ISSN 1831-9424

European Commission

# Physico-chemical criteria to support Good Ecological Status in Europe

*Report to ECOSTAT by the Task Group on Supporting Physico-Chemical Elements*
*2024*

Kelly, M., Teixeira, H., Lyche Solheim, A., Free, G., Phillips, G., Salas Herrero, M.F., Kolada, A., Varbiro, G., Poikane, S.

2024

Joint Research Centre

EUR 31832 EN

**Contact information**
Name: Sandra Poikane
Address: European Commission, Joint Research Centre (JRC), Ispra, Italy.
Email: sandra.poikane@ec.europa.eu (or ENV-Water@ec.europa.eu)

**EU Science Hub**
https://joint-research-centre.ec.europa.eu

JRC136407

EUR 31832 EN

PDF ISBN 978-92-68-12038-5 ISSN 1831-9424 doi:10.2760/355815 KJ-NA-31-832-EN-N

Luxembourg: Publications Office of the European Union, 2024



How to cite this report: European Commission, Joint Research Centre, Kelly, M., Teixeira, H., Lyche Solheim, A., Free, G., Phillips, G., Salas Herrero, M.F., Kolada, A., Varbiro, G. and Poikane, S., *Physico-chemical criteria to support Good Ecological Status in Europe*, Publications Office of the European Union, Luxembourg, 2024, https://data.europa.eu/doi/10.2760/355815, JRC136407.

# Contents

## Abstract

This report summarises approaches to setting robust criteria to achieve Water Framework Directive (WFD) objectives for the Annex V physico-chemical supporting elements: transparency, oxygenation, temperature, salinity and acidification. Nutrients have been considered in previous reports. This work is intended to complement existing guidance on statistical approaches to setting thresholds. A general conclusion is that thresholds that were set for older directives (e.g. Freshwater Fish Directive) may not be sufficiently stringent to protect Good Ecological Status and that Member States should check the validity of these. The current approach to data collection and aggregation may also need to be revisited. For example, for oxygen or salinity, concern is warrented for both persistent chronic impacts and episodic acute incidents which are becoming more frequent with climate change events such as heatwaves. We tested the latest version (2023) of the toolkit for setting thresholds on transparency and oxygenation conditions using EU-wide data demonistrating its potential. We identify situations where it may be possible to use historical time series, ecotoxicological data and peer-reviewed literature to set thresholds. However, thresholds are highly context specific, so it is rarely possible to give specific guidelines that apply across the EU.  Each chapter ends with a summary table to help guide on the appropriatness of the different approaches to set thresholds.


.

## Acknowledgements

We are very grateful to individuals and organisations who allowed us to use figures in this report. Stuart Warner (UNEP) and Geog Wolfram (DWS Hydro-Ökologie GmBH) made valuable contributions to the chapter on salinity. We thank all member states for comments that improved this document.

.

### *Authors*

Martyn Kelly: Bowburn Consultancy, Durham, UK

Heliana Teixeira: University of Aveiro, Portugal

Anne Lyche Solheim: NIVA, Norway

Gary Free: EC JRC, Ispra, Italy

Geoff Phillips: University of Stirling, UK

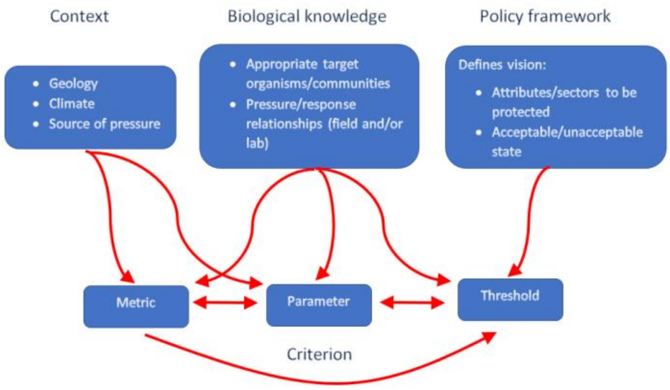
Fuensanta Maria Salas Herrero: EC JRC, Ispra, Italy

Agnieszka Kolada: Institute of Environmental Protection, Warsaw, Poland

Gabor Varbiro: Centre for Ecological Research, Debrecen, Hungary

Sandra Poikane: EC JRC, Ispra, Italy

## Executive summary




- This report summarises approaches to setting robust criteria for physico-chemical supporting elements in order to achieve Water Framework Directive (WFD) objectives. It is designed to complement guidance on statistical approaches to setting thresholds, recognising that there are situations where thresholds cannot be derived from monitoring data, and also circumstances where the entire criterion (not just the threshold) needs to be re-evaulated.

- Here the concept of criteria is used to encompass the main components of threshold setting: parameter, metric and threshold (see above figure). For example for salinity, either chloride or conductivity may be selected as the representative parameter and expressed as a metric such as an annual mean or 95th percentile. A threshold or boundary is then selected – for example to support good ecological status.

- We address physico-chemical supporting elements identified in Annex V of the WFD with the exception of nutrients (which has been addressed in detail already). These are: transparency, oxygenation, temperature, salinity and acidification. Salinity and acidification are not specifically mentioned in relation to transitional and coastal waters; however, we have included some examples where these may affect ecological status in these habitats and, therefore, where Member States may need national standards in order to protect sensitive ecosystems and achieve WFD objectives.

- A recurring theme is the influence that climate has on the aquatic environment, and the implications that this has for WFD objectives. Definitive guidance will be included in the updated CIS Guidance Document 24 ("River Basin Management in a Changing Climate"). Meanwhile, we have identified some areas where approaches to criteria-setting may need to be reviewed.

- A general conclusion is that thresholds that were set for older directives (e.g. Freshwater Fish Directive) may not be sufficiently stringent to protect Good Ecological Status. At the very least, Member States should check the validity of these using their own national data.

- We also recognise situations where the current approach to data collection and aggregation may need to be revisited. In the case of oxygenation, for example, concern is shifting from persistent chronic impacts due to heterotrophic activity to episodic acute incidents related to heatwaves and high primary productivity or anoxia in the bottom waters of lakes after collapse of massive algal blooms. More use may need to be made of continuous monitoring in the future.

- The toolkit for setting nutrient thresholds is currently being upgraded and the new version (available in 2023) will be able to set thresholds for other supporting elements too. We have tested this on transparency and oxygenation conditions using EU-wide data and believe that this holds real potential for setting thresholds in the future. Some preliminary results have been included; however, we believe that this will be most effective when national data are combined with knowledge of the local flora and fauna.

- There are, however, situations where analysis of historical time series and the use of ecotoxicological data may be more appropriate.

- It may also be possible to use peer-reviewed literature to set thresholds. However, thresholds are highly context specific, so it is rarely possible to give specific guidelines that apply across the EU. In particular, many older studies (or recent studies from other parts of the world) do not evaluate properties that readily translate to ecological status classes. Published studies need to be reviewed for relevance by experts with knowledge of the situations under which they will be used.

# 1 Introduction

After two decades of implementing the Water Framework Directive (WFD), there is widespread awareness that many waterbodies across Europe still do not meet the criterion of "good ecological status" (GES). There are myriad reasons for failure, one of which is elevated levels of physico-chemical pollutants. These should, in theory, be recognised by Member States as part of their national monitoring programmes and managed via Programmes of Measures. This, however, presumes that thresholds for these pollutants are set at levels that are sufficiently stringent to protect GES. Evidence gathered by ECOSTAT for inorganic nutrients suggests that this is often not the case (Kelly et al., 2022; Teixeira et al., 2022) and this report extends the analysis beyond nutrients to embrace a range of other physico-chemical elements likely to influence ecological status.

Supporting element criteria are a key link between policy objectives and desired outcomes (for example GES). They enable political ambition to be expressed in tangible and practical terms. Criteria combine an appropriate parameter (e.g. "conductivity"), metric (e.g. "annual mean") and threshold (e.g. "1000 µS cm-1") (Poikane et al., 2019). The threshold represents the point on a stressor gradient that differentiates an "acceptable" from an "unacceptable" state and, as environmental data are intrinsically variable, also incorporates an appropriate degree of precaution. Such criteria can be used to identify water bodies in need of restoration, prioritise those with the greatest needs, design restoration strategies and measure progress towards these objectives. They are, consequently, fundamental to safeguard and sustain water quality and ecosystem health and need to be set using the best scientific knowledge, reflecting links between the stressor, degraded ecosystems and ecosystem services.

There is, however, considerable inertia involved too: once a threshold has been set, regulators are often reluctant to change it without compelling evidence. They may have used these thresholds, for example, to set targets for discharges of pollutants. Companies responsible for these discharges will, in turn, have based capital investment plans on these targets. Other sectors, such as agriculture, may resist tighter thresholds unless there is compensation for lost revenue.

However, there is also a strong case for revisiting thresholds over time. Not only does legislation change, but the water environment also changes, often as a positive response to earlier legislation. In some cases, better evidence becomes available that challenges existing thresholds. Finally, almost all thresholds assume a straightforward stressor-response relationship whereas recent research has highlighted the importance of interactions amongst a number of stressors. In particular, all parts of Europe are now feeling the consequences of climate change which is altering the way that aquatic ecosystems are functioning, creating yet another reason why existing physico-chemical thresholds need to be revisited.

Whilst the focus of this report is on thresholds, there are also cases where the changing water environment requires all aspects of a criterion to be reconsidered. Dissolved oxygen in rivers presents a good example: many countries still used thresholds set out in the Freshwater Fish Directive (78/659/EEC) which was introduced in an era when many rivers still received substantial inputs of untreated or partially-treated organic matter from wastewater treatment works. A simple annual average of oxygen concentration or saturation was, therefore, a useful indicator of the condition of a water body. Since this time, however, conditions in rivers have improved, due both to EU legislation (e.g. Urban Wastewater Treatment Directive: 91/271/EC) and efforts by national governments. However, whilst organic loading has decreased, nutrient concentrations are still a concern, leading to diel swings in dissolved oxygen due to primary productivity, particularly in summer. The risk, in many cases, is no longer a long-term chronic oxygen deficit but short-term acute episodes. Many Member States, however, still report oxygen thresholds based on annual averages. Evaluation of thresholds in isolation would, in this case, be of limited benefit without a broader consideration of the criteria that Member States use when evaluating dissolved oxygen conditions in rivers.

This report is part of a larger project that considers all the chemical and physico-chemical elements that support the biological elements specified in Annex V of the WFD. Nutrients have been considered elsewhere (Kelly et al., 2022; Teixeira et al., 2022; Phillips et al., 2023); those considered here are:

- Transparency
- Thermal conditions
- Oxygenation conditions
- Salinity
- Acidification status

As thermal and oxygenation conditions are tightly linked, these are considered together in a single chapter.

All these chapters synthesise discussions during a series of workshops held in January and February 2022 for experts nominated by Member States along with the JRC staff and members of the project team. Each chapter starts with an introduction laying out the principles underpinning each supporting element before considering approaches to setting criteria in greater depth.

## 1.1 Approaches to setting thresholds: general points

In all cases we stress that there is no single "right" approach to deriving a criterion. A range of options have been attempted by Member States, each "tuned" to specific situation (depending on water category, the types of water bodies, data availability and the types of relationships obtained). We have tried to point out the limitations, where appropriate, and also to highlight best practice.

Some general starting points are:

- Does a criterion for this supporting element exist already? If it was developed prior to the WFD/MSFD, then the criterion should be checked for its ongoing applicability. If there is no criterion in place, then this will need to be derived from scratch.
- What data are available from which a new threshold can be developed? Are these sufficient to allow trends within waterbodies to be followed and, in particular, are there reliable records from at least a few water bodies from the period before the stressor was artificially elevated?
- Does the supporting element have chronic or acute effects on the biota? Each of these may require different approaches to measurement as well as to setting appropriate criteria.
- Have any ecological tipping points been identified for this supporting element beyond which changes will be very difficult to reverse?
- A preliminary literature and/or expert workshop, is recommended before starting any new work, in order to understand the main issues in relation to the region(s) where the criteria are to be applied and to identify knowledge gaps.

We have evaluated approaches to setting thresholds for each supporting element at the end of the respective chapter using a "traffic light" system to provide a succinct visual summary, as follows:

✗ not recommended for this supporting element;

✓ may be appropriate for this supporting element but proceed with caution; and,

✓ A good approach for this supporting element

**As a general rule, we encourage the derivation of thresholds either from national monitoring data or from ecotoxicological studies using taxa and performed under conditions that are relevant to the region in which the threshold will be used.** We recognise, however, that this is not always possible and have therefore presented other options too. Where there is a consensus on appropriate thresholds in the peer-reviewed literature, this can be used. We have also, where possible, offered some likely thresholds based on analysis of EU-wide datasets. These are useful for demonstrating the shortcomings of older legislation such as the Freshwater Fish Directive (78/659/EEC) but may not be sufficiently tuned to local conditions to be sure that they are protecting good status. We recommend, therefore, that these indicative thresholds are tested using local data.

The following guidelines apply to all supporting elements when setting thresholds using published literature:

1. Use thresholds derived from Species Sensitivity Distributions (SSDs) where possible. Make sure that taxa used to construct the SSD are indigenous to your region, and that a range of organism groups are represented. In particular, make sure that sensitive taxa indicative of high/good status are included. Check that the most sensitive life-stages are included and that the criteria used to evaluate the tests are broadly consistent with "good ecological status".

2. Separate chronic and acute criteria may be appropriate (depending on the parameter). Acute criteria should consider both the intensity and duration of the effect.

3. If it is not possible to use a SSD, compile ecotoxicology results for sensitive taxa from your region, and/or keystone/foundation species, and/or species that are essential for key ecosystem services. Check that test conditions are equivalent to those encountered in your region.

4. Validate likely thresholds using monitoring data.

Guidelines for developing thresholds for supporting elements from monitoring data are under development. Until these are available, methods described in Phillips et al. (2021) and Kelly et al. (2022) can be used. Note that the Shiny App cannot be used for supporting elements other than nutrients and that R scripts may need to be modified before they can be used.

**The recommended approaches for deriving thresholds for supporting elements were introduced at the ECOSTAT workshop in April 2023.**

1. Categorical methods are likely to be the most useful when applied to supporting elements other than nutrients. Binomial logistic regression is the recommended approach (guidance is under development, currently with a draft version available: Phillips et al.,2023) but other categorical methods can also be used meanwhile For example using box plots of supporting SE concentrations in water bodies with high and good versus less than good status for relevant BQEs.

2. Make sure that there is a significant difference between Good and Moderate status before proceeding. Note that changes along supporting element gradients are not necessarily linear, and it may be appropriate to group status classes differently (e.g. combining High, Good, Moderate and Poor status and comparing with Bad, in the case shown in Fig. 4.1, because there is no response to DO between Good and Moderate or even Good and Poor).

3. Validate likely thresholds using published values in the literature.

## 1.2 How does climate change influence criteria?

It became clear during the preparation of this report that climate change presents some significant challenges when setting physico-chemical criteria. The significance of climate change has escalated in the period since the WFD was adopted. The current guidance on handling climate change-related issues within WFD implementation (CIS Guidance Document 24: River Basin Management in a Changing Climate) was published in 2009. This assumed that implications for WFD objectives from climate change would be minor; however, this is clearly not the case, with many examples of impacts on the aquatic environment, and with interactions with other stressors. The question of adjusting criteria explicitly to deal with climate change was not addressed so, in this report, we have summarised those aspects of the criterion-setting process that need to consider climate change.

**A revised version of Guidance Document 24 is in preparation and will be reviewed by ECOSTAT during 2023. Text below is based on the 2009 version and will be updated as the recommendations of the revised draft become clear. Early indications are that option 1 is likely to change (i.e. Member States are likely to have opportunities to review reference conditions and, therefore, adjust ambition). This could have implications for option 3 (will revised ambitions still protect key ecosystem services?) (see below).**

Climate change is a pressure faced by surface water bodies and in that context Member States, would therefore, have an obligation to “prevent deterioration” (Quevauviller, 2011; Wilby et al., 2006). Moreover, climate-driven ecological changes may result in alterations to ecological communities that exceeds any reasonable interpretation of “low levels of distortion resulting from human activity” (Annex V Table 1.2). However, at the same time, it is not a pressure that can be managed locally or reversed solely by setting and implementing robust thresholds for physico-chemical criteria.

This raises two questions:

1. Which aspects of the accommodation/derogation of climate change effects into supporting element criteria setting should be dealt with by Member States and which aspects does the whole EU need to take together?

2. What is the most appropriate form of this accommodation?

**Options for handling climate change within existing WFD mechanisms include:**

1. **Changing the "expected condition" or type of a water body** (i.e. a shallow lake may be switched from a "high alkalinity" type to a "brackish" type due to rises in conductivity associated with reduced rainfall or saline incursions). This could, potentially, lead to more lenient nutrient targets and a de facto reduction in ambition (because deviation is now being measured against a different benchmark).

CIS Guidance Document #24 states (p. 49): "Although climate change has the potential to impact on virtually all quality elements included in the definition of WFD ecological status, this does not affect the principles of water status assessment, which remain valid", and (p. 55) "In general, reference conditions and default objectives should not be changed due to climate change projections over the timescales of initial WFD implementation (up to 2027) unless there is overwhelming evidence to do so." This implies that, in general, ambition for any individual water body should not change as a result of global warming.

However, the report also includes a hypothetical example (p. 53) of a very shallow, high alkalinity lake close to the coast which, due to a series of stormy winters resulting from a changing climate, becomes brackish in character after coastal defences are breached. Restoring coastal defences may be deemed "infeasible or disproportionately expensive" leading to the necessary re-classification of this lake as "brackish, very shallow" (change of type) and even, over time, as a "transitional lagoon" (change of water category). Dependence on sea defences in this case may mean that the freshwater "lake" was, itself, dependent on artificial sea defences so there is a logic underlying reclassification.

2. **Retaining the original type designation for a waterbody** (i.e. acknowledge that the change is an alteration from the natural state) and use existing instruments within the WFD (i.e. Less Stringent Objectives: Article 4 paragraph 5). Ambition has not changed (deviation is measured against the same benchmark), but there is now tacit admission that this will not be achieved, along with a reason (e.g. "infeasible or disproportionately expensive").

CIS Guidance Document #24 ("River basin management in a changing climate") says (p. 57/58) that climate change should not be used as a general justification for relaxing objectives whilst at the same time recognising that the use of exemptions is an integral part of river basin management under the WFD. It goes on to say: "exemptions without justification in line with the Directive cannot be seen as a general strategy to cope with the consequences of climate change. At the same time, the use of exemptions can have negative consequences for making water resources more resilient to climate change impacts" and,

"Where climate change is brought forward as the underlying reason for exemption due to excessive cost or unfeasibility, a clear and robust evidence base as for exemptions in other cases and consistent with other aspects of the approach to climate change should be provided. Within this evidence, DETECTION of a trend alone will be insufficient to invoke a change of policy and process, and ATTRIBUTION of the trend to anthropogenic climate change will be required."

This clearly recognises situations where objectives can be relaxed. One question that ECOSTAT may, therefore, wish to address is what a "clear and robust evidence base" means in practical terms. We suggest that this includes appropriate methods for detecting trends, data collection at a frequency that ensures sufficient statistical power and appropriate links to climate drivers in order to demonstrate causation. Ideally supporting data should be provided to enable linkages with other MS evidence for climate driven trends.

3. **Retaining the original type designation and setting new objectives for supporting elements** in order to mitigate, as far as possible, against the effects of climate change. Although climate change is not a pressure that can be readily addressed via Programs of Measures, it may be appropriate to revise thresholds for other stressors to ensure that the overall condition of a water body does not change (e.g. Spears et al., 2022).

For example, warmer summers are likely to result in greater frequencies of cyanobacterial blooms (Richardson et al., 2019, Kakouei et al., 2021) and more stringent measures to protect key ecosystem services such as recreation (and associated economic benefits) may require stricter nutrient thresholds (Huo et al., 2019).

None of these are questions that can be answered without further consideration; the purpose of this document is to highlight the need for a dialogue both within ECOSTAT and beyond, in order to develop a workable framework for incorporating climate change effects into ecological status class assessments and setting supporting element thresholds. For some supporting elements (e.g. salinity, particularly in southern Europe) the effects are already advanced and joined-up thinking on approaches is now urgent. Biogeographical differences across Europe mean that generalisations are very difficult. Management decisions need to be based on a good understanding of local water bodies and, in particular, an awareness of the extent to which key parameters have changed in recent decades.

It is unlikely that current BQEs alone will be adequate for monitoring trends and setting criteria. It is also important to recognise "tipping points" beyond which changes will be very difficult to reverse (Klose et al., 2021). Recognising such "tipping points" from empirical data is likely to be very challenging (Hillebrand et al., 2020) so it may be necessary to pay special attention to keystone species whose loss may precipitate ecological cascades. There may be situations when it is more appropriate to base criteria on the requirements of one or a few species rather than on the aggregated response of a whole community. As ever, the most appropriate response will vary between water categories and types and will require a good understanding of how ecosystems function.

# 2 Transparency

## 2.1 Background

Water transparency is one of the supporting physico-chemical quality elements specified in the WFD for the classification of ecological status in lakes, transitional and coastal waters (WFD, Annex V). Transparency is also one of the criteria for the assessment of the environmental status of coastal and marine waters in the Marine Strategy Framework Directive under Descriptor 5 Criteria 4 ("the photic limit of the water column [...] due to increases in suspended algae"). Though not a required supporting element for rivers, several countries measure Secchi depth, turbidity and/or suspended solids in order to achieve WFD objectives; several of the same issues apply to rivers and these are also considered in this chapter. However, we have assumed that suspended solid thresholds are linked to impacts on sedimentation, which is beyond the scope of this report. The issue of transparency is only related to deeper rivers, which are represented by a subset of existing river types. High concentration of suspended solids in smaller and more shallow rivers is also a natural trait of clay-rivers found in lowland areas in some Northern European countries. Clay-rivers pose a special challenge to WFD classification due to this natural phenomenon affecting phytobenthos as well as other BQEs (e.g. Schneider & Skarbøvik 2022, Eriksen et al. 2015).

