SAC*	Status	Europ*
TRaC	Suspended solid*	Special area* of conservation	Achieve*	EU
Transitional	SS	SPA*	Potential*	
Brackish	Turbidity	Special protection area*	Pass	
Headwater*	Macronutrient*		Fail	
Estuar*	Macro-nutrient*		Compl*	
Pond*	Eutroph*		Favourable condition*	
Stream*				
Channel*				
Loch*				
Lough*				
Macroinvert*				
Fish				
Diatom*				
Macrophyte*				

** Acts as a wildcard, and will pick up variations on the key word, for example River* will also pick up Rivers and Riverine.*

2.1.2 Unstructured searches

Unstructured searches were also conducted in Google Scholar in order to provide a sense check that the most relevant literature was obtained in the structured review, also providing an opportunity to identify any relevant academic articles that may have been missed (e.g., where the search terminology occur in the main body of the article, but not in the title, abstract or keywords). Searches were also conducted for relevant grey literature using Google. Literature was also requested from relevant organisations and public authorities and reviewed for relevance.

2.1.3 Article review

All articles from the structured search deemed ‘relevant’ at the abstract filtering stage were reviewed, along with selected articles from the unstructured searches completed and certain articles provided by interviewees or workshop participants (see Section 2.2). Literature was also obtained from the OEP, contacts of the OEP and the wider project team. Additional papers from the top 1000 most relevant articles (identified through the Web of Science search; Section 2.1.1) that were deemed highly relevant were also reviewed in full. In some cases, literature cited (and of potential relevance) in the articles reviewed (and so on) were also reviewed and selected relevant information extracted.

Selected information deemed relevant to the research questions addressed in this study was extracted and presented within this document. From the literature reviewed, a ‘core’ evidence base of 40 articles from the academic literature relevant to the second of the research questions (see Section 1.2) was established. Selected information from these articles has been summarised in a database (6.Appendix B).

2.2 Evidence gathering from specialists

2.2.1 Interviews

Interviews with personnel who have knowledge/ experience pertinent to this study were undertaken. The choice of interviewees was informed by the literature review process and input from the OEP, and interviews were held online using Teams:

- Prof. Mike Bowes (UK Centre for Ecology and Hydrology)
- Prof. Pippa Chapman (University of Leeds)
- Prof. Penny Johnes (University of Bristol)
- Prof. Martyn Kelly (Bowburn Consultancy)
- Wendy McKinley (The OEP College of Experts)
- Prof. Rupert Perkins (Cardiff University)
- Prof. Geoff Phillips (Stirling University)
- Dr. Marc Stutter (James Hutton Institute)
- Dr. Savannah Worne (Loughborough University)

The interviews were semi-structured in that a list of core questions (below) were derived and formed the basis of the interviews (having been submitted to interviewees in advance). These covered topics pertinent to this review around which conversations flowed. Often, discussions spanned across the topics included in multiple questions. Additional questions, to those included in the list (below), may also have been asked, for example relating to the interviewee’s specific expertise. Notes from each interview are included in Appendix C.

Core Questions

Question number	Research question
1	Do you think the current nutrient EQSs applied under the WFD are fit-for-purpose, i.e., are they protective of ecological health? ▪ If no, can you expand upon why you believe this?

2	Do the current EQSs applied under the WFD cover the correct nutrients and the correct nutrient fractions to reflect ecological responses? ▪ If not, what nutrients/ nutrient fractions do you feel should be included and why?
3	Is the methodological approach used to derive current nutrient EQSs applied under the WFD appropriate to drive ecological improvements? ▪ If not, what alternative approach would you be in favour of and why?
4	Have there been methodological advancements/ improvements in scientific understanding in recent years which you feel should be used to inform updated nutrient EQSs which provide a closer link to ecological responses?
5	Are there any examples of best practice from elsewhere in Europe, or further afield, which the UK should be adopting in respect of assessing nutrient risk to ecological health? ▪ Which aspects of these approaches are better than the current UK approach in your view? Are these approaches delivering better outcomes, i.e., improved ecological health?

2.2.2 Workshop

Following the completion of all nine interviews, a virtual Workshop was arranged; its purpose was to facilitate sharing of views and discussion on topics pertinent to the project. Interviewees were provided with the option to attend, whilst representatives from regulatory and non-departmental bodies in Northern Ireland, Wales, Scotland and England as well as members of UK and international research institutions attended the Workshop.

The Workshop contained three focus areas: (1) Statistics used to set EQSs, (2) Nutrients of focus, and (3) Methodological advancements. Using Mentimeter responses to targeted questions relating to each focus were captured, whilst there was also an open discussion at the end of each focus area.

For persons who were invited to the Workshop but did not attend, the opportunity was provided for views to be shared by completing a 'standalone' survey (again via Mentimeter). The survey included the presentation from the Workshop, which included the questions which were asked and the opportunity to submit responses to these. Similarly, the survey was also provided to those who needed to start/ leave part-way through the Workshop, giving the opportunity to share views on the full complement of questions.

Meeting notes from the Workshop, which capture all responses received (i.e., via workshop meeting and survey), are included in Appendix C.

2.3 Research boundaries and limitations

A limitation of using a search string is that potentially relevant literature may not be picked up given that this is strictly defined. However, additional unstructured searches were completed to mitigate this.

Findings included within this report are limited to selected information that was deemed relevant to the research questions being addressed. Owing to the breadth of information available and additional complexities to be considered decisions needed to be made based on professional judgement as to the information reported on.

A comprehensive assessment of nutrient standards / methodologies in other countries within the same climate zone as England and NI was beyond the scope of this review, though select information on the nutrient standards/ methodologies used in other countries was extracted from the literature reviewed and included in this report. To complete a comprehensive assessment, targeted searches for literature associated with nutrient standards in other countries would be required.

Also outside the scope of this report are the procedures associated with protected sites. Protected sites include, for example, Sites of Special Scientific Interest (SSSIs) (England), areas of special scientific interest (ASSI) (NI), Special Areas of Conservation (under the Habitats and Birds Directives) and the standards and procedures for protected areas are set out in the legislation establishing the protected areas. The Habitats Directives, for example, has differing objectives and approaches to the WFD, such that, no straightforward read-across is possible between WFD status and Habitats Directive condition (JNCC, 2015). Thus, to appropriately assess standards and procedures for protected areas a separate review is required.

3. The Water Framework Directive Regulations

3.1 Background

The WFD Regulations were developed to 'transpose' (to put into domestic law so as to give effect to) the EU Water Framework Directive (WFD). The WFD at EU level (Directive 2000/60/EC)[2] and parts of the EQS daughter directive (Directive 2008/105/EC)[3], are transposed into English and Northern Irish law by the Water Environment (WFD) (England and Wales) Regulations (2017)[4] and The Water Environment (WFD) Regulations (NI) (2017)[5]. EU Directive 2013/39/EU[6] amended the EQS Directive concerning EQS values and those changes were transposed in England through the Water Environment (Water Framework Directive) (England and Wales) (Amendment) Directions 2015[7] and in Norther Ireland through the Water Framework Directive (Classification, Priority Substances and Shellfish Waters) 2015[8].

Following the UK's exit from the EU, the WFD Regulations acquired the status of 'retained EU law' under the European Union (Withdrawal) Act 2018[9], however government has the power to modify these regulations (until June 2026). The Water (Amendment) (NI) (EU Exit) Regulations 2019 ensures that the Water Framework Directive (as transposed in 2017) continues to operate in NI following the departure of the UK from the EU. At the time of writing, there is general alignment between the EU EQS values for nutrients and those included in the WFD Regulations for England and Wales and NI.

In this Chapter, sufficient background on the WFD regulations is provided for the reader to understand the role of nutrients in Ecological Status of a surface waterbody. Details associated with the existing EQSs for nutrients (in England and NI) pertinent to this study are summarised and 'current' status for nutrients in the surface water bodies of England and NI is presented.

3.2 Water body classification

The Environment Agency in England and the Northern Ireland Environment Agency are the main bodies responsible for implementing the WFD Regulations. Among their specified functions, the

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[2] Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy [2000] OJ L 327/1. Available at: https://eur-lex.europa.eu/resource.html?uri=cellar:5c835afb-2ec6-4577-bdf8-756d3d694eeb.0004.02/DOC_1&format=PDF

[3] DIRECTIVE 2008/105/EC of the European parliament and of the council. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:02008L0105-20130913&from=EN

[4] The Water Environment (Water Framework Directive) (England and Wales) Regulations 2017 (legislation.gov.uk). Available at: https://www.legislation.gov.uk/uksi/2017/407/made

[5] The Water Environment (Water Framework Directive) Regulations (NI) 2017 (legislation.gov.uk). Available at: https://www.legislation.gov.uk/nisr/2017/81/contents/made

[6] Directive 2013/39/EU of the European Parliament and of the Council of 12 August 2013 amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32013L0039

[7] The Water Environment (Water Framework Directive) (England and Wales) (Amendment) Regulations 2015 (legislation.gov.uk). Available at: https://www.legislation.gov.uk/uksi/2015/1623/resources

[8] The Water Framework Directive (Classification, Priority Substances and Shellfish Waters) Regulations (NI) 2015 (legislation.gov.uk). Available at: https://www.legislation.gov.uk/nisr/2015/351/contents/made

[9] Ss.2-4, European Union (Withdrawal) Act 2018

relevant environment agency assesses the condition of water bodies, classifies their status and proposes objectives and programmes of measures.

The relevant environment agency must classify water bodies in accordance with the approach set out in the WFD[10]. For surface waters, the system, which is summarised in Figure 3.1[11], classifies each water body in terms of its ecological and chemical status based on tests for various parameters or 'elements'. Results for different quality elements are combined to form the overall ecological classification, ranging from 'High Ecological Status' (which means unaffected or virtually unaffected by human activity) to 'Bad Ecological Status' (meaning severely damaged) (Figure 3.1).

3.2.1 Ecological status classification

Ecological status classifications can be composed of up to four different assessments (Environment Agency, 2011) (Figure 3.1):

- An assessment of status indicated by a biological quality element (such as fish, macrophytes and phytobenthos, invertebrates or algae). The presence of invasive species is covered by a separate test.
- An assessment of compliance with environmental standards for supporting physico-chemical conditions, such as **phosphorus** or **nitrogen**, dependant on the category of surface water body, i.e., rivers, lakes or transitional and coastal (TraC) waters (see Section 0 for more detail).
- An assessment of compliance with environmental standards for concentrations of specific pollutants, which include **unionised ammonia (as nitrogen)**, dependant on the category of surface water body, i.e., fresh water or salt water (as for unionised ammonia, as nitrogen) (see Section 3.2.1.2 for more detail).
- In determining high status only – a series of tests to make sure that hydromorphology is largely undisturbed.

Ecological status is recorded as 'high', 'good', 'moderate', 'poor' or 'bad'. High represents 'largely undisturbed conditions', whilst other classes show increasing deviation from undisturbed or reference conditions. Deviation is required to be expressed as an ecological quality ratio (EQR) which ranges from one at the 'high status end' to zero at the 'bad status end' (Environment Agency, 2011).

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[10] Reg 6, WFD Regulations.
[11] Further technical detail can be found in Annex 4 of the OEP report 'A review of implementation of the water framework directive regulations and river basin management planning in England' (OEP, 2024a)

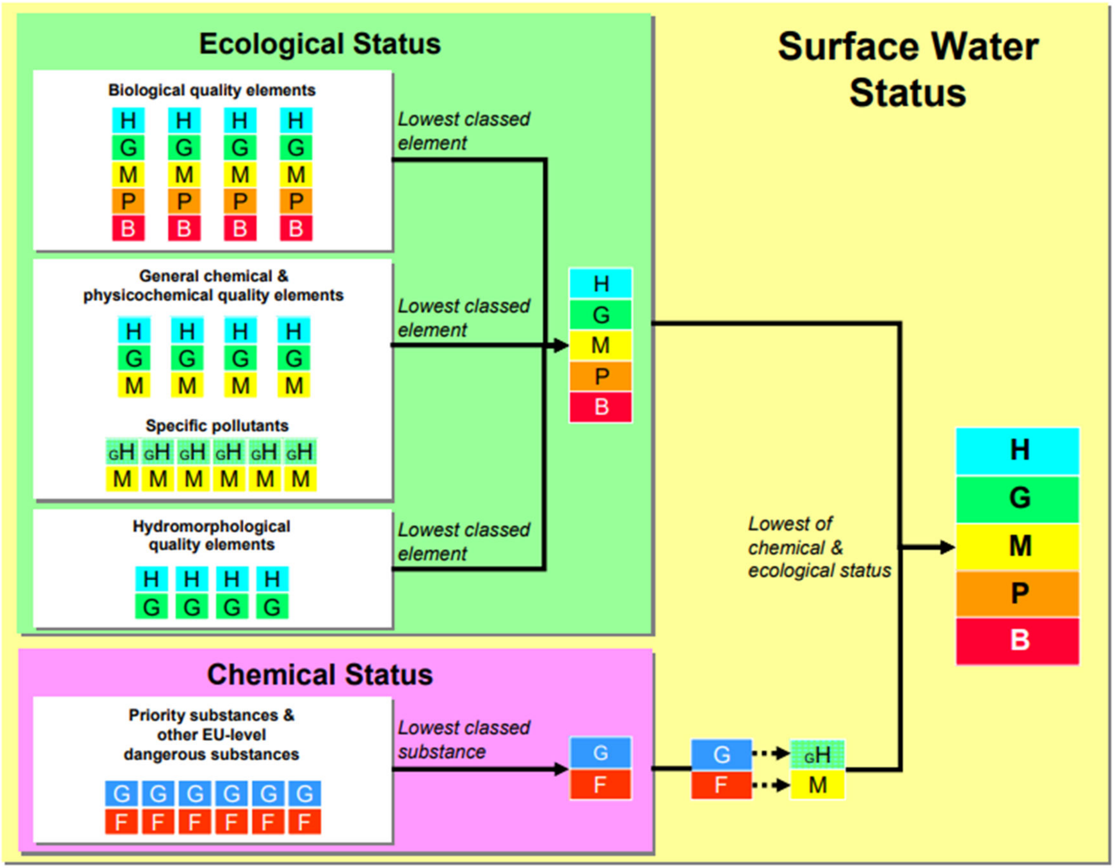

Figure 3.1. Classification of surface water bodies under the WFD Regulations (Source: UKTAG, 2007a). Note: 'H' means High; 'G' means Good; 'GH' means Good or better; 'M' means Moderate; 'P' means Poor; 'B' means Bad; and 'F' means Failing to achieve Good Surface Water Chemical Status.

3.2.1.1 Role of physico-chemical supporting quality elements in ecological status

Supporting elements are the physico-chemical factors, such as nutrients, which are required to support a functioning ecosystem. Class boundary values have been developed for these supporting elements which correspond to high, good, moderate, poor and bad status. Nevertheless, in classification supporting elements are only able to influence status down to moderate[12]; it is only biological elements that can determine poor or bad status (Environment Agency, 2011). The nutrients which are physico-chemical quality elements that are used in producing classifications for each water category are presented in Table 3.1.

Table 3.1. The nutrients which are physico-chemical quality elements used in producing classifications for each water category

Quality element	Rivers	Lakes	TraC
Ammonia (total as nitrogen)	✓	✓	×
Dissolved inorganic nitrogen	×	×	✓
Total nitrogen	×	✓	×
Phosphate (reactive phosphorus (unfiltered orthophosphate)	✓	×	×
Total phosphorus	×	✓	×

3.2.1.2 Role of specific pollutant sub-element in ecological status

Specific Pollutants, which include unionised ammonia (as nitrogen), are defined as substances that can have a harmful effect on biological quality. The UK was responsible for developing EQSs for Specific Pollutants, which include substances commonly encountered when issuing environmental permits/ consents and substances listed under earlier EU Directives (such as List II under the Dangerous Substances Directive). Specific Pollutant sub-element classification is based on individual Specific Pollutant elements using the one out all out process with Specific Pollutant elements classed as either ‘high’ or ‘moderate’. Thus, if a single Specific Pollutant element is moderate, i.e., concentration exceeds the EQS (mean and/ or 95th percentile), then the sub-element classification will also be moderate[13].

3.2.1.3 Heavily modified and artificial water bodies

For surface water bodies that are artificial (created by man where no water body previously existed) or heavily modified, the classification is based on ecological ‘potential’ rather than ‘status’. Recognising that these water bodies, given their nature, cannot necessarily be expected to offer or achieve the same conditions as other surface water bodies, they have different assessment approaches for biological and hydro-morphological quality elements (The OEP, 2024). Nevertheless, environmental standards for physico-chemical quality elements and for specific pollutants are used in

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[12] Though regulatory agencies require indicative boundaries below moderate status for management purposes (WFD UKTAG, 2019a).

[13] https://environment.data.gov.uk/dataset/82f053d7-7b6b-4b41-87eb-7d56077ed65a (05/03/2025)

classifying the ecological potential of heavily modified and artificial water bodies in the same way that they are used for classifying ecological status[14].

3.3 Existing water quality standards for nutrients

Nutrients which are physico-chemical quality elements (Section 3.2.1.1) or Specific Pollutants (Section 3.2.1.2) and are used in producing ecological status classifications (in England and Northern Ireland) are described in this section. In Table 3.2., details associated with the existing EQSs for these nutrients, pertinent to this study, are summarised. These include the statistic and 'type' of standard, as well as associated variables, and the methodology used to determine the threshold for the Good-Moderate (G/M) class boundary. Aspects which have been explored in our review of the standards (see Chapter 3.3.1).

Information associated with existing standards has principally been taken from reports published by the UK Technical Advisory Group (UKTAG) on the WFD. The UKTAG is a working group of experts drawn from UK environment agencies and conservation agencies and includes representatives from the Republic of Ireland (RoI). The UKTAG's role includes provision of technical advice on environmental standards for achieving WFD status and how they may be used for river basin planning (WFD UKTAG, 2014a). Unless otherwise specified, information presented in Table 3.2. is believed to apply to England and NI.

The current status for nutrients, as defined below, in the water bodies of England and NI are presented in Section 3.3.1:

- **Total Nitrogen (TN)** is the sum of all forms of soluble and particulate nitrogen present in a water sample and includes nitrate (NO_3^-), nitrite (NO_2^-), unionised ammonia (NH_3), ammonium (NH_4^+) and organic nitrogen.

- **Dissolved Inorganic Nitrogen (DIN)** is the sum of all forms of soluble inorganic nitrogen present in a water sample and includes nitrate (NO_3^-), nitrite (NO_2^-), unionised ammonia (NH_3) and ammonium (NH_4^+).

- **Total ammonia (total as Nitrogen)** is the sum of unionised ammonia (NH_3), which is the fraction toxic to fish and macro-invertebrates, and ammonium (NH_4^+).

- **Total phosphorus (TP)** is the sum of all forms of soluble and particulate phosphorus present in a water sample and includes orthophosphate ($H_2PO_4^-$, HPO_4^{2-}), dissolved organic phosphorus and phosphorus bound to particulate matter.

- **Reactive phosphorus (RP)** is the concentration of orthophosphate species ($H_2PO_4^-$, HPO_4^{2-}) and is determined using the molybdenum blue colourimetric method following settling instead of filtration and therefore includes loosely bound and available phosphate. For reference, soluble reactive phosphorus (SRP) is commonly obtained after filtration (0.45µm filter).

[14] Except where a quality element is so affected by the use and modified characteristics of that body as to make it inappropriate to do so.