Water transparency is an indicator of the quality of surface waters, with higher transparency linked to better condition of the water body. Light limitation is the most important driver of macrophyte degradation (Carstensen *et al*., 2011). The most common and widely used method to assess transparency is through Secchi depth measurement, which provides an indirect indicator of the depth of the euphotic zone. The euphotic zone is the upper part of the water column in lakes and marine waters where light is sufficient for photosynthetic activity, i.e., the depth where photosynthetically available radiation (PAR) is reduced to 1% of the value measured just below the surface (Aarup, 2002). The average light (PAR) remaining at Secchi depth equates to approximately 10 to 20% of surface light (Lorenzen, 1978, 1980; Preisendorfer, 1986) but this varies with several factors, such as light and water surface conditions, observer, etc. The euphotic zone corresponds, roughly, to twice the depth of the disappearance of the Secchi disk (i.e. Secchi depth) in many waters, but there is much variation from this. The ratio of Secchi depth / euphotic depth varies depending on the optical properties of water, for example relative concentrations of particles versus coloured dissolved organic matter (CDOM). If CDOM is dominant, the ratio is lower than if suspended particles are dominant.

Water transparency is decreased due to both light scattering and light absorbance. Light scattering is due to suspended particles and can be assessed via measurements of suspended solids (SS, in mg $L^{-1}$) or by turbidity (see below). Light is absorbed by coloured dissolved organic matter (CDOM) and by suspended particles, including phytoplankton cells and is measured spectrophotometrically at various wavelengths. A well-designed typology should account for coloured water, when this is a natural feature, so that transparency measurements are better indicators of factors related to anthropogenic effects.

Lake transparency is closely related to recreational user's perception of water quality and aesthetics with highly coloured and turbid waters considered less desirable for bathing (Smith *et al*., 1995). Collecting measurements of Secchi depth is easily implemented by citizen science initiatives and can provide large datasets as well as mobilizing interest in water quality (Lottig *et al*., 2014).

Turbidity is an alternative measure for assessing this supporting element. This is an optical determination of water clarity usually measured in Nephelometric Turbidity Units (NTU) or Formazin Nephelometric Units (FNU), with higher values associated with poorer ecological quality. Turbidity indicates the loss of water transparency due to the presence of suspended particles in water (e.g., inorganic and organic matter such as clay, silt, detrital particles (particulate organic carbon), phytoplankton and other microscopic organisms), which can originate from different sources, for example, elevated nutrients in water, by shoreline/catchment erosion, resuspended bottom sediments, or by detritus from stream and/or water discharges.

Many studies have shown there is a strong relationship between turbidity and Secchi depth, but the nature of this relationship (typically a hyperbolic curve) means that low Secchi depth values can correspond to a wide range of turbidity values. While both phytoplankton and macrophytes may respond directly to light attenuation, clear relationships with other BQEs are complicated particularly when increasing turbidity is due to the presence of fine particulate suspended material. Suspended particulates can affect fish gill structures and function, while settlement of the suspended particles of silt, clay, and other organic materials can suffocate bottom-dwelling organisms as well as newly hatched larvae or eggs, and lead to substantial changes of the structure of benthic habitats. For this reason, these measurements provide complementary information, given that different BQEs (e.g., macrophytes, fish, invertebrates) will each show different sensitivity.  Whilst the scope of this report does not extend to considering impacts of silt in surface waters in general, it is important to recognise this as a potential interaction under some circumstances.

**Table 2.1.** Terminology associated with the assessment of transparency.

| Term | Meaning |
|---|---|
| Colour | In ecological contexts, this is a proxy measure of the concentration of humic compounds capable of absorbing light in the blue wavelengths, with the remaining wavelengths reflected in the red-green parts of the light spectrum and the water looks brown. |
| Suspended solids | Solid material left on a filter of known pore size after a water sample has passed through it.  In the context of this report, it may be a proxy for turbidity in some circumstances. High concentrations of suspended solids in rivers and shallow lakes and estuaries cause the water to look milky with shades of grey, while glacial silt particles can cause the downstream lake water to look turquoise. |
| Transparency | The transmission of light into a body of water; it is attenuated by both absorbance and scattering |
| Turbidity | The amount of light that is scattered by suspended material when a light is shined through a liquid. |

## 2.2 Approaches to setting thresholds

### 2.2.1 Starting points

Issues to consider:

- Most of the literature considers transparency primarily as an expression of eutrophication and as an influence on photosynthetic biota. However, there are also cases where turbidity and suspended solids may influence other elements of the biota (e.g. fine particulate matter may damage fish gills and cause unsuitable habitats for benthic fauna).

- There is a general consensus that transparency is not a significant issue where Secchi depth > water depth (i.e. you can see the bottom). This may however influence the estimation of trends and seasonal patterns of Secchi depths and prevent trends detection in some BQE. For example, some species of rooted macrophytes may have light requirements such that their depth limit may be lower than Secchi depths, which may lead to a poor characterization of the light regime needed to support this BQE (Carstensen 2010).

- There is a risk of circularity if transparency thresholds are set using chlorophyll concentrations or phytoplankton metrics (as transparency is often another expression of phytoplankton concentration). By contrast, low transparency is a driver of macrophyte status, so it may be more appropriate to develop thresholds using macrophytes (or attached algae) rather than phytoplankton. Macrophyte species adapted to deeper water, such as some species of *Chara* and *Nitella,* are sensitive to minor changes in phytopankton abundance given the exponential decline in light with depth (Blindow 1992; Free et al. 2006). However, further research may be required to contrast boundaries calculated from several BQEs given variation in sensitivity and to evaluate appropriateness.

- If there are sufficient sites in reference conditions, reference values could be set from values recorded at these locations.

- Is the intention to use a transparency /turbidity threshold to influence classifications or as supporting information? The role that the threshold will play in monitoring the impact of activities (e.g. construction) in the watershed also needs to be considered. This may also require role of pulses of turbid water to be considered as well as longer term "chronic" effects.

- If zebra mussels or other filter-feeding non-native fauna are present, then the influence of these on transparency needs to be considered (Karateyev *et al*., 2002). Their presence may lead to higher than predicted transparency for a given level of source pressure (Cunha *et al*., 2019) so applying a threshold developed using data from lakes where non-native fauna are present to lakes where they are absent may lead to erroneous classifications. Similarly, acidification can result in higher transparency (e.g. Yan, 1982) so, again, inappropriate thresholds may be developed if the dataset includes acidified as well as non-acidified lakes.

- We recommend keeping issues associated with "transparency" and "suspended solids" separate, as far as is possible, although there will always be areas of overlap between these.

### 2.2.2 The influence of types

Most countries that have developed transparency and/or turbidity thresholds to support GES have recognised that water bodies differ in their clarity, even in the absence of stressors, and therefore separate thresholds may need to be developed for different types within a water category. In this respect, the situation is similar to that for nutrients (Kelly et al., 2022; Teixeira et al., 2022); however, some types of water body can present particular challenges:

- When colour due to natural factors (e.g. humic materials) is naturally high;
- Waterbodies influenced by glaciers, where there may be high natural concentrations of fine inorganic particles;
- Highly calcareous lakes where high calcite concentrations may impart a milky turbidity to the water;
- Saline lakes, where coagulation of fine particulates may reduce transparency; and,
- Tidal ecosystems, such as deltas and estuaries, high energy coasts and coastal waters under the influence of large rivers discharge are systems with naturally turbid waters because of the strong tidal currents and high hydro-dynamism that control suspended particulate matter.

These situations (particularly humic and glacial influences, as well as naturally high turbidity conditions) may be influenced by climate change (Steinberg, 2003; Gaiser *et al*., 2009; Robins *et al*., 2016), so care needs to be taken when establishing baselines based on contemporary data.

The national typology for the water category in question should be a good starting point for developing transparency and/or turbidity standards; however, it may be possible to combine some of the less-well represented types with others where more data are available in order to produce datasets that have the gradient length and data quantity to allow thresholds to be inferred.

### 2.2.3 Determinands and parameters

In general, thresholds for turbidity are less widely reported than those for Secchi depth, and this is reflected in WFD reporting (Table 2). Turbidity measures (e.g., FNU, NTU or other) are only used by France (FNU) and Spain (NTU), in some national coastal water types.  Secchi depth (SD) transparency is used by more countries as it provides a quick, cheap, and straightforward measurement that offers useful supplementary information to BQEs and other supporting elements.  “Suspended solids” also influence the light environment and, as a result, photosynthetic communities although they can also have other ecological effects (e.g., causing gill damage, sedimentation). Increased cattle access leading to sedimentation has been shown to threaten achievement of high and good status in rivers (Conroy *et al*., 2016). However specific work on sedimentation is beyond the scope of this chapter but is it something that merits attention in the future.

**Table 2.2.** Overview of parameters and metrics used to monitor transparency conditions (Kelly *et al.*, 2022b; Teixeira *et al.*, 2022). Numbers refer to the number of countries reporting standards that use a particular parameter/metric combination. Metrics have been split into those that measure the central tendency (e.g., mean, median) and those measuring a more extreme statistic (e.g., percentiles, maximum and minimum).

| **Supporting element** | **central tendency** | | **Percentile** | | **other** |
|---|---|---|---|---|---|
| | **annual** | **seasonal** | **annual** | **seasonal** | |
| **Lakes** | | | | | |
| Secchi disk depth (m) | 13 | 2 | 1 | | |
| **Rivers** | | | | | |
| Secchi disk depth (m) | 2 | | 1 | | |
| Suspended solids (mg/L) | 3 | | | | |
| **Transitional** | | | | | |
| Secchi disk depth (m) | 2 | 1 | 1 | | |
| **Coastal** | | | | | |
| Secchi disk depth (m) | 2 | 2 | 1 | | |
| Turbidity (NTU or FNU) | 1 | | 1 | | |
| Suspended solids (mg/L) | 1 | | | | |

EEA data showed strong relationships between Secchi depth associated with different status classes for both phytoplankton and macrophytes in inland waters (Kelly *et al*., 2022b), suggesting that current thresholds are broadly consistent with good status. However, there are still strong differences between countries and some countries report thresholds below 1 m for some types, which may not support good status for these BQEs (Fig 2.1). Turbidity is not detailed in the Supporting Elements technical report for TRAC and marine waters (Teixeira *et al*., 2022), but boundary values in use in coastal waters also vary greatly (9 – 36 NTU), possibly driven by a combination of typology and the use of annual means vs percentiles. (For scale purposes: deionised water generally has a turbidity of 0.02 NTU, drinking water is usually between 0.02 and 0.5 NTU, and untreated wastewater can fall anywhere between 70 and 2000 NTU). Once again, there are major differences between countries.

Generally, turbidity and suspended solids may be better suited to monitoring around localized inputs than for routine or surveillance monitoring. It is possible to automate these measurements and collect continuous data via sensors. When assessed using a nephelometer, Turbidity (NTU) measures reflected light (at right angle) and therefore has the benefit of not being influenced by absorption by for example dissolved humic substances, an important consideration in northern Europe. However, this type of monitoring still needs to be related to ecologically-relevant targets.

Overall, differences between standards may be due to one or more of the following factors:

- the way that the standards were set (for example, expert judgment rather than derived empirically using relationship with BQEs);
- the summary statistics used;
- adjustments for other factors (for example, reference conditions for different types).

Countries with particularly lenient thresholds should check these against national data on relationships with sensitive BQEs to validate whether these are sufficiently protective.

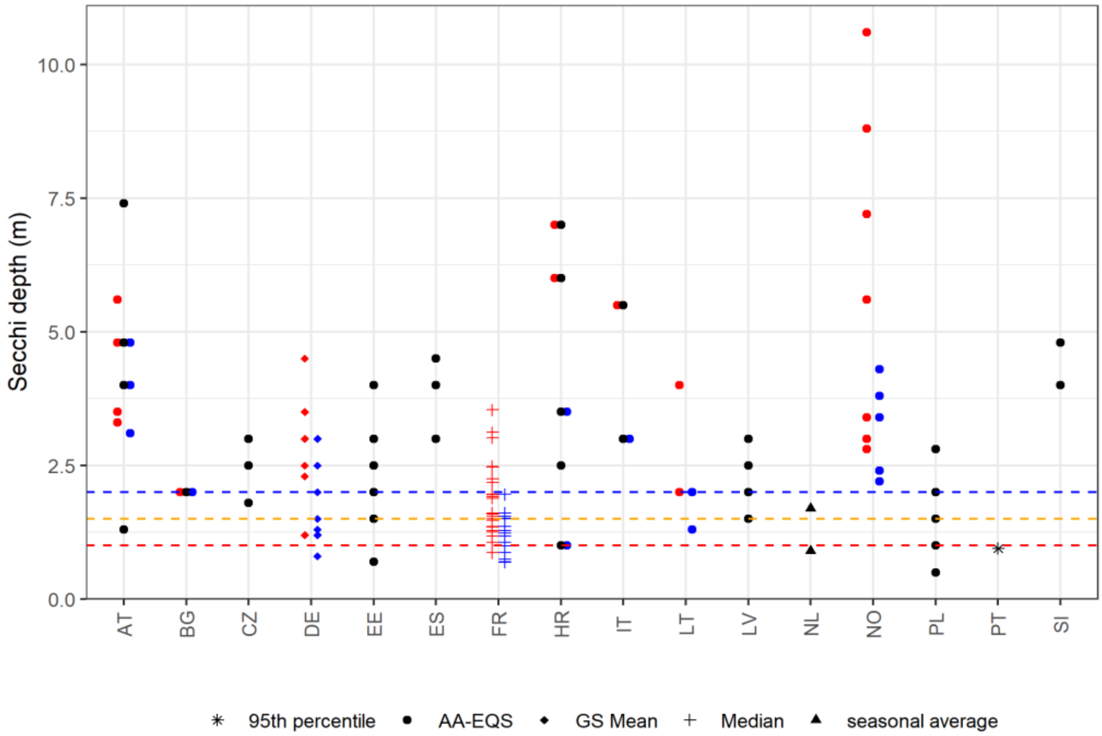



**Figure 2.1.** Secchi disk depth standards for lakes by country (single value black, minimum blue, maximum red). Dotted lines show interquartile range of mean or median values, (10th percentile=red, 25th percentile=orange, 50th percentile = blue.). Reproduced from Kelly et al. (2022).

### 2.2.4 Empirical approaches, expert judgement or historical data?

Examples of both empirical analysis of contemporary data and of the use of historical data are available. Historical data can be powerful, so long as measurements started before the onset of eutrophication (e.g. Bodensee: Murphy *et al.*, 2018). Analysis of historical data gives an excellent insight into the pre-eutrophication transparency for the lake in question; a question that arises is how confidently such data can be extrapolated to other lakes of a similar type.

Empirical approaches carry the same advantages and challenges as for establishing nutrient thresholds (Kelly *et al.*, 2022a). So long as the gradient based on contemporary measures spans the boundaries of interest, there is no reason why methods used to develop nutrient thresholds cannot be used for transparency too. As for nutrient thresholds, extrapolation beyond the limits of the data is not recommended.

Linear regression and categorical methods have both been used. Denmark used multivariate regression with chlorophyll, colour and depth as independent variables from which a predicted Secchi Depth at the good/moderate boundary was derived. Not all lake types had sufficient data, and others did not have strong relationships. However, relationships were established for four types, covering 80% of Danish lakes (Fig. 2.2: Søndergaard et al., 2019). An advantage of this approach was that specific boundaries for individual lakes could be derived, rather than relying on a general standard applicable to an entire type.

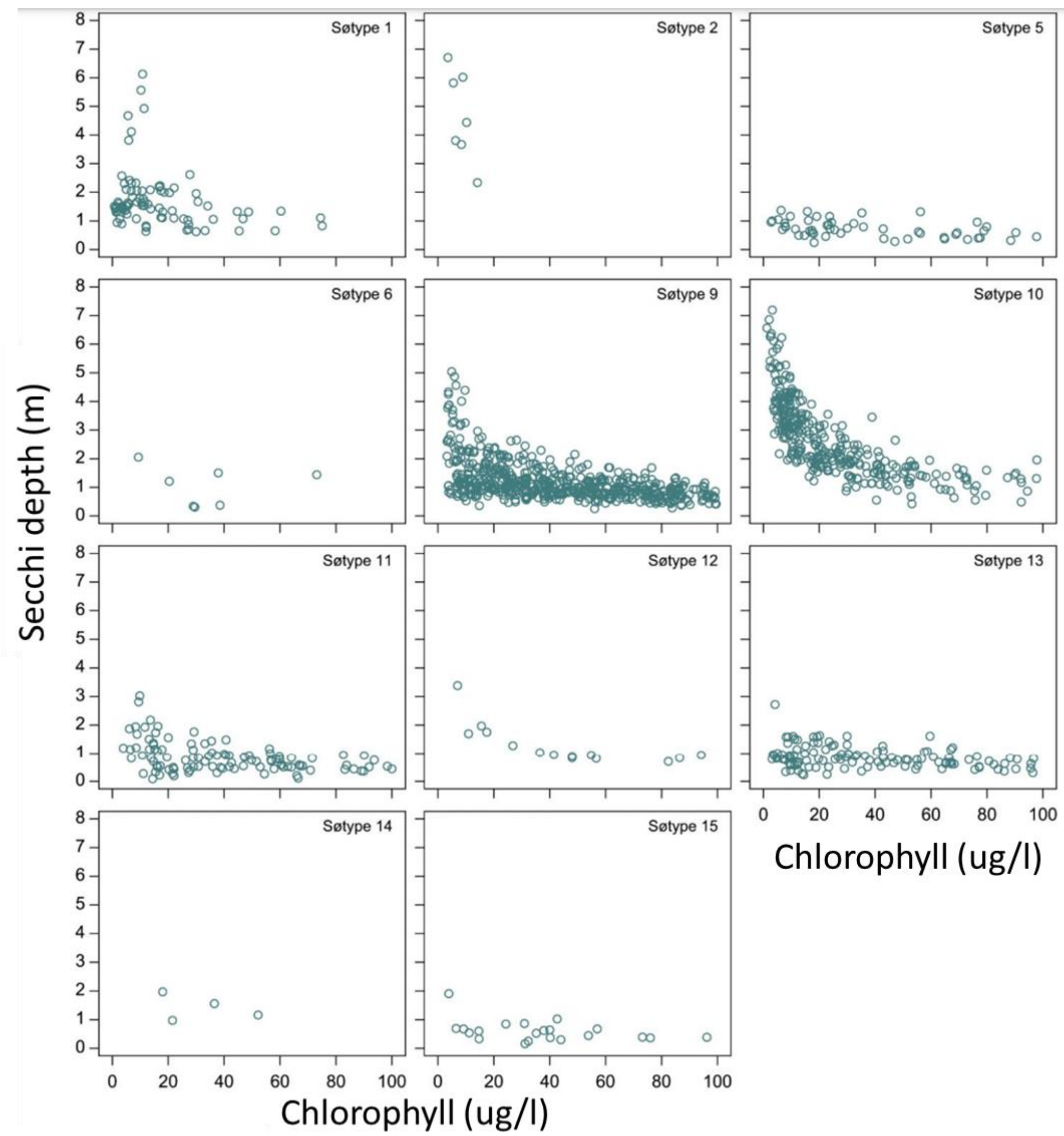



**Figure 2.2.** Relationship between Secchi depth and chlorophyll a in all eleven Danish lake types. From Søndergaard et al. (2019).

An approach using multiple regression has also been developed for Norwegian lakes (Phillips, 2013; Direktoratsguppen vanndirektivet, 2018). First, the relationship between Secchi depth, chlorophyll and humic materials (measured as colour in mg Pt/L) was established as:

Secchi depth = (ln(95)-ln(20))/[(0.037×A^0.60 )+(0.02×chla)] ($r^2 = 0.79$; $p < 0.001$)

Where: "95" = 95% of incident light (assuming 5% is reflected);
"20" = light at Secchi depth (assumed to be 20% of incident light);
"chla" = chlorophyll a concentration (µg/L); and,
"A" = colour (mg Pt/L).

With this equation, Secchi depth can be calculated for any value of chlorophyll and colour. By substituting intercalibrated chlorophyll boundaries and measured colour into the equation, threshold values for Secchi Depth can be predicted for any lake (Fig. 2.3), thus having the same advantage as the Danish method.

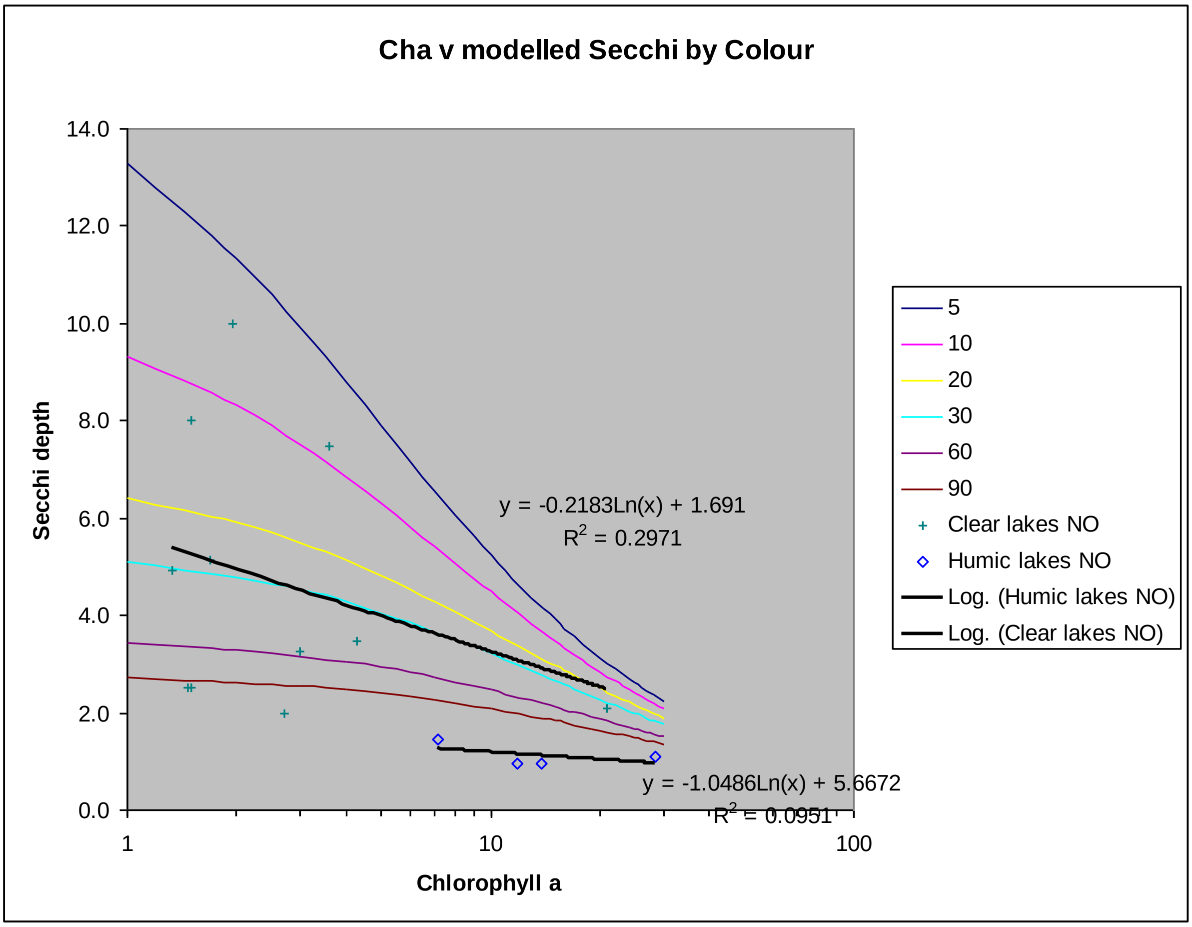



**Figure. 2.3.** Non-linear model fits between chlorophyll a and Secchi depth for colour values of 5, 10, 20, 30, 60 and 90 mg Pt $L^{-1}$. Points mark mean chlorophyll and Secchi depth values for NO lakes. Lines show linear fit for clear and humic lake types (Phillips, 2013).

For glacial lakes, a model developed in Alaska was used (Edmundson & Koenings, 1986; Koenings & Edmundson, 1991):

Log euphotic depth = 1,2270 - 0,6635 Log NTU ($r^2$=0.94)

Euphotic depth = 4.2 x Secchi depth.

Boundaries were then estimated as the reference value for Secchi depth in a glacial lake (from these equations), multiplied by the threshold EQR for the respective boundary in a very clear mountain lake (Table 3).

**Table 2.3.** Boundary values for Secchi depth in two low alkalinity Norwegian lake types. The EQR-values for the clear highland lake type L-N7 are included as an illustration of how targets are set for glacial lakes (such as Lake Gjende). More information is given in the text. (Lyche Solheim et al. 2017).

| NGIG type | Type description | Colour (mg Pt/L) | Ref. | H/G | G/M | M/P | P/B |
|---|---|---|---|---|---|---|---|
| L-N2a | Clear, shallow, lowland | 5 | 11.4 | 8.8 | 7.2 | 4.4 | 2.4 |
| | | 10 | 8.3 | 6.9 | 5.8 | 3.8 | 2.3 |
| | | 20 | 5.9 | 5.1 | 4.5 | 3.2 | 2.0 |
| | | 30 | 4.8 | 4.3 | 3.8 | 2.9 | 1.9 |
| L-N7 | Very clear, highland, shallow or deep (non glacial), | - | 13.8 | 12.3 | 10.6 | 7.2 | 4.6 |
| | | | EQRs: | 0.89 | 0.77 | 0.52 | 0.33 |
| Glacial | Example: Lake Gjende | - | 3.1 | 2.7 | 2.4 | 1.6 | 1.0 |

In theory, expert judgement could be used, through judicious review of literature and/or importing standards developed for other purposes (the Freshwater Fish Directive, for example, required inland waters to have mean suspended solids ≤ 25 mg/L). However, these would then need to be validated against contemporary monitoring data. Equally, the process could be inverted so that expert judgement becomes a means of validating thresholds derived by other approaches. The best approach will depend upon individual circumstances, but the primary stages of threshold setting should use the most data-rich option with especial consideration given to the number of distinct water bodies represented, rather than simply the number of data points. The less data-rich option then becomes an opportunity for independent validation of the thresholds. We also recommend that a consistent approach should be adopted across a country. With these conditions in mind, expert judgement is likely to be more useful as a means of validation than for setting thresholds in the first place. Similarly, thresholds derived from historical data need to be checked against contemporary monitoring data for the same boundary, and vice versa.

## 2.3 Role of Climate change

Of particular concern, within the context of this report, is the effect that climate change has on the background concentrations of organic and inorganic particles. There are many reports of increasing "brownification" of inland waters due to increased loads of humic material (Forsberg, 1992; de Wit et al., 2016; Lipczynska-Kochany, 2018; Monteith et al., 2023) as well as of changing concentrations of inorganic particles associated with glacial retreat (Marín *et al*., 2013).

Likewise, turbidity conditions are likely to be affected by altered river flows including floods and more extreme rain events with shifts in storm surge regimes as well as sea-level rise under climate change scenarios (Robins *et al.*, 2016). Climate-induced lowering of water levels can also lead to decreased transparency through increased resuspension of fine sediments due to wind action (Ludovisi & Gaino, 2010). In addition, naturally turbid waters are already more frequently exposed to hypoxia events (dissolved oxygen < 2 mg $L^{-1}$), particularly during the warmer seasons (Schmidt *et al.*, 2019) and thus are more likely to be affected by increased water temperatures predicted for Europe's temperate ecosystems under climate change scenarios.

These changes need to be considered within the framework outlined in chapter 2.

## 2.4 Likely ranges of Secchi depth in European lakes and other water categories.

There has been extensive research on the relationship between Secchi depth and other eutrophication parameters (e.g. OECD, 1982). The problem, however, is that results are often difficult to relate to WFD status classes. We therefore recommend that published values are evaluated very carefully for relevance to both local lake types and WFD objectives before being adopted.

Strong relationships between Secchi depth and biological parameters means that it should be possible to estimate the transparency required to support good ecological status using the boundary prediction model of Phillips et al. (2023) and an EU-wide dataset. As aquatic plants respond directly to reductions in transparency whereas as phytoplankton abundance contributes to reduced transparency macrophyte data were used to predict boundaries, to reduce the risk of circular reasoning. However, further research should explore the contrasting boundaries calculated from different BQEs. There were sufficient data to run exploratory analysis to predict boundaries for just five broad types. LA-3 (low-mid altitude calcareous (including humic) shallow, stratified lakes) are shown as an example (Fig. 2.4) and summary results for other types are presented in Table 2.4. However, none of these would be acceptable for use, for example LA-01 (very large, stratified lakes), is based on a limited dataset from just two countries, resulting in wide confidence limits despite an apparently strong model (AUC = 0.842). LA-02 similarly has a low number of lakes (24) upon which to set boundaries. In addition LA-03 included a mix of humic and non-humic lakes and these should be partitioned given the importance of colour on transparency. These results are included to demonstrate the potential – and highlight the pitfalls – of using binary logistic models for predicting thresholds for transparency, and they cannot yet provide guidance on acceptable values.

Preliminary results suggest that this approach also works on coastal and transitional waters (for both Secchi depth and turbidity) but it is premature to include guiding threshold values in this report. The report "Establishing Supporting Physical Chemical Element Standards: A revised approach applied to transitional and coastal waters" (Teixeira et al., 2023) will contain a more detailed consideration of TRAC waters".

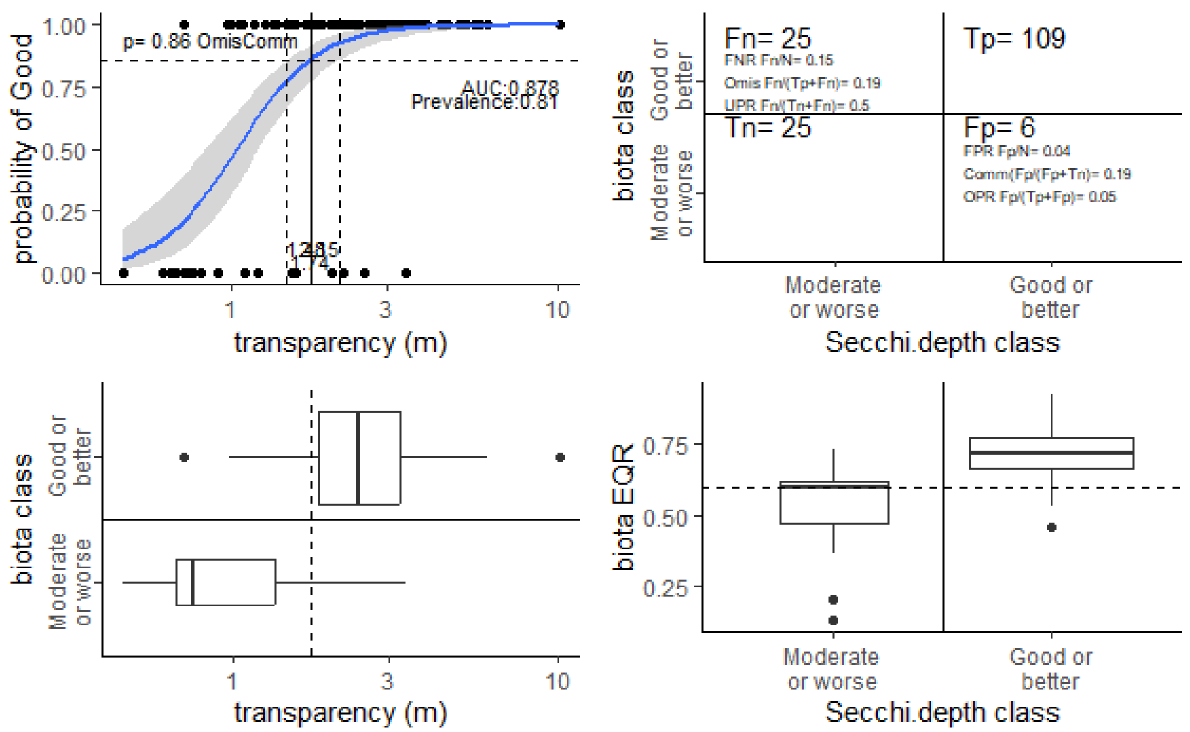



**Figure. 2.4.** Use of Binary Logistic Models to determine the good/moderate boundary for Secchi depth in low-mid altitude calcareous (including humic) shallow stratified lakes.

**Table 2.4.** Predicted values of Secchi depth to support Good Ecological Status in European lakes. Values available for five broad types. Key: lcl / ucl = upper/lower confidence intervals of predicted boundary; auc = area under curve; n = number of samples on which prediction is based; countries = source(s) of data. -The final recommendation to use is based on a combination of size of dataset, number of countries contributing data and strength of model.

| Broad type | Predictions (m) | | | auc | n | Countries | use? |
|---|---|---|---|---|---|---|---|
| | Boundary | lcl | ucl | | | | |
| LA-01 very large, stratified | 2.5 | 8.5 | 1 | 0.842 | 23 | FI, NO | ✗ |
| LA-02 lowland, calcareous, very shallow, unstratified | 3.7 | 3.6 | 4.5 | 0.600 | 24 | FI, NO, SE, UK | ✗ |
| LA-03 low-mid altitude, calcareous (incl. humic), shallow, stratified | 1.7 | 2.2 | 1.5 | 0.878 | 165 | BE, NO, PL, | ✗ |
| LA-04 lowland-mid altitude, humic (& siliceous) | 0.9 | 1.1 | 0.73 | 0.807 | 74 | PL | ✓ |
| LA-05 lowland, siliceous | 1.8 | 2.5 | 1.3 | 0.724 | 25 | FI, NO, SE | ✗ |

## 2.5 Conclusions

- Transparency data often shows strong relationships with principal stressor gradients, so it should be possible to derive thresholds using approaches developed for establishing nutrient thresholds.

- It may be beneficial to use BQEs that respond to transparency, rather than those that contribute to transparency, when deriving thresholds. In lakes, for example, macrophytes may be more appropriate than phytoplankton. Although, further work comparing boundaries derived from different BQEs is recommended.

- This consideration of transparency as a supporting element overlaps with other topics of interest to ECOSTAT including sediment and invasive organisms.

- Likely approaches are summarised in Table 2.5.

**Table 2.5.** Summary guidance for setting ecological thresholds for transparency in lakes. See section 1.1 for explanation of use recommendations (✗ and ✓).

| Level | Threshold | Source | Use? |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | | Not available | ✗ |
| 2. Threshold based on published literature: general prescription | | OECD (1982) | ✗ |
| 3. Threshold based on analysis of WISE/SoE data | See Table 2.4 | | ✗ |
| 4. Threshold based on published literature: targeted | See 1.1 | | ✓ |
| 4. Threshold based on national data | See 1.1 | | ✓ |

# 3 Oxygenation and thermal conditions

## 3.1 Background

Dissolved oxygen is, of course, essential for aquatic life. The concentration present depends, in part, on the water temperature, on diffusion within the water column and at the air-water interface and on biological activity. The interaction with temperature means that oxygenation conditions will be influenced by global warming with oxygen solubility decreasing as temperature increases. As oxygenation and thermal conditions are so closely-linked in terms of ecological effects they are considered together in this report.

The approach to measuring temperature and oxygenation will vary depending on water category and on local biogeography. For example, it may be appropriate to treat most stretches of river as spatially homogeneous when measuring oxygen, but many lakes, transitional and coastal waters will experience stratification. In such cases, decisions about which part of the water column to sample will need to be taken. In addition, the solubility of oxygen decreases as salinity increases, meaning that sea water holds almost 20% less dissolved oxygen than freshwater at the same temperature- and pressure. This will affect assessments of transitional and coastal water categories.

Two contrasting trends set the challenges involved in establishing oxygenation and thermal criteria into perspective: the first is a gradual improvement in oxygenation conditions in many water bodies globally due to targeted interventions by regulators including reduced loading from wastewater treatment plants (see, for example, Steckbauer *et al*., 2011). The second is the risk that benefits of such improvements are being offset by rising global temperatures across all water categories (e.g., Altieri & Gedan, 2014; Altieri *et al*., 2017; Borja *et al*., 2006 , Jeppersen et al., 2017; Palmer et al., 2008; Trolle et al., 2011; Kaushal et al., 2014). The oxygenation and thermal regimes cannot be considered in isolation, as they interact with many other elements that influence ecological status (e.g., Moss *et al*., 2011; Schmidt *et al.,* 2019). It has long been recognised that tropical lakes have higher productivity for a given nutrient concentration than temperate lakes (Lewis, 1996; 2011; Chen et al., 2021) and this geographical trend is a portent of temporal trends within individual lakes due to climate warming. It is likely that, in order to achieve WFD objectives, other criteria (including, importantly, those for nutrients) may need to be tightened if ambitions for sustainable water resources are to be achieved.

The interaction between temperature, oxygen and nutrients is a classic case of a multistressor situation which exhibits a complexity that embraces entire ecosystems. In the case of Austrian lakes, for example, warmer weather extends the length of thermal stratification leading to longer periods of oxygen depletion in late summer, with concomitant phosphorus release from sediments (Ficker *et al*., 2017; Luger *et al*., 2021). The 2018 heatwave in northern Europe led to high temperatures in Norwegian lakes (Woolaway *et al*., 2020), leading to salmonids seeking cooler water at depth, with impacts on pelagic food webs (Lennox *et al*., 2021). Recent projections estimate that lakes will get warmer for longer, with heatwaves possibly spreading across several seasons (Woolway *et al*., 2021). In the aftermath of such heatwaves, an interruption by cold and cloudy weather systems can lead to a sudden decline in temperature and a period of low irradiance, leading to a rapid decline in primary production. The net-negative ecosystem production that results can lower oxygen concentrations and result in fish fills (Jeppesen *et al*., 2021). Marine heat waves, similarly, have been shown to have catastrophic effects on kelp forests, and the ecosystems associated with these (Wernberg *et al*., 2016). Marine phytoplankton are also affected (Remy et al., 2017; Battem et al., 2022), with impacts exacerbated by elevated nutrients (Hayashida et al., 2020). Ecological cascades resulting from these can affect higher trophic levels too (Roberts et al., 2019; Piatt et al., 2020). These examples highlight the importance of following not just trends in average temperature, but also the magnitude and duration of extreme events.

Although temperature and oxygen availability are linked, temperature exerts direct effects too, and these also need to be considered when setting criteria. Temperature, for example, influences growth rates and life-cycles as well as other aspects of ecological functioning (e.g. production and respiration rates, phenology changes, plant germination/ flowering period, fish reproduction ability, larval survival and development, etc.). Similarly, low dissolved oxygen is rarely the only stressor influencing a water body and can also influence redox processes at sediment-water interfaces (leading to the release of hydrogen sulphide and methane for example). When evaluating impacts and setting protective thresholds, the role of other stressors also needs to be considered.

All of these points will influence how oxygen and thermal criteria are set. In principle, both are relatively straightforward measurements; however, appropriate criteria for any particular water body will depend on a reading of local circumstances and may lead to differences in frequency and locations of measurements that complicate comparisons between countries. The focus in this chapter, therefore, is not to prescribe a single approach, but to draw out some principles that should ensure that protective thresholds can be set regardless of circumstances.

## 3.2 Approaches to setting thresholds

### 3.2.1 Starting points: general considerations

Temperature and oxygenation conditions are different, but inter-related, stressors. Many of the issues that need to be considered are common to both, but there are also issues particular to each stressor.

- In both cases, local circumstances need to be considered. Criteria are often set by Member States to apply across a territory, albeit with variations depending on water body types and with a view to protecting "good ecological status". However, the most appropriate criterion will also depend on the nature of the stressor. Both temperature and oxygenation conditions can have chronic and acute effects, depending on local conditions. Each of these may require a different criterion, as well as different measuring regimes. **For oxygenation, in particular, we suspect that many criteria currently in force are not fit-for-purpose as they do not address the most dangerous manifestations of the stressor**.

- There is a general consensus that standards for both temperature and oxygenation conditions need to be type-specific. This should be self-evident, given the wide range of latitudes straddled by the EU, and the tight coupling between temperature and dissolved oxygen concentration. For rivers, we anticipate that fast-flowing rivers, containing sensitive species will be more impacted by a loss of oxygen than slow-flowing rivers whilst humic lakes will require different oxygenation standards to those with clear water.

- Criteria for "acute" effects will need to consider both the level of the stressor and the duration of the effect.

- Whilst there is clearly a need to set boundaries that protect existing BQEs, oxygenation and temperature criteria may need to consider other ecosystem properties and will also need to consider "tipping points" that result in ecological cascades ("domino effects": Klose et al., 2020).

- For both oxygenation and thermal conditions, there are strong cases for moving away from spot measurements and towards continuous monitoring in many situations.

### 3.2.2 Starting points: oxygenation conditions

- Many criteria for oxygenation criteria were derived in an era when wastewater treatment was less effective than it is now. They were designed for waters with high levels of heterotrophic activity and, as water quality across the EU improves, the nature of the problem has changed. This means that there are many situations where the criteria themselves may no longer be fit-for-purpose. For inland waters, in particular, we noted many countries still use the thresholds set by the Freshwater Fish Directive. It is not always clear whether the efficacy of these thresholds for achieving WFD objectives has been tested, rather than just assumed.

- What are the most likely sources of problems? Is there a general depression of dissolved oxygen (DO) due to heterotrophic activity, occasional risks of deoxygenation (e.g. due to episodic storm sewer overflows), night-time anoxia caused by high plant biomass, low DO due to heat waves? Under some circumstances (e.g., downstream from outlets from hydropower plants), supersaturation of oxygen can kill fish directly via "fish bubble disease" (Machova *et al*., 2017; Weitkamp *et al*., 2003). Each situation presents a different type of challenge.

- “Percent oxygen saturation” (“%sat”) is widely used as a measure of oxygenation conditions. However, as this depends upon temperature, it will not reflect impacts due to warmer climates and, for that reason, its use as a supporting element should be discouraged. Whilst we recognise that %sat is easy to understand and keeping this variable offers historical continuity, the focus should shift to using DO concentration as the basis for criteria. So long as temperature and altitude are known, it should be possible to estimate oxygen saturation from concentration to ensure continuity with older records.

- Biochemical Oxygen Demand (BOD) is used in some water categories (in particular in rivers), and is one of the EEA’s core set of indicators reported annually by 24 countries for more than 10,000 monitoring sites (Oxygen consuming substances in European rivers (europa.eu)). The EEA has identified 5 classes (<1.4, 1.4-2.0, 2.0-3.0, 3.0-4.0, >4.0), but these are not linked to biological quality elements. This may give a better indication of chronic impacts of oxygenation than criteria based on small numbers of spot measurements of DO. However, it will not be appropriate in all circumstances.