Table 3.2. Existing Environment Quality Standards for nutrients in surface water (England and Northern Ireland, unless specified otherwise)

Nutrient	Surface Water Category	Standard Statistic	Type of standard	Variables	Biological indicator used to develop G/M thresholds	Methodology used to develop G/M thresholds	Range of threshold values for G/M	References
Reactive Phosphorus	Rivers	Annual mean	Site-specific	Alkalinity, Altitude	Lowest scoring of either macrophytes or phytobenthos Environmental Quality Ratio (EQR)	Regression analysis between reactive P and biological EQR.	28 – 98 µg/L[15]	WFD UKTAG (2012), WFD UKTAG (2013)
Total Ammonia (as Nitrogen)	Rivers (England and NI), Lakes[16] (England)	90th percentile	Site-specific	Alkalinity, Altitude	Macroinvertebrate communities	Developed on the basis of the relationship between macroinvertebrate communities and ammonia as N.	0.3 – 0.6 mg/L[17]	WFD UKTAG (2008a)
Total Phosphorus	Lakes	Annual mean	Site-specific	England: Altitude, Alkalinity and Depth. Northern Ireland: Altitude, Alkalinity, Depth and Humic substances (colour)	Biological elements such as the biomass of phytoplankton (as chlorophyll a), the taxonomic composition of macrophytes, and changes to diatoms preserved in sediments of lakes were considered.	A model using the Morpho Edaphic Index (Phillips and Pitt, 2016) was used to predict reference P values linked to catchment geology and topography (alkalinity and depth). Pragmatic expert judgements were used to determine the proportion of change for P for each threshold, and for each reference type.	**England**: 10 - 53 µg/L[18] **Northern Ireland**: 10 – 62 µg/L[19]	WFD UKTAG (2008b), WFD UKTAG (2016)
			Type-specific	Geological category, Depth, and for England only Geographical region			**England:** 8 – 49 µg/l[20] **Northern Ireland**: 8 – 31 µg/l[19]	
Total Nitrogen	Lakes	Annual mean	Type-specific	Depth, Humic substances (colour)	Phytoplankton EQR	Regression analysis based on the relationship of TN to phytoplankton (along with depth and humic type). The model produced a relatively strong correlation coefficient, with $R^2 = 0.747$.	0.74 - 1.46 mg/L[21]	WFD UKTAG (2019a), WFD UKTAG (2019b)
Dissolved Inorganic Nitrogen	TraC	England: Winter mean or 99th percentile[22] NI: Winter mean	Type-specific	England: Salinity, Turbidity (annual mean suspended particulates) Norther Ireland: Salinity	-	The deviation of the boundary for good and moderate is a general increase of 50% in the reference baseline.	**England**: 30 – 270 µM[23] (transitional) **England**: 18 – 270 µM[23] (coastal) **Northern Ireland**: 18 µM (TraC)	WFD UKTAG (2008b)
Unionised ammonia (as Nitrogen)	TraC (Northern Ireland) Salt water (England)	Annual mean	Single value (for salt water)	N/A	Fish	EQS was based on a combination of acute toxicity data to fish and threshold concentrations inferred from field data.	21 µg/L	WFD UKTAG (2007b)

[15] Calculated using the maximum and minimum values for altitude and alkalinity as described in the WFD (2015), covering alkalinities 2 - 250 mg/l $CaCO_3$, altitude 10 – 355 m and for an EQR of 0.532 (good biological threshold).
[16] No information was found in relation to the biological indicator / methodology used to develop G/M thresholds for Lakes.
[17] Calculated with alkalinities 10 - 100 mg/l $CaCO_3$ and altitudes 50 – 500 m.
[18] Calculated with alkalinities 10 - 100 mg/l $CaCO_3$, altitude 50 – 100 m, depth 2, 10, 50 m, and for surface and deep-water samples.
[19] Range of threshold values across all lake types as included in The Water Framework Directive (Classification, Priority Substances and Shellfish Waters) Regulations (Northern Ireland) 2015.
[20] Range of threshold values across all lake types as described by WFD UKTAG (2016).
[21] Range of threshold values across all lake types as described by WFD UKTAG (2019a).
[22] In England, the winter mean (1st November to 28th February) is used for clear waters and the 99th percentile is used for intermediately turbid, turbid or very turbid waters. In Northern Ireland the winter is defined as 1st December to 28th February.
[23] Standards for turbidites ranging from clear to very turbid.

3.3.1 Current status

The recent/ current nutrient status for water bodies in England in 2022[24] (Environment Agency, 2025) and NI in 2024 (correspondence with the Northern Ireland Environment Agency, (NIEA)) are shown in Figure 3-2. Unionised ammonia, which is a specific pollutant in salt waters, is not monitored as a nutrient and is therefore not shown.

In England, 92% of rivers and 98% of lakes achieved a Good or High status for ammonia (total as N) and 40% of lakes achieved a good or high status for TN. Both rivers and lakes performed less well for phosphorus, with only 39% of river achieving a Good or High status for RP and only 28% of lake for TP. The coastal waters performed better than the transitional waters for DIN, with 63% achieving a Good or High status compared to only 12% for transitional waters.

In Northern Ireland, 97% of rivers achieved a Good or High status for ammonia (total as N). Both rivers and lakes performed less well for phosphorus, with only 59% of river achieving good or high status for RP and only 43% of lake for TP. The coastal waters performed much better than the transitional waters for DIN, with 79% achieving a good or high status compared to 0% for transitional waters. In NI, ammonia (total as N) is not monitored, and TN is monitored but classifications are yet to be set, which is why there are gaps in Figure 3.2b.

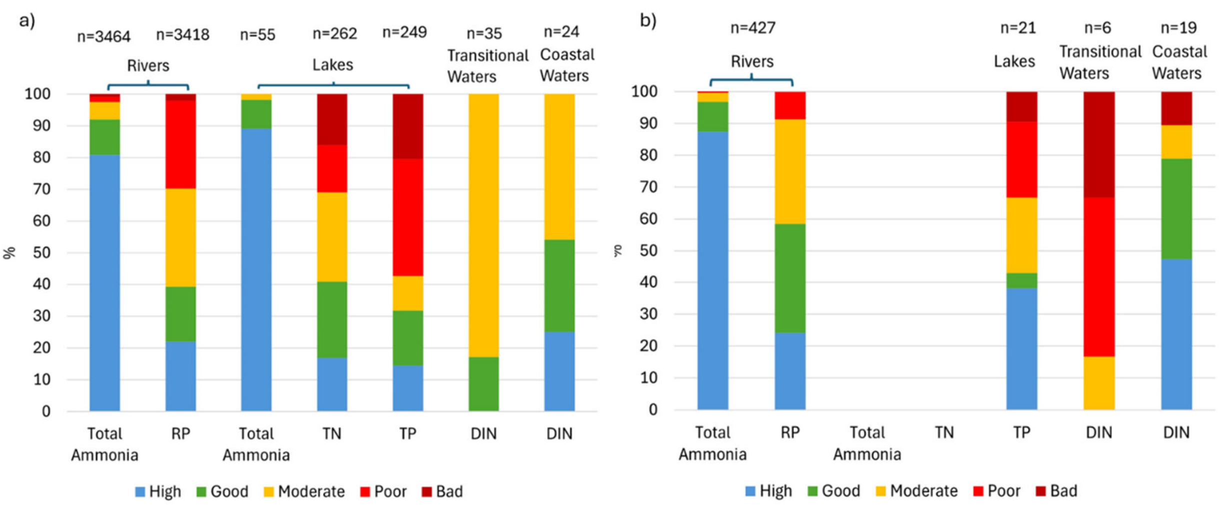

Figure 3.2. Status of nutrient fractions in surface water bodies in a) England (2022) and Northern Ireland (2024).

[24] Most recent publicly available dataset.

4. Findings and Discussion

The findings and discussion section begins with a brief description of the evidence base (Section 4.1) and a summary of how nutrient standards (and the methodologies used to determine them) vary across EU Member States (MS) (Section 4.2). Sections 4.3 to 4.9 aim to address a series of specific questions pertinent to the project aim, namely:

- 4.3 Do the current nutrient standards cover the correct nutrient fractions?
- 4.4 Are the current threshold values effective to protect ecological health?
- 4.5 Are the type- and site-specific components effective, and should other factors be considered?
- 4.6 Do current standards use the most appropriate statistical metric (e.g., annual mean, percentiles)?
- 4.7 Are current standards developed using the most appropriate biological metric (e.g. when used to determine pressure-response relationships)?
- 4.8 Are current thresholds developed using the most appropriate statistical methodologies?
- 4.9 What methodological advancements could be considered if developing new thresholds?

Addressing each of these questions in turn, the information included in Sections 4.3 to 4.9 typically includes:

- A description of current practices in England and NI.
- Comparison of the above with current practices in EU MS, largely informed by the review published by Poikane et al. (2019a).
- Commentary from the interviewees and workshop respondents.
- Evidence from the literature that supports the current approach.
- Evidence from the literature that suggests alternative/ additional approaches and may provide a closer link to ecological health.
- Summary box containing key conclusions and/or recommendations based on the evidence.

4.1 The core evidence database

From the academic literature reviewed, a 'core' evidence base of 37 articles were identified as relevant to the second of the research questions – to identify if the current nutrient EQSs are protective of ecological health and function in the surface water bodies of England and NI. As previously mentioned, selected information from these articles has been summarised in a database (6.Appendix B). Most of these studies focused on rivers (n = 17) and/or lakes (n = 18), with fewer relevant studies identified for transitional (n = 7) and coastal waters (n = 3) (Figure 4.1a). The majority of these articles addressed phosphorus (P) (n = 29) and/or nitrogen (N) (n = 25), with fewer discussing suspended sediments (n = 5) (Figure 4.1b). Similarly, the expertise of the interviewees and workshop participants predominantly covered rivers and lakes, and P and N. It is therefore acknowledged that the findings and discussion relating to TraC waters and suspended sediments are based on a relatively limited evidence base, and this has been highlighted as a recommendation for future work.

Most of the studies in the database (6.Appendix B) were based in the UK (n = 15) or focused on European-scale meta-analysis of data from multiple MS. There were also several articles from the Republic of Ireland and Germany, and a few from Spain, Poland, Lithuania and France (Figure 4.1c).

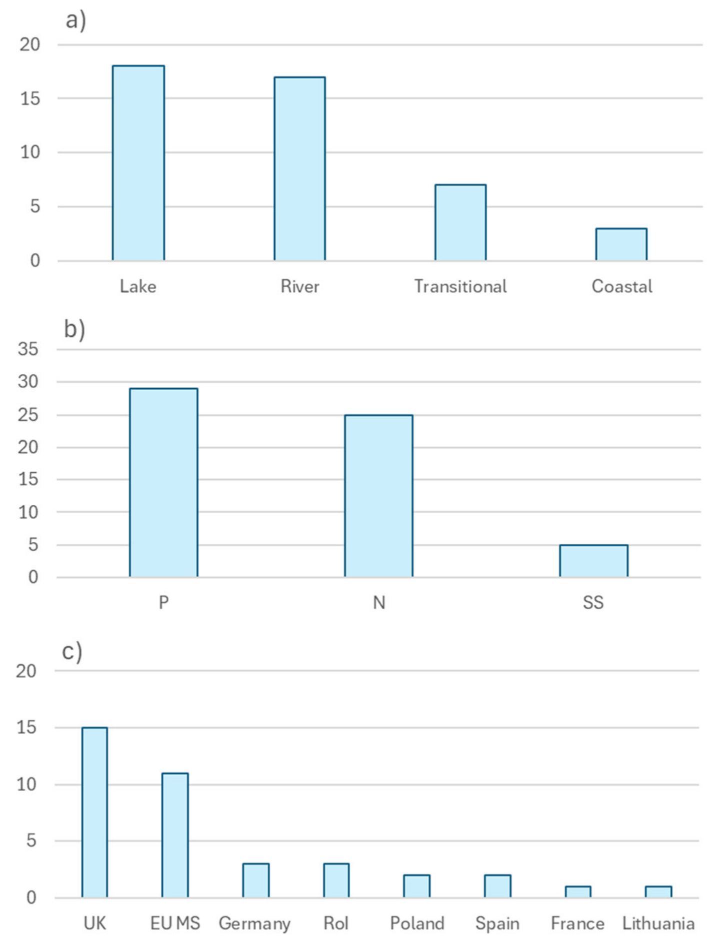

Figure 4.1. The number of articles included in the core evidence database focusing on (a) different water body types, (b) phosphorus (P), nitrogen (N), and suspended sediment (SS), and (c) the geographical representation of the studies.

4.2 How do nutrient standards and the methodologies used to derive them vary between England and NI and other countries within the same climate zone?

The WFD sets a common approach for managing water bodies across the EU, however it does not specify nutrient standards or targets for the whole continent. Despite nutrients being crucial for determining the ecological health of waters, there are no unified standards to assess the actions needed to achieve GES across Europe (Nikolaidis et al., 2022). Instead, the evaluation of water body status is based on the way the WFD is implemented, which varies between member states (MS) (Carré et al., 2017). Each MS creates its own national targets for N and P to achieve GES.

Extensive intercalibration work has been undertaken to ensure that the concept of ecological status is transferable between organisms (phytobenthos, macrophytes, invertebrates and fish) in freshwater environments and between EU MS. However, the same cannot be said for the transitional and coastal waters, or for the supporting elements such as nutrients, and therefore MS can interpret the WFD in a myriad of ways, producing a large range of thresholds, even for comparable water body types (Poikane et al., 2019).

Whilst a comprehensive review of the approaches employed to determine nutrient standards in other countries was beyond the scope of this review, a recent study by Poikane et al. (2019a) provides a review on the nutrient criteria used in Europe, under the WFD, including the various nutrient fractions, metrics and methods for determining threshold values employed across EU MS. Relevant differences identified between the nutrient fractions and threshold-setting methodologies used in the UK and in the EU MS are summarised at the beginning of each of the following sections in this Chapter.

Throughout the discussion, the focus will largely be on defining the Good-Moderate (G/M) boundary owing to the significance of this boundary in achieving the overall goal of the WFD (at least 'Good'status). This boundary is also more problematic than the High-Good (H/G) boundary as the definitions of 'Good' and 'Moderate' allow a wide scope for interpretation (Bennion et al., 2014).

4.3 Do the current nutrient standards cover the correct nutrient fractions?

The Redfield ratio was developed in 1954 and provides a consistent atomic ratios of N, P and carbon in marine phytoplankton, but is commonly applied across a range of ecosystems to determine which nutrients may be limiting in a localised system, even though it is not applicable to all organisms or ecosystems. This has formed the basis for the assumption that P is the limiting nutrient in freshwater and N is the limiting nutrient in saline waters which is reflected in the nutrient fractions that are currently used under the WFD.

Rivers in England and NI currently have EQSs for Reactive P (RP) and Total ammonia. In the review of nutrient thresholds by Poikane et al. (2019), the UK and the Republic of Ireland (RoI) were the only two countries/ MS to set standards in terms of RP, with the majority instead using TP (Figure 4.2a). Total ammonia is considered a Specific Pollutant in terms of compliance. Total ammonia is considered as part of this review due to its link to Biochemical Oxygen Demand (BOD) and the negative impacts it has on invertebrates (6.C.1), however, due to the dominance of research into N species and P fractions it will not be a key focus of the discussion. England and NI do not currently have an N-based EQS for rivers, whereas some EU MS have adopted an EQS for nitrate-N or TN (Poikane et al., 2019).

In contrast, lakes in England and NI currently have EQSs for TP, TN and total ammonia, with TP and TN being the most common nutrient fractions used within EU MS (Figure 4.2a, Figure 4.2b) (Poikane et al., 2019).

TraC waters in England and NI have standards for winter DIN and Unionised Ammonia but no P-based EQS (Table 4.1). In contrast, most of the other MS do have a P standard for TraC waters, with the most common choices in coastal waters being summer TP, annual SRP and winter SRP, and predominantly annual SRP in transitional waters (Figure 4.2d,c). DIN was also the most common choice in TraC waters in EU MS (Poikane et al., 2019a). Unionised Ammonia standards for TraC waters are based on the toxic effects it has on fish (WFD UKTAG, 2007b); however this is not a driver of eutrophication and thus is not a key focus of the discussion.

All of interviewees thought that England and NI do not currently have EQSs for all the nutrient fractions needed to be fully protective of ecological health. A number of interviewees believe that a comprehensive understanding of nutrient dynamics requires consideration of all nutrient fractions and their interactions with biological communities. This holistic approach can lead to better management practices and improved ecological outcomes. However, several interviewees and workshop participants noted concerns over the practical (cost) implications of measuring more fractions. Increased nutrient fractions would likely result in less sites and/or a reduced sampling frequency (unless a larger budget was available), and this would likely be more detrimental to ecological health. Therefore, the marginal gains achieved by increasing the number of nutrient fractions may not be offset by the additional cost, or the loss of temporal and spatial coverage. There is also an important consideration around consistency in assessment and the ability to monitor progress/decline effectively, so any change in monitored fractions would need to be done in a staged way to ensure comparability.

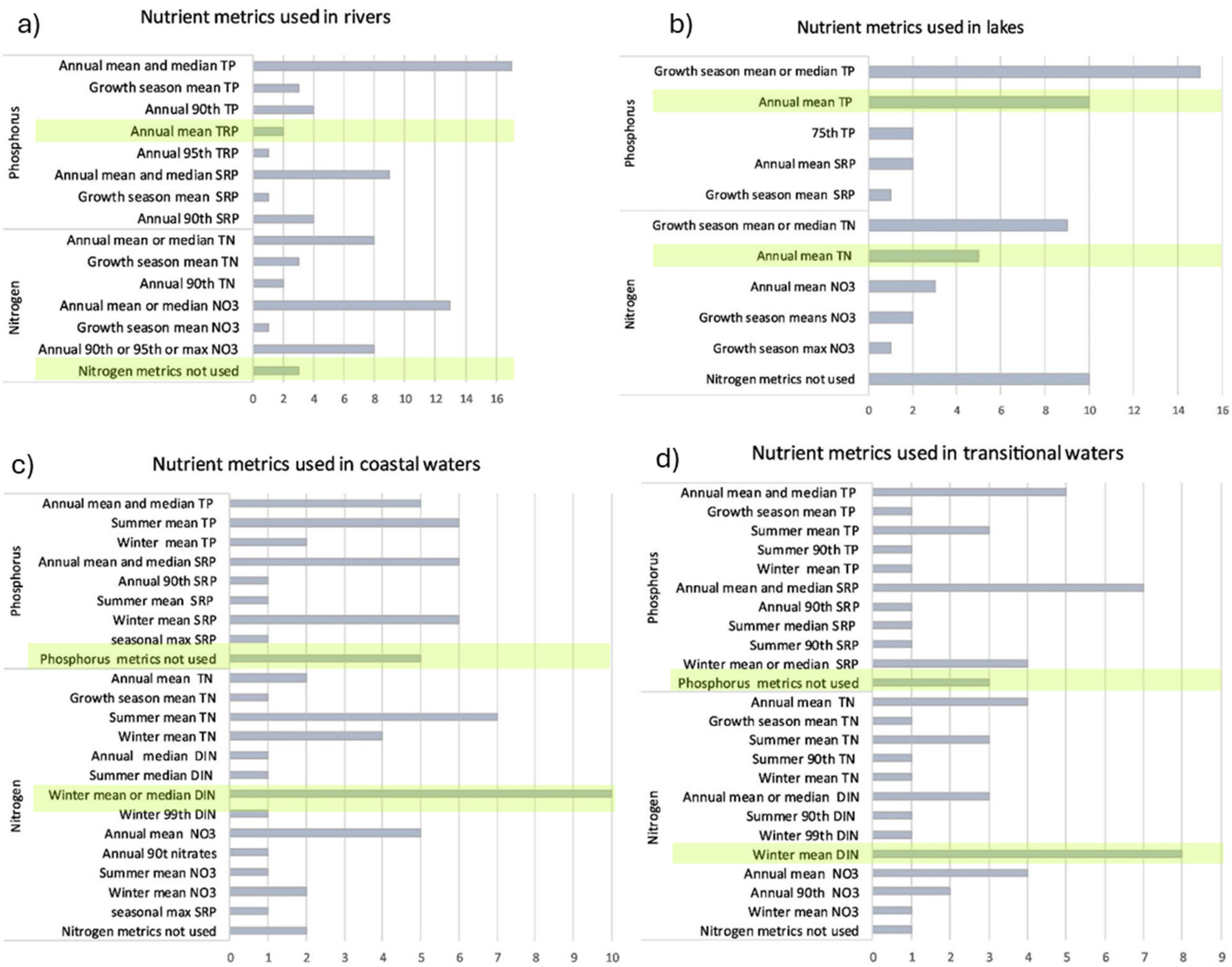

Figure 4.2. Metrics used to specify N and P EQS under the WFD for ecological classification in the European Union Member States (inclusive of the UK) for a) lakes, b) rivers, c) coastal and d) transitional waters, adapted from Poikane et al. (2019a). Current nutrient fraction and metrics used in England and NI are highlighted. Note that when compiling these Figures, the UK did not have a TN standard for lakes, so this is not reflected in the reported statistics in (b).

4.3.1 Rivers

4.3.1.1 Are the current reactive P (RP) standards appropriate?

Rivers in England and NI only have standards for RP. SRP is typically measured, while RP (current EQS metric) includes loosely bound and available phosphate, making it a more complex metric (6.C.6). Unpublished results indicate that there are no significant differences between RP compared to filtered samples (6.C.8). However, others have noted that the samples should be filtered (thereby measuring SRP) as the current method (settling, not filtering) can incur significant error, especially in cases where there are substantial concentrations of algae (6.C.5).

Whilst RP (or SRP) is easy to measure, it is not necessarily the most useful in terms of threshold setting as the relationship between trophic diatom response and SRP concentration can vary widely (6.C.7). In areas with high SRP ranges, the relationship is clearer, but in regions with lower concentrations, the data may be more scattered. In saturated systems, SRP works well as a standard, however, in more sensitive systems, other forms of P and lower concentrations need to be considered to protect the ecology (6.C.7). The response of algae and diatoms to P levels are also not linear and may plateau after a certain point, (e.g. >100 µg/L SRP), displaying no significant response in the algae/ diatoms (Poikane et al., 2022; 6.C.7, 6.C.5).

Several interviewees felt that RP is a useful metric but only if used alongside TP. SRP is generally considered the most available form of P and is used by all plankton/algae. However, some cyanobacteria, for example Microcystis which is a key toxin producing species in the UK, can use less bioavailable forms of P, including organic P. Measuring SRP alone does not capture those compounds, and therefore does not represent all available P. This is particularly true in level dependent river systems (and lakes) with internal loading of sedimentary P, where cyanobacteria can grow in SRP-replete conditions by utilising these other less bioavailable forms (6.C.4).