- There may be some types where it is not appropriate to include oxygenation criteria when the “one out, all out” rule is applied. Examples include some Italian lakes with very slow turnover (see Box 1). In such situations, it is better to consider oxygenation alongside other criteria (e.g. nutrients, Secchi depth).

**Box 3.1. The Italian deep subalpine lakes - cascading effects from increased winter temperatures, reduced mixing and altered nutrient dynamics.**

The deep subalpine lakes are of key economic importance in northern Italy and their size and depth makes them a key regional water resource requiring priority management (Premazzi et al., 2003; Regione del Veneto, 2018). A warming trend has been detected in the lakes with annual average surface temperatures increasing 0.017 $\circ$C $yr^{-1}$ and 0.032 $\circ$C $yr^{-1}$ in summer (Pareeth et al., 2017). This has led to more stable stratification and increasing isolation of the hypolimnion from the epilimnion, with no complete mixing since 2006. This has led to a gradual decrease in oxygen concentrations in the hypolimnion with the result that climate now exerts more control on oxygen than trophic status (Rogora et al., 2018). This has also reduced nutrient transfer from the hypolimnion to the epilimnion resulting in alterations to phytoplankton composition (Salmaso et al., 2018). Most studies have attributed the cause to long-term climate change and fluctuations in large-scale regional climate drivers such as the East Atlantic and the North Atlantic Oscillation (EA, NAO: Rogora et al., 2018; Salmaso et al., 2018). Lower surface water temperatures and sinking of colder water in winter leading to destabilisation of stratification is one of the principal mechanisms contributing to deep lake mixing (Horne & Goldman, 1994; Woolway et al., 2019) and positive values in the EA and NAO, resulting in higher winter temperatures, are likely to critically alter this process.

The Water Framework Directive (WFD) requires that lakes are assessed using biological quality elements including phytoplankton (chlorophyll-a as well as taxonomic composition). Ecological status is measured as deviation from reference condition which varies depending on lake type and geographic region (Council of the European Communities, 2000, 2013; Järvinen et al., 2013). The system adopted for alpine lakes included only altitude, mean depth, alkalinity and lake size (Wolfram et al., 2009) with the mixing characteristics of the lake were included as optional type parameters (Council of the European Communities, 2000). Climate change may therefore create pressure to alter management targets and strategies as the lake typology and, therefore, ecological status boundaries may effectively change, presenting a stark choice between setting unobtainable goals and the need to protect and improve water quality (Cardoso et al., 2009). While chlorophyll-a may decline with continued stratification, superficially indicating an improvement, other biological quality elements such as macroinvertebrates in the sub-littoral and profundal zones may deteriorate given the lower oxygen concentrations below the thermocline (Rossaro et al., 2007). Climate change may therefore test whether the intentions of the WFD to appraise the structure and functioning of freshwater ecosystems can be realized given current approaches.

### 3.2.3 Starting points: thermal conditions

- The wide latitudinal variations across Europe mean that only general principles can be offered here. The focus is on localised warming, as this will have immediate consequences for RBMPs; however, it is also important to consider long-term trends, as criteria for other SEs may need to be adjusted to mitigate against consequences of this.

- Whilst seasonal and vertical heterogeneity ("stratification") of water bodies is widely recognised, there are aspects of spatial heterogeneity that are still poorly understood. There are dangers in extrapolating temperature trends between water bodies, and also in using air temperature as a proxy for water temperature (Bonacina et al., 2023).

- Rising temperatures are also conducive to the survival and spread of invasive non-native species which may alter the tipping points for ecosystem collapse and recovery (Reynolds & Aldridge, 2021; Schiel *et al*., 2018)

### 3.2.4 Oxygenation and temperature: parameters and metrics

Both dissolved oxygen and temperature are amongst the most straightforward physico-chemical elements to measure in the field and there is unlikely to be any serious concerns about methods. In rivers Biochemical Oxygen Demand (BOD) is also widely used, mostly based on 5-day incubations (a few countries use 7-day incubations) and methods are, again, well established. All thresholds reported for BOD were based on central tendencies (means, medians). A few countries also use Chemical Oxygen Demand (COD) but there were too few to allow meaningful comparisons. However, BOD is no longer measured in several countries (e.g. Denmark, France, the Netherlands, Sweden), presumably due to better urban wastewater treatment or the expense in relation to the perceived low level of unique information provided by this parameter. However, in light of the problems with direct measurement of DO because of diurnal fluctuation, there may be a case for retaining BOD, especially where heterotrophic activity is likely to be important.

However, these measurements will only provide useful information if appropriate sampling strategies and summary metrics are used. The long-standing concern about the impact of reduced DO concentrations arising from heterotrophic microbial activity from relatively constant point-source inputs of organic pollution may explain the widespread use of central tendencies (means or medians) to summarise oxygenation parameters (Tables 3.1, 3.2). However, as organic loadings have decreased, due to improved urban wastewater treatment, the relevance of these measurements has also declined and focus should shift to detecting extreme events (e.g. $O_2$ decline due to sewage overflows, night-time anoxia leading to fish kills in polluted rivers, or specific zones coming under increasing pressure such as anoxia developing in bottom waters of stratified lakes in late summer (or ice-covered lakes in late winter). The likelihood of events such as these are better captured through the use of lower percentiles although the relevance of such metrics will also depend on the frequency of sampling and the time at which the sample is collected. Upper percentiles are also quite widely used. These, presumably, are measures of the secondary effects of eutrophication, implying that diurnal cycles may result in night-time anoxia. Reliable estimates of both lower and upper percentiles will require more measurements than needed for measures of central tendency, resulting in greater costs.

Furthermore, as both temperature and oxygen concentration vary across a diurnal as well as an annual cycle, the timing of spot samples is critical and the reliance on manual sampling will create an inherent bias (i.e. criteria are likely to be based on conditions experienced during "office hours"). **Criteria based around infrequent measurements of inherently variable parameters are unlikely to be effective indicators of conditions that will support good ecological status.** Continuous monitoring seems to be a better way forward, but this comes at a price.

**Table 3.1:** Overview of parameters and metrics used to monitor oxygenation conditions in inland waters (from Kelly *et al*., 2021). Numbers refer to the number of countries reporting standards that use a particular parameter/metric combination. Metrics have been split into those that measure the central tendency (e.g. mean, median) and those measuring a more extreme statistic (e.g. percentiles, maximum and minimum).

| **Supporting element** | **central tendency** | | **percentile** | **other** |
|---|---|---|---|---|
| | **annual** | **seasonal** | | |
| **Lakes** | | | | |
| Dissolved oxygen (mg/L) | 3 | 3 | 3 | 1 |
| % oxygen saturation | 3 | 1 | 1 | 1 |
| **Rivers** | | | | |
| Dissolved oxygen (mg/L) | 9 | | 7 | 1 |
| % oxygen saturation | 3 | 1 | 8 | |
| BOD5 | 13 | | 7 | 1 |

**Table 3.2:** Overview of parameters and metrics used to monitor oxygenation conditions in transitional and coastal waters (from Teixeira *et al.*, 2022). Numbers refer to the number of countries reporting standards that use a particular parameter/metric combination. Metrics have been split into those that measure the central tendency (e.g. mean, median) and those measuring a more extreme statistic (e.g. percentiles, maximum and minimum).

| Supporting element | central tendency | | percentile | | other |
|---|---|---|---|---|---|
| | annual | seasonal | annual | seasonal | |
| **Transitional** | | | | | |
| Dissolved oxygen (mg/L) | 1 | | 3 | 1 | |
| % oxygen saturation | 2 | 1 | 1 | 1 | |
| BOD5 (mg/L) | | | 1 | | |
| **Coastal** | | | | | |
| Dissolved oxygen (mg/L) | 2 | 1 | 2 | 1 | |
| % oxygen saturation | 2 | 1 | 2 | | |

Duration of low DO or high temperature events also needs to be considered. For Italian transitional waters (such as the Venice Lagoon), periods of anoxia (< 1.0 mg $L^{-1}$ DO) exceeding 24 hours result in a downgrading of status from good to moderate whilst periods of hypoxia (≥ 1, < 2 mg $L^{-1}$) exceeding 24 hours prompt a two-year period of macrozoobenthos monitoring to confirm status (decree 260/2010: https://www.gazzettaufficiale.it/eli/id/2011/02/07/011G0035/sg).

Oxygen debt and oxygen consumption, a proxy for oxygen budget, along with the depth of hypoxia are complementary measures used in the Baltic (Stoicescu *et al.*, 2019). Season and environmental conditions determine the adequacy of each of these measures, but natural variability makes it a challenge to detect anthropogenic causes driving variation in oxygen patterns and values.

### 3.2.5 Approaches to setting thresholds

***Use of field data***

Experiences on setting oxygenation standards from contemporary monitoring data are mixed. In estuaries in northern Spain, strong relationships between dissolved oxygen and metrics based on fish and crustaceans were derived (Uriate & Borja, 2009; Borja *et al.*, 2006), from which thresholds could potentially be derived. However, other studies have had less success. In the case of Luxembourg rivers, a capacity to differentiate bad status from other classes was observed, but not between good and moderate status (Kelly & Birks, unpublished data: Fig. 3.1). This is more likely to relate to the limitations of the measurement of the supporting element (as discussed above) than to a lack of an effect. These data suggest that a threshold closer to 10 mg $L^{-1}$ would be appropriate to protect good status yet, at the same time, the lack of differentiation between high, good, moderate and poor status also suggests that oxygen concentrations are not the primary criterion determining ecological condition in this region.

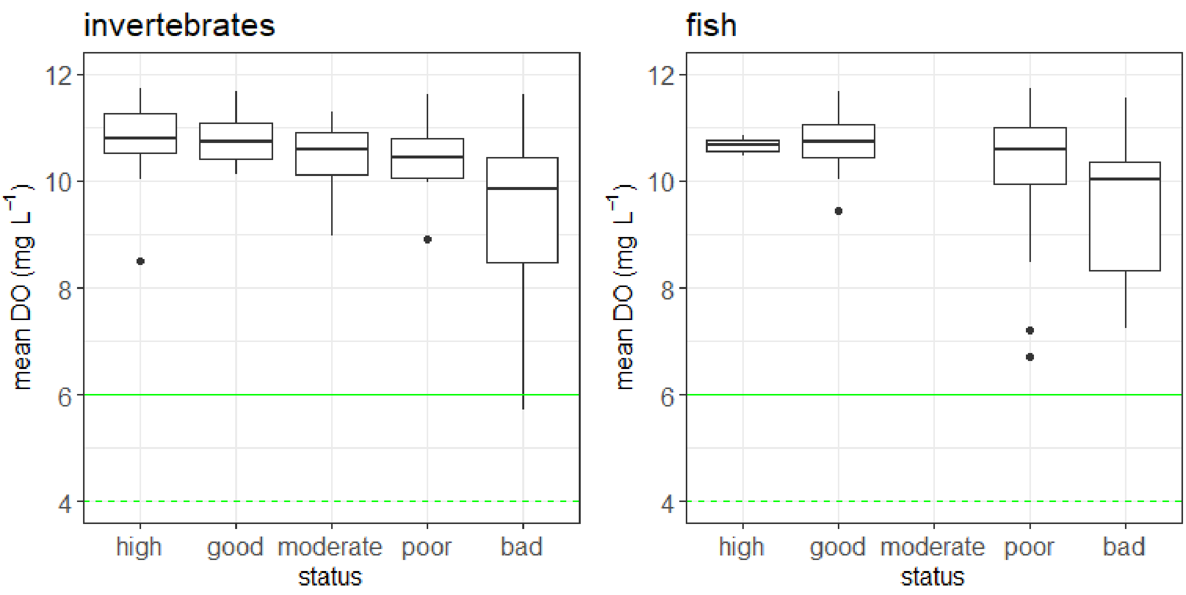



**Figure. 3.1.** Relationship between ecological status and mean dissolved oxygen concentrations in Luxembourg rivers (2015-2017). The green lines show limits mandated by the Freshwater Fish Directive (2006/44/EC) for salmonid (solid) and cyprinid (dashed) fish. Unpublished data from Ministère de l'Environment, du Climat et du Développement durable, le Gouvernement du Grand-Duché de Luxembourg.

Analysis of EU-wide data submitted to WISE supports this assertion. Preliminary visual examination of invertebrate-based classifications from lowland rivers suggested some national and type variation in mean dissolved oxygen concentration albeit with considerable noise and a few outliers (e.g. Mediterranean regions of Spain: Fig. 3.2). Using the boundary prediction model of Phillips et al. (2023), a good/moderate boundary of about 10 mg $L^{-1}$ was estimated for lowland rivers (Fig. 3.3). A broad conclusion from this analysis is that the limits set by the Freshwater Fish Directive are likely to be too lenient in many cases. Validation of this general limit against national data is recommended as there will be exceptions where different limits are required.

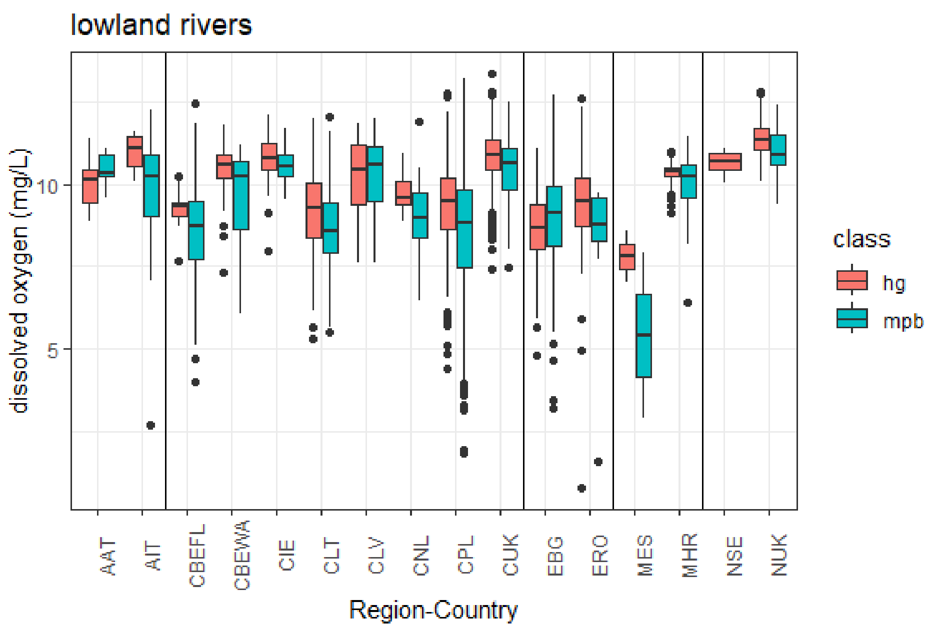



**Figure. 3.2**. Range of dissolved oxygen concentrations measured when invertebrates are at high or good status (“hg”) or moderate, poor or bad status (“mpb”) in national data submitted to the WISE database. Region-Country labels are two-letter national codes preceded by their geographical region (A = Alpine; C = Central-Baltic; E = Eastern Continental; M = Mediterranean; N = Northern). Data from Belgium are further subdivided into Flanders (“BEFL”) and Wallonia (“BEWA”).

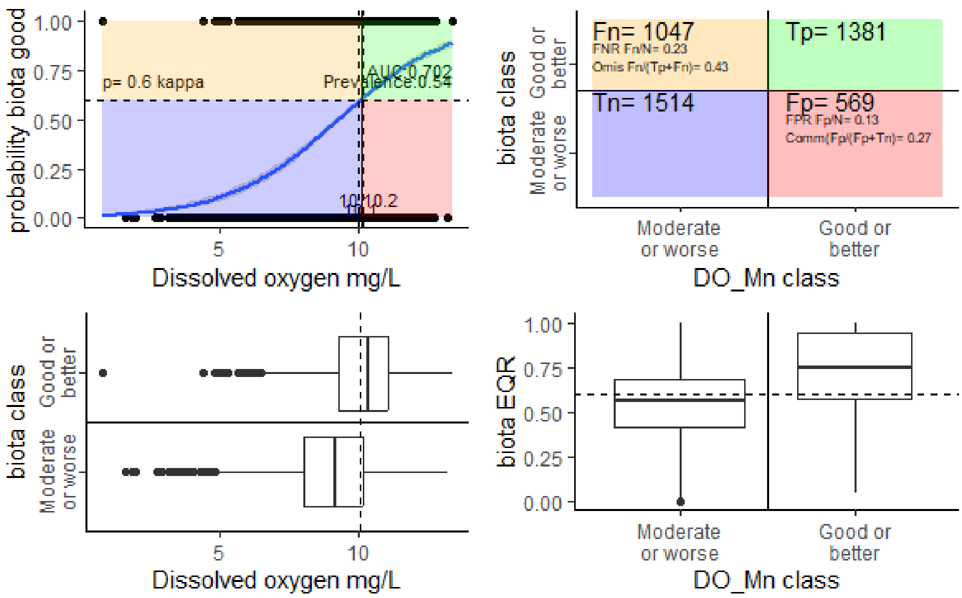



**Figure. 3.3.** Results of fitting binary logistic model to invertebrate data from lowland rivers submitted to WISE: a) scatter plot with model fit and predicted boundary concentrations for p threshold determined by kappa; b) confusion matrix showing number of true and false records and measures, c) boxplots showing range of DO concentrations for waterbodies classified by biota. d) boxplots showing range of EQR for waterbodies classified using the predicted DO boundary. dotted lines show boundary values.

A similar analysis performed using mean BOD5 rather than dissolved oxygen also suggested some national and type variation along with considerable noise (Fig. 3.4). An analysis using a subset with only those national datasets with a significant difference between high/good and moderate/poor/bad resulted in a relatively weak model (AUC[1] below the recommended threshold of 0.7) suggested a good/moderate boundary of 1.8 mg $L^{-1}$ $O_2$ for lowland rivers (Fig. 3.5) which, as for DO, is more stringent than the limits set by the Freshwater Fish Directive. However, it is likely that stronger models could be developed using datasets focussed on particular countries or regions.

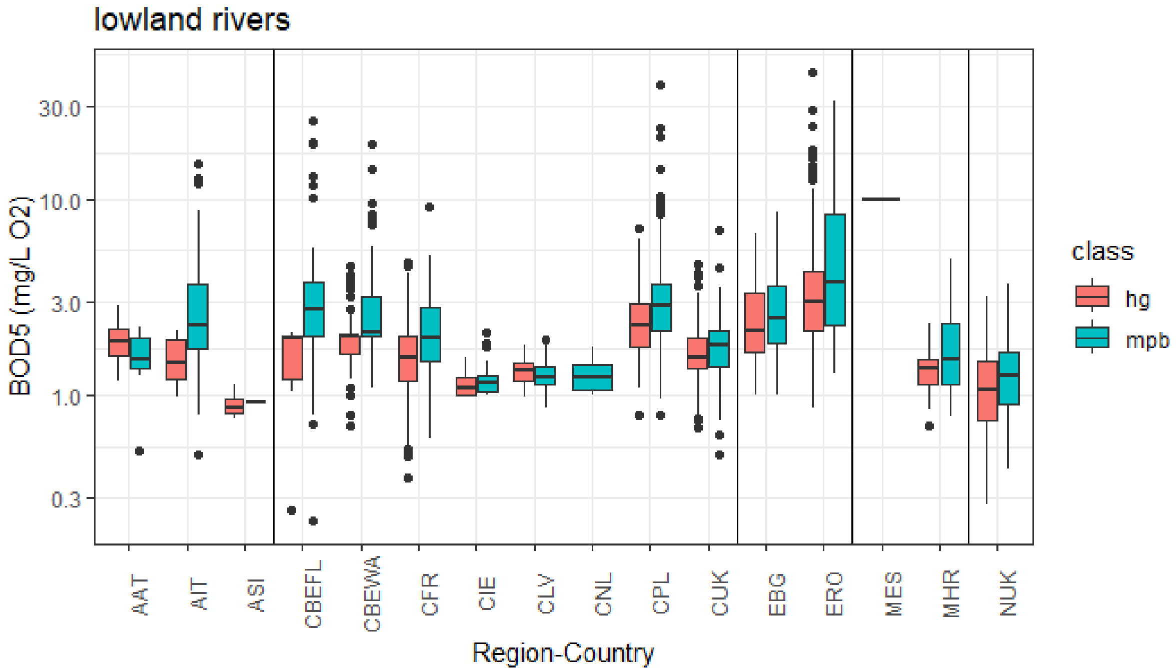



**Figure. 3.4.** Range of biochemical oxygen demands (as mg $L^{-1}$ $O_2$) when invertebrates are at high or good status ("hg") or moderate, poor or bad status ("mpb") in national data submitted to the WISE database. Region-Country labels as in Fig. 3.2.

[1] AUC = "area under the curve", a summary statistic used in binary logistic regression, referring to the "receiver operating curve, ROC, which plots the true positive rate as a function of the false positive rate. See Phillips et al. (2023) for more details.

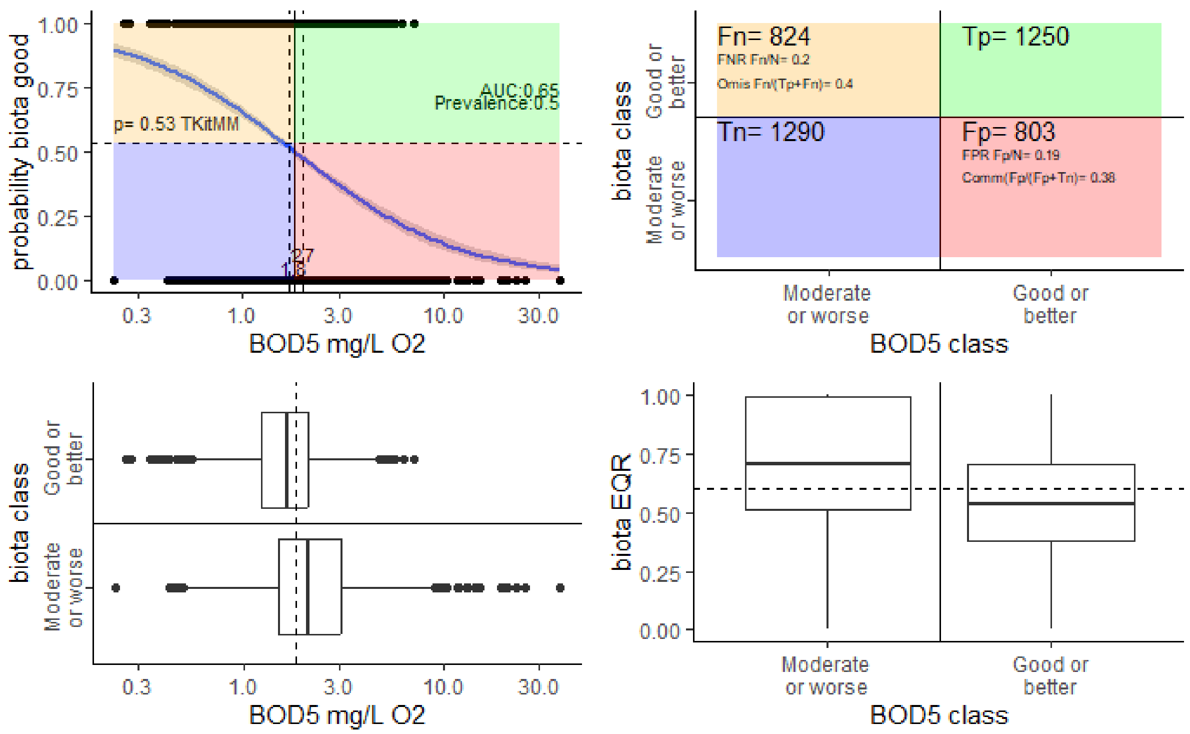



**Figure. 3.5.** Results of fitting binary logistic model to invertebrate data from lowland rivers submitted to WISE: a )scatter plot with model fit and predicted boundary concentrations for p threshold determined by omission-commission; b) confusion matrix showing number of true and false records and measures, c) boxplots showing range of BOD5 for waterbodies classified by biota. d) boxplots showing range of EQR for waterbodies classified using the predicted BOD5 boundary. Dotted lines show boundary values.

Where historical data are available, it may be possible to use this to establish criteria. This has been done for estuaries in northern Spain, where a relationship between severe hypoxia and fish diversity is apparent in a dataset spanning 30 years (Fig. 3.6). A further example, for the Baltic Sea, is described in Helcom (2013) and Phillips et al. (2018).

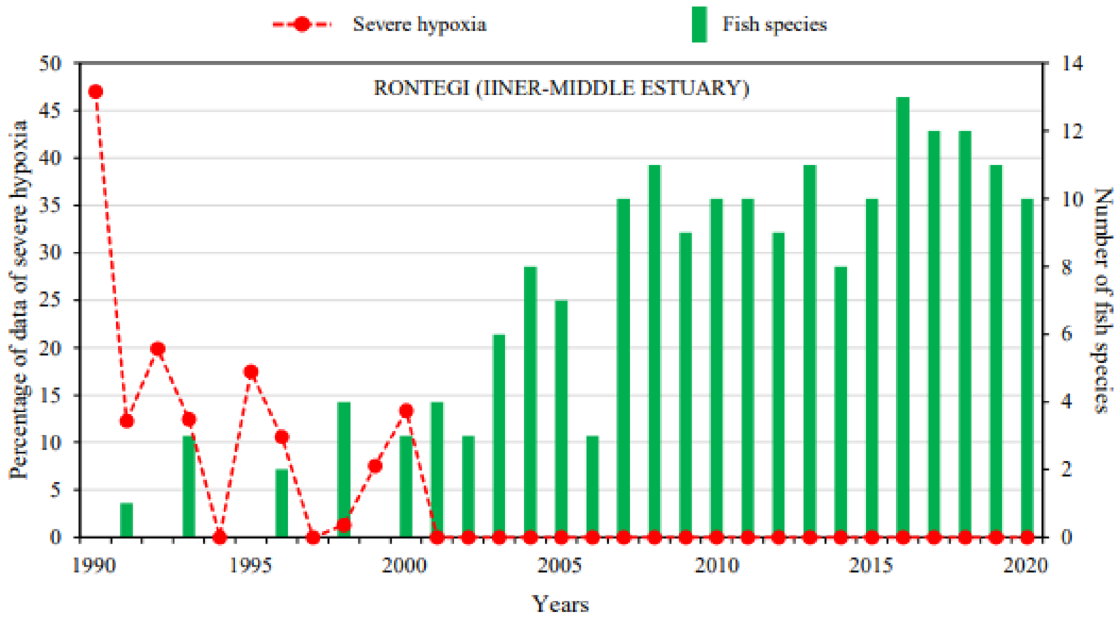



**Figure. 3.6.** Relationship between fish diversity and severe hypoxia in the inner-middle estuary of the Nerbioi River, northern Spain (source: Franco & Borja., in Lyche Solheim et al., (2021); data collection funded by: Consorcio de Aguas Bilbao Bizkaia).

***Ecotoxicology***

A very large number of studies on the effect of low dissolved oxygen concentrations on aquatic life have been performed, although it is often difficult to relate this to thresholds set by Member States due to the wide range of approaches to sampling and data aggregation. Rivers represent one of the most straightforward situations and allow data collected by Member States to be compared directly with results from ecotoxicology experiments.

Fig. 3.7 shows the range of dissolved oxygen concentrations associated with invertebrates at high or good status. Many of the national standards reported in Kelly *et al.* (2022) were at or close to levels set in the Freshwater Fish Directive (6 mg $L^{-1}$; represented by a dashed line). Results from ecotoxicology experiments (summarised by US EPA, 1986; see also Alabastar & Lloyd, 1980) show a responses on both sides of this line. If, for ease of interpretation, we interpret “slight production impairment” to be roughly compatible with “good ecological status”, then 6 mg $L^{-1}$ will not protect early life stages of salmonids (particularly as DO concentrations in gravels will be substantially lower than in the water column) and will just protect adult salmonids. By contrast, this criterion should protect most non-salmonid fish. Questions must remain, however, about how limits derived from these thresholds compare with the parameters and metrics used by Member States for their DO assessments and, in particular, how sensitive these are to cyclical fluctuations in DO.

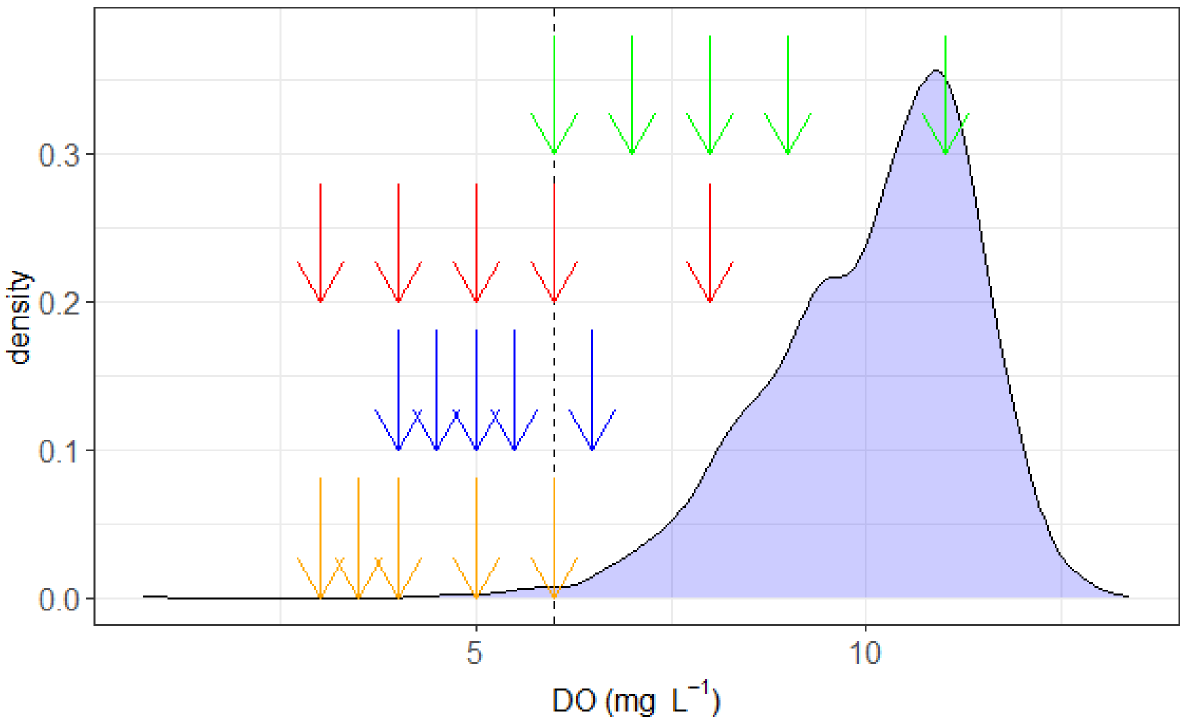



**Figure. 3.7**. Density plot of dissolved oxygen concentrations associated with invertebrate assemblages at high or good status overlain with threshold values derived from experimental studies. Green = salmonids (early life stages); red = salmonids (other life stages); blue = non-salmonids (early life stages); orange = non-salmonids (other life stages). Arrows indicate, from left to right, limit to avoid acute mortality, severe production impairment, moderate production impairment, slight production impairment, no production impairment. Dashed line = lower limit prescribed by Freshwater Fish Directive (2006/44/EC) for Salmonid waters. Density distribution based on data from WISE, ecotoxicology limits from US EPA (1986).

## 3.3 Range of ecology-related thresholds

### 3.3.1 General comments

The range of oxygenation criteria used within the EU are discussed in Kelly et al. (2022) and Teixeira et al. (2022). The wide range reported is due to the following:

- the purpose (assessing chronic effects will require different criteria to assessment of acute effects);
- the way that the standards were set (for example, based on ecotoxicology end-points rather than derived empirically);
- the assessment concept (for example, surface water or bottom water depending on which part of the water column is being assessed);
- the summary statistics used;
- adjustments for other factors (for example, salinity range, reference conditions for different types);
- different aggregation rules; and,
- the regulatory mechanism within which the standards are used.

### 3.3.2 Oxygenation in inland waters

A general conclusion for oxygenation in inland waters was that national thresholds are often lower than field data suggests (Kelly et al., 2022). This, however, assumes that DO is driving classifications whereas, in many cases, low dissolved oxygen is one of a number of stressors, with phosphorus and ammonium explaining a greater portion of the variation within the data than DO (Kelly et al., 2022). Thus, DO thresholds are, at best, "failsafes" for situations where other stressors are not detected by routine monitoring programmes. Whether existing thresholds are adequate for this purpose is a question that will have very context-specific answers. Our view is that the discussion needs to consider more than just thresholds, as the ability of monitoring programmes to detect short-term acute episodes will become more important in the future, and many national programmes are not yet equipped to meet this challenge.

Many national standards for BOD also seem to be derived from the Freshwater Fish Directive (3 mg $L^{-1}$ $O_2$ for salmonid waters; 6 mg $L^{-1}$ $O_2$ for cyprinid waters) (Fig. 3.8). The 75th percentile for BOD in rivers reported to WISE where the invertebrates are at least good status is 2.3 mg $L^{-1}$ $O_2$ (see above). This suggests that some national standards may need to be tightened, especially for salmonid rivers. However, interpretation of relationships with BOD are complicated by correlations with phosphorus and ammonium (Kelly et al., 2022), and there are also likely to be type-specific effects, meaning that checks would need to be conducted within each country. Many thresholds for BOD cited in the literature (e.g. Kristensen & Hansen, 1994; Mladenović-Ranisavljević et al., 2018; Vigiak et al., 2019) are not derived from ecological criteria, and should be treated with caution. Recent studies where ecology has been used have either not focussed specifically on WFD criteria or were concerned with defining "reference conditions". Jones et al. (2008), for example, recommended 1.8 – 2.0 mg $L^{-1}$ $O_2$ as thresholds for Special Areas of Conservation in the UK whilst Pardo et al. (2012) proposed average concentrations of 2.8 mg $L^{-1}$ $O_2$ and 90th percentiles of 3.6 mg $L^{-1}$ $O_2$ for screening reference sites in Central-Baltic GIG. Other GIGs used similar thresholds (Table 3.3). Friberg et al. (2010) concluded that "important macroinvertebrate taxa are reduced at concentrations of $BOD_5$ that are normally perceived as indicating unimpacted stream conditions". They supported this with graphs showing, in many cases, exponential curves showing the response of sensitive invertebrate taxa to BOD, suggesting that the gap between "unimpacted" (reference conditions, high/good boundary) and a "slight" impact will be narrow.

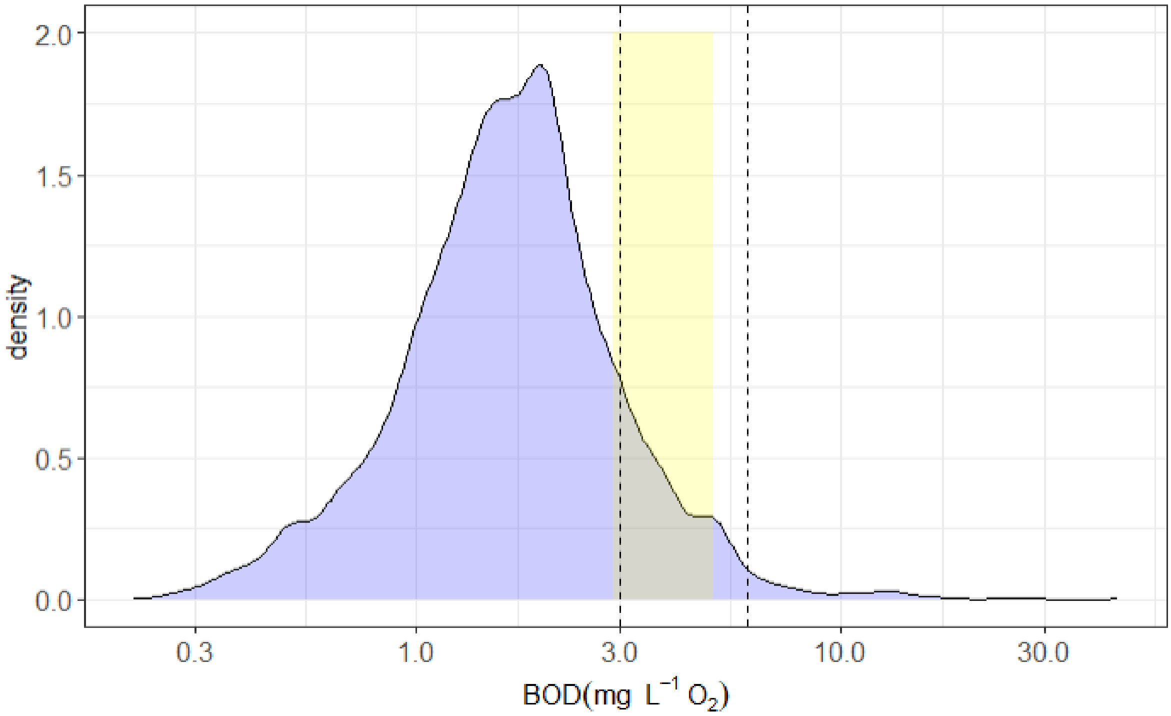



**Figure. 3.8.** Density plot of 5-day biochemical oxygen demand (BOD) associated with invertebrate assemblages at high or good status overlain with threshold values from the Freshwater Fish Directive (3 & 6 mg $L^{-1}$ $O_2$ for Salmonid and Cyprinid waters respectively). The yellow rectangle represents the range between 25$^{th}$ and 75$^{th}$ percentiles of national standards for rivers reported in Kelly et al. (2022). Density distribution based on data from WISE.

**Table 3.3.** Threshold concentrations for BOD5 used to screen for reference conditions in the intercalibration exercises for rivers. BOD was not used as a screening criterion in Mediterranean GIG (Feio et al., 2013).

| GIG | BOD Threshold (mg $L^{-1}$ $O_2$) | Note |
|---|---|---|
| Alpine | Mean: < 2<br>90$^{th}$ percentile: < 2.75 | |
| Central-Baltic | Mean: < 2.4<br>90$^{th}$ percentile: <3.6 | (R-C3:<br>mean < 2;<br>90$^{th}$ percentile < 2.75) |
| Eastern Continental | Mean < 2<br>Mean < 2.4 | (R-E1a and R-E1b types)<br>(other types) |
| Northern | Mean < 1.6 | (UK and IE only) |

A comparison with oxygenation thresholds set in countries in the G20 indicates that thresholds for DO concentrations tend to be more stringent in the EU than elsewhere, despite evidence above that suggests that they are not yet strict enough to protect good status. On the other hand, thresholds for percent saturation in lakes seem to be more relaxed (Fig. 3.9). However, this comparison is complicated by the lack of detail about sampling regimes: lower values for percent saturation could arise if, for example, countries in the EU were mostly sampling from the hypolimnion. Many national websites do not present separate DO criteria for lakes and rivers, and we suspect that many are set with rivers, rather than standing waters, in mind.

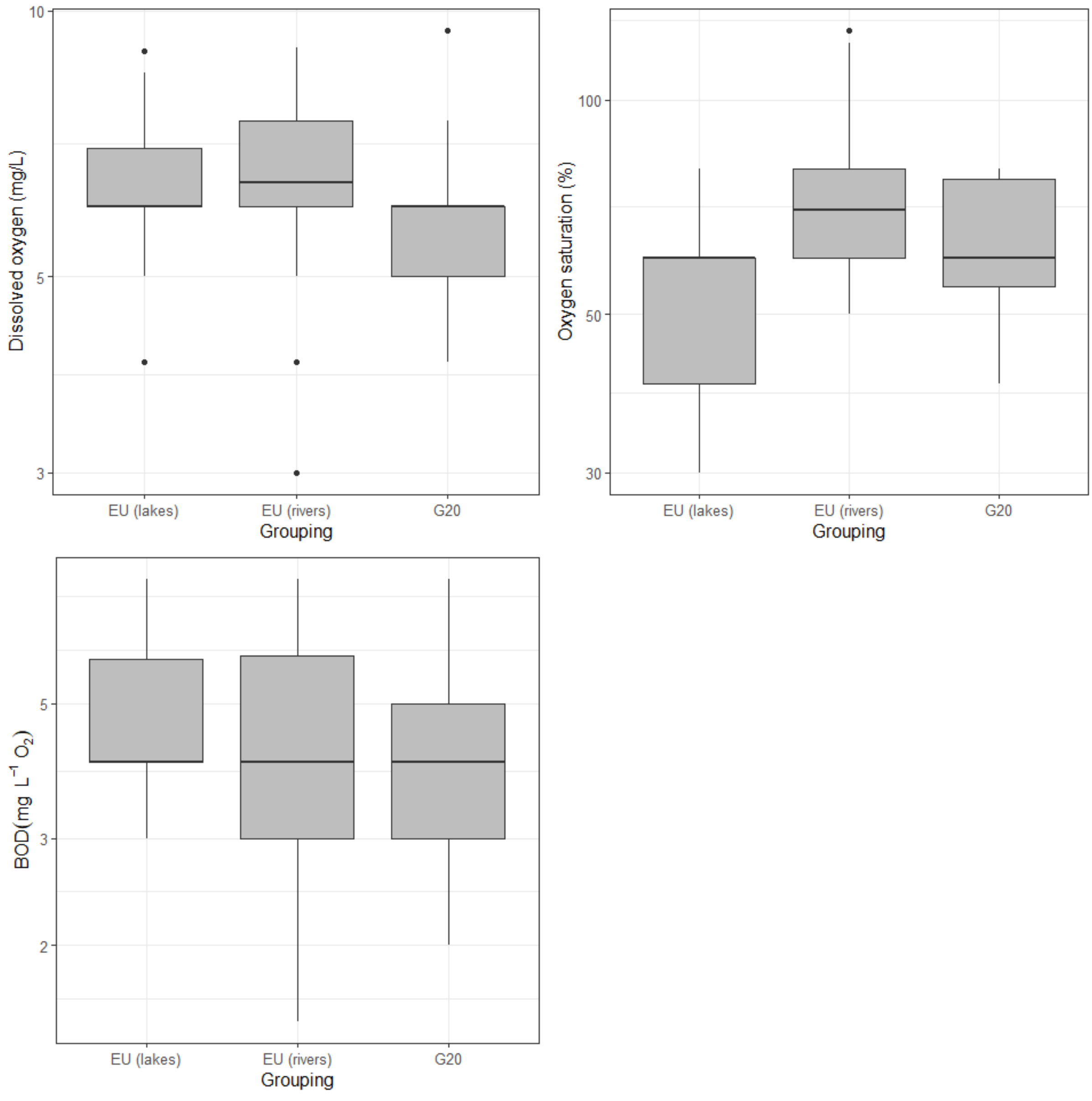



**Figure. 3.9**. Comparison between national oxygenation thresholds for EU Member States and G20 countries excluding those in the EU. EU data from Kelly et al. (2022); G20 from various internet sources.