The amount of P in a water body as particulate P can also be significant, particularly in rivers and streams impacted by agriculture or where sediment loss is high (6.C.9; 6.Appendix D). Another challenge associated with RP/ SRP is being able to connect diffuse pollution science, which often deals with particulate P, to SRP standards in water bodies. This involves understanding the conversion of particulate P in the stream bed to SRP and predicting the impact of land measures like buffer strips and erosion control (6.C.7).

TP can be a key indicator of freshwater eutrophication, as it represents both available soluble P but also the particulate bound P water column, including the phytoplankton (EA, 2012). In some cases lakes, ponds and rivers with the biggest eutrophication problems exhibit very little SRP as it is all contained in the algae, therefore both SRP and TP should be measured to gain a comprehensive picture of water quality (6.C.5, 6.C.2). As TP encompasses all bioavailable forms of P (inorganic P, dissolved organic P and particulate P), a number of interviewees and workshop participants suggested that TP should have an EQS and be measured alongside SRP.

Following recommendations from the Water Target Advisory Group (Defra, 2022), the UK government introduced legally binding environmental targets, under the UK Environment Act 2021, including specific goals related to TP pollution (as opposed to just phosphate). This produces a misalignment between the objectives of the Environment Act and the monitoring undertaken by the Environment Agency under the WFD, and updated EQS including TP could address this.

However, it is important to recognise that TP may also not be suitable for all environments. In low alkalinity rivers Poikane et al. (2021) found that there were stronger (more significant) relationships between primary producers (macrophytes and phytobenthos) and SRP (r^2 = 0.40-0.65, $p < 0.001$) compared to a weak or non-detectable relationship with TP (R^2 = 0.09-0.20, $p<0.001$). Nikolaidis et al. (2022) assessed river and lake nutrient targets to support GES across the EU and suggested annual mean values of 11-105 µg/L for the river TP G/M boundary.

4.3.1.2 Should there be an N standard for rivers?

The assumption, based Redfield ratio, that P is the limiting nutrient in rivers is considered by many to be flawed and outdated. Extensive evidence from bioassays and correlation analysis indicates that both P and N can limit primary production in rivers (e.g. Dodds and Welch, 2000; Dodds and Smith, 2016; Jarvie et al., 2018) and therefore both N and P should be considered when attempting to restore a river to GES (Dodds and Smith, 2016).This is supported by current research in the Wye Valley which indicates that N fractions (particularly ammonium and nitrate) are the key drivers of ecological degradation (6.C.3). Whilst the UK does not currently have an N-based river standard, a number of EU Member States surveyed by Poikane et al. (2019a) do, with the majority using nitrate-N (20 countries) followed by TN (13 countries).

Most interviewees (7/9) suggested that there is an urgent need to improve river N regulations with the introduction of an N-based river standard. However, opinions were divided over which fraction would be the most suited for rivers. From the interviewees, the majority (4/7) felt that nitrate would be the preferred choice, with others recommending DIN, TN and organic N. Workshop participants were asked to rank N fractions in order of importance for inclusion as a river EQS; TN was the most popular choice, followed by total ammonia (for which an EQS already exists), DIN and nitrate.

It has been suggested that ecological standards for N, reflecting relatively unpolluted conditions, should be implemented and are required to be much lower than those currently used for human health protection (11.3 mg N/L (or 50 mg NO_3/L) in the drinking water standard) (WHO, 2024). Pan-European and globally representative studies have identified a threshold of 1.5 to 2 mg/L for TN, above which ecosystems shift from organic to nitrate dominance (Wymore et al., 2012; Durand et al., 2011), however this threshold does not account for technical feasibility of achieving these thresholds in anthropogenic impacted systems in the UK. Nikolaidis et al. (2022) developed EU-wide and regional nutrient targets to define GES for river and lake TP and TN and suggested annual mean values of 0.5 – 3.5 mg/L for the river TN G/M boundary. Similarly, Poikane et al. (2021) found that the combined macrophyte/ phytobenthos models (calculated as the minimum of the EQRs of the two organism groups) for low alkalinity lowland rivers (Type R-C1) produced the G/M boundary at 1.63 mg/L TN (range: 0.71–4.19 mg/L).

Several interviewees believe that a nitrate standard for rivers would be the preferred choice (over TN) as nitrate is considered to be more significant in driving an ecological response and is more widely measured, thus having greater potential for inclusion as a standard (6.C.6). Introducing a nitrate standard would likely result in more investment in mitigation methods and would promote nutrient trading (e.g., investment in buffer strips may increase to help meet targets) (6.C.7). Jarvie et al. (2018) found that there was a greater potential for P limitation in rivers and N limitation in headwater streams. There was also a greater potential for P and N co-limitation in headwater streams than rivers, especially in the Upland-Low-Alkalinity streams, suggesting that managing both P and N inputs may be required to minimise risks of degradation of these sensitive headwater stream environments. Jarvie et al. (2018) identified an ecologically limiting threshold for nitrate of around 0.4 mg N/L in headwater streams.

Previous work by G. Phillips (sponsored by the Environment Agency, unpublished) demonstrated a clear biological impact of elevated nitrate concentrations, with macrophytes more responsive to nitrate than phosphorus (6.C.6). In addition, data from the EU showed that while high P concentrations were always associated with high N concentrations (as both are associated with waste/effluent), high N concentrations could occur at low P concentrations due agricultural impacts. This end of the scale (low P, high N) is problematic for ecological health but is not covered by the current EQSs. Although nitrate is likely to be high in winter and low in summer, it would be best to use a measure of central tendency (e.g., annual median) to define thresholds (6.C.6).

Evidence suggests there is a significant pressure-response relationship (Spearman Rank correlation coefficients (rs)) between macroinvertebrates (based on the Irish Quality Rating System which used as a surrogate for ecological status to the EQR) and nitrate + nitrite ($rs = -0.526$, $p<0.0001$) (Donohue et al. 2006), which also supports the use of nitrate as a standard. It has also been suggested that if EQSs for N fractions are developed, the methods used to determine thresholds must account for changes in concentrations due to flow rates and rainfall patterns (Appendix 6.C.3). It was noted when the latest P standards were updated, the farming community also registered significant concerns regarding the implementation N standards (6.C.8).

4.3.1.3 Should there be organic N and P standards for rivers?

Although inorganic nutrient fractions (e.g. nitrate, ammonium) are considered to drive water bodies ecological responses, organic nutrients can form a significant fraction (>50%) of the total nutrient pool particularly in upland and/ or mixed land-use systems, where they can play a critical role in ecological response (Mackay et al., 2020; 6.C.7; 6.C.2). All nutrient forms (organic and inorganic) are bioavailable and in complex ecosystems different organisms have evolved to use different nutrient forms (phosphate, nitrate, ammonia, organic molecules) based on accessibility and preference. To protect ecosystems effectively organic nutrient fractions should be fully considered (6.C.2). However, it was recognised that the current methods used to measure organic nutrients have a degree of uncertainty and need development, something which would need to be considered in terms of the appropriateness of setting EQSs. Despite this several interviewees felt that the analysis of organics still provides important information, even if not used directly for regulatory purposes (Appendix 6.C.3, 6.C.4).

4.3.1.4 Should there be a sediment standard for rivers?

Source apportionment studies conducted by Defra (2018) concluded that ~70% of suspended sediment as well ~50% nitrate and 25% of P were sources from land classified as agricultural. In addition to acting as a substrate for nutrient delivery from the surrounding catchment, the quantity, quality and dynamics of sediment in rivers can influence the ecological status via hydromorphological changes, stage-discharge relationships and flood risk (Slater et al. 2015) and can also have negative impacts on the spawning of salmonoid fish. Research has also demonstrated clear relationships between suspended sediment data and filter feeders/ invertebrate metrics (6.C.7; Stutter et al., 2007).

Under the WFD, hydromorphological quality elements should be assessed as a supporting element of the classification process, however hydromorphology only impacts the classification of High status and the assessment/ monitoring of sediment transport is not required (Nones et al., 2017). Some researchers believe that if both sediment transport and hydromorphological quality elements are not effectively considered, then it is likely that rivers will obtain a misleading and optimistic assessment of ecological status (Nardini et al., 2008). In their work Nones et al. (2017) found that only 12 out of 20 countries monitored sediment transport, hydromorphological changes, and biota simultaneously. This may result in undetected deterioration of ecological health, which undermines the objectives of the WFD (Nones et al., 2017).

Whilst there are currently no standards for sediment in rivers, the Environment Act 2021 does specify legally binding targets for sediment reduction (e.g., at least 40% reduction by 2038, using 2018 as the baseline year). However, without a driver for compliance monitoring (under the WFD) it is unclear if there will be sufficient monitoring, reporting and mitigation to effectively meet these targets. Nones et al. (2017) suggest that future monitoring programs should include hydromorphology and sediments at the same level as biological and physico-chemical elements. They suggest that national legislation and EU guidance requires updating to reflect the causal interrelationships between hydromorphological alterations, sediment transport and the biological (ecological) status of freshwater ecosystems.

4.3.2 Lakes

4.3.2.1 Are the current TP and TN standards for lakes appropriate?

In Lakes, England and NI currently have standards for TP and TN (and total ammonia). Total values have been used because the longer residence times (compared to most rivers) mean that soluble nutrients can be incorporated into algal and other plant biomass, thus very low concentrations of soluble nutrients (particularly in the summer months), will not be reflective of the true nutrient status (UKTAG, 2019a).

Mellios et al. (2020) assessed the relationship between cyanobacterial biomass (CBB) and nutrients in 822 lakes across northern Europe (including the UK). Classification and Regression Tree analysis indicated that when considering the whole dataset (including all ten lake groups), TP played the most significant role towards the prediction of CBB, while TN was influential only for the subset of samples (n = 276) where TP was larger than 89.75 µg/L. Similar results were found by Macintosh et al. (2019) who studied lakes in the RoI and found that TP exhibited the largest relative influence on EQR scores for phytoplankton and macrophytes. This suggests that in lakes it is important to have both TN and TP (which was echoed by most interviews and workshop participants), thus supporting the current approach.

4.3.2.2 Should there be additional inorganic and/or organic nutrient standard for lakes?

When workshop participants were asked which N factions should be included for lakes, the respondents (n=9) ranked the current standards (TN and total ammonia) as the most important, followed by nitrate and DIN. One interview also noted that it was important to include DON for both lakes and rivers (6.C.4). However as described above (Section 4.3.1.3) there are limitations associated with the accuracy of analytical methods required to measure organics, and there are also cost implications (sampling and analysis) associated with having an increased number of standards. No further information was found from the literature relating to this topic.

4.3.3 TraC

TraC waters in England and NI currently have standards for winter DIN and unionised ammonia, but no EQS for P. In contrast, the majority of MS in the review by Poikane et al. (2019) do include P (except from of the Northeast Atlantic Sea region in France and the Netherlands (transitional and coastal) and Ireland (coastal)).

When asked which (if any) P-based EQS should be introduced for TraC waters, 75% of the Workshop respondents (n=9) ranked RP first and 16% of the workshop respondents (n=2) ranked SRP first

(6.Appendix D). One Workshop respondents commented that TP EQS should be included for TraC as it can also be limiting in these environments and should ideally be based on a summer mean.

4.3.3.1 Should there be a P standard for TraC waters?

In saline waters, N is often seen as the key nutrient involved in the formation of algal blooms, but P may also be important in some estuarine situations (EA, 2022). Both DIN and SRP monitored as part of the Convention for the Protection of the Marine Environment of the North-East Atlantic (or OSPAR) eutrophication assessment. In the recent OSPAR assessment, the biodiversity outcomes indicated that decreases in nutrient concentrations, particularly for P entering the North Sea, may be driving downward trends in phytoplankton biomass across the Greater North Sea. The disequilibrium between nitrate and P has increased, resulting in an imbalance of nutrients which can negatively affect phytoplankton biomass. This is projected to continue owing to the success of phosphorus reduction (OSPAR, 2023) and thus supports the inclusion of a P-based standards to protect ecological health in the UK.

Friedland et al. (2019) investigated drivers of eutrophication in coastal lagoons spanning the German/ Polish border and found the most significant relationships were between TP and chlorophyll a, and GES was associated with TP concentrations of 71 µg/L TP. It was found that significant reductions in N and P loading to the lagoons (30% TN and 70% TP) resulted in improved ecological status in the large lagoon however no improvements were observed in the small lagoon, owing to long residence time, sediment resuspension and reduction in submerged vegetation. This highlights the complexity associated with improving water quality in transitional waters, whereby reaching the desired status is not always possible by reducing nutrient loads alone.

4.3.4 Summary

4.3 Do the current nutrient standards cover the correct nutrient fractions?

General points

- Research suggests that colimitation (N and P) is more common than previously assumed so the use of a single nutrient criterion should always be questioned (Poikane et al., 2019).
- The consensus was that all water bodies should have standards for at least TN and TP standards (in addition to current fractions). Total measurements help identify all sources of contamination, including livestock, which contribute significantly to nutrient pollution. Understanding total nutrient fluxes can enable effective mitigation measures to be adopted.
- The development of additional nutrient fractions should be avoided if the resultant increase in resources would lead to a decrease in the spatial and/ or temporal frequency of monitoring by the Environment Agency and NIEA.

Rivers

- The consensus across academic literature, interviews and workshop input is that reactive P is a useful metric, although this should be in the form of SRP (filtered).
- For SRP to be effectively utilised for the protection of ecological health, TP should also be included, which would account for inorganic and particulate fractions, which better represent major sources of bioavailable P to UK rivers and align with targets set by the Environment Act.
- Most interviewees and workshop participants believe rivers should have an N-based standard. The preference would be TN or nitrate (or both), with some support for including organic fractions. Some participants noted the benefit of collecting information on organic fractions to help inform mitigation measures, even if this information is not used for regulatory purposes.
- Some research suggests that a sediment-based EQS for rivers would improve the overall assessment of ecological health, however this warrants further review.

Lakes

- The current TN and TP standards are largely considered to be fit-for-purpose, however some interviewees and workshop participants felt that additional fractions (e.g. DIN) should also be included.

Trac

- The consensus across academic literature, interviews and workshop input was that the current DIN standard for TraC waters is appropriate, however ideally there would also be TN standard.
- Most interviewees and workshop participants felt that TraC waters should also have a P standard, either as SRP or TP.

4.4 How do current G/M thresholds compare with values in the literature?

Individual site- and type-specific factors can significantly impact the nutrient concentration thresholds for each water body type. The range of N and P concentrations used for the G/M threshold by broad type across EU MS are presented in Figure 4.3, and ranges for water bodies in England and NI are summarised in Table 4.1. To assess how protective the current threshold values are likely to be, they have been compared with G/M thresholds from comparable water bodies in countries with similar bioclimatic conditions. This is not intended to provide a comprehensive assessment of the individual threshold values (which is beyond the scope of this review) but aims to contextualise current values for England and NI against other countries' thresholds and thresholds determined through academic review.

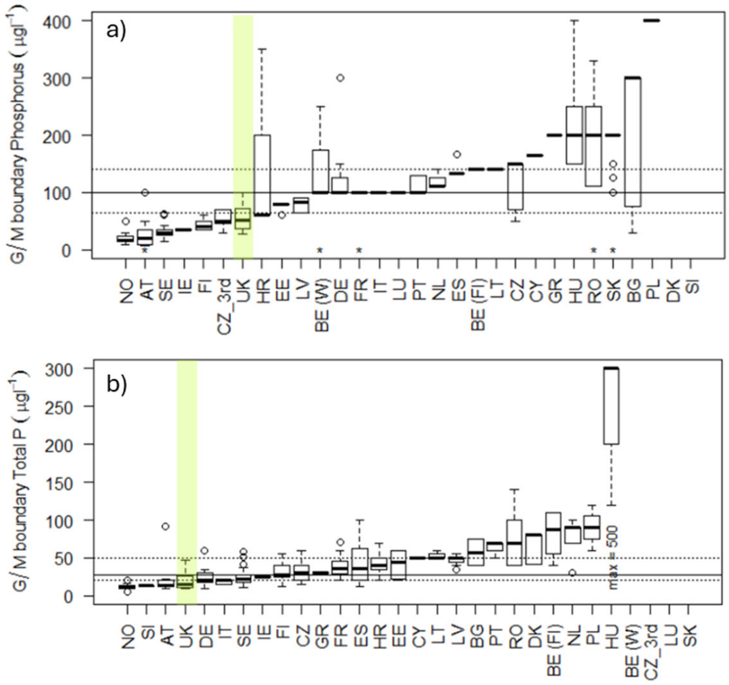

Figure 4.3. Range of reported G/M thresholds arranged by median value of boundaries for each country for (a) river P and (b) Lake TP. Lines mark the 25th, 50th and 75th percentile values for all countries (for rivers, the * identifies 90th percentile metrics that were halved) (adapted from Phillips and Pitt, 2016).

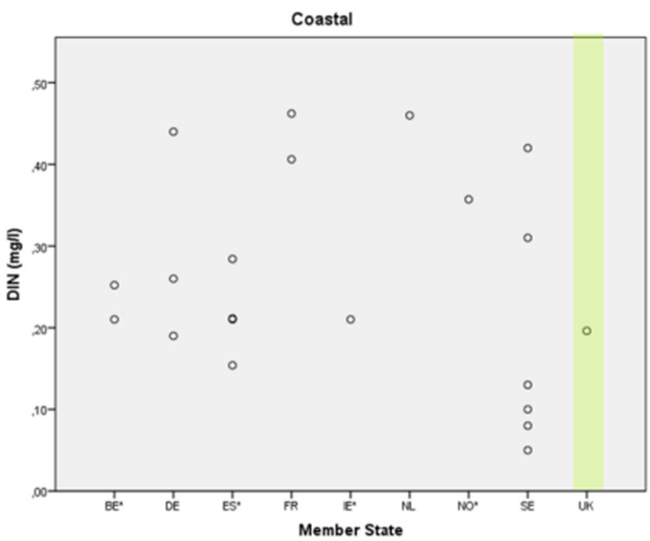

Figure 4.4. G/M boundary values for winter DIN in coastal waters in the Northeast Atlantic (Belgium (BE), Denmark (DE), Spain (ES), France (FR), Ireland (IE), Norway (NO), Sweden (SE)

and the United Kingdom (UK). Figure only shows the UK threshold for clear waters (Dworak et al., 2016).

Table 4.1. Summary of current G/M thresholds ranges per water body type

	Rivers	Lakes	Transitional	Coastal
Phosphorus				
Total Phosphorus (TP)		10 - 40 µg/L^^		
Reactive Phosphorus (RP)	28 – 98 µg/L^			
Nitrogen				
Total Nitrogen (TN)		0.74 - 1.46 mg/L^^^		
Dissolved Inorganic Nitrogen (DIN)			30 – 270 µM**	18 – 270 µM**
Ammonia (Total as Nitrogen)	0.3 – 0.6 mg/L*	0.3 – 0.6 mg/L*		
Unionised Ammonia (as Nitrogen)			21 µg/L	21 µg/L

^Standards for rivers were calculated using the maximum and minimum values for altitude and alkalinity as described in the WFD (2015), covering alkalinities 2 - 250 mg/l $CaCO_3$, altitude 10 – 355 m and for an EQR of 0.532 (good biological threshold).
^^Standards for lakes with alkalinities 10 - 100 mg/l $CaCO_3$, altitude 50 – 100 m, depth 2, 10, 50 m, and for surface and deep-water samples.
^^^range of thresholds across all classes as described by WFD UKTAG (2019a).
*Standards for rivers and lakes of alkalinities 10 - 100 mg/l $CaCO_3$ and altitudes 50 – 500 m.
** Standards for turbidites ranging from clear to very turbid. Standards are presented in µM, as per the WFD.

4.4.1 Rivers

It is not possible to directly compare the current thresholds used in England and NI with many of the thresholds used in other countries and reported in the literature, as most countries use a different P fraction (mostly TP). Nevertheless, the range of river RP concentrations used to define the G/M threshold in England and NI appear more precautionary than the thresholds used by many EU MS (Figure 4.3a) and are broadly in line with those proposed for SRP in the Central Baltic Region of Europe, based on recent guidance for calculating thresholds (Poikane et al., 2019) (Table 4.2.).

However, some evidence suggests that for some river types, the current RP standards in England and NI may be too elevated to protect the ecology. Under the existing approach, once calculated, site-specific RP standards for high ecological status can be significantly higher than natural background levels, making them not fit for purpose (6.C.2). Natural (unpolluted) waters typically have around 10 µg/L of P, whilst current site-specific standards can be significantly higher (e.g. ~55 µg/L of P for high alkalinity, low altitude rivers) (Mouchos et al., 2022). Thus some researchers feel that elevated P concentrations are causing ecological damage before reaching the current calculated P thresholds for GES (6.C.2).

Evidence suggests that reducing P to 30 µg/L would be more effective at protecting the ecological health of rivers than the current target of ~50 µg/L (depending on site-specific factors) which is only temporarily limiting. A reduction from 100 µg/L RP to 50 µg/L RP will likely result in a response from the algae, however this is likely only a transient response, and within a few years the original assemblage will be reinstated. Flume experiments conducted by CEH (unpublished) have demonstrated that algal communities adapt to the P concentration, and significant reductions in P are needed to see a permanent improvement in ecological status (6.C.5).