### 3.3.3 Oxygenation in transitional and coastal waters

The situation in TRAC waters is complicated by the variety in approaches to assessing dissolved oxygen, and also by interactions between oxygen and salinity. There are fewer direct references to oxygenation in TRAC than in inland waters in existing Directives, with only the Shellfish Waters Directive (2006/113/EC) citing mandatory thresholds. These are average values, expressed as percent saturation and, as such, have the problems discussed earlier. They also apply only to a limited range of relatively shallow waters and are designed to protect only one component of the biota. Best et al. (2007) offer an alternative set of criteria for surface waters ("atmospherically ventilated layers") that will also serve in situations where bottom water is replaced rapidly. These are expressed as concentrations of dissolved oxygen and are adjusted for salinity. Thresholds for "good ecological status" range from 5.0 mg $L^{-1}$ at the freshwater limit to 4.0 mg $L^{-1}$ for full-strength seawater. The metric in this case is the 5$^{th}$ percentile, which implies a need for at least 20 measurements per year. Caution is needed, however, as evidence elsewhere in this chapter suggests that historic freshwater thresholds may be too lenient (Figs 3.1, 3.3, 3.5). Best et al. (2007) also recommend a "fundamental intermittent standard" of 2 mg $L^{-1}$ with a return period of six years. This recognises the uncertainty in the data from which thresholds are determined, and acts as a failsafe to catch episodic events.

The MSFD requires that, in developing their marine strategies, Member States use existing regional sea conventions to co-ordinate amongst themselves. Of the four sea conventions that apply to EU Member States (and Norway), OSPAR (NE Atlantic) and HELCOM (Baltic Sea) have developed criteria for "oxygen deficiency" (or "oxygen debt") in the bottom layers of stratifying water. OSPAR (2005) use a universal threshold of 6 mg $L^{-1}$ whilst thresholds for HELCOM are defined as the 95$^{th}$ percentiles during the period before 1940 (6.37 mg $L^{-1}$ for Bornholm Basin; 8.66 mg $L^{-1}$ for the Baltic Sea proper (Helcom, 2013). To the best of our knowledge, no similar guidance yet exists for the Mediterranean (Barcelona Convention) or Black Sea (Bucharest Convention).

### 3.3.4 Temperature

Setting criteria for temperature presents a substantial challenge. Realistically, many changes observed in recent years are due to regional and global factors that cannot be controlled via Programmes of Measures, questioning the relevance of "thresholds" as understood in WFD implementation. There are, however, sufficient instances of localised warming (occasionally cooling) due to human activities for thresholds to have a role. In practice, thresholds need to be closely-linked to local conditions. Consequently, Europe-wide comparisons are impossible, and temperature was not included in either Kelly *et al*. (2022) or Teixeira *et al*. (2022). Upper percentiles from monitoring data (ideally spanning several years) should give a good indication of expected temperatures for river types in different seasons, providing thresholds against which localised warming can be assessed.

## 3.4 Effect of climate change

For both oxygenation conditions and temperature, it is clear that an approach that relies solely on annual means based on spot samples from surface waters at a relatively low frequency is unlikely to provide sufficient protection. Whilst these approaches should detect long-term trends in average conditions, and provide continuity with historical measurements, they are inadequate for detecting acute events. For these, thresholds based on lower percentiles, and a greater frequency of sampling (especially at times when risk is greatest) will be necessary. There is likely to be greater reliance on continuous monitoring for both oxygenation conditions and temperature in the future, although this will also increase costs. Also, the limitations of current measurement approaches means that it is unlikely to be possible to derive thresholds for acute events from monitoring data.

The scientific community recognizes also that climate change poses extra challenges to routine measurements of dissolved oxygen as the detection of e.g. oceanic and coastal deoxygenation trends (Grégoire *et al*., 2021; Pitcher *et al*., 2021) requires data with higher accuracy (i.e., the difference between the measurement and a true value) and precision (i.e., the repeatability between measurements of a same sample) than that for estimating the oxygen levels or detect hypoxia events in a system.

It is also noticed that, for instance, reductions in nutrient loadings might have a larger effect on oxygen conditions than the effects of climate change, despite the fact that climate change will partially reduce the gains in oxygen from nutrient management (e.g., Irby *et al*., 2018 in Pitcher *et al*., 2021).

## 3.5 Conclusions and recommendations

- Thresholds for dissolved oxygen concentrations that are based on values in the Freshwater Fish Directive are unlikely to be sufficiently protective to achieve WFD objectives in rivers. Alternative values are presented in Table 3.4 but Member States are also encouraged to derive appropriate thresholds from their own data.

- It is harder to generalise for lakes, as Member States collect data in a number of different ways. Once again, we encourage Member States to derive appropriate thresholds from their own data.

**Table 3.4.** Summary guidance for setting ecological thresholds for dissolved oxygen concentration in rivers. See section 1.1 for explanation of use recommendations (× and ✓).

| Level | Threshold | Source | Use? |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | 6 mg $L^{-1}$ (salmonid waters)<br>4 mg $L^{-1}$ (cyprinid waters) | Freshwater Fish Directive (78/659/EEC | ✗ |
| 2. Threshold based on published literature: general prescription | 9 mg $L^{-1}$ (salmonid waters)<br>5.5 mg $L^{-1}$ (cyprinid waters) | US EPA (1986) (values are for "slight production impairment") | ✓ |
| 3. Threshold based on analysis of WISE/SoE data | 10 mg $L^{-1}$ (lowland rivers) | Analyses of invertebrate data for this project | ✓ |
| 4. Threshold based on targeted published literature: | See notes below | | ✓ |
| 5. Threshold based on national data | See notes below | | ✓ |

- Thresholds for biochemical oxygen demand for inland waters that are based on values in the Freshwater Fish Directive are also unlikely to be sufficiently protective to achieve WFD objectives in some rivers types (particularly for cyprinid rivers). Alternative values are presented in Table 3.5 but Member States are encouraged to derive appropriate thresholds from their own data.
- The wide range of biogeographical conditions combined with the array of chronic and acute problems associated with these supporting elements means that it may only ever be possible for Member States to conform to a set of broad guiding principles for setting thresholds to protect good status in some water categories. In stratified lakes, in particular, an understanding of fish habits will be necessary in order to set appropriate thresholds. A species that inhabits the hypoliminion, for example, will not be directly protected by a criterion based on measurements close to the surface.
- Most existing measurements, and thresholds set using these, are of only limited value in protecting against the effects of climate change. More detailed measurements, and a better understanding of the role of extreme events, will be needed to set protective standards. These events will need to be defined in terms of both their magnitude and duration. This, in turn, will have practical implications, in terms of frequency of sampling (particularly during summer months) and time of day. Greater use of continuous monitoring is envisaged in the future.
- Warming due to climate change may mean that oxygen concentrations fall even though percent saturation remains unchanged. The use of criteria based on percent saturation should be discouraged in the future.
- For both oxygenation conditions and temperature we recommend a move away from spot measurements and towards continuous measurements, to better characterise the extreme events that are likely to cause ecosystem damage in the future.

**Table 3.5.** Summary guidance for setting ecological thresholds for biochemical oxygen demand in rivers. See section 1.1 for explanation of use recommendations (× and ✓).

| **Level** | **BOD Threshold** | **Source** | **Use?** |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | 3 mg $L^{-1}$ $O_2$ (salmonid waters)<br>6 mg $L^{-1}$ $O_2$ (cyprinid waters) | Freshwater Fish Directive (78/659/EEC | × |
| 2. Threshold based on published literature: general prescription | Values > 3 mg $L^{-1}$ $O_2$ unlikely to be sufficiently protective in either salmonid or cyprinid waters | Friberg et al. (2010) | ✓ |
| 3. Threshold based on analysis of WISE/SoE data | 1.8 mg $L^{-1}$ $O_2$ (lowland rivers) | Analyses of invertebrate data for this project | ✓ |
| 4. Threshold based on published literature: targeted | Not possible to use ecotoxicological approaches; few published criteria are derived from ecology | | × |
| 5. Threshold based on national data | See guidelines above. | | ✓ |

**Table 3.6.** Summary guidance for setting ecological thresholds for dissolved oxygen concentration in well-mixed TRAC waters. See section 1.1 for explanation of use recommendations (✗ and ✓).

| Level | Threshold | Source | Use? |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | ? | None present values as concentration; Shellfish Waters Directive presents % sat but not recommended | ✗ |
| 2. Threshold based on published literature: general prescription | | Values from Best et al. need further local validation | ✓ |
| 3. Threshold based on analysis of WISE/SoE data | | Not possible within current work package – see Teixeira et al. (2023) | ✓ |
| 4. Threshold based on targeted published literature: | See 1.1 | | ✓ |
| 5. Threshold based on national data | See 1.1 | | ✓ |

**Table 3.7** Summary guidance for setting ecological thresholds for oxygen debt below the halocline in stratified TRAC waters. See section 1.1 for explanation of use recommendations (✗ and ✓).. See OSPAR (2005) and Helcom (2013) for further details on oxygen debt.

| Level | Threshold | Source | Use? |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | NE Atlantic 6 mg $L^{-1}$<br>Baltic Sea: various values | OSPAR (2005)<br>HELCOM (2013) | ✓ |
| 3. Threshold based on targeted published literature: | See 1.1 | | ✓ |
| 4. Threshold based on national data | See 1.1 | | ✓ |

**Table 3.8.** Summary guidance for setting ecological thresholds for temperature in rivers. See section 1.1 for explanation of use recommendations (✗ and ✓).

| Level | Threshold | Source | Use? |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | Increase downstream of thermal discharge<br>1.5 °C (Salmonid waters)<br>3 °C ( (cyprinid waters)<br>Temperature must not exceed:<br>21.5 °C (Salmonid waters)<br>28 °C ( (cyprinid waters)<br>(10 °C during breeding periods of species which need cold water for reproduction) | Freshwater Fish Directive (78/659/EEC) | ✗ |
| 2. Threshold based on published literature: general prescription | Not possible to make a general recommendation | | ✗ |
| 3. Threshold based on published literature: targeted | Not possible to make a general recommendation. | | ✗ |
| 4. Threshold based on national data | Upper percentiles from monthly monitoring data (ideally spanning several years) | Hale & Müller (2014; 2017) | ✓ |

# 4 Salinisation

## 4.1 Background

Salinisation, defined as the increase in total concentration of major ions in a water body, is a long-standing challenge in inland waters which, with the rapid advance of global warming, is becoming more widespread, with implications for their biota (Fig. 4.1) and for the ecosystem services that water bodies provide (Kaushal et al., 2018; Jeppesen et al., 2020; Cunillera-Montcusí et al., 2022). Salinisation also influences physical and physico-chemical processes such as thermal stratification and the capacity of water to retain dissolved oxygen. It is also a direct pressure defined in Annex V of the Water Framework Directive (EU, 2000). However, fewer countries in the European Union have protective standards than for other quality elements specified in Annex V of the WFD (Kelly et al., 2022b). Moreover, salinity interacts with other stressors, with many examples of both additive and antagonistic relationships on organisms (Rotter et al., 2013, Velasco et al., 2019), complicating or disguising the scale of effects.

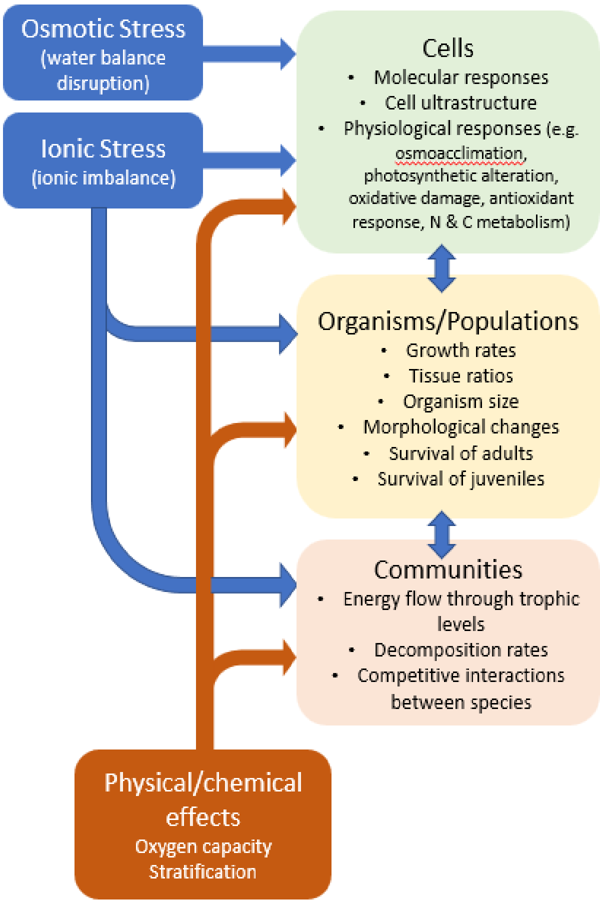



**Figure. 4.1.** Interactions between stresses and biological processes which determine ecological responses to salinisation of inland waters. Adapted from Sandoval-Gil et al. (2023).

There are several reasons for salinisation of inland waters:

- Hydrological alterations due to climate change. These include:
    - saline incursions in coastal areas due to rises in sea level;
    - changes in precipitation patterns and in natural hydrological regimes. Many countries in southern Europe are predicted to receive less precipitation in the coming decades, meaning less runoff and potentially leading to salinisation of lakes and rivers through evaporation; this will be exacerbated by changes in agricultural practices (e.g. increased demand for water leading to greater abstraction).
- point source discharges from mining and industrial processes (e.g. potash mining, leather industry); and,
- diffuse runoff from salted roads during winter ($10 \times 10^6$ tonnes annually in France: Barbier et al., 2018; $0.5 \times 10^6$ tonnes in 2019/20 in Poland: Szklarek et al., 2022);

One result of this is an upward trend in salinisation parameters such as chloride concentration in many surface waters (Box 1), with implications for the biota and the services that these provide to society. There is a long history of using organism's responses to salinity as markers for tracking climate change (Reed, 1998; Fritz et al., 1999) and, more recently, a growing awareness that salinisation of inland waters is a widespread and inevitable outcome of global warming (Cañedo-Argüelles et al., 2013; Short et al., 2016; Kaushal et al., 2021; Cunillera-Montcusí et al., 2022). Although largely considered to be a problem for inland waters, greater evaporation and abstraction from inland waters has also contributed to increased salinity in some parts of the Mediterranean Sea. Desalinisation plants, too, discharge brine which can also contribute to localised increases in salinity, with potentially damaging ecological effects (Garrote-Moreno et al., 2014; Gacia et al., 2007).

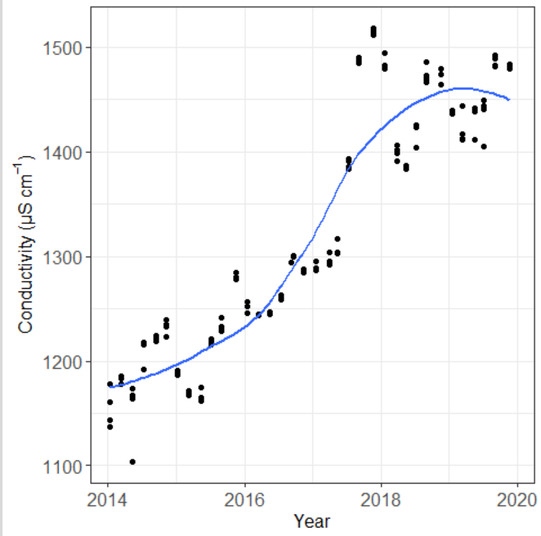
There are, however, considerable gaps in our knowledge of how this will affect ecosystems and how it interacts with other stressors. Also missing, is how the impact of salinisation will be addressed by regulators to guarantee sustainable water resources. Which chemical criteria should be used? What are the key thresholds? How well do criteria established for other purposes (irrigation, drinking water) protect ecology? Do these thresholds need adjustment if other major pressures such as nutrients are also present (and vice versa); and what biological criteria should also be used? There is no simple answer to any of these questions. The nature of the pressure (point / diffuse), the geographical region (temperate, semi-arid), local regulatory regimes and societal expectations all combine to produce answers that are tailored to particular circumstances (e.g. Griffith, 2014).

Our understanding the role of salinisation as a stressor is further complicated by the role of salinity as a typological factor. For transitional and coastal waters, it is an obligatory factor whilst in inland waters, geology contributes to ionic strength of water and, as such, is reflected by some of the parameters used to assess salinisation.

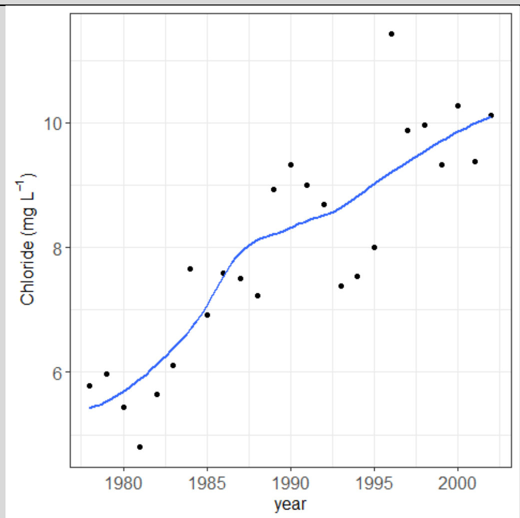
**Box 4.1. Trends in salinity in European inland waters**

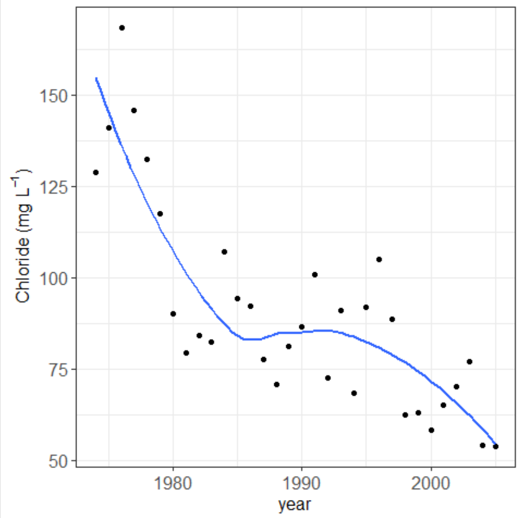
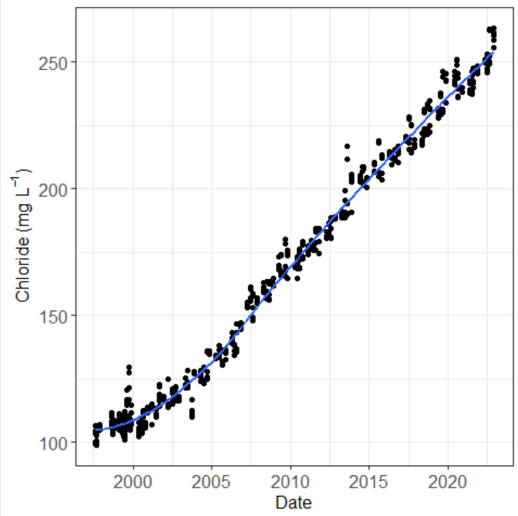
Different waterbodies will show different trends in salinity parameters, depending on location and the types of pressures encountered. The graphs below show data from four contrasting waterbodies to demonstrate this variability and to emphasise the importance of local knowledge in understanding trends and effects.

Lago di Trasimeno (a shallow lake in central Italy): conductivity has varied considerably over recent decades due to a combination of climate and management interventions (Ludoivici & Gaino, 2010). However, since 2014 there has been a consistent upward trend in conductivity to a point where some brackish taxa are beginning to appear. Data from ARPA, Umbria.




The River Rhône (Switzerland) shows a consistent upward trend in chloride concentrations since 1980 although concentrations are still well below those likely to have significant effects on the biota. Data from GEMstat.



| | |
|---|---|
| The River Mersey in north-west England, by contrast, shows a general decrease in chloride concentrations from levels where they were very likely to have influenced the biota to current levels that are below likely thresholds for chronic effects. Falls in chloride concentration reflect improvements in water quality treatment along with closures and relocation of many of the traditional industries of the region (see: Holland & Harding, 1984; Burton, 2003). Data from GEMstat. |  |
| Upward trend in chloride concentration in a groundwater-fed lake near Vienna, located close to a busy road. Values are now in the range where chronic effects of salinity are likely to be exhibited. Georg Wolfram, unpublished data. |  |

## 4.2 Why are salinity criteria necessary?

Policy objectives are written statements expressing ambition, which then need to be translated into measurable evidence-based criteria related to desired outcomes. Each criterion combines an appropriate parameter (e.g. “conductivity”), metric (e.g. “annual mean”) and threshold (e.g. “1000 µS cm$^{-1}$”) (Poikane et al., 2019a). The threshold represents the point on the stressor gradient that differentiates an “acceptable” from an “unacceptable” state and, as environmental data are intrinsically variable, it defines a summary metric rather than an absolute value, generally incorporating an appropriate degree of precaution. Such criteria are used to identify water bodies in need of restoration, prioritise those with the greatest needs, design restoration strategies and measure progress towards these objectives. As a result, they need to be set using the best scientific knowledge of the links between the stressor(s), degraded ecosystems and ecosystem services.

Salinity criteria falls into two categories:

- Use-related criteria designed to protect specific uses / ecosystem services: provision of water for domestic use, for agriculture (e.g. irrigation and watering livestock), aquaculture, energy production and industry (Fig 4.2).
- Ecology-related, intended to protect biodiversity and aquatic life. In this case, criteria are based on the ecosystem response to the increase of salinity. Different biological communities have different sensitivities, and the criteria are usually derived to protect the most sensitive components (Hart et al., 1991).