The site-specific P threshold values may also be too high (not protective enough) for shallow rivers with high alkalinity in NI (6.C.6). This is supported by the findings from Poikane et al. (2021), who found that river types with high alkalinity displayed weak correlation between primary producers and nutrients (SRP, TP and TN) (r^2 = 0.03 - 0.27, $p < 0.001$) (compared to more significant relationships (r^2 = 0.40 - 0.65, $p < 0.001$) for SRP/TN in low alkalinity streams), which suggests that thresholds (and the methodologies employed to set them) could warrant review for these river types.

Table 4.2. Summary of SRP and TP concentrations reported in the literature for the G/M boundary in rivers

Typical SRP/TP ranges	Typology	Description/location	Reference
Current RP thresholds (µg/L) for England and NI			
28 – 98	Range covers alkalinities 2 - 250 mg/L $CaCO_3$, altitude 10 - 355 m	Calculated using the combined macrophyte/phytobenthos* linear regression model	WFD UKTAG (2013)
SRP (µg/L) thresholds in the literature			
32 (18 –58)	Low alkalinity, lowland rivers	Calculated using the combined macrophyte/phytobenthos* linear regression model (or the Central-Baltic region of Europe)	Poikane et al., 2021
40 (19–78)	Low alkalinity upland rivers		
21 (17–28)*	Low alkalinity, lowland rivers	Calculated using the combined macrophyte/phytobenthos* minimization-of-mismatch' approach (for the Central-Baltic region of Europe)	
32 (28–38)	Low alkalinity, lowland rivers		
TP (µg/L) thresholds in the literature			
35		Republic of Ireland	EPA, 2024
40–105 47-70 11-27	Lowland Mid-altitude Highland	G/M boundary suggested to support GES across EU board types	Nikolaidis et al. (2022)
** Calculated as the minimum of the EQRs of the two organism groups*			

4.4.2 Lakes

4.4.2.1 Lake TP standards

The G/M threshold for TP in lakes in England and NI typically ranges between 10 – 40 µg/L. Evidence from the literature suggest that the current thresholds are broadly in line with those used in other countries or determined through academic review (Table 4.3.), with findings from Phillips and Pitt (2016) suggesting the UK's thresholds are more stringent compared to many EU MS (Figure 4.3b).

Table 4.3. Summary of TP concentrations reported in the literature for the G/M boundary in lakes

Typical TP ranges (µg/L)	Typologies	Description/location	Reference
10 – 40*	Covering a range of typical alkalinities, depths and altitudes*	Current thresholds used in England and NI	WFD UKTAG (2016)
17 - 60 17 10	Lowland Mid-altitude Highland	G/M boundary suggested to support GES across EU broad types	Nikolaidis et al. (2022)
14 - 27	Covering a range of lowland and mid-altitude typologies	TP ranges determined using phytoplankton (Northern region).	Poikane et al., 2022
20		Concentration used in the 2006 Republic of Ireland (RoI) legislation	Donohue et al., (2006)
20		TP concentrations >20 µg/L resulted in a reduction in sensitive species relative to increased tolerant species, which was suggested as an ecologically meaningful definition of the G/M threshold.	Bennion et al. (2014)
48 – 53 58 – 78	High alkalinity shallow High alkalinity very shallow	Major productive shallow lake types in Europe using data from the Central Baltic region	Poikane et al., (2019b)
16 – 30	All lake types combined	70 surveillance lakes in the RoI	Free et al., (2016)

**Standards for lakes with alkalinities 10 - 100 mg/l $CaCO_3$, altitude 50 – 100 m, depth 2, 10, 50 m, and for surface and deep-water samples.*

4.4.2.2 Lake TN standards

TN G/M thresholds for lakes in England and NI range from 0.74 - 1.46 mg/L and are broadly in line with concentrations identified in the literature (Table 4.4.). A survey of nutrient standards in use for the WFD across European MS (Phillips and Pitt, 2016) also showed that, despite a range of concentrations being in use for the G/M boundary, for most lake types the median values were all within the range derived by UKTAG (2019a). Threshold values also closely aligned with values used in parts of the UK for designation of eutrophic lake nitrate vulnerable zones (NVZs) under the Nitrates Directive in England (threshold values 1-2 mg/l for total nitrogen, Defra 2016 in WFD UKTAG, 2019a), and target values adopted for the UK Common Standards Monitoring Guidance for Lakes (generic target 1.5 mg/l, site specific targets 0.4 – 1.5 mg/l applied for some lakes in England, JNCC, 2015 in WFD UKTAG, 2019a).

Table 4.4. Summary of TN concentrations reported in the literature for the G/M boundary in lakes

Typical TN ranges (mg/l)	Typologies	Description/location	Reference
0.74 - 1.46	Range of threshold values across all lake types	Current thresholds used in England and NI	WFD UKTAG (2019a)
0.5-1.8 0.5 0.5	Lowland Mid-altitude Highland	G/M boundary suggested to support GES across EU board types	Nikolaidis et al. (2022)
1.1–1.2 1.0–1.4	High alkalinity shallow High alkalinity very shallow	Major productive shallow lake types in Europe using data from the Central Baltic region	Poikane et al. (2019b)
0.6–0.9	Covering a range of lowland and mid-altitude typologies	Thresholds determined using phytoplankton in the Northern Region of the EU	Poikane et al. (2022)

4.4.3 TraC

4.4.3.1 DIN

DIN G/M thresholds for TraC waters in England and NI range from 30 – 270 µM, depending on turbidity. This range is broadly in line with concentrations identified in the literature, however no threshold values were identified as high as the current thresholds for 'very turbid' waters meaning it is unclear if these are stringent enough (Figure 4.4; Table 4.5.).

Table 4.5. Summary of DIN concentrations reported in the literature for the G/M boundary in TraC waters

Typical DIN ranges (µM)	Typologies	Description/location	Reference
Transitional (salinity 25)			
30 70 180 270	Clear (mean) Intermediate turbidity* Turbid* Very turbid*	Current thresholds for England and NI	WFD UKTAG (2008b)
38.1	Set for summer Secchi depth of 1.7 m (Sagert et al., 2008)	Small lagoon spanning the German/Polish boarder	Schernewski et al. (2015) Friedland et al. (2019)
Coastal (salinity 32)			
18 70 180 270	Clear (mean) Intermediate turbidity* Turbid* Very turbid*	Current thresholds for England and NI	WFD UKTAG (2008b)
74.5**	All water typologies (clear to very turbid)	Determined using the minimisation of mismatch approach	Salas Herrero et al. (2019)

79.7**	Determined using binomial logistic regression***
212**	Determined using quantile regression models

**99th percentile*
***using UK data as part of a wider data set representing the North Atlantic Estuaries broad type*
**** with a 50% probability of being in either good/moderate category*

4.4.3.2 Unionised ammonia

Uriarte and Borja (2009) found that increases >10 µM ammonia (~140 µg/L ammonia as N) produced a rapid decrease in fish quality. This is significantly higher than England and NI's current EQS suggesting the current standard is protective of fish health in TraC waters. However, the UKTAG suggested that ammonia standards may warrant review, as there is some evidence of under-protection of fish (WFD UKTAG, 2014b).

4.4.4 Summary

4.4 Are the current threshold values effective to protect ecological health?

Rivers
- Current RP standards in England and NI may not be stringent enough to protect ecology health.
- Threshold values may also be too high (not protective enough) for shallow rivers with high alkalinity in NI.

Lakes
- Lake TN and TP thresholds are broadly in line with those used across EU MS.

Trac
- The range of DIN threshold values for G/M are broadly in line with concentrations identified in the literature, however no thresholds values were identified as high as the current threshold values for 'very turbid' waters, meaning it is unclear if these are stringent enough.
- Limited evidence suggests that the unionised ammonia standard is protective of fish heath, however further evidence is required.

4.5 Are the type- and site-specific components effective, and should other factors be considered?

Environmental components relating to each water body type have been incorporated when deriving thresholds to make them better suited to that particular water body, or to a given typology (Table 4.6.).

River RP, lake TP and river/ lake total ammonia standards are site-specific, meaning each water body will have bespoke thresholds. This approach means threshold concentrations are calculated on a site-by-site basis which provides the benefit of not requiring river to be classed in "types" (Kelly et al., 2022) and is considered a major advantage of the UK's current approach to threshold setting over many EU MS (6.C.8).There are also type-specific thresholds for some lake typologies (e.g., marl lakes) and for TraC TN and DIN (Table 4.6.).

Table 4.6. Type-specific and site-specific components incorporated in the current nutrient EQSs for England and NI

Water body/ nutrient type	Type-specific and site-specific components incorporated in current EQS
River TP	Site-specific: alkalinity, altitude
River/ lake total ammonia	Site-specific: alkalinity, altitude
Lake TP	England: Site-specific (geographical region, geological category, depth) or Type-specific* NI: Site-specific: colour and depth
Lake TN	Type-specific: depth, humic substances
TraC DIN	Type-specific: salinity, turbidity (annual mean suspended particulates)
TraC unionised ammonia	No type- or site-specific components, just a single value (for salt water)

** Where lake (or part of a lake) has been categorised as Marl, or where there is insufficient data*

Whilst site and type specific aspects are largely considered a strength of the current approach; additional variables have also been suggested for inclusion. Physical parameters like flow, residence time, temperature, sunlight and exposure to other chemicals often have a greater impact on algal blooms than nutrient concentrations alone. Many of these aspects are also impacted by climate change resulting in varying flow conditions that affect P concentrations (and therefore eutrophication), even when the inputs to the river remain constant (6.C.9).

However, it is also important to consider that including some of these variables would require additional monitoring which may result in a reduced spatial and temporal resolution of sampling due to cost implications. Whilst there was support from the interviews and workshop attendees for exploring further options for type- and site-specific variable, there was a general consensus (amongst interviewees) that amendments to the current approach should be avoided if they would negatively impact the resolution of monitoring.

Furthermore, it is important to remember that the site-specific factors that are included in the threshold calculations are determined based on statistical analysis performed on national-scale datasets. Therefore, including additional site-specific factors may not be viable, if their inclusion does not improve the strength of pressure-response relationships.

4.5.1 Rivers

In the UK, P standards (for rivers and lakes) are based on models which use the alkalinity and altitude of the site. Altitude acts as a proxy for population density, with more rural catchments occurring at higher altitude. Alkalinity reflects pH which impacts flocculation and suspension of P and sediment in the water column. An increase in pH can promote precipitation of P with iron, aluminium and calcium.

4.5.1.1 Does alkalinity compromise the P standard in rivers?

One potential issue with incorporating alkalinity when calculating site-specific thresholds is that care must be taken when applying such models in regions where river alkalinity is artificially elevated. This may be the case in areas where calcium carbonate (lime) is applied to agricultural land or added to the river directly to mitigate acidification in low alkalinity rivers, or where a high proportion of flow is derived from effluent (Tappin et al., 2016).

Tappin et al. (2016) investigated if the use of alkalinity compromises the P standard in a catchment in North Devon. They found that P standards for rivers are dependent on alkalinity and altitude and in the case of reference sites (exhibiting minimal anthropogenic impact) they explained most of the variance in RP. This relationship is used to calculate the expected concentration of RP for each site (and therefore the class boundaries). However, effluent contributes significant alkalinity to rivers, and therefore in effluent influenced rivers the calculated reference conditions and ecological status boundaries may be incorrect. The size of the discrepancy is dependent of the proportion of river alkalinity derived from effluent compared to 'natural' alkalinity sources.

This may mean that the calculated concentration of RP at a given boundary may be higher in effluent impacted rivers than in those with a lower proportion of treated effluent rivers, which may provide less stringent thresholds, and therefore a misleading representation of ecological status. However, this study focussed on a single catchment with limited groundwater infiltration (base index flow ~0.45), and the result may not be representative of other catchments. The impact of artificially elevated alkalinity on the calculated baseline (and thus the thresholds produced) therefore warrant further review across a range of catchments.

4.5.1.2 Potential additional type/site-specific factors that could be explored for rivers

When developing the current thresholds UKTAG identified a significant amount of unexplained variance in the model, which was largely due to biological responses to P being affected by factors other than site altitude and alkalinity. They suggested that in the future it would be useful to quantify these sources of error and explore alternative statistical approaches for defining boundaries that may capture additional elements of complexity in this system (WFD UKTAG, 2012).

Rivers have fewer site-specific categories than lakes, and many of the interviewees felt there was a need to explore the potential of including additional factors like the degree of modification, flow, and climate. Clarity of water may also be important, for example using suspended sediment or turbidity. Dissolved organic carbon (DOC) concentrations have been observed to be increasing globally in recent years, the cause of which is subject to academic debate. Increases in the flux of DOC to aquatic environments increases the attenuation of UV rays and may contribute to warming of rivers and lakes, resulting in an environment conducive to eutrophication (6.C.9). Workshop participants were asked to consider which variables were important for determining the impact of nutrients on ecology in rivers (and should be considered when developing future standards). Flow regime, altitude, alkalinity and temperature were all considered important and to a lesser extent suspended sediment and shading.

In the Wye catchment, the narrative around eutrophication is usually focused on P, whereas the P has decreased at many sites in recent years, and it is likely to be flow, water temperature and sunlight which are the main divers of eutrophication (Perkins et al., 2023; 6.C.5). This means that riparian zones for shading and flow control are crucial forms of mitigation (6.C.3). However, it has also been suggested that due to the range of sites used in determining the pressure-response curves between the chemical measure (e.g. TP or SRP) and biological responses (e.g., changes in UK Trophic Diatom Index (TDI) or chlorophyll-a) that site-specific factors such as light, temp, flow, other pollutants etc., are already incorporated into the relationships, and explain the observed scatter (6.C.7).

4.5.2 Lakes

At the time of standard development, the UKTAG did not consider a site-specific approach to the TN standards as the 'simpler' type-specific approach provided a sufficiently robust relationship between phytoplankton status and TN concentration (R^2 = 0.747) (UKTAG, 2019b). Several interviewees felt that the UK has categorised lakes well, considering factors like size, depth, alkalinity, and altitude. However, some believe there is a need to include physical modification typologies in the models (6.C.4). For example, lake stratification (influenced by flow and temperature) affect nutrient dynamics and algal growth. Flow and temperature should therefore be considered within the typology classifications used for monitoring and assessment. This is particularly relevant for understanding internal nutrient loading and algal blooms (6.C.4). However, data on thermal stratification is not available at most lakes and would require considerable resources to gather.

Workshop participants selected depth as the most important variable for lakes followed by alkalinity, humic substances, residence time, temperature, and geology. Other variables that were mentioned included water body size and lake water stratification.

Humic substances are included as a type-specific factor for TN in lakes but are not included for TP in lakes in England (but are for NI). Vinogradoff and Oliver (2015) suggest that water colour could be a useful additional parameter to improve model precision when predicting 'near natural' conditions, or reference conditions in carbon-rich lakes. They found that a simple parameter (derived from UV-Vis spectroscopic relationship) linked to water colour and humic substances was a better predictor of TP than the currently employed models ($r^2 = 0.585$ vs $r^2 < 0.01$). This suggests that a water colour-related parameter should be incorporated into an updated predictive model for lakes from peat or carbon (C) rich soil areas (Vinogradoff and Oliver, 2015).

Some lake type standards may need more development than others. Research on lakes in Scotland found that current P prediction models (used to determine reference conditions) can have low precision in C-rich areas and suggests that amendments to predictions of reference conditions may be needed for C-rich lakes (see Section 4.5.2). Marl lakes also represent a more sensitive lake subtype requiring derivation of separate, more appropriate EQS, however further work may be needed in this regard (Free et al., 2016).

4.5.3 TraC

TraC DIN standards are dependent on salinity and turbidity. The UKTAG derived a salinity gradient from the freshwater to the salt-water end of a water body. Coastal waters are defined as located within 1 or 3 nautical miles of the coast or having a salinity of 30 to 34.5 ppt. Transitional waters (estuaries) are generally described by a salinity of <30 ppt. Recognising that nutrient enrichment is controlled by the attenuation of light within the water body, which in turn is controlled partly by the amount of suspended matter in the water column, the UKTAG established nutrient thresholds for three types of water bodies, based on the level of turbidity (WFD UKTAG, 2008b). The thresholds based on winter mean nutrients would be assessed first. If these are met the status of the water body is at least good. If the threshold for good status is exceeded for a transitional water, then the turbidity related value is brought in and the water body downgraded to moderate only if this too is failed (WFD UKTAG, 2008b). No further information was found in the literature review pertaining to type-specific factors used to calculate DIN in TraC waters.

4.5.4 Summary

4.5 Are the type- and site-specific components effective, and should other factors be considered?

Rivers

- In some catchments, the alkalinity component may provide misleading calculations of reference conditions for rivers that are highly impacted by effluent, which may provide a misleading representation of ecological status. However, this warrants further review owing to the limited information reviewed on this topic.
- The consensus across the evidence base collated was that the inclusion of additional factors like modification impact, flow, climate and clarity (e.g., turbidity/suspended sediment) should be explored.

Lakes

- The type-specific approach for lake TN standards is generally considered effective, however it may be worth exploring the impact of including physical modification typologies in the models.
- Including a metric of colour/humic substances for the TP lakes in England may improve thresholds for lakes from peat or C rich soil areas.

Trac

- No information was found relating to the type-specific factors used to calculate TraC DIN standards.

4.6 Do current standards use the most appropriate metric?

The statistics/ metrics used for the nutrient standards in England and NI are summarised in Table 4.7. For P standards, most EU MS surveyed by Poikane et al. (2019) used mean/ median metrics for rivers, growth season mean/ median for lakes and a variety of metrics (annual, summer and winter mean) for TraC waters (Poikane et al., 2019). For N standards, the annual mean was most common in rivers whereas the growth season mean was the most common metric applied in lakes. In TraC waters winter mean DIN and summer mean TN were most common (Poikane et al., 2019).

Of the workshop participants that were confident to answer (i.e. did not select 'not sure') 53% (n=15) felt the annual mean was an appropriate statistic for P, and 73% (n=15) felt the annual mean was an appropriate statistic for N.

Table 4.7. Statistics (metrics) currently used for each standard in England and NI

Water body/ nutrient type	Statistic used to set thresholds
River RP	Annual mean
River and lake Total Ammonia	90^{th} percentile
Lake TP	Annual mean
Lake TN	Annual mean
TraC DIN	Mean (clear waters) or the 99^{th} percentile (intermediately turbid, turbid or very turbid waters)
TraC unionised ammonia	the long-term mean (with no definitions of long-term provided in the WFD 2015 Regulations)

4.6.1 Is the 90th percentile suitable for the total ammonia standards?

Ammonia standards were developed on the basis of ammonia conditions being associated with macro-invertebrate communities. The frequency of high, potentially toxic concentrations was considered a better indicator of ecological risk which is better represented by a percentile than mean values that can be heavily influenced by a small number of samples with a high concentration. Data for thousands of sites of 'Good' biological quality were used and the value achieved by 90% of the sites (the 90th percentile) was selected as the standard (WFD UKTAG, 2008a).

Some concerns have been raised about how appropriate the 90th percentile metric is when used for classification based on a low sampling frequency (6.Appendix C6.Appendix D). The WFD Regulations do not set a minimum requirement for the number of samples which underpin the classification. Whilst samples have typically been collected monthly, there may be occasions when samples are only collected quarterly, however the 90th percentile is typically recommended for sample sizes over 30, and not suitable for sample sizes below 10. The approach was developed on the basis that compliance is assessed over 3 years with monthly sampling which equates to 36 samples, however sampling frequency has declined in the last 15 years.

4.6.2 Annual mean compared to growing season means

The use of an annual mean statistic for lake TN reflects overall conditions in the lake (WFD UKTAG, 2019b). While loadings to the lake are likely to vary seasonally, it is not just summer nutrients which drive growth, since the residence time of water in many lakes means that inputs prior to the growing season will still be available in spring (WFD UKTAG, 2019b).

Whilst river P EQS are determined based on annual mean concentrations, many studies highlight the importance of P to periphyton and benthic diatom growth during the low flow, summer months (Tappin et al., 2016). Some workshop participants argued that whilst the annual mean is effective for regulatory and analytical purposes, it may not be the best metric for determining ecological risk, which is more dependent on peak and seasonal P concentrations. Therefore, using a summer mean for P EQSs (lakes and rivers, and potentially future TraC) instead of an annual average (as is currently used for river TRP and lake TP) may be more relevant to the ecological response, however this was explored (and discounted) by UKTAG during the 2012 update.

Under the current sampling regime, EQSs tend to be set in terms of measures of annual central tendency owing to the limited sampling frequency of the Environment Agency dataset. However, seasonal variations in nutrient concentrations and ecological growth periods means that nutrient pressure is not uniform all year round. For example, nutrient concentrations during low flows in the summer months may be more important as they are more likely to trigger algal blooms (6.C.3). However, restricting assessment to summer concentrations under current monitoring approach, would mean they were based on a more limited dataset and therefore incur more uncertainty (6.C.8, 6.C.6). Using the summer mean would therefore only be statistically viable if the Environment Agency sampling frequency was increased and potentially focused on the growing season. However, it was also recognised that focusing on the growing season (summer) may miss the crucial winter P loading, which significantly affects nutrient concentrations and primes the system for summer (6.C.6,6.C.4).

4.6.3 Potential of nutrient ratios for setting thresholds

Some interviewees stated that individual nutrient concentrations are not fully informative (e.g., regarding biologically available P). The ratios of different nutrient fractions (e.g. C:N, N:P) are critical as small changes in these ratios can drive significant ecological shifts. Interpreting these ratios can therefore improve understanding relating to ecological shifts (6.C.3). Changes in nutrient ratios can also affect toxin production and community composition. However, others feel that whilst there is value in calculating nutrient ratios, they are only relevant where there are very low nutrient concentrations, and therefore absolute concentrations should remain the focus (6.C.5).

4.6.4 Would load-based standards provide a valuable addition?

The current approach relies on linking a nutrient concentration to a biological response. However, it has been suggested nutrient loads (flow volume multiplied by nutrient concentration) could provide a better understanding of nutrient dynamics and their ecological impacts than concentrations alone, as this considers the total amount of nutrients entering a system rather than just their concentration at a specific time (6.C.9; C.4).

Accurate measurements of flow rates at the time of nutrient concentration measurement are essential for calculating load estimations (6.C.3). Total maximum daily loads (TMDLs) are currently used in the USA, which may be more suited for protecting coastal water bodies or lakes (6.C.7). A TMDL can be thought of as an estimate of the total amount of pollution a waterbody can assimilate without exceeding water quality standards. However, it is important to recognise that with a low sampling frequency, as is common in some Environment Agency datasets used for EQS assessment, there can be significant uncertainty in load estimation (e.g. Lloyd et al., 2016; Skeffington et al., 2015; Halliday et al., 2014). This would need to be considered in detail before any load based EQSs could be derived. Furthermore, this approach would likely require a significant increase in flow monitoring, which is unlikely to be financially viable.

4.6.5 Summary

4.6 Do current standards use the most appropriate metric (e.g., annual mean, percentiles)?

- Whilst summer mean nutrient concentrations may be more relevant to the ecological response, they would only be statistically viable if the Environment Agency sampling frequency was increased.
- Load-based thresholds may provide a valuable addition to concentration-based thresholds for some river types; however, this may not be financially viable if a significant increase in flow monitoring is required.

4.7 Are current standards developed using the most appropriate biological metric?

Under the WFD, boundaries for “supporting elements” such as nutrients should ideally be linked to boundaries between ecological status classes for one or more biological quality element (BQE) (Kelly et al., 2022). A significant body of research has contributed to developing and intercalibrating biological indicators to assess the impact of eutrophication in rivers, lakes and in TraC waters (Borja et al., 2013; Garmendia et al., 2013; Marbà et al., 2013). It is generally accepted that photosynthetic biota (e.g., phytoplankton and macrophytes) represent the most suitable BQEs for the assessment of eutrophication as they typically display the most direct responses to nutrient conditions compared to higher trophic levels (Poikane et al., 2019a). However, in some cases heterotrophic organisms (e.g.

benthic invertebrates, fish or zooplankton) might also prove effective, as they may respond better to secondary effects of eutrophication such as hypolimnetic deoxygenation or habitat alteration.

To classify the BQE, an Environmental Quality Ratio (EQRs) is used. The EQR incorporates the key WFD requirements for ecological classification: typology, reference conditions, and class boundary setting and calculates the relationship between pristine (type-specific) reference conditions and observed values (with 1 being high status and close to zero being bad status). One of the pre-requisites for the intercalibration of ecological EQRs is that they should show a significant relationship with a pressure gradient (i.e. nutrient gradient), thus it would also imply that the same relationship could be used to set nutrient thresholds that support ecological health (Philips and Pitt, 2016). However, Kelly et al. (2022) raised the question of whether metrics that are developed as broad indicators of ecological integrity (e.g., BQEs) are appropriate for deriving nutrient standards. Whilst the relationship between nutrients and ecological status cannot be ignored (as they provide the basis for WFD ecological assessment), Kelly et al. (2022) propose that there is a case for developing alternative metrics focussed on individual stressors. It is possible that variants of metrics would be capable of filtering out some of the “noise” and permit purer insights into biology-nutrient relationships.

The BQEs are also important for determining reference conditions. In some water body types (e.g. lakes) the sediment record can be used to establish historical baselines that serve as reference points (Bennion et al., 2004); however, such opportunities are rare and it is often easier to identify modern locations which are close to their natural state (e.g., catchments with low population density and an absence of significant human activity) (Pardo et al., 2012). However, finding unaltered/ pristine water bodies (lakes, rivers and TraC waters) in England is challenging.

Phytoplankton (using diatoms as a proxy) are currently used for calculating the lake TN and TraC DIN EQSs, and for rivers either macrophytes or phytobenthos EQR (whichever is lowest scoring) are used to determine RP thresholds (Table 4.8.).

Table 4.8. A summary of the biological indicator used to develop current thresholds

Water body/ nutrient type	Biological indicator used to develop current thresholds
River RP	Lowest scoring of either macrophytes or phytobenthos EQR
River Total Ammonia	Macroinvertebrate communities
Lake TP	No pressure-response was used, however expert judgement considered biological elements such as the biomass of phytoplankton (as chlorophyll a), the taxonomic composition of macrophytes, and changes to diatoms preserved in sediments of lakes.
Lake TN	Phytoplankton EQR
TraC unionised ammonia	Fish. EQS was based on a combination of acute toxicity data to fish and threshold concentrations inferred from field data.

4.7.1 Rivers

In a UK-based study, Willby et al (2012) found correlations between the river macrophyte nutrient index and SRP ($r^2 = 0.48$) and Total Oxidised Nitrogen (TON) ($r^2 = 0.58$). Similarly, Donohue et al. (2006) found relatively strong Spearman Rank correlation coefficients (rs) between river macroinvertebrates (Irish Quality Rating System) and SRP (rs = -0.587, p<0.0001). Evidence from the literature broadly supports the use of diatoms as well as the combined approach of diatoms and macrophytes. In low alkalinity streams, Poikane et al. (2021) found fairly strong relationships between SRP concentration and macrophytes (r^2 = 0.41 and 0.42 respectively), which increased to r^2 = 0.48 using the current combined methods (lowest scoring) and to r^2 = 0.49 when using the average of both scores.

However, in some cases the relationship between diatom metrics and nutrients can also be weak. Poikane et al. (2021) found weak correlation with biology and SRP in high alkalinity streams and Jüttner et al. (2012) found weak correlations between TDI and NO_3-N (r^2 = 35.7, $p<0.001$) and PO_4-P (r^2 = 15.6, $p<0.05$) in the River Taff and Ely catchments which was similar to the relationships between diatom EQR and NO_3-N (r^2 = 36.3, $p<0.001$) and PO_4-P (r^2 = 17, $p<0.05$). Poikane et al. (2019) found that there was a high amount of unexplained variability in the macrophyte-nutrient relationship owing to intrinsic factors like water body alkalinity, depth, size, colour, and inter-annual climate fluctuations. Furthermore, it is recognised that hydromorphological factors and other stressors can impact macrophyte communities directly, which adds to the uncertainty in setting nutrient thresholds using this BQE (Poikane et al. 2019).

Poikane et al. (2021) found that the combined macrophyte/ phytobenthos models (calculated as the minimum of the EQRs of the two organism groups) gave the most stringent predictions for the G/M threshold. This supports the current method employed in England and NI for determining the river P thresholds. However, when applying the revised reference model for diatoms Kelly et al. (2020) argue that averaging the two sub-elements of the “macrophytes and phytobenthos” biological quality element is a more realistic option than the current approach of taking the lower of the two assessments.

The reference model underlying the UK phytobenthos (diatom) tool for WFD assessments was revaluated by Kelly et al. (2020). They proposed a new approach which uses quantile regression to predict the lowest values of the Trophic Diatom Index (equating to the best available condition) at any level of alkalinity. Whilst a reference model based on least disturbed or minimally impacted conditions would be preferable in theory, in practice the absence of lowland high alkalinity streams in a minimally impacted condition in the UK precludes the use of these approaches (Kelly et al., 2020). The revised reference model for diatoms therefore provides a better sense of the expected ecological response to nutrients and therefore a better indication of what an appropriate EQR should be, however this was

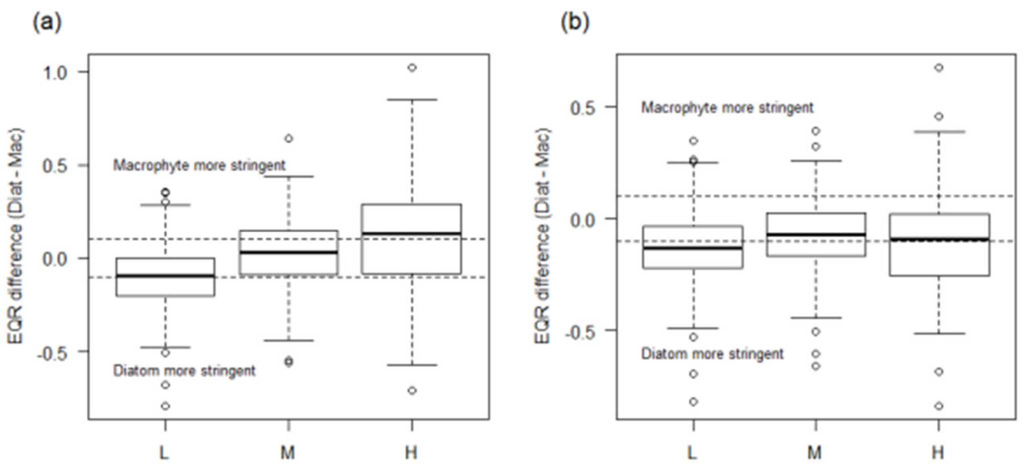

Figure 4.5. Difference between diatom (Diat) and macrophyte (Mac) EQR values using (a) current TDI reference and (b) new TDI reference, split by alkalinity type (L = < 75 mg $CaCO_3$ L^{-1}, M = 75 - 125 mg $CaCO_3$ L^{-1} , >125 mg $CaCO_3$ L^{-1}). Horizontal lines mark ±0.1 EQR units i.e. 1 WFD class (Kelly et al., 2020).

developed too late to be included when developing the current standards. This means there is an opportunity to use this model to provide better calibrated methods for determining river P thresholds (in terms of setting future thresholds or revalidating current thresholds) (6.C.8).

Whilst this approach may produce slightly stronger correlations with P, it may also result in slightly less stringent threshold values. However, the main rationale for using the updated reference model is to make the thresholds more meaningful across a range of alkalinities. Using the current UK TDI reference model, diatoms tended to be more stringent at low alkalinity and macrophytes at high alkalinity (Figure 4.5a). However, the new diatom reference model shifts this balance, leading to consistently more stringent classifications being obtained using diatoms across the entire alkalinity range (Figure 4.5b). Therefore, applying the new reference model would likely result in only diatoms being used (following the current approach of using the most stringent of the two organism groups), therefore making the macrophytes redundant, unless they are averaged. Kelly et al. (2020) demonstrate an uplift in model sensitivity from considering both macrophytes and diatoms together when using the updated diatom reference model.

In many ecosystems, nutrients are not the only factors affecting biological communities. This can be especially significant in rivers, where phytobenthos and macrophytes respond to many other pressures (e.g., grazing, shade, and hydromorphology) meaning that rivers have a less well-defined coupling between nutrient concentrations and ecological response than lakes. This means that simple pressure-response models (e.g., linear regression models) will result in threshold concentrations with very large uncertainty bands (Kelly et al., 2022). G. Phillips acknowledged that the process of developing the current river P thresholds was very challenging as it was difficult to obtain relationships between P and biology. Therefore the relationships used to determine the thresholds were fairly weak and had a significant amount of noise (6.C.6).

4.7.2 Lakes

Evidence supports the use of phytoplankton/ diatom metrics when determining nutrient thresholds in lakes, with chlorophyll a, Plankton Trophic Index (PTI), and cyanobacterial biovolume exhibiting strong relationships with TP concentration (Thackeray et al., 2013). For 70 surveillance lakes in Ireland, the relationships were most significant with TP and phytoplankton (r^2 = 0.65) followed by, macrophytes (R^2 = 0.65), phytobenthos (r^2 = 0.59) and fish (r^2 = 0.32). Following normalisation of the EQRs (to a 0-1 scale) a higher correlation was achieved (r^2 = 0.84) by averaging together the results for phytobenthos, phytoplankton and macrophytes (Free et al., 2016).

In Europe’s central Baltic regions, significant relationships were observed between macrophyte status and nutrient concentrations (TP and TN) by using univariate and multivariate models (combining TP and TN) (Poikane et al., 2019b). However, the relationships are generally weaker than for phytoplankton metrics (Carvalho et al., 2013; Dolman et al.,2016; Lyche-Solheim et al., 2013). Poikane et al. (2014) suggest class boundaries for determining the ecological status of lakes should align with specific thresholds in mean growing season chlorophyll-a (relating to phytoplankton), which would put the G/M boundary at 10-12 µg/L in moderately deep lakes (mean depth 3-15 m) and at 21-23 µg/L in shallow lakes (<3m). Such thresholds have robust ecological consequences for lake functioning and which, therefore, provide strong and objective targets for sustainable water management in Europe.

Although the current approach to threshold setting for lake TP was not based on a pressure-response relationship with a biological metric, evidence suggests this may be achievable (dependent on match data availability). Bennion et al. (2014) investigated the relationship between lake trophic diatom index

(LDTI[25]) and TP. LTDI score and nutrient variables were strongly correlated in medium alkalinity and high alkalinity lakes (TP: r = 0.46 and 0.68, respectively (p<0.01)), but weakly correlated in low alkalinity lakes (TP: r = 0.29, p<0.01), where the LTDI was more closely associated with pH and alkalinity (r = 0.55 and 0.40, respectively (p<0.01)). The authors suggested setting the G/M boundary at the point at which nutrient-sensitive and nutrient-tolerant taxa were present in equal relative abundances. For example, they found that TP concentrations over 20 µg/L resulted in a reduction of sensitive taxa and a relative increase in tolerant taxa in medium alkalinity lakes (although in low alkalinity lakes diatoms showed little response along the TP gradient). However, the relationship between diatoms speciation/abundance with silicate can be important in some lakes which may undermine the general applicability of diatoms indices to lake classification.

Some authors support the use of diatoms over macrophytes when determining eutrophication impacts of ecological health. This is because indices derived from macrophytes may not provide a direct indication of water-column nutrient concentrations because most macrophytes derive the majority of their nutrients from the sediments, whereas the epiphytes rely on nutrients in the littoral water and are considered the primary scavengers of water column nutrients (Wetzel 2001). Furthermore, macrophytes responded more slowly to in nutrient load reductions compared to phytoplankton and provided significantly lower lake quality ratings than phytoplankton in German lakes (Eigmann et al., 2016).

4.7.3 TraC

Evidence supports the use of phytoplankton for determining DIN thresholds in TraC waters, however there may be updated metrics/ methods that could be applied if conducting future threshold updates. Increased nutrient input to estuarine environments can promote phytoplankton blooms but may also modify the phytoplankton community structure which can have adverse ecological impacts throughout the food chain (Ní Longphuirt et al., 2019). Ní Longphuirt et al. (2019) used data from the RoI to develop a new multi-metric index for assessing the status of phytoplankton communities which was designed to encompass not only biomass and bloom frequency but also community structure (diversity and evenness) and abundance. The new index performed well against current methods to determine ecological status and provided improved agreement with other physico-chemical and biological WFD parameters, allowing a more detailed assessment of the impact of disturbance on the system. The inclusion of community structure acknowledged the imbalances in the phytoplankton communities of some systems even when frequent blooms are not evident.

The current unionised ammonia standard was based on a combination of acute toxicity data to fish and threshold concentrations inferred from field data, and as a Specific Pollutant it is determined on a pass/fail basis and does not correspond to a range of ecological classes (e.g. High, Good, etc.). The AZTI's Fish Index (AFI) was found to have a significant negative correlation with mean bottom ammonia concentration explaining a significant amount of variance ($r^2 = 0.69 – 0.78$, $p<0.00001$) (Basque country, Spain), with ammonia concentrations over 10 µmol/L found to produce a rapid decrease in quality (Uriate and Borja, 2009). However, a linear relationship cannot be used establish thresholds, but instead thresholds should be determined after intercalibration with other methodologies, as has been undertaken with benthic communities.

[25] LDTI was developed based on the trophic diatom index and used to generate EQRs for the low medium and high alkalinity lakes in the study (228 UK lakes).

4.7.4 Summary

Are current standards developed using the most appropriate biological metric?

Rivers

- An updated reference model for diatoms is now available which could provide better calibrated methods for determining river P thresholds than the current approach.
- Evidence suggests that averaging the two sub-elements of the macrophytes and phytobenthos BQE is preferable over the current approach of taking the lowest score of the two assessments, particularly if using the updated reference model.

Lakes

- Evidence supports the current approach using of phytoplankton/ diatom metrics when determining nutrient thresholds in lakes.

Trac

- Little evidence was obtained for TraC waters which may warrant further review.

General points

- Whilst the BQE EQRs provide a likely candidate when developing a pressure-response relationship, they may not provide the most statistically robust option for determining thresholds, and therefore other metrics (or combinations of metrics) should be explored when developing (or revising) thresholds.

4.8 Are current thresholds developed using the most appropriate statistical methodologies?

Determining effective boundary conditions (especially for the G/M boundary) is key to the effective protection of ecological health. Nutrient thresholds are linked to the regulatory regime, and significant financial implications necessitate establishing approaches that are statistically robust and understandable at all organisational levels and by the wider public (Kelly et al., 2022). If thresholds are set too low, water bodies may appear to have GES based on nutrient concentrations but fail ecological surveys (false positives). Conversely, if thresholds are too high, water bodies may seem to fail based on nutrient concentrations, even if there is no detrimental impact on the ecology (false negatives). This misalignment can divert investment away from critical areas to those where remediation may have a negligible impact on ecological health.

There are numerous methods that can be employed to estimate and define nutrient thresholds, and it is therefore important to consider the advantages and limitations associated with the different approaches (Poikane et al., 2021). Setting effective boundary conditions relies on three main criteria (Poikane et al., 2021): (1) the nutrient fraction having a suitably strong relationship with the EQR (and/oror other biological metrics); (2) appropriate statistical methods are used to determine boundaries; and (3) the development of a clear understanding of the uncertainty associated with the chosen methodology, and the likelihoodof misclassification. Poikane et al. (2019) highlighted that nutrient thresholds do not represent a ‘line in the sand’ but rather a zone in which the confidence of achieving a prescribed outcome (e.g. GES) varies.

Setting nutrient targets for aquatic systems is a complex process, especially in dynamic environments such as rivers and TraC waters, where multiple pressures impact ecology (Salas Herrero et al., 2019). Applying different approaches to the same dataset can yield a wide range of potential threshold values with varying implications for regulators. Therefore, developing nutrient thresholds must include rigorous validation steps to ensure regulatory boundaries are robust (Kelly et al., 2022). These steps may involve checking threshold estimates against published values, boundaries used by other countries with similar water bodies, and examining the condition of other biota components (Kelly et al., 2022; Piroddi et al., 2021). The data that underpins threshold development often contains considerable uncertainty and heteroscedasticity[26], complicating the use of simple statistical methods. It is therefore important to develop an understanding of underlying ecological processes and consider multiple stressors in setting nutrient thresholds (Kelly et al., 2022).

A wide variety of methods are used to establish threshold values across EU MS. These can be broadly divided in data-driven (regression, modelling, distribution of classified water bodies) and expert-judgement-based (e.g. as arbitrary divisions of distribution of nutrient concentrations in all water bodies) (Poikane et al., 2019). Ecology-based approaches that examine the statistical relationship between nutrient concentration and biological variables, via statistical techniques such as univariate regression and mismatch analyses can enhance sustainable river management, especially where nutrients are the main obstacle to achieving GES (Poikane et al., 2019).

Meta-analysis of thresholds for similar water body types found that the highest (least protective) threshold values were typically determined for lake and river P when assessing the distribution of nutrient concentrations in all water bodies, followed by expert judgement. In contrast, the lowest (most protective) thresholds were devised from regression analysis and modelling (Poikane et al., 2019). Recent EU technical guidance enables countries to review/establish the thresholds they use for P and N to achieve GES (Phillips et al., 2024). Poikane et al. (2019) have simplified the various approaches to threshold setting into six broad categories which are summarised below. A summary of methodologies employed to derive the current standards in England and NI is presented in Table 4.9.

1. Regression analysis between nutrient concentrations and biological response. This was the most common approach for lakes in EU member states (Philips et al., 2018, 2024a).

2. Modelling (e.g., using alkalinity and depth to predict lake TP).

3. Distribution of nutrient concentrations in water bodies classified (using ecological criteria) as high, good and moderate status (Phillips et al., 2018).

4. Distribution of nutrient concentrations in all water bodies – whereby nutrient criteria are defined from an arbitrary percentile of the distribution of nutrient concentrations from all water bodies (Dodds and Welch, 2000).

5. Expert judgement, which may take inspiration from older directives (e.g., the use of 5.65 mg/l of nitrate in freshwater bodies which was derived from drinking water standards). This was the most common approach for rivers and transitional waters in EU member states.