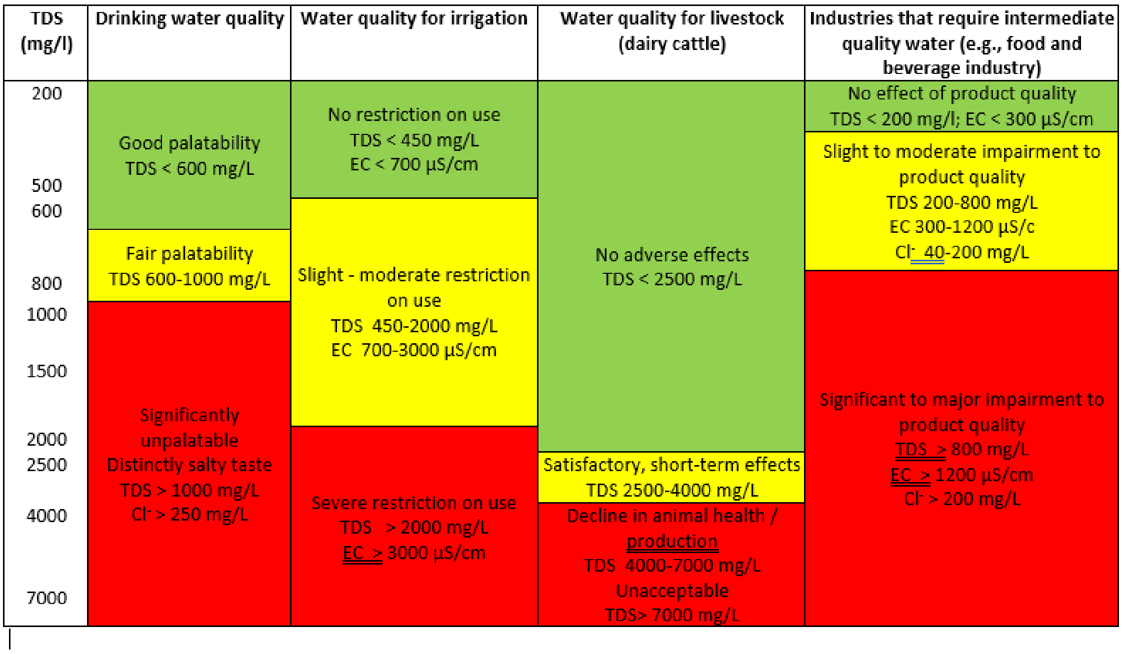



**Figure. 4.2.** Overview of the effect of salinity on water uses. Green = conditions will support the use; yellow = some effects may be observed; red = likely to be detrimental to use. EC = electric conductivity, TDS = total dissolved solids, $Cl^-$ = chloride concentration. Based on data from WHO (2022) [drinking water quality], Ayers and Westcot (1985) [irrigation], DWAF (1996) [industrial water use], ANZECC (2000) [water for livestock].

In general, development and implementation of use-related salinity criteria is more advanced than that for ecology-based criteria and these are often included in primary legislation (e.g. EC Drinking Water Directive), guidelines (e.g. FAO Water Quality for Agriculture) or regulations. They may be set by national governments or at regional or international levels. The World Health Organization, for example, has established international water quality guidelines for drinking water and other household uses (WHO, 2022). The criteria of 250 mg $l^{-1}$ of chloride and 200 mg $l^{-1}$ of sodium linked to a detectable salty taste are widely accepted and adopted in a number of national and regional legislative acts and guidelines (Fig. 4.2). Similarly, the authoritative international water quality guidelines for agricultural uses were developed for the Food and Agricultural Organization (Ayers & Westcott, 1985). The guidance is provided both for irrigation water and livestock drinking water and is based on conductivity, total dissolved solids, sodium and chloride concentrations.

In contrast, salinity criteria for ecosystems are still largely lacking. There are three principal reasons for this:

- Ecosystem complexity, leading to limited understanding of their functioning and difficulties in establishing causal relationships between increasing salinisation and ecological effects;
- Other pressures (nutrients, hydromorphology, invasive species) are often seen as a greater priority for policymakers (Poikane et al., 2020); and,
- Economic and administrative constraints, such as a lack of resources for monitoring and ecosystem health. This, in turn, is part of a broader issue of cross-sectoral integration (Pittock et al., 2013; Milhorance et al., 2021; Carvalho et al., 2019) across the “climate-energy-land-water nexus” (Vinca et al., 2021).

Therefore, while there is a general consensus on salinity criteria for major water uses such as human consumption, agriculture and industries, the situation is much less clear when the objective is protection of aquatic life. It appears that many countries have either not set such criteria or that information is not in readily-accessible forms. There are many reasons behind this lack of consensus, including different approaches used to set criteria and different definitions of “sufficient” protection. Meanwhile, however, the effects of salinisation on inland waters continue to be reported (Table 4.1), in some cases deserving to be considered as ecological disasters (Free et al., 2023). This emphasises the need for “safe operating limits” for salinity to be defined (or reassessed) in order to protect ecosystems from human-induced salinisation.

**Table 4.1.** Case studies of impacts of salinity on biota across ecosystems and communities

| Ecosystem | Community | Impacts | Range of salinity | Reference |
|---|---|---|---|---|
| Werra river, Germany | Benthic invertebrates | Severe degradation of communities, decrease of biodiversity, dominance by three halophilic neozoic species | EC 506–5,220 µS $cm^{-1}$<br>$Cl^-$ 39-1,500 mg $l^{-1}$<br>$SO_4^{2-}$ 61-343 mg $l^{-1}$ | Arle & Wagner, 2013<br>Braukman and Böhme, 2011 |
| River Meurthe, France | Benthic invertebrates | Decrease in taxonomic richness, change in taxonomic composition, increase in invasive taxa | Salinity 0.21-2.6 g $l^{-1}$<br>EC 277–3,422 µS $cm^{-1}$ | Piscart et al., 2005 |
| Ponds, Southern Poland | Benthic invertebrates | Decrease in taxa diversity and richness, colonization by halotolerant species, invasion of non-native taxa | EC 220–42,400 µS $cm^{-1}$<br>TDS 100–21,100 mg $l^{-1}$<br>$Cl^-$ 8–7,292 mg $l^{-1}$ | Sowa et al., 2020 |
| Oder river, Germany /Poland | Phytoplankton | Massive blooms of *Prymnesium parvum,* massive fish kills | EC 470–7,290 µS $cm^{-1}$ | Free et al., 2023 |
| Rio Grande, USA/ Mexico | Fish fauna | Functional and taxonomic homogenization of fish fauna | EC 1,160–3,440 µS $cm^{-1}$ | Miyazono et al., 2015 |
| Coastal lakes, New Zealand | Zooplankton | Decrease in taxonomic richness and abundance | Salinity 1.2-4.7 psu | Schallenberg et al., 2003 |
| Onondaga Lake, US | Zooplankton | Decrease in taxonomic richness and density, change in composition, decrease grazing pressure and resulting phytoplankton blooms | Salinity 1–3‰ | Siegfried et al., 1996 |
| Lake Toolibin, Australia | Vegetation | Decline in the wetland vegetation | TDS 300–20,000 mg $l^{-1}$ | Froend et al., 1997 |

| | | | | |
|---|---|---|---|---|
| Wipper river, Germany | Macrophytes, benthic invertebrates | Mass development of salt-tolerant macrophyte species *Stuckenia pectinata*, resulting in oxygen depletion during night-time<br>Degradation of benthic community | EC 766-5,439 μS $cm^{-1}$<br>$Cl^-$ 46-1,490 mg $l^{-1}$<br>$SO_4^{2-}$ 98-496 mg $l^{-1}$ | Feld et al., 2023 |
| Lippe river, Germany | Diatoms, benthic invertebrates | Distinct shift in community composition, dominance of invasive species | EC <645 μS $cm^{-1}$ to >3,134 μS $cm^{-1}$ | Schröder et al., 2015 |
| Hun-Tai River Basin, northeast China | Periphyton, benthic invertebrates, fish | Marked decline in functional diversity and community diversity, simplified trophic links | $SO_4^{2-}$ 10-200 mg $l^{-1}$ | Zhao et al., 2021 |
| Llobregat basin rivers, Spain | Benthic invertebrates, riparian vegetation | Deterioration of the riparian vegetation, extensive depletion of benthic fauna | EC 1,400-132,400 μS $cm^{-1}$ | Ladrera et al., 2017 |

EC = electric conductivity, TDS = total dissolved solids, $Cl^-$ = chloride, $SO_4^{2-}$ = sulphate concentration

## 4.3 Approaches to setting thresholds

### 4.3.1 Starting points

- Although salinisation is defined as an increase in total concentration of major ions in a water body, this chapter focusses on instances where this increase is due to processes associated with changes in evaporation and precipitation patterns, leading to elevated sodium and chloride concentrations, rather than on changes in rock weathering rates (see Gibbs, 1970) and proxy processes such as addition of lime to fields to boost agricultural productivity (where over 20% can be lost to surface water: Cuttle and James, 1995). The latter is, strictly, "alkalinisation" (Kaushal et al., 2013) and though also a problem, with ecological consequences in some regions (Arts, 2002; Free et al., 2009), is out with our scope.

  As a result, there is less justification for type-specific thresholds for salinity **within** a region than for other supporting elements, as our working definition only encompasses situations outside the range of geological weathering. The nature of salinisation problems will, however, vary considerably **between** regions in Europe, and this means that a variety of approaches will be needed and Member States will need the capacity (and data) to develop and validate criteria according to their own needs.

- All recent monitoring data should be assumed to be influenced by climate change, unless there is strong evidence to the contrary. Ideally, thresholds should be derived from historical data or validated against an historical baseline

- It is quite likely that many water bodies will have salinity at levels not yet approaching thresholds but which do show systematic trends away from the baseline state. Detecting these trends, and ensuring that catchment management does not exacerbate them (and, where possible, mitigates against them), will be a major role of monitoring for this supporting element.

- Denmark has made a decision not to set salinity criteria. In this low-lying country, many rivers are subject to tidal incursions and high salinity is regarded as a “natural” phenomenon. These incursions are very variable in both frequency and intensity of their impacts. Occasional saline incursions can lead to losses of common organisms which can influence metrics and, as a result, classifications. Where there is a geographical and data-based presumption that failure to fulfil environmental objectives is due to a saltwater impact, the reason for failure is stated as “natural conditions.

### 4.3.2 Parameters

Salinity is the sum of the concentrations of all major ions. In general, different anthropogenic sources of salt pollution are associated with different sets of ions with different environmental and toxicological consequences (Griffith, 2014). Therefore, ideally, all dominant ions should be measured and ion-specific limits devised (Cañedo-Argüelles 2016a; Schuler et al., 2018). However, measuring all ions on a routine basis across all sites in a monitoring network is often impractical and salinity tends to be expressed via a number of proxy measurements, of which conductivity and chloride concentration are used most widely. Each has advantages and disadvantages. Currently, only information on approaches to evaluation of salinity in inland waters are available.

- **Total Dissolved Solids (TDS)** is the most direct measure of salinisation. Ideally, TDS is measured by gravimetry; however, it is sufficiently closely related to conductivity that it is often inferred from a conversion factor (TDS in mg $L^{-1}$ can be estimated as two thirds of conductivity in µS $cm^{-1}$, Flanagan, 1990). This, however, raises questions about why a straightforward conductivity measurement cannot be used. TDS is reported by four G20 countries but not (as far as we know) by any EU countries.

- **Conductivity** is an easy measurement to make, with robust and relatively cheap instruments and is, as a result, widely used. It is the parameter recommended for SDG Indicator 6.3.2 and it is reported by seven EU countries. The principal issue with conductivity is that measurements also reflect influences of local geology, with inland waters in calcareous regions often having natural conductivity values close to some of the proposed thresholds. However, so long as users are informed by the local geochemical context, then conductivity is an excellent means of acquiring data at a relatively low cost.

- In many respects, **chloride concentration** is a better measure of salinisation than conductivity because it is a direct measure of one of the ions responsible for physiological effects and is less confounded by interactions with local geology. It is, however, a more time consuming (and expensive) measurement which may make it less attractive than conductivity. It is possible, again, to estimate chloride concentration from conductivity if you have either a general (Herbert et al., 2015) or a locally specific (Schulz & Cañedo-Argüelles, 2018) understanding of their relationship (see Box 2). Chloride concentration is also unsuitable as a salinity measure where other ions predominate (e.g. Soucek & Kennedy 2005, Buchwalter et al. 2019) and there are situations where these other ions should be measured (Cañedo-Argüelles et al., 2016a; Schuler et al., 2019).**Practical Salinity Units** (PSU) are used by one EU country. These are derived from the Practical Salinity Scale developed by oceanographers and based on the ratio of the conductivity of a sample to that of a standard potassium chloride solution (Fofonoff, 1985). PSUs are, in other words, also derived from conductivity so have limited benefit for inland waters over a straight conductivity measurement unless there is a need to relate values to conditions in seas and estuaries.

This overview recognises a range of plausible options. Conductivity, though not perfect, is adequate if the relationship with toxic ions in the system being assessed is well understood. It is also a robust measurement that is relatively cheap and straightforward to apply. As analysis of trends is likely to be important, especially where climate change is implicated, there is little reason to change a method if historical records extend back into the past. Chloride may be a better predictor of ecological effects (see below) but it is unlikely that benefits outweigh advantages of moving from conductivity.

### 4.3.3 Metrics

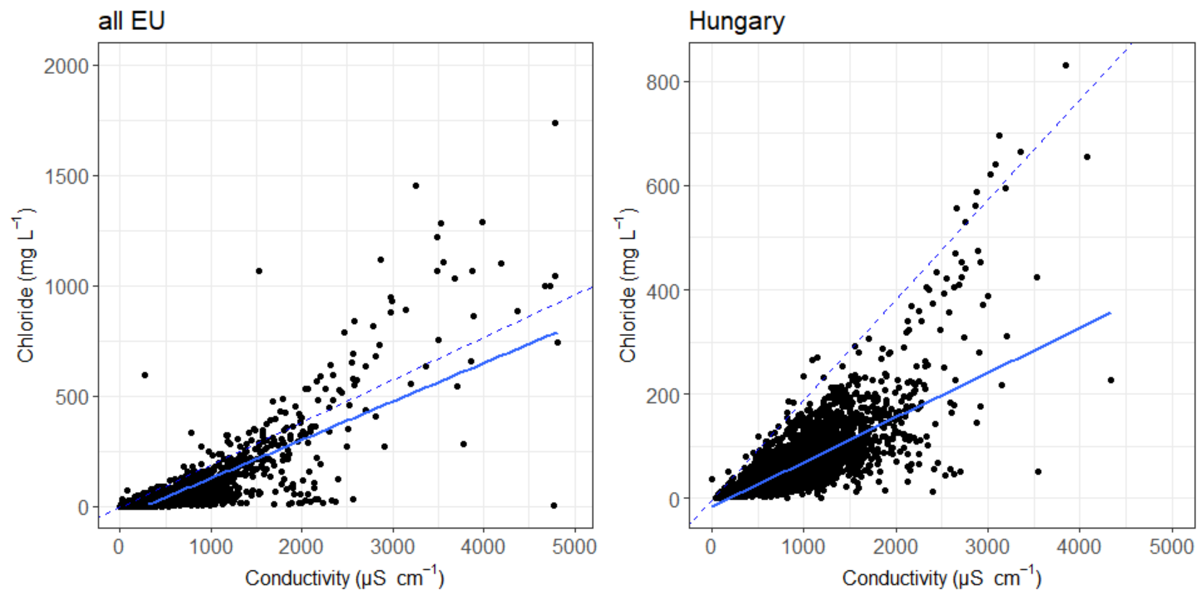
As environmental measurements are inherently variable, basing decisions about water body management on a single measurement is discouraged. Broadly speaking, chronic environmental stresses and long-term change are best evaluated using measures of central tendency (mean, median) whilst extreme values (maximum or upper percentiles) may be more appropriate for stressors with short-term acute effects. In practice, salinity effects can be chronic or acute, so both central tendencies and upper percentiles have roles, depending on circumstances. The frequency of sampling determines the uncertainty in the metric and this will be greater when dealing with upper percentiles than with central tendencies. An annual mean based on two or three samples – as is often only possible in countries with limited financial resources – may not allow assessment with adequate certainty. This may, however, still provide valuable information on trends if continued for many years especially if done with sufficient regularity to control for seasonal variation. Conversely, it would have little ability to detect a short extreme change, which could exert substantial environmental damage. Ideally, means based on monthly samples should be used to assess the risk of chronic exposure. This also increases the chance of capturing short-term peaks, such as those that may occur after winter salt application to highways, and thus provides an opportunity to use upper percentiles for the assessment of acute toxicity due to short-term exposure.

A comparison of the approaches used in different countries gives a very heterogeneous picture, with much information only available in national documents and reports or cited in scientific papers without specifying the metric to be calculated or the required monitoring frequency (CCME, 2011; Vosylienė et al., 2006; Arle & Wagner, 2012). Some countries use both mean and maximum values (Austria, Czech Republic), others have included only the maximum value (Lithuania) or the 90th percentile (Slovakia) in their regulations. Canada and Austria differentiate between chronic and acute exposure. The ratio of for acute and chronic exposure derived from bioassays varies between roughly two and eight (Elphick et al., 2011; Hassell et al., 2006; Paradise, 2009).

## Box 4.2. Are conductivity and chloride criteria interchangeable?

Conductivity is widely collected during field monitoring campaigns but much of the evidence suggests that chloride is a better measure of toxic effects. How closely are these two variables related? The figure below shows this relationship for two datasets: one using data from the entire EU (from WISE) and one using just data from Hungary.

In both cases, linear regressions are significant and conductivity explains a considerable amount of the variation in chloride ($r^2$ = 0.55 and 0.57 respectively); however, there is also considerable scatter that increases with conductivity. The effect is more pronounced for Hungary (note how the regression based on the 95$^{th}$ percentile diverges from the linear regression as conductivity increases). Better relationships for Hungary can be obtained by limiting analyses to particular types ($r^2$ = 0.62 and 0.71 for small-medium size low altitude and mid altitude streams respectively).




**Relationship between conductivity and chloride concentration for a dataset encompassing the entire EU (left; mean values; n = 4536 records) and one showing just data from Hungary (right; spot measurements; n = 9105 records). Solid lines show the line of best fit based on linear regression and dashed lines show the 95$^{th}$ percentile of the data.**

Using the linear regression equations, the chronic and acute thresholds based on SSDs from which Austrian thresholds are derived, expressed as conductivity (µS cm$^{-1}$) are as follows:

| Threshold | EU dataset | Hungarian dataset |
|---|---|---|
| Chronic | 1076 | 1521 |
| Acute | 3482 | 4668 |

Although these demonstrate that it is possible to change between conductivity and chloride concentration, we do not intend that these values are used as thresholds. Rather, we recommend that thresholds can be obtained from relationships based on local data. Nonetheless, these plots do suggest that the widely-used threshold of 1000 µS cm$^{-1}$ is broadly in line with ecotoxicological evidence for chronic effects although we cannot rule out the possibility that some inland water types will show symptoms at lower values.

Whilst Horrigan et al. (2005, 2007) showed stronger correlations between laboratory and field tolerance results for the mean salinity rather than single, measurements Kefford et al. (2007) stressed that high temporal variability through multiple pulses may be especially critical for benthic invertebrates. Therefore, in the future, continuous monitoring using online probes may provide better protection for aquatic life than single measurements. This will create opportunities for defining new metrics for salinity thresholds, which better take account of the relationship between exposure time and concentration in some regions. Another largely unresolved question concerns how best to capture temporal dynamics of salinity and better understand their potential effect (e.g. for aquatic insects, a salinity peak which occurs when adults are emerging may be less toxic than one which occurs when larvae are developing (Moyano Salcedo et al., 2022).

### 4.3.4 Thresholds: Empirical approaches, expert judgement or ecotoxicology?

Thresholds link the parameter and metric to the policy objectives and should be set at levels where significant changes in the ecological response or restrictions of human use can be observed. Physico-chemical thresholds to protect ecosystems can be derived in two ways (Fig. 4.3): either the threshold is set as the upper limit that supports the acceptable condition (left hand curve in Fig 4.3) or it is the lower limit of the unacceptable condition (right hand curve in Fig. 4.3). The former is the approach widely adopted for the implementation of the EU WFD where the objective is to protect “good ecological status” and involves evaluating the response of the “natural” biota to the pressure. (see Poikane et al., 2019; Kelly et al., 2022a for nutrients, but this can also be applied to other stressors). The second approach is the classic ecotoxicological approach where thresholds are derived under controlled experimental conditions, either based on lowest effect concentrations plus a safety factor or based on species sensitivity distributions (SSD) by establishing a percentage of sensitive species potentially impacted (Elphick et al., 2011). In theory, both should yield similar results and analysis of datasets derived from field surveys should validate results of ecotoxicological trials and *vice versa.* However, the two approaches reflect the outcome of different processes: interactions amongst species across varied environmental conditions in the former and direct physiological effects on individual species in the latter. Impaired fitness of some species due to the stressor should result in detectable changes in communities, linking the two approaches together. In both cases, allowances need to be made to account for statistical uncertainty, interactions with other stressors and also for how the criteria are used in decision-making (Kelly et al., 2022a).