6. The OSPAR Comprehensive Procedure is applied for TraC waters, whereby waterbodies are considered as ‘Eutrophication problem areas’ if their actual status deviates 50% or more from reference conditions. This was the most common approach for coastal waters in EU member states.

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[26] Heteroscedasticity (also know as heterogeneity of variance) describes a situation where the variance of the errors (or residuals) in a regression model is not constant across all levels of the independent variable(s).

4.8.1 Best Practice for establishing nutrient EQSs in the EU

To help MS achieve GES in surface waters, various statistical approaches were proposed in the Joint Research Centre (JRC) Best Practice Guide for Establishing Nutrient Concentrations to Support Good Ecological Status (henceforth referred to as the guidance) (Phillips et al., 2018). This guidance complements previous EU common implementation strategy (CIS) guidance on eutrophication assessment (European Commission, 2009) by providing more targeted advice on how to link nutrient concentrations in surface waters to specific policy objectives. The guidance is designed to assist regulators when determining appropriate concentration of P and N that are likely to support GES and can be applied to setting new boundaries as well as validating existing thresholds.

The second edition (Phillips et al., 2024a) sets out a five-stage workflow (1. setting objectives, 2. feasibility check, 3. boundary estimates, 4. assess misclassification rates and 5. validation) and includes more guidance on understanding the influence of other stressors on biological status, including how to determine which stressors within a dataset explain most variation in the biology, and also suggests some approaches for setting thresholds in the face of other stressors. Specifically, it incorporates a new statistical methodology which uses binary logistic regression to set nutrient thresholds (dependent on data characteristics) and applies a confusion matrix to compare observed and predicted classifications (Phillips et al., 2024b).

The second edition of the guidance only focuses on two approaches for calculating thresholds (Type II Regression and Binary Logistic Regression) but acknowledges that the other approaches described in the 2018 guidance may still be appropriate in certain circumstances (Phillips et al., 2024a). The guidance is supported by a toolkit (Excel Workbook) which contains R-scripts to facilitate the necessary statistical analysis and visualisations (Phillips et al., 2025) and is also supported by a web-based 'Shiny' application which provides an interactive interface to the statistical models used to estimate boundaries (JRC, 2024).

The statistical point at which thresholds are set (e.g. mean or quantiles/confidence intervals) can impact the effectiveness of the G/M threshold. Common practice is to set the most probable threshold at the point where the biological threshold intersects the chemistry (mean) (e.g., Figure 4.6a Figure 1a below), however there are two additional approaches which can provide contrasting advantages and limitations (Kelly et al., 2022):

1. **Most probable threshold (current approach)**: This target is based on regression best fit lines or the mismatch approach and sets the mean nutrient concentration as the threshold (e.g., Figure 4.6a). It has a 50% likelihood of achieving GES but poses a moderate risk of downgrading water bodies even if their biological status is good, due to the 'one out, all out' principle.

2. **Less stringent threshold**: This target is determined using either the upper quantile of linear regression residuals or a higher probability value from logistic regression (e.g., Figure 4.6b). It classifies only 25% of water bodies as not achieving GES based on nutrients when their biological status is good. While it reduces unnecessary downgrades, it offers low precaution. This approach is ideal for regions with many water bodies not meeting good status, prioritising remediation and highlighting the importance of nutrients relative to other pressures. However, it is less suitable for preventing deterioration in water bodies already at good status.

3. **Most stringent threshold**: This target uses lower quantiles of linear regression residuals, quantile regression, or lower probability values from binary logistic regression e.g., Figure 4.6c). It ensures that most water bodies within a type achieve good status but may result in

unnecessary downgrades due to the 'one out, all out' principle, affecting expenditure on measures unless additional safeguards are implemented.

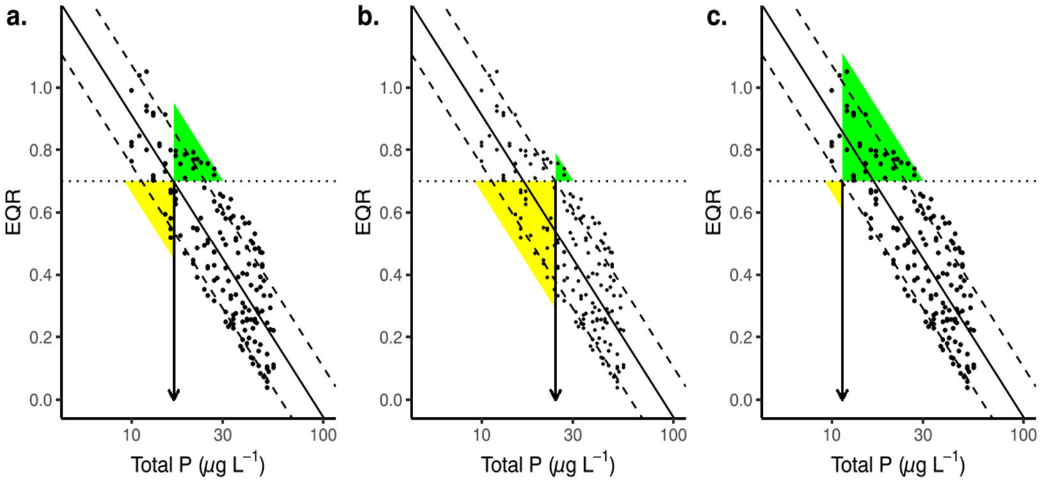

Figure 4.6. Hypothetical relationship between TP and biological EQR, showing the regression line (solid black line) and the confidence intervals (dotted lines). Horizontal dashed line shows the biological G/M boundary (0.7 in this case), and the vertical arrow highlights the intersection of this biological boundary with (a) the regression line, (b) the upper confidence interval and (c) the lower confidence interval. Highlighted areas show where classification mismatch occurs (green: biology Good, but TP Moderate and yellow: biology less than Good but TP good) (Kelly et al., 2022).

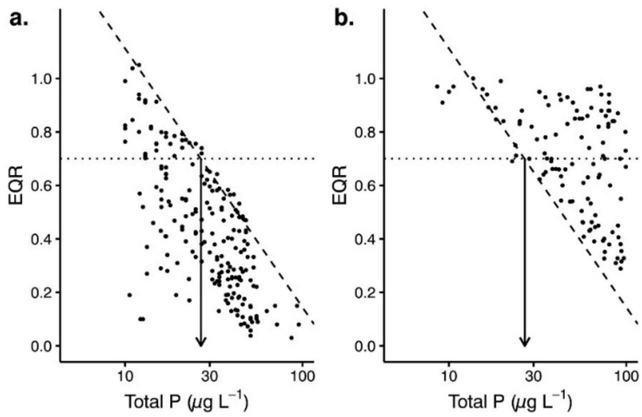

Figure 4.7. Hypothetical relationship between TP and biological EQR where multiple pressures occur. Horizontal line shows the biological G/M boundary (0.7 in this case), and the vertical line highlights the intersection of this biological boundary with (a) upper quantile (95th percentile), (b) the lower quantile (5th percentile) (Kelly et al., 2022).

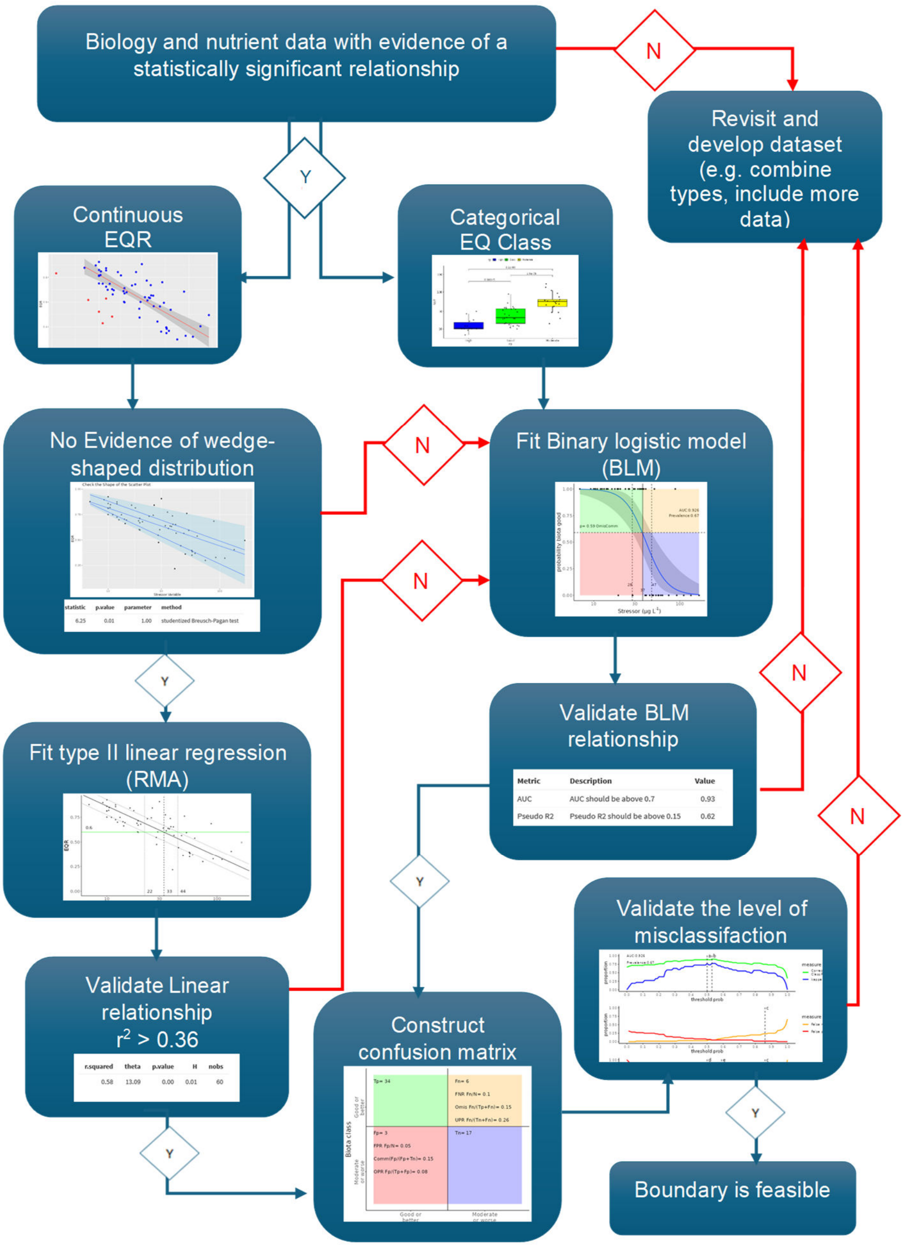

Figure 4.8. Guide to selecting appropriate methods for deriving nutrient boundaries to support good ecological status. Note that other methods may be appropriate for specific circumstances (e.g. when several stressors are present) (Phillips et al., 2024a).

4.8.2 Methodologies used to derive current nutrient thresholds in England and Northern Ireland

For rivers in the UK, P standards are based on models which use the alkalinity and altitude of the site, along with the regression between reactive phosphorus (annual mean) and biological EQR (using the lowest scoring of either macrophytes or phytobenthos). The P standards were developed in line with the 2018 guidance, albeit with an inverted type I regression, which was more precautionary than the normal approach in the guidance (type II regression) (6.C.6).

Within lakes, relationships between nutrients and ecology are fairly strong as nutrients typically represent the principal pressure. This means boundary setting is typically more robust for lakes than other freshwater environments (Kelly et al., 2022).

Lake TP standards (annual mean) were developed using a mathematical model that aimed to predict the reference (natural) level of P for the lake. A factor was then applied (representing the degree of change from reference conditions) to determine threshold values. This was based on expert judgement/ understanding relating to the way biological elements (e.g., biomass of phytoplankton as chlorophyll a, the taxonomic composition of macrophytes, and changes to diatoms preserved in sediments of lakes) respond to increased P. However, this meant the lake thresholds values are not based on ecological principles with biological significance.

The Lake TN thresholds were developed in 2019 and are therefore not included in WFD 2015 Regulations (WFD, 2015). Threshold values were determined using a regression model, based on the linear regression relationship between TN and phytoplankton EQR, including the variables depth (very shallow/ shallow/ deep) and humic type (humic/ polyhumic and clear, based on measured colour). Lake TN standards were developed using the EU Common Implementation Strategy (CIS) guidance toolkit (2009), although there have now been two subsequent iterations of EU-based guidance in this field (Phillips et al., 2018 and Phillips et al., 2024b).

The G/M boundary for DIN in TraC waters is defined by a general increase of 50% in relation to the reference baseline conditions. This relationship was extended to derive boundaries between moderate and poor status, and between Poor and Bad, with all thresholds for TraC waters. UKTAG also suggested that these (thresholds for Poor and Bad) are used for guideline purposes to prioritise action and that they should not be used to classify ecological status - secondary biological effects should be considered before the final status is declared. The thresholds for DIN are therefore not based on ecologically relevant nutrient concentrations.

4.8.3 Alternative methodologies that may improve threshold setting

Advantages and limitations of the current approaches used to set thresholds in England and NI are summarised in Table 4.9 alongside recommendations specific to the given standard. General principles/ recommendations that could be applicable to multiple water bodies and/ or standards are summarised below.

4.8.3.1 Confusion Matrix

The implications associated with the statistical point at which the threshold is set (e.g., Figure 4.6 and Figure 4.7) will become more apparent as the predictive power of the regression equation decreases. This is especially true in rivers and TraC waters where many additional pressures (other than nutrients) impact the ecology. In these cases, the pressure-response relationship between nutrient concentration and biological status have a very high level of uncertainty and likely display a ‘wedge’ shape when plotted (e.g. Figure 4.7). In these cases, regulators often set thresholds using the upper quantile however this typically results in an elevated boundary that is not precautionary (Phillips et al., 2024a).

The updated guidance emphasises the importance of quantifying this uncertainty by using a 2 x 2 matrix (a "confusion matrix”) (Phillips et al., 2024a). The confusion matrix is a visual tool that offer a means of summarising the implications of a proposed boundary quickly and efficiently by quantifying the number of misclassifications, splitting these into cases where the biology is High or Good status but chemistry predicts lower status (false negative) and also where biology is less than Good status but chemistry predicts Good or better status (false positive) (Figure 4.9).

A logical next step following this review would be to evaluate the current thresholds using the recent guidance[27]. To protect ecology, it has been suggested that it is more important to adopt the most precautionary standards instead of the standard most likely to be the same as the biological status (6.C.6). For example, Poikane et al. (2021) applied both linear regression and the 'minimisation-of-mismatch' approach (as per the 2018 guidance) to define the G/M boundary for low alkalinity rivers (lowland and upland) in the Central Baltic region of Europe. In both cases the 'minimisation-of-mismatch' approach made the thresholds significantly more precautionary (changing from 32 µg/L to 21 µg/L for lowland rivers and from 40 µg/L to 32 µg/L for upland rivers) and therefore made the thresholds more protective for ecological health.

4.8.3.2 Recalculate thresholds following most recent guidance

Alternatively, the thresholds could be recalculated, using the most recent data and following the full approach set out in the guidance. Broadly speaking this involves using either Ranged Major Axis Regression (RMA) for linear relationships or Binary Logistic Models (BLM) for non-linear or categorical approaches), followed by a confusion matrix visualisation. G. Phillips suspects that new analysis would not have a significant impact on the majority of site-specific river P threshold values. However, he believes that some of the river P threshold values could be slightly too elevated (not stringent enough) and therefore would benefit from being tested with the new assessment methods that have been developed in the most recent guidance (6.C.6; Phillips et al, 2024a).

Salas-Herarro et al. (2019) tested the applicability of the recent (2018) guidance for deriving nutrient thresholds in TraC waters. The methods tested and the thresholds they produced are described in Table 4.10, which exemplifies the range of thresholds that can be generated depending on the method employed. In summary, the complexities introduced by multiple pressures in TraC water preclude the use of linear regression methods and require the use of categorical approaches (Figure 4.10).

4.8.3.3 Consider the application of multivariate modelling

Multivariate modelling also offers potential for improved modelling and threshold setting. Poikane et al. (2019) demonstrate that multivariate models, which include both TP and TN, explain more variability in macrophyte status than univariate models, highlighting the importance of considering multiple nutrients in ecological assessments. Multivariate models with both TP and TN as predictors had higher r^2 values for high alkalinity shallow ($r^2 = 0.50$) and very shallow ($R^2 = 0.49$) lakes relative to the use of TN ($r^2 = 0.29–0.37$) or TP singly ($r^2 = 0.41–0.46$). Poikane et al. (2022) also found that multivariate ordinary least squared (OLS) regression tended to have higher r^2 values (ranging from 0.37 to 0.80; $P < 0.001$) than univariate regression. Kelly et al. (2020) highlight that until we can improve conceptual models (e.g., by using multivariate modelling approaches) to incorporate multi-stressor effects, we need to recognise and manage the large uncertainty associated with thresholds set by simple regression-based approaches, which result in an increased risk of false positive/ negative classifications.

[27] Note that G. Phillips has offered to run this analysis on behalf of the Environment Agency, if provided with the data, and able to share results with the JRC (6.C.6).

Table 4.9. A summary of the different approaches employed to determine nutrient thresholds in England and NI, along with associated advantages and limitations, and recommendations for future improvements.

Standard (date developed/ updated)	Methodologies employed to derive thresholds	Brief description of the methodologies employed to derive the current thresholds	Advantages of the current methodology	Limitations of the current methodology	Recommendations
River P (2012/13)	Regression analysis	Thresholds were determined using models which incorporate site-specific variables (alkalinity, altitude) against a statistical pressure-response relationship (regression) between reactive P and biological EQR (using the lowest scoring of either macrophytes or phytobenthos).	Whilst both the interviewees who were involved in developing the P standards felt that the current standards were initially fit-for-purpose and were well calibrated using the data available at the time, they believe that the standards are now in need of review due to changing conditions, particularly regarding climate change (6.C.6, 6.C.8).	It has been suggested that a significant proportion of the match data (macrophyte, diatom, and water quality samples, within 500 m of each other) used to develop the river P standards was from NI (around one third - 221 out of 620) (6.C.1). This may have limited how representative the dataset was for the entire UK, especially in some river typologies (e.g., chalk streams and headwater-type streams). River P EQSs are currently set at the 'midway position' representing a concentration at which there is equal statistical confidence of the nutrient and biology EQR being in adjacent classes (e.g. Figure 4.6a) (WFD UKTAG, 2013), however this approach does not necessarily produce the most protective thresholds. The relationships used to determine the thresholds were fairly weak and had a significant amount of noise (6.C.6).	Recalculate thresholds using the most recent data, and ensuring fair geographical representation across broad typologies, which may produce more effective thresholds for certain typologies. Assess how protective current thresholds are using the confusion matrix, as described in the updated Best Practice Guide (Phillips et al., 2024a).
Lake P (2012/13)	Modelling and Expert Judgment	Thresholds were calculated using a mathematical model for predicting reference P values linked to catchment geology and topography (alkalinity and depth), which acted as a proxy for natural background levels. Pragmatic expert judgements were used to determine the proportion of change (e.g., in EQR) for P for each threshold, and for each reference type.	The site-specific approach used in the UK to derive standards for P in lakes is considered to be more precautionary compared to methods used in other EU MS (6.C.6).	Lake P standards were not established from a pressure-response relationship owing to a lack of comparative data at the time. At the time of developing the lake TP EQS, UKTAG commented that if data had been available for thousands of lakes (like rivers), then the same methods used to initially derive standards for rivers (WFD UKTAG, 2008a) would also have been applied to lakes (WFD UKTAG, 2008b).	Recalculate lake P thresholds using the most recent data and define using a pressure-response relationship (if the necessary data is available).
Lake TN (2019)	Regression analysis	Thresholds were determined using a statistical pressure-response regression model, based on the relationship of TN to phytoplankton (along with depth and humic type).	Advantages included the paleoecology techniques as well as the rigorous modelling efforts, the use of large datasets and the incorporation of paleoecology for determining baselines for lakes (6.C.4). The relationships between nutrients and ecology being fairly strong in lakes, have resulted in a model with a relatively strong r^2 of 0.747 (UKTAG 2019b)	Data from NI's lakes were excluded because TN data were not available for the relevant time period (UKTAG 2019a) and therefore the thresholds may be less suitable to lakes in NI than in the England. Consideration of the mismatch at the G/M boundary and the percentage of classifications agreeing to within one class indicated that N performs slightly better than the lake P classification for phytoplankton, but slightly less well for macrophytes (UKTAG 2019a).	If thresholds are recalculated, date from NI should be included if available. Assess how protective current thresholds are using the confusion matrix, as described in the updated Best Practice Guide (Phillips et al., 2024a).
Trac DIN	Distribution of nutrient concentrations in all water bodies	The deviation of the boundary for Good and Moderate is a general increase of 50% in the reference baseline.	The current thresholds have been amended (made slightly more protective) in order to align with the OSPAR convention such that the WFD G/M boundary also represents the difference between a 'nonproblem area' and a 'potential problem area' for OSPAR's Comprehensive Procedure (OSPAR, 2002) .	The mixing line (saline to freshwater) used to calculate thresholds applies to TraC waters in the UK with salinity of 25 and above but should not be used for the upper end of estuaries as it does not reflect any relationship that might apply between biology and nitrogen in freshwater.	Further work may be required for the low salinity parts of estuaries (WFD UKTAG, 2008b). Thresholds may benefit from revaluation in line with the most recent guidance (e.g., using ecologically relevant thresholds determined using binomial logistic regression).

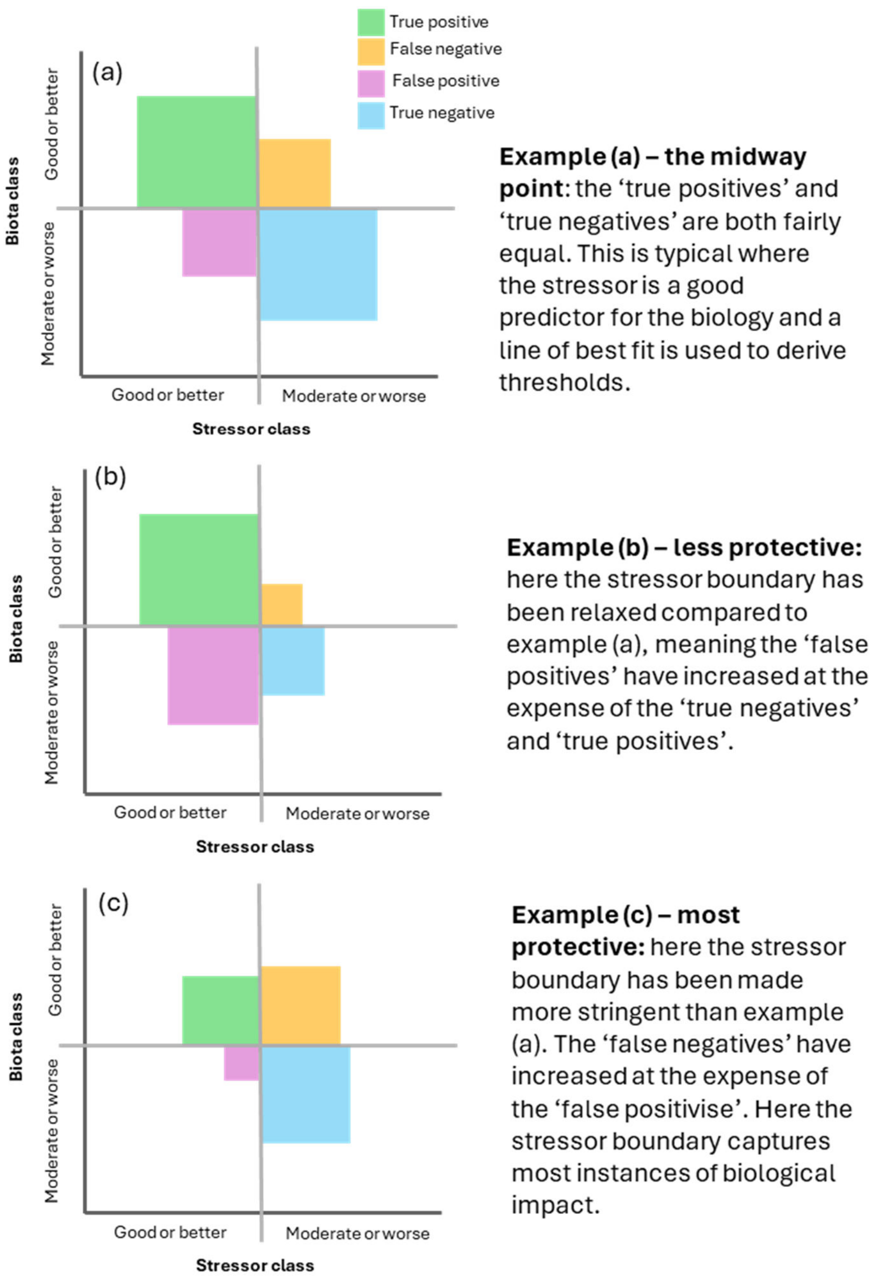

Figure 4.9. Example of a confusion matrix visualisation (adapted from Phillips et al., 2024a).

Table 4.10. Different statistical methods employed to determining G/M thresholds for Transitional waters in the North Atlantic Region by Salas-Herrero et al., 2019.

Method	G/M thresholds	Advantages of method	Issues with method
Linear regression (**Figure 4.10**a)	n/a	n/a	Linear regression was not appropriate for calculating the TraC DIN thresholds owing to the low correlation between DIN and BQE ($R^2 = 0.21$) (see wedge-shaped scatter in Figure 4.10a). In this case only categorical approaches should be adopted for deriving nutrient boundaries.
Binomial logistic regression (**Figure 4.10**b)	79.7 µM*	This is the most reliable and flexible categorical method included in the toolkit, when linear modelling is not appropriate.	Boundary estimates determined by this method may still be unreliable if other pressures are operating.
Quantile regression (**Figure 4.10**c)	212 µM (using the 0.7 quantile)	This method can be applied to datasets displaying a wedge-shaped scatter.	The 95% confidence intervals obtained for the G/M boundary were too wide; indicating that large EQRs variation (ranging from 0.39 to 0.81) could be expected at this concentration of DIN.
Minimise-the-mismatch approach (**Figure 4.10**d)	74.5 µM (mismatch rate of 28%)	This method can be applied to datasets displaying a wedge-shaped scatter and is the least sensitive method to outliers and non-linear relationships.	

** This value represents a 50% probability of being less than good. The more precautionary boundary would be set at a 25% probability of being less than good (32.3 µM) and a less precautionary boundary would be set at 75% chance of less than good (196.4 µM) (*Figure 4.10*).*

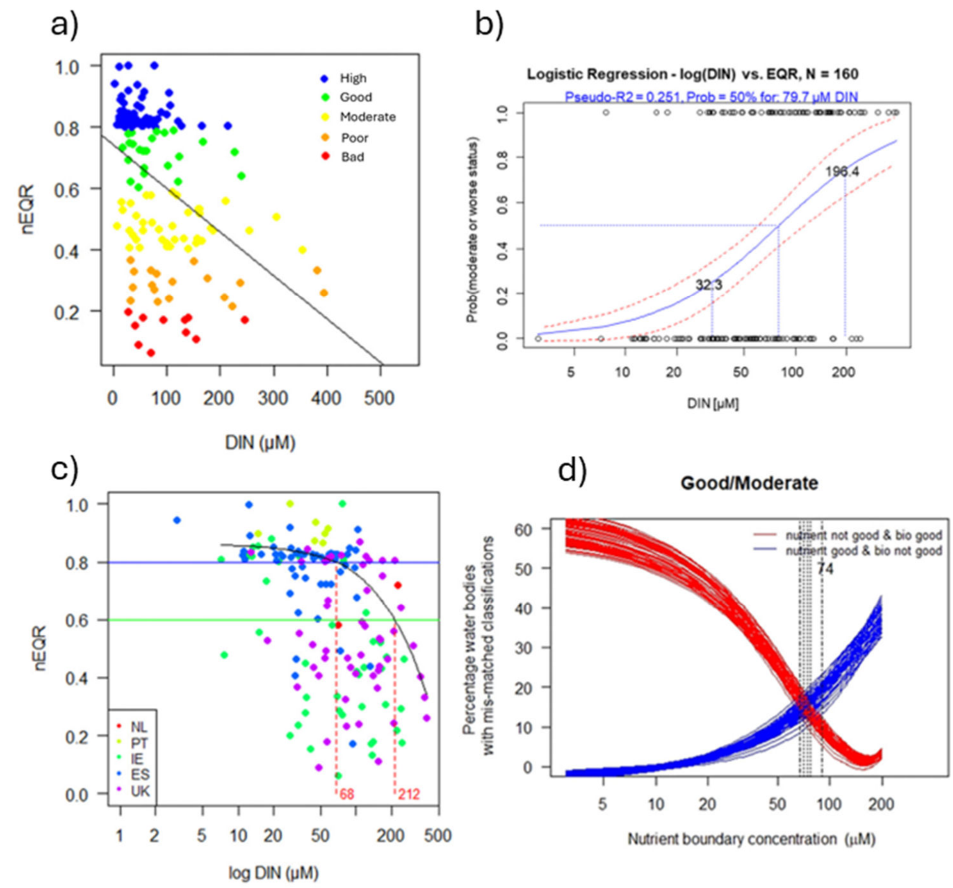

Figure 4.10. (a) the relationships of DIN and intercalibrated EQRs for the NE Atlantic Estuaries, showing a wedge shaped scatter plot and linear trend coloured by ecological status; (b) shows binomial logistic regression, with lines showing potential boundary values at different probabilities of being less than good; (c) quantile regression fit, where horizontal lines indicate EQR boundary at H/G (blue) and G/M (green) and vertical lines show the corresponding nutrient thresholds at the 70th quantile; and (d) shows the relationship between the mis-classified records for the biological and nutrient classifications with vertical lines representing the point where the misclassification is minimised (the G/M boundary) (adapted from Sallas-Herrero et al., 2019).

4.8.4 Summary

Are current thresholds develop using the most appropriate statistical methodologies?

General points

- Since the current thresholds were developed, updated EU guidance has been produced to help MS produce ecologically relevant nutrient thresholds. The current thresholds could therefore benefit from being re-evaluated in line with the most recent Best Practice Guidance.
- Current thresholds could be tested using a ‘confusion matrix’ to determine how precautionary they are.

Rivers

- River P EQS are currently set at the midway position, however they could be more protective for the ecology if set at the 25th percentile.

Lakes

- Lake TP standards are based on a mathematical model, however if the data is available, they may warrant recalculating based on ecological principles (i.e. pressure-response relationships)
- Lake TN standards are based on pressure-response relationships and display relatively strong correlation. However, data from NI was not included in the calculations, as the data was not available at the time. If available, thresholds could be recalculated with the inclusion of NI data.

Trac

- Thresholds for DIN are not based on ecologically relevant nutrient concentrations. Evidence suggests that simple pressure-response relationships (linear regression) cannot be applied to these environments due to multiple stressors. However, other classification-based statistical approaches (as per the EU guidance) could be employed to determine more meaningful thresholds.

4.9 Methodological advancements or improvements that could be used to inform updated nutrient EQSs?

Based on the literature review and interviews undertaken, a number of potential methodological advancements or improvements that could be used to inform updated nutrient EQSs to better reflect ecological health were identified. Within the workshop these were then considered to explore what advantages they may offer, as well as identifying any limitations or caveats around their use/development (Figure 4.11Figure 4.11Figure 4.11).

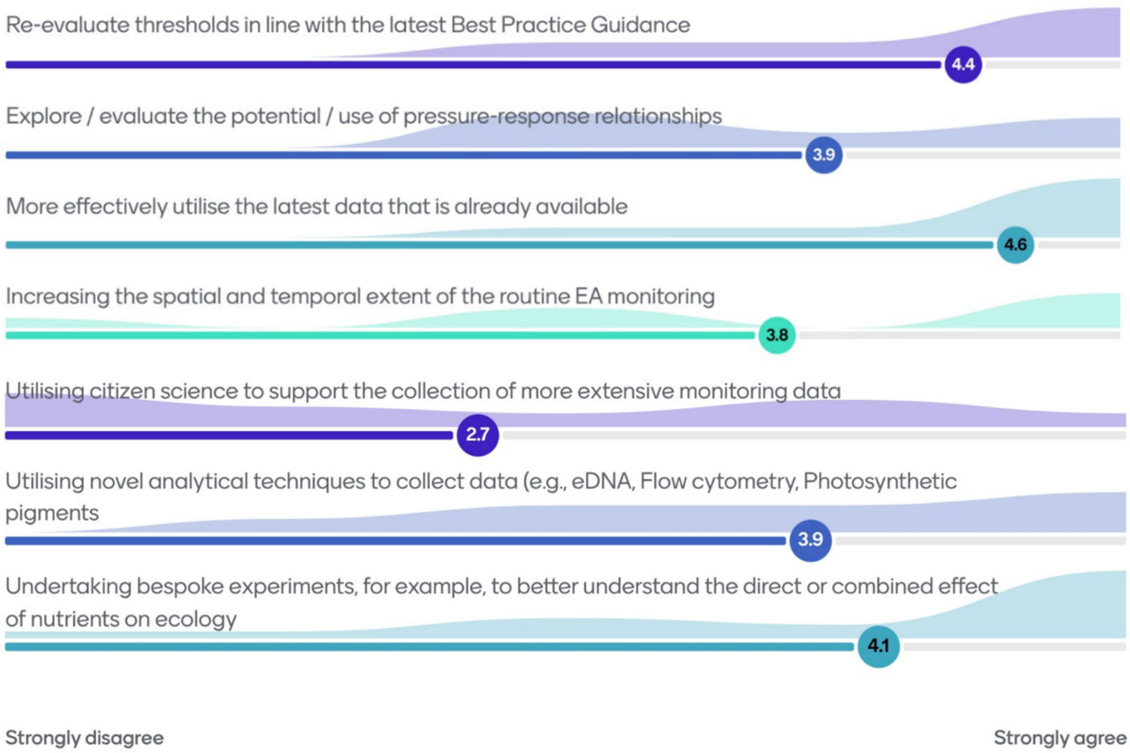

Figure 4.11. Output of the workshop where participants were asked to score a number of potential options to consider when updating/developing future thresholds (with 1 being strongly disagree and 5 being strongly agree).

4.9.1 Re-evaluate thresholds using the most recent data and guidance

Whilst the current standards were developed using the best available data at the time, it is highly likely that more extensive, geographically representative data sets now exist which may be utilised to derive new and/ or updated thresholds. This may take the form of standard regulatory monitoring data (form the Environment Agency, NIEA and SEPA) as well as data generated by academic studies and through consultancies. One problem however is that many research findings remain unpublished due to time constraints and changes in journal publication standards and requirements.

Greater collaboration and data sharing among researchers, private companies, and regulatory bodies could enhance the effectiveness of nutrient standard development and monitoring efforts. Before recalculating thresholds/ developing new standards, it would be worthwhile engaging with researchers to access unpublished data that could inform standard setting (6.C.5)6.C.5. Data from various sources could be combined to create a comprehensive (centrally managed) dataset (6.C.4). Recalculating thresholds with the most recent data would also provide the opportunity to employ the current Best Practice Guidance from the EU relating to setting effective nutrient thresholds (Phillips et al., 2024a).

Another important reason to reassess standards with the most recent data is the impact that moving baselines may have on the classification of water bodies. Anthropogenic climate change results in increased frequency and magnitude of extreme climatic events (e.g., droughts, storms, floods and strong winds) which are likely to exacerbate nutrient pollution effects by increasing the load delivered to the aquatic environment (Malta et al., 2017). Extreme heat events, which were not a major consideration when developing the P standards in 2012/13, are now a significant factor impacting eutrophication (6.C.8). These events contribute to algal blooms, particularly in lakes, however the only practical mitigation is to reduce nutrient levels, making it harder for algae to thrive. Lake nutrient EQSs therefore warrant a review to determine if they are still effective under the current (and future) climate conditions (6.C.8).

It is therefore plausible that previous research may not be comparable to present-day water chemistry and ecology. This moving baseline means it is important to revisit the standards on a regular basis, to make sure they are effective under current conditions (6.C.9). It is also important to remember that nutrient thresholds set in current conditions should not be communicated as static limits as they might need adjustments in the future to counteract the impact of these additional stressors and protect from such future scenarios, which are likely to intensify in the coming years (Kelly et al., 2022; Phillips et al., 2024a).

4.9.2 The potential of laboratory and field scale experimental analysis to improve threshold setting methodologies

Where there are gaps in current data sets (published and unpublished), some interviewees felt that the regulators (or UKTAG) should consider commissioning bespoke experiments to gain the information needed to establish more accurate and realistic standards.

There is a particular need for bespoke experiments to better understand the impact of P on river ecology, as current broad-brush statistical approaches are considered insufficient (6.C.5). One option would be to monitor upstream and downstream of wastewater treatment works following upgrades to enhance their P removal (acting as a natural lab). Assessing around 20 sites in this way, that cover a range of environments would be a valuable way to validate statistical approaches and provide confidence that the P targets are correct (6.C.5).

Experimental analysis such as floating flume experiments and mesocosum experiments (e.g., UKCEH AQUA-REP facility) have also been used to test the biological response (algal growth) to a range of P and N concentrations. Experimental set-ups (i.e. flume and AQUA-REP) could be manipulated to produce wide ranges in P, N and metals, and could be used to cover a range of national river typologies. Another approach to generate robust evidence on TN and TP is to develop sentinel catchments with daily sampling and rigorous QA/ QC. This approach would provide reliable data to demonstrate policy effectiveness (6.C.2).

Nevertheless, results from these types of experiments still need to be tested in relation to the large datasets to determine if the findings can be scaled up (6.C.8). Incorporating new methods and ensuring their relevance at a national level can be challenging and there is a need for data sets that provide comprehensive coverage which can demonstrate the relevance of these methods (6.C.7).

4.9.3 Suggested modifications to the implementation of water legislation

A common theme amongst the interviewees was that the current implementation of the WFD (specially relating to sampling frequency) is insufficient to generate meaningful water body classifications that are representative of ecological status. Although it is widely acknowledged that significantly increasing the budget to allow for increased sampling frequency is unlikely, it is worth discussing, should this become an option in the future.

4.9.3.1 Potential addition of a further “very bad” category

It is important to consider the upper end of nutrient ranges (e.g., the later summer/ early autumn peaks), especially in hyper-eutrophic systems. Once a water body goes over the threshold for Poor/ Bad (e.g. TP of > 140 µg/L), then there is no scaling of how bad it gets. Lakes with 150 µg/L or 1000 µg/L are both categorised as Poor/ Bad, when clearly the one with 1000 µg/L is going to need much greater intervention to improve quality. Given that so many water bodies are now eutrophic/ hypereutrophic, it has been suggested that additional categories may be needed to outline the extent of Poor/ Bad, for example a “Very bad” category (6.C.4).

4.9.3.2 Sampling frequency

There were a number of concerns about the reliability and consistency of data collection. Regulatory data may be patchy, with gaps in monitoring and inconsistencies in methods. This affects the robustness of the standards and the ability to prove their effectiveness (6.C.7).

Adequate temporal and spatial resolution are essential for accurate reporting of ecological health. There are currently no specific requirements (in England or NI) to capture different flow conditions (rivers) when taking water quality samples for classification, which may lead to a lack of variability in data for the full range of flow conditions. The nutrient concentrations obtained via monthly sampling can vary considerably depending on whether samples are collected at periods of high or low flow. This may have an impact on the overall classification of a water body, especially in rivers where flow will have a significant impact.

The low frequency of sampling by the Environment Agency can also bias the data, especially towards fair weather (low flow) conditions which can affect the accuracy of the collected data on P loads and distribution. Understanding transfers of P during high flow events (e.g., heavy rainfall) is crucial as these events can lead to P being deposited in riverbeds and later released, impacting the availability of P to the ecology (6.C.7).

When deciding when to sample, it is also important to consider internal nutrient loading and its impact on concentrations and models. Internal loading typically occurs through the summer (with seasonal anoxia and productivity-driven changes in pH and alkalinity). If the system is stratified, this will be mixed into the water column after the stratification period (varies but typically Oct/ Nov in the UK) (6.C.4). If the lake is shallow and well-mixed, internal loading can impact nutrient concentrations all year. Research at Rutland Reservoir has demonstrated TP values ranging from 34 µg/L in the spring (start of the growing season) and up to 1100 µg/L in the late summer/early autumn when internal loading spikes and algal blooms break down). This means that depending on when the sample is taken, this water body could either be classed as Good/ Moderate or Poor/ Bad (6.C.4).

The ideal solution to address these concerns would be to increase the frequency of regulatory sampling, however this may be unlikely due to financial implications. There may however be alternative methods that could be employed to provide data that is more representative of site conditions in a more cost-effective way, for example using continuous monitoring, improved technologies (e.g., automated sample collection at high flow) and/ or citizen science, which are discussed in more detail below.

4.9.4 New analytical methods that could be utilised

Advances in technology, such as probes for in-situ nutrient measurement, DNA techniques for diatom analysis and flow cytometry/ pigment analysis to determine abundance of blue-green algae could enhance monitoring efforts and inform future nutrient thresholds. These technologies can provide more accurate and comprehensive data, allowing for better understanding and management of nutrient levels (6.C.4). Once the regulators have collected several years of data, there is the potential to investigate the potential for using new datasets to develop and set meaningful thresholds.

4.9.4.1 The potential of high frequency data to improve threshold setting

Studies have found a high degree of statistical bias when low frequency P concentration data were related to ecological thresholds, compared with high-frequency (hourly) data (Jung et al., 2020; Fones et al., 2020). The introduction of high-frequency nutrient monitoring will potentially provide a more accurate methodology for a statutory control of G/M boundaries in streams and rivers due to their ability to capture extreme events (Halliday et al., 2015; Wade et al., 2012).

This can be achieved using in-situ sensors for continuous or high-resolution monitoring of dissolved oxygen, pH and temperature which can improve data quality and resolution. Advances in continuous water quality monitoring technology is ongoing with current technologies utilising a mixture of ion selective electrodes, UV and fluorescence spectroscopy to monitor selected nutrient fractions (mainly ammonium and nitrate). Development in both microfluidic and bank side digestion technologies also facilitate the in-situ measurements of total nutrient pools (TN and TP). Owing to recent developments of in-situ sensors capable of monitoring single and multiple excitation emission wavelength pairs, fluorescence spectroscopy has emerged as a powerful tool for the in-situ measurement of organic matter fluorescence, also providing insights into photosynthetic pigments (discussed further in Section 4.9.4.4) providing insights into algal groups and nutrient relationships. Such probes are becoming routine and more affordable and can measure blue-green algae as well as chlorophyll (6.C.4).

Whilst high-frequency sensor data can provide detailed insights, the datasets are associated with significant uncertainties and therefore a combination of sensor data and laboratory analysis is recommended for robust evidence. Effective utilisation of near real-time water quality monitoring presents challenges as continuous data cannot be applied to standards based on spot samples and therefore new standards would likely need to be derived if real-time monitoring data was to be used for classification (6.C.1).

4.9.4.2 Environmental DNA (eDNA)

eDNA techniques are becoming more routine and can offer a more comprehensive view of biodiversity with reduced subjectivity (6.C.4). eDNA analysis can provide rapid and informative insights into water quality and offers significant value by allowing multiple analyses from a single sample. It can detect indicator species, invasive species, and target species, providing better spatial and temporal resolution than traditional methods (6.C.3,6.C.4). In the United States (US), the EPA and New York Water Bureau have embraced biological monitoring and eDNA more than the UK (US EPA, 2024).

The use of DNA for diatom and invertebrate analysis could provide more rapid and less subjective assessments compared to traditional taxonomy. After compiling several years' worth of data, this could be used to aid the development of nutrient thresholds (6.C.3,6.C.4). Combining eDNA with water chemistry and environmental data can also enable the development of predictive models for water quality. Network analysis can help understand relationships between nutrients and biological communities. Machine Learning and artificial intelligence (AI) may enhance these models for site-specific analysis and allow site-specific EQSs to be developed with meaningful thresholds (6.C.3).

However, it is important that future methods remain comparable to past data to monitor changes effectively. eDNA will give very different values than traditional taxonomic techniques. DNA methods for diatoms are promising but there is the need for continued development and comparison with traditional methods to refine these techniques (Appendix C.4). Additionally, the process may not be able to deliver quick results due to the need for large batch processing (6.C.8).

4.9.4.3 Flow cytometry

The traditional focus was on the relationship with P and chlorophyll a (which is sometime monitored by the Environment Agency) however most problems are associated with blue-green algae (which are not captured by chlorophyll-a) and are not currently assessed. Flow cytometry provides a cheap, easy and efficient method to assess blue-green algae (can easily run 100 samples per day) as it has therefore been suggested that cell counts should be routinely monitored by the regulators (6.C.5). Recent development in flow cytometry technology means that is also possible to take photos of each algae which can then be classified. This process will likely become more accurate and efficient with AI.

Once a few years' worth of flow cytometry data has been accumulated, it could be highly beneficial when developing future thresholds. Furthermore, it may be possible to identify thresholds in light, flow, temperature and nutrient concentrations that initiate and terminate algal blooms, using the Eutrophication Risk Model (UKCEH, 6.C.5).

4.9.4.4 Pigment analysis

Blue-green algae produce different pigments compared to chlorophyll-a and there is potential for pigment techniques to be used to monitor both P and N-related algal blooms. Strong correlations have been found between N and pigments like phycocyanin which is the typically measured pigment in waters and is indicative of total cyanobacteria (blue-green pigment). In lake sediments and palaeo data, pigments such as zeaxanthin, canthaxanthin, myxoxanthophyll and echinenone can be used (which are related to different types of cyanobacteria) (Appendix C.4). This approach can provide insights into the types of algae present and their ecological impacts and is a valuable tool for understanding algal dynamics. There is currently no routine evaluation of cyanotoxins as part of the WFD, despite being hazardous to human and animal health.

Advances in liquid chromatography methods now allow rapid and high-throughput measurement of cyanotoxins, which can be measured easily from the filtrates of water samples. This is routinely done in marine/coastal settings in shellfish areas and is being led by CEFAS. Establishing baselines for cyanotoxin concentration, rather than cyanobacterial counts, would be a far more effective assessment of potential ecological impact of harmful algal blooms. This is work that has already begun at CEFAS. Though pigment analysis provides good potential for monitoring algal blooms, there is a need for ground truthing and the building of comprehensive datasets (Appendix C.4).

4.9.5 Potential for citizen science?

One of the major problems with the current approach is that the frequency of sampling, particularly for freshwater algae, is that it is considered too low to detect short-term events. Similarly, standard macrophyte assessment is done once per season and only conducted once every three years at a particular site, so if there is a problem occurring for a relatively short period (e.g., in the summer) then the methods are not responsive enough. More frequent and responsive methods are therefore needed to pick up these events (6.C.8).

One potential solution is to utilise citizen science approaches to help fill data gaps. Standard Environment Agency techniques are resource intensive and under increased financial pressure, whereas the citizen science approaches are designed to be more cost-effective. There is potentially an optimal zone in the middle-ground between these two approaches, and it has therefore been suggested that the Environment Agency could look to utilise some citizen science principles in order to make sampling more responsive to rapidly changing conditions. This could result from a partnership with the citizen scientists, and/ or the development of a rapid Environment Agency assessment methodology using citizen science approaches (6.C.8).

Citizen science has shown promising results in the river Wye for capturing intra-annual variation in biomass and the impact of major floods (Kelly et al., 2016). The Big Windermere Survey provides an example of successful citizen science utilisation and demonstrates how engaging the public in data collection can provide valuable insights and increase monitoring coverage (Appendix C.4). Citizen science was successfully integrated into the system health monitoring program in Queensland, Australia. This program trained citizens to collect robust data and increased involvement and ownership of environmental issues. It also used innovative sampling methods for chemicals and nutrients. The Riverfly initiative in the UK aimed to train assessors for macroinvertebrate monitoring. This initiative helped increase public involvement in water quality monitoring 6.C.7) and the AQuA Project[28], based at the University of York and funded by the Natural Environment Research Council (NERC) is looking at the role citizen science can play in water quality sampling (6.C.9).

Whilst citizen science can increase data resolution and engage the public, there are challenges related to data quality and consistency. Proper training and quality control measures are essential to ensure reliable data and further challenges lies in collating and maintaining the data for long-term use.

4.9.6 The importance of effective communication and decision making

Some interviewees felt that there is a disconnect between academic research and practical application in the water industry. One potential solution is for academic papers to be translated into more user-friendly formats for industry professionals. Better communication can increase the impact and practical use of academic research (6.C.3).

The need for transparent communication and stakeholder engagement when developing standards is also important, especially given the political and economic implications of changing standards. M. Kelly advocates for setting honest, evidence based environmental standards and having sensible derogations where needed. The consultation process for the most recent P standards was mostly a technical conversation with water industry scientists and others. M. Kelly suggests that future consultations should involve a wider range of stakeholders (for example Non-Governmental Organisations (NGOs) such as the Rivers Trust), given the increased public interest in water issues (6.C.8).

Having accurate standards and targets is important, but excessive focus on perfection can inhibit action. Decision makers often seek black-and-white answers but taking action that will fix 75% of the problem is better than inaction. Achievable and affordable improvements should be prioritised, even with only moderate confidence in the data, rather than striving for perfection. For example, implementing measures that will benefit the environment, are less costly and could be implemented easily should be prioritised without the need for the same level of excessive evidence required to implement costly measures (6.C.1).

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[28]The Aqua Project. Available at: https://aqua.york.ac.uk/home

4.9.7 Summary

Methodological advancements or improvements that could be used to inform updated nutrient EQSs

Determining thresholds

- Whilst the current standards were developed using the best available data at the time, it is highly likely that more extensive, geographically representative data sets now exist which may be utilised to derive new and/ or updated thresholds.
- Before calculating thresholds (new/ updated), the most recent data sets should be reviewed to identify gaps (e.g., relating to specific typologies or environmental conditions) that may limit its applicability to threshold setting.
- Gaps in data should be filled by using published and unpublished datasets, bespoke experiments and/ or additional sampling campaigns.

Improving the evidence base to support delivery in the water environment

- Additional categories may be needed to outline the extent of Poor/Bad, for example a “Very bad” category.
- There was a consensus amongst interviewees that an increase in the frequency of regulatory sampling would likely improve threshold setting and aid the protection of ecological health, however this may be unlikely due to financial implications.
- Alternative methods could be employed to provide data that is more representative of site conditions in a more cost-effective way, for example using continuous monitoring, improved technologies (e.g., automated sample collection at high flow) and/or citizen science.
- Novel analytical techniques (e.g. eDNA, flow cytometry, pigment analysis) offer the potential for developing more ecologically relevant nutrient thresholds in the future, after several years of data collection.

5. Conclusions

Conclusions relating to each of the two overarching research question investigated in this study are presented below.

RQ1: What are the current EQSs for nutrients in England and Northern Ireland?

- Nutrients with EQSs and which are currently used in producing ecological status classifications for surface water bodies include nitrogen and phosphorus.
- The fraction of nitrogen or phosphorous which standards are associated with varies depending on surface water body type, as does the statistic which has been adopted.
- Standards are generally either site-specific or type-specific and require the use of various variables to determine the values applicable to a surface water body.

RQ2: Are current nutrient EQSs protective of ecological health and function in the surface water bodies of England and Northern Ireland?

There was a consensus that EQSs require regular updates and re-evaluation considering new data and methods to remain up-to-date and to address the evolving challenges posed by climate change and other factors. Whilst the potential scale of the ecological impact arising from updating current thresholds is unclear from this review, it has been suggested that the any minor changes in threshold values (for most waterbodies) are unlikely to result in significant improvements to ecological health (6.C.6).

Significant improvements are more likely to result from systemic changes that result in a more holistic approach to threshold setting coupled with the more comprehensive use of the data to better inform remediation and mitigation measures (6.Appendix C). Further work is required to determine the costs and potential benefits associated with many of the recommendations and suggestions that have resulted from this review. Key conclusions in relation to the sub-set of questions explored are included below.

Do the current nutrient standards cover the correct nutrient fractions?

General points

- Research suggests that colimitation (N and P) is much more common than previously assumed, so the use of a single nutrient criterion should be questioned. This was echoed by the interviewees and workshop participants, whereby there was a consensus that all water bodies should have standards for at least TN and TP (in addition to current fractions).
- Implementing N and P limits for all water bodies is likely to result in an improved understanding of ecological health and actions required to improve it compared to tweaking current threshold values, however this may also result in greater political and financial ramifications.
- However, the development of additional nutrient fraction EQSs should be avoided if the resultant increase in resources would lead to a decrease in the spatial and/ or temporal frequency of monitoring by the Environment Agency.

Rivers

- Most interviewees and workshop participants believe there should be an N-based standard for rivers. The preference would be TN, nitrate, or both, with some support for including organic fractions.
- Some participants note the benefit of collecting information on organic fractions to help inform mitigation measures, even if this information is not used for regulatory purposes.

- Limited evidence was identified that suggests that a sediment-based EQS for rivers would improve the overall assessment of ecological health, however this warrants further review.

Lakes

- The current TN and TP standards are largely considered to be fit-for-purpose, however some interviewees and workshop participants felt that additional fractions (e.g. DIN) should also be included.

TraC

- The consensus was that the current DIN standard for TraC waters is appropriate, however ideally there would also be a standard for TN.
- Most interviewees and workshop participants felt that TraC waters should also have a P standard, either as SRP or TP.

Are current G/M threshold values protective of ecological health?

General points

- Nutrient threshold values currently employed in England and NI are broadly in-line (or at times more stringent) that those used in EU.

Rivers

- River RP thresholds appear to be stringent compared to EU MS and broadly align with thresholds determined through academic review.
- However, evidence from the interviewees suggest that the RP threshold values in England and NI may be too high (not stringent enough) to protect ecological health, this may be especially true in for shallow rivers with high alkalinity in NI.

Lakes

- Lake TN and TP thresholds are broadly in line with those used across EU MS.

TraC

- The range of DIN threshold values for G/M are broadly in line with concentrations identified in the literature. Limited evidence suggests that the Unionised Ammonia standard is protective of fish heath, however this warrants further review.

Are the type- and site-specific components effective, and should other factors be considered?

General points

- The potential of including additional/ alternative site-specific factors when calculating EQS could be explore for all thresholds and water bodies. However, it is important to remember that the suitability of type- and site-specific factors are determined based on statistical analysis performed on national-scale datasets. Therefore, including additional site-specific factors may not be viable, if their inclusion does not improve strength of pressure-response relationships.

Rivers

- Limited evidence suggests that the alkalinity component may produce misleading calculations of reference conditions for rivers that are highly impacted by effluent, which may provide a misleading representation of ecological status, however this warrants further review.
- There was consensus that the inclusion of additional site-specific factors like the degree of modification, flow, climate and clarity (e.g., turbidity/ suspended sediment) should be explored when developing thresholds.

Lakes

- The type-specific approach for lake TN standards is generally considered effective, however it may be worth exploring the impact of including physical modification typologies as well as flow and temperature (as proxies for stratification) in the models.

- Including a metric of colour/ humic substances for the TP lakes may improve thresholds for lakes from peat or C rich soil areas.

Trac

- No information was found relating to the type-specific factors used to calculate TraC DIN standards.

Do current standards use the most appropriate metric (e.g. annual mean, percentiles)?

General points

- Whilst summer mean nutrient concentrations may be more relevant to the ecological response, they would only be statistically viable if the Environment Agency sampling frequency was increased.
- Some evidence suggests that load-based thresholds may provide a valuable addition to concentration-based thresholds for some river types, however this may be financially unviable dues to the required increase in flow monitoring and thus warrants further review.
- The use of the 90th percentile for the total ammonia standards (in rivers and lakes) is rational, however it may represent a fairly artificial statistic if calculated on limited samples (e.g., quarterly sampling).

Are current standards developed using the most appropriate biological metric?

General points

- Whilst the BQE EQRs provide a likely candidate when developing a pressure-response relationship, they may not provide the most statistically robust option for determining thresholds, and therefore other metrics (or combinations of metrics) should be explored when developing (or revising) thresholds.

Rivers

- An updated reference model for diatoms is now available which could provide better calibrated methods for determining river P thresholds than the current approach.
- Evidence suggests that averaging the two sub-elements of the macrophytes and phytobenthos BQE is preferable over the current approach of taking the lowest score of the two assessments.

Lakes

- Evidence supports the current approach using of phytoplankton/ diatom metrics when determining nutrient thresholds in lakes.

TraC

- Little evidence was obtained for TraC waters which may warrant further review.

Are current thresholds develop using the most appropriate statistical methodologies?

General points

- Since the current thresholds were developed, updated EU guidance has been produced to help MS produce ecologically relevant nutrient thresholds. The current thresholds could therefore benefit from being re-evaluated in line with the most recent Best Practice Guidance.

Rivers

- The relationships used to determine the current thresholds had a significant amount of noise.
- River P EQS are currently set at the midway position, however they could be more protective for the ecology if set at the 25th percentile.

Lakes

- Like TP standards are based on a mathematical model, however if the data is available, they may warrant recalculating based on ecological principles (i.e. pressure-response relationships).

- Lake TN standards are based on pressure-response relationships and display relatively strong correlation. However, data from NI was not included in the calculations, as the data was not available at the time. If available, thresholds could be recalculated with the inclusion of NI data.

Trac

- Thresholds for DIN are not based on ecologically relevant nutrient concentrations. Evidence suggests that simple pressure-response relationships (linear regression) cannot be applied to these environments due to multiple stressors. However, other classification-based statistical approaches (as per the EU guidance) could be employed to determine more meaningful thresholds.

Methodological advancements or improvements

Determining thresholds

- Whilst the current standards were developed using the best available data at the time, it is highly likely that more extensive, geographically representative data sets now exist which may be utilised to derive new and/ or updated thresholds.
- Before calculating thresholds (new/ updated), the most recent data sets should be reviewed to identify gaps (e.g., relating to specific typologies or environmental conditions) that may limit its applicability to threshold setting.
- Gaps in data should be filled by using published and unpublished datasets, bespoke experiments and/ or additional sampling campaigns.

Improving the evidence base to support delivery in the water environment

- Additional categories may be needed to outline the extent of poor/ bad, for example a “very bad” category.
- There was a consensus amongst interviewees that an increase in the frequency of regulatory sampling would likely improve threshold setting and aid the protection of ecological health, however this may be unlikely due to financial implications.
- Alternative methods could be employed to provide data that is more representative of site conditions in a more cost-effective way, for example using continuous monitoring, improved technologies (e.g., automated sample collection at high flow) and/ or citizen science.
- New sampling techniques (e.g. eDNA, flow cytometry, pigment analysis) offer the potential for developing more ecologically relevant nutrient thresholds in the future, after several years of data collection.

6. Recommendations

What can be done to improve the current standards using existing approaches?

- Employ the confusion matrix (as per the 2024 EU Guidance) to examine the misclassification rates associated with all current thresholds. This is a straightforward, low-cost assessment of how effective the current thresholds are.
- If new/ updated thresholds are being developed, then the most recent EU guidance and UK data should be used.
- There was a consensus that the current RP standards for rivers (which were developed in 2012) now warrant review to bring them up to date.
- If nutrient thresholds for rivers are recalculated or developed in the future, then the following additional recommendations should be considered/explored:
    - Consider changing the TP methodology (settling) to reflect SRP (filtering).
    - Pay particular attention to high alkalinity streams, with a focus on NI as these may be less well served by the current thresholds.
    - Use the updated reference model for phytobenthos.
    - Explore the potential improvement in model sensitivity achieved by averaging the two sub-elements of the macrophytes and phytobenthos BQE instead of taking the lowest score of the two assessments (particularly applicable if using the updated reference model).
- Re-evaluate lake P thresholds and define using a pressure-response relationship (if possible).
- Future updates to lake TN standards should aim to include data from NI (which was not included when developing current TN standards).

What additional standards should be considered for development?

- There was strong evidence and support for introducing TN and TP standards for rivers to better reflect ecological responses. There is currently a misalignment between the objectives of the Environment Act (2021) and the monitoring undertaken by the relevant environment agencies under the WFD, and updated EQS including TP could address this.
- Limited evidence suggests that a sediment-based standard may be important for protecting ecological health in rivers, however this warrants further review.
- There was strong evidence and support for introducing a P-based standard for TraC waters, either as SRP or TP.
- Explore the potential of analysing organic fractions for informative purposes, alongside regulatory-derived nutrient fractions for all water bodies.

What wider approaches would merit consideration?

- Explore the potential (statistical viability) of including more site/ type-specific factors when developing thresholds. For lakes these may include flow and temperature in the typology classification. Furthermore, it may be beneficial to include a water colour-related parameter into an updated predictive model for lake TP from peat or C rich soil areas. For rivers these may include the degree of modification, flow, climate and clarity (e.g., turbidity/ suspended sediment).
- The use of the growing season mean could be explored if regulatory sampling frequency is increased in the future.
- BQEs may not be the most suitable metrics for determining thresholds and therefore other metrics (or combinations of metrics) should be explored.

What monitoring/ data would be required to improve current or future thresholds?

- Where thresholds are being developed, the most recent data sets should first be reviewed to identify gaps (e.g., relating to specific typologies or environmental conditions) that may limit its applicability to threshold setting. The following recommendations could be employed where gaps are identified:
    - Engage with researchers to access unpublished data that could inform standard setting (and consider the development of a centrally managed repository for relevant environmental data).
    - Commission bespoke experiments.
    - Develop sentinel catchments with daily sampling and rigorous QA/ QC to generate robust evidence on TN and TP, and on the error/ uncertainty associated with varying sampling frequency.
    - Employ continuous monitoring and/ or new analytical approaches to gain better insight into pressure-response relationships.
- Explore the potential (cost-benefit) of updating the monitoring requirements under the WFD (or relevant legislation), this may include:
    - Increasing regulatory sampling frequency.
    - Adding additional analytical techniques (alongside current methods) to determine suitability for future regulatory monitoring and threshold development, for example eDNA, chlorophyll-a (specifically for larger river sites), cell counts (flow cytometry) and cyanotoxin pigments. These approaches have the potential to aid the development of more ecologically relevant nutrient thresholds in the future; however several years' worth of data would be required before these could be utilised effectively.
    - Exploring the potential of citizen science (or citizen science-inspired rapid assessment methodologies) to provide higher frequency information regarding nutrient concentrations and ecological health.

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Appendix A. Literature search results and filtering

Provided as separate document.

Appendix B. Database

Provided as separate document.

Appendix C. Interview Meeting Notes

C.1 Interview with Wendy McKinley

Provided as separate document.

C.2 Interview with Penny Johnes

Provided as separate document.

C.3 Interview with Rupert Perkins

Provided as separate document.

C.4 Interview with Savannah Worne

Provided as separate document.

C.5 Interview with Mike Bowes

Provided as separate document.

C.6 Interview with Geoff Phillips

Provided as separate document.

C.7 Interview with Marc Stutter

Provided as separate document.

C.8 Interview with Martyn Kelly

Provided as separate document.

C.9 Interview with Pippa Chapman

Provided as separate document.

Appendix D. Workshop Meeting Notes

Provided as separate document.

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