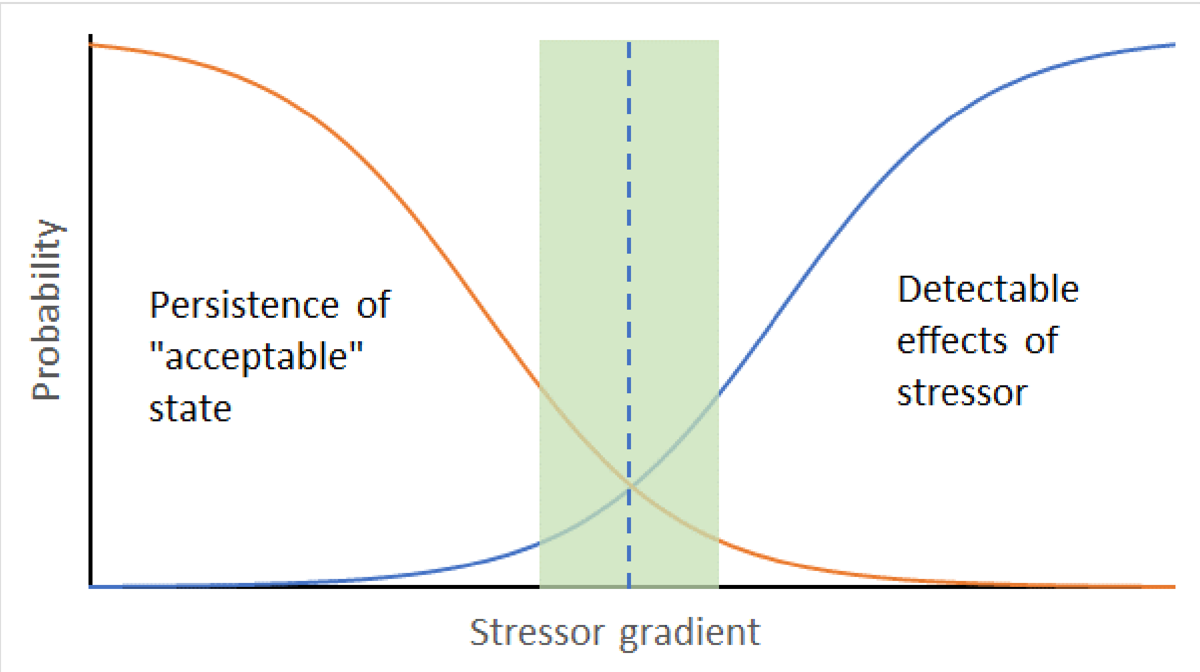



**Figure. 4.3.** Theoretical representation of approaches to derivation of criteria (represented by a dashed line) to protect ecosystems against stressors: defined in relation to the “acceptable” state (left hand curve); 2. defined to minimise the impact of the stressor on organisms (right hand curve). The dotted line represents a hypothetical “true” threshold (the position of which will depend on the precise wording of legislation; the green box is a reminder that any derived boundary will also have a range of uncertainty associated with it.

#### *4.3.4.1 Use of field data*

A well-established protocol exists for deriving ecological metrics from community changes along environmental gradients using spatio-temporal substitution (e.g. characteristics of a dataset derived from spatially-separated samples are used to indicate likely changes at a single site as a pressure changes over time: Pickett, 1989). The relationship between these metrics and the stressor of interest can then be used to derive thresholds (e.g. Poikane et al., 2019, Kelly et al., 2022b).

In practice, salinity is rarely the only stressor impacting water bodies, and interactions with other stressors can confound predictions of threshold concentrations (Phillips et al., 2019). Several metrics that purport to measure "general degradation" show correlations with conductivity / chloride as well as with other water quality variables (e.g. Dell'Uomo, 1998; Prygiel & Coste, 1998; Schulz & Cañedo-Argüelles, 2019). When salinity thresholds for rivers in Europe were compared with actual monitoring data, biological impacts appeared to occur at lower concentrations than these thresholds (Fig. 4.4). However, most of the metrics used in these assessments were evaluating the overall condition of the biota and the most likely explanation is that salinity parameters were very rarely determining the condition of the flora or fauna in these rivers. These interactions are a major impediment when using biological data to set thresholds.

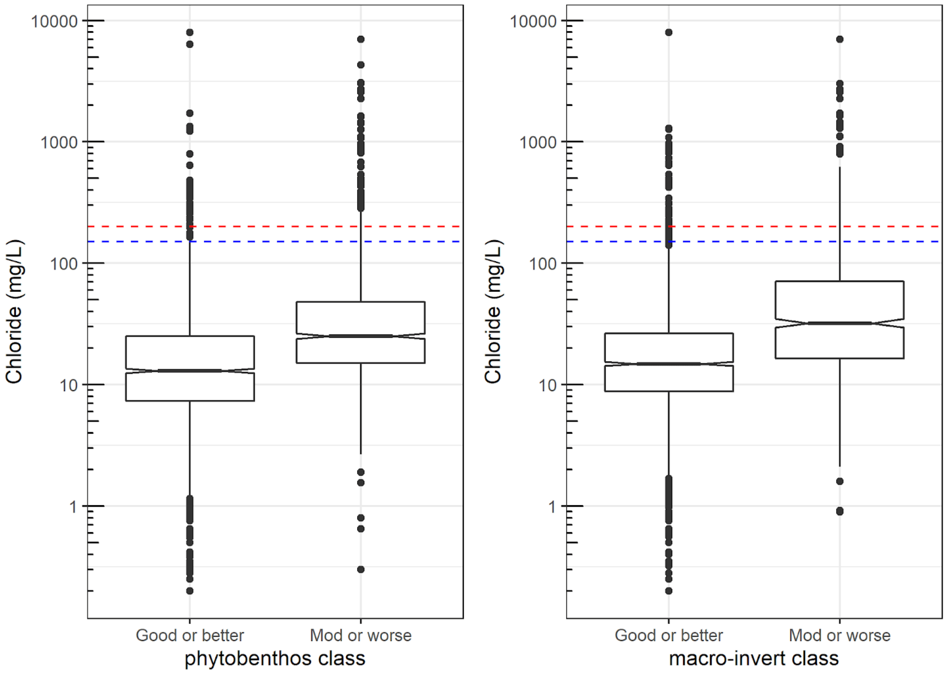



**Figure. 4.4.** River chloride standards used in the EU (dotted lines) overlain on box plots showing the range of chloride concentrations of sites classified by phytobenthos and macro-invertebrates. (90th percentile=red, median=blue). Mod = Moderate. From Kelly et al. (2022).

Relatively few specific metrics have been developed to assess biotic responses to salinity in inland waters, meaning that it is difficult to develop strong stressor-response relationships. Examples where salinity metrics have been developed include Ziemann (1999)'s Halobic index for diatoms and Habach's Haloindex for benthic invertebrates (Haybach). Interpretation is, however, confounded in the presence of other stressors (Fig. 4.5) creating problems when setting boundaries using field data. Challenges when assessing the biological effects of salinity are discussed in more detail by Ziemann and Schulz (2011).

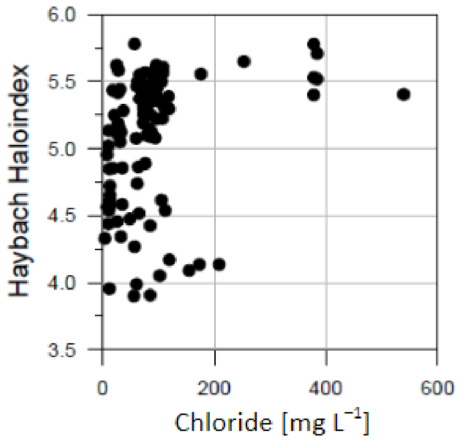



**Figure. 4.5.** Relationship between Heybach's Haloindex and chloride concentration (left) and the Saprobic index. There is a weak correlation with chloride, but a strong correlation with the saprobic index, meaning that it would be unwise to interpret the Haloindex solely in terms of salinity. G. Wolfram, unpublished data.

Most of the proposed indices focus on species turnover and diversity. Diversity, by itself, is not an unambiguous indicator of saline effects but it may point to elevated salinity, if associated with elevated numbers of brackish taxa (Fig. 4.6: see Kelly et al., 2023). Diversity, however, needs to be interpreted with care as our understanding of changes in communities in response to slight increases in salinity is limited. Even small changes can potentially exert a selective pressures, affecting diversity and community network stability (Mo et al., 2021). There is clearly a need to determine causal relationships rather than just rely upon correlations between salinity parameters and biological metrics.

Another approach is to focus on salinity preferences for key species. Hartman et al. (2021), for example, used a threshold of 1500 µS $cm^{-1}$ to identify and map sites at risk from *Prymnesium parvum* blooms in the Ohio River basin, and this threshold was also used, along with nutrients and non-marine sulphates, to screen European rivers (Free et al., 2023).

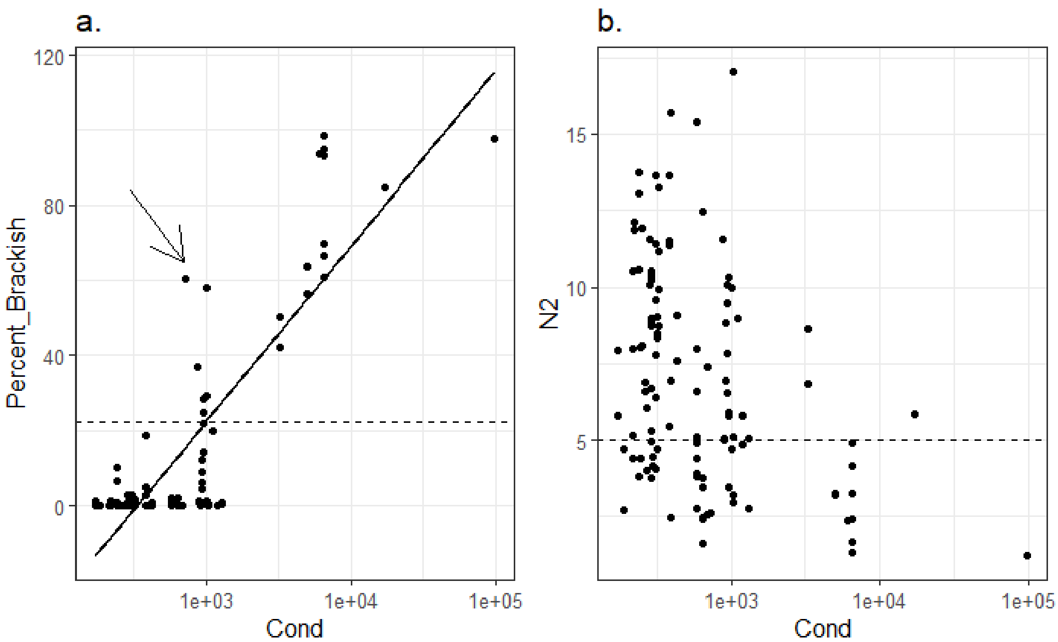



**Figure. 4.6.** Relationship between a) percent brackish diatoms and b) Hill's N2 diversity and conductivity in Greek lakes. In the case of diversity, the 10th percentile of lakes with an a priori designation of high or good status was taken as a threshold, but this is not a reliable indicator of saline effects unless associated with an elevated proportion of diatoms belonging to brackish taxa. The strong nutrient gradient across these lakes will confound interpretation of a dedicated salinity metric. From Kelly et al. (2023).

#### 4.3.4.2 *Use of ecotoxicology*

The challenges involved in assessing the scale of salinity impacts from field data mean that field data are rarely used to set thresholds. The alternative is to base thresholds on ecotoxicological data. Single species tests have a role, particularly when protecting economically-important or keystone species but the trend, in recent years, has been towards using information from many species. In particular, broadly applicable thresholds can be set using species sensitivity distributions (SSDs: Fig. 4.7). These integrate the results of ecotoxicological tests on several organisms, resulting in a more broadly applicable threshold than is obtained from single-species tests (Elphick et al., 2011). It is also important that SSDs do not focus only on mortality data but also assess sublethal effects using biomarkers, growth rates, predation efficiency, behavioural changes (Hassel et al., 2006; Hoover et al., 2013; Leite et al., 2022) otherwise the SSD approach may underestimate the whole range of effects of salinisation on the ecosystem (Cañedo-Argüelles et al., 2016b). When the SSD approach was applied in Austria, chronic thresholds of 100–120 mg $L^{-1}$ Cl and acute thresholds of 590–670 mg $L^{-1}$ were obtained, leading to proposed good/moderate class boundaries of 150 mg $L^{-1}$ (based on annual average chloride concentration) and 600 mg $L^{-1}$ for short-term (3 day) exposure (Wolfram et al., 2014). The chronic threshold roughly aligns with current national thresholds for chloride (Fig. 4.4).

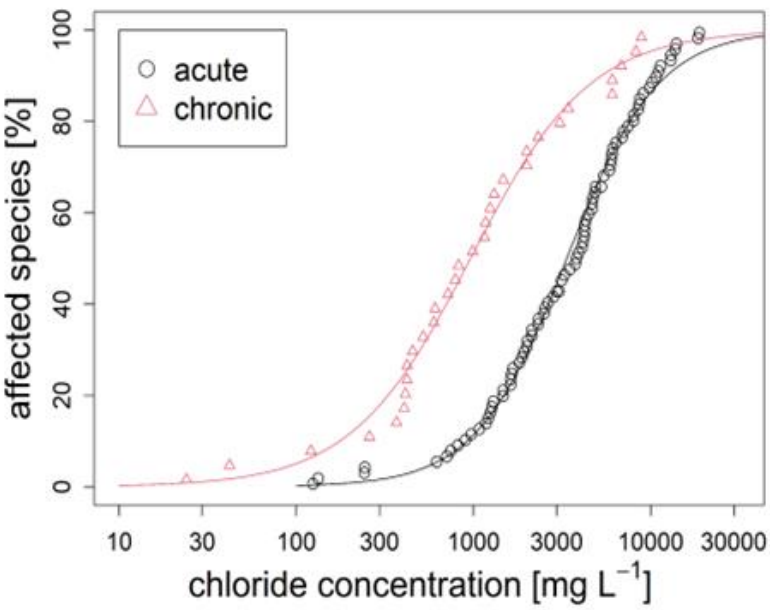



**Figure. 4.7.** A species sensitivity distribution plot summarising results of single-species tests for acute (n=83) and chronic (n=32) toxicity, based on data in Wolfram et al. (2014). The Austrian Water Quality Guideline was set at concentrations corresponding to the intersection with the 5$^{th}$ percentile of test data.

While SSD have proven valuable for deriving thresholds in many cases (Posthuma et al., 2019), they have drawbacks. The determination of a potentially affected fraction of sensitive species (typically 5%) to predict a critical concentration remains subjective but has a large impact on the threshold to be determined due to the low slope of the logistic curve in the lower part of the range. Adding confidence limits can help to evaluate the uncertainty of the shape of the statistical distribution especially at the tails of sensitivity. Even so, a particularly high weight is placed on a few species in the lowest concentration range, while the distribution of tolerant species in the upper range has little influence. A further problem is that the use of laboratory organisms will not take account of differences between populations that might arise from adaptation and/or rapid evolution (Sala et al., 2016; Jeremias et al., 2018).

Belanger et al. (2016) emphasise the importance of the taxonomic variety and the need to focus on sensitive groups and taxa whilst Wheeler et al. (2002) suggest the use of a modest “safety factor” (up to 10). Field studies (e.g. Mo et al., 2021) also emphasise how selective pressures can act, even at low salinity levels, to alter community structure. The danger of SSDs is that they provide an illusory “safety in numbers”, whereas the bulk of the tests are performed on species known to grow well in laboratory conditions rather than assembled to understand the sensitivity of particular ecosystems. Dickey et al. (2021), coming at the problem from a completely different direction (concerned with “freshening” of saline environments rather than salinisation of freshwater habitats), focussed on one keystone predator whose loss would have effects that cascaded through several trophic levels. In any case, the relevance of a threshold derived from SSD with a limited dataset for the assessment of an entire ecosystem needs to be critically evaluated, taking account of water body types, geological background, and geographical differences of the taxa used.

#### *4.3.4.3 Range of ecology-related thresholds*

Information on how countries set salinity criteria is difficult to find, but “expert judgement” is widely used for other stressors (Poikane et al., 2019a) and we suspect that this is also the case for salinity. Where information is available, laboratory tests and field data have been used, with the former generally preferred (Table 4.2). This is despite evidence that laboratory tests show greater tolerance to toxic pollutants compared to the results acquired from field data or mesocosm studies (Clements et al., 2013; Arnott et al., 2020; Hintz et al., 2022). However, field studies also have disadvantages, associated with confounding variables and multiple stressors (see above).

Neither field-based methods nor ecotoxicology are perfect and one result is a wide range of thresholds (Table 4.3). Field-based approaches embrace the natural complexity of ecosystems with a result that thresholds obtained are compromised by interactions with other stressors. On the other hand, laboratory-based approaches sidestep this complexity with the result that they potentially offer a spurious precision that may be attractive to regulators. Phillips et al. (2019), writing about nutrient thresholds, emphasise the need to validate thresholds produced by independent means, irrespective of the approach adopted whilst Clements and Kotalik (2016) demonstrate the potential for mesocosms to provide ecologically-realistic environments within which confounding factors can be controlled.

**Table 4.2.** Approaches for setting salinity criteria intended to protect aquatic life. Note that different regions/states/provinces may have different approaches within a single country.

| Country | Thresholds | Method | Reference |
|---|---|---|---|
| Australia | Low-risk trigger values<br>EC 30-5000 µS $cm^{-1}$* | Field data: 80th percentile of the reference systems distribution (unmodified or slightly modified ecosystems) | ANZECC, 2000 |
| Canada | Long-term threshold<br>$Cl^-$ 120 mg $l^{-1}$ | Aquatic toxicity tests:<br>SSD method using 28 species (invertebrates, fish, aquatic plants and algae) | CCME, 2011 |
| | Short-term threshold<br>$Cl^-$ 640 mg $l^{-1}$ | Aquatic toxicity tests:<br>SSD method using 51 species invertebrates, fish) | |
| China | Long term threshold<br>$Cl^-$ 200 mg $l^{-1}$ | Aquatic toxicity tests: SSD method using 20 species (invertebrates, fish, algae) | Hong et al., 2023 |
| Poland | Good-moderate class threshold<br>EC 300-850 µS $cm^{-1}$* | Field data: relating EC to good status thresholds using biological quality elements (macrophytes, phytobenthos and macroinvertebrates) | Kolada et al., 2018 |
| South Africa | TDS should not be >15% comparing with unimpacted conditions | Comparison of actual concentration to background levels | DWAF, 1996 |
| United States | Criteria continuous concentration (CCC)<br>230 mg $Cl^-$ $l^{-1}$ | Aquatic toxicity tests: acute toxicity tests using 12 genera (invertebrates, fish) and toxicity percentage ranking method | US EPA, 1988 |
| | Criteria maximum concentration (CMC)<br>860 mg $Cl^-$ $l^{-1}$ | Acute value divided by the acute-to-chronic ratio | |

* for different water body types

**Table 4.3.** Salinity thresholds defined using response of biological communities to changes in salinity (in an increasing order).

| Region, waters | Philosophy | Threshold | Reference |
|---|---|---|---|
| Rivers in Germany | Ecological change points for benthic invertebrate taxa | $Cl^-$ 25 mg $l^{-1}$ | Sundermann et al., 2015 |
| Rivers in Ontario, Canada | Critical change point for diatom communities | EC 250-400 µS $cm^{-1}$<br>$Cl^-$ 35 mg $l^{-1}$ | Porter-Goff et al., 2013 |
| Rivers of Central Appalachia, USA | Greatest cumulative benthic invertebrate community diversity loss | EC 283 µS $cm^{-1}$<br>$SO_4^{2-}$ 50 mg $l^{-1}$ | Bernhardt et al., 2012 |
| Germany, different river types | Thresholds of good to the moderate status according to the EU WFD | EC 400-1000 µS $cm^{-1}$<br>$Cl^-$ 40-90 mg $l^{-1}$ | Halle and Müller, 2013 |
| Streams in Queensland, Australia | Most dramatic shift in benthic invertebrate composition | EC 800-1000 µS $cm^{-1}$ | Horrigan et al., 2005 |
| River Lippe, Germany | Major changes in community composition of benthic invertebrates and diatoms | EC 900-1000 µS $cm^{-1}$ | Schröder et al., 2015 |
| Australian streams and wetlands | Direct adverse biological effects | EC 1500 µS $cm^{-1}$<br>TDS 1000 mg $l^{-1}$ | Hart et al., 1991 |
| Streams in south-east Australia | Decline in invertebrate species richness | EC 1500 µS $cm^{-1}$ | Kefford et al., 2011 |
| River Wipper, NE Germany | Limnetic (α-oligohalobic) diatom assemblages | $Cl^-$ <400 mg $l^{-1}$ | Ziemann et al., 2001 |
| Wetlands in Western Australia | Decline in non-halophilic species richness<br>Decline in total species richness of invertebrate fauna | TDS 2600 mg $l^{-1}$<br>TDS 4100 mg $l^{-1}$ | Pinder et al., 2005 |

Interpretation of current criteria is complicated by the range of parameters, metrics and thresholds that are in use. Conductivity is the most widely used parameter, and EU and G20 countries typically have conductivity thresholds around 1000 µS $cm^{-1}$ (Fig. 4.8a) which roughly aligns with the transition from the dominant source of minerals as rock weathering to evaporation (Gibbs, 1970) and also approximates to the equivalent for chloride toxicity thresholds (but see caveats in Box 1). In order to support countries reporting on SDG indicator 6.3.2 that were unable to define a target threshold value for salinity, UNEP suggested an optional target value of 500 µS $cm^{-1}$ in 2020 (UNEP, 2020). This target value aligned with work of Carr and Rickwood (2008) and Srebotnjak et al. (2012) that included a review of threshold values used globally. Of 58 countries that provided information on the salinity threshold values they used in their indicator calculation in 2020, ten had adopted this 500 µS $cm^{-1}$ value (UNEP, 2021). Based on toxic levels inferred from chloride-conductivity relationships (Box 2), this threshold is probably too stringent for many European countries, and is in the range where conductivity will be influenced by weathering as well as by evaporation.

Generally, thresholds in the eight EU countries which use chloride were lower (mostly < 200 mg $L^{-1}$) than those in G20 countries (Fig. 4.8b). Values for chronic toxicity derived from SSDs tend to be at the lower end of these relationships (120 mg $L^{-1}$ for Canada; 150 mg $L^{-1}$ for Austria).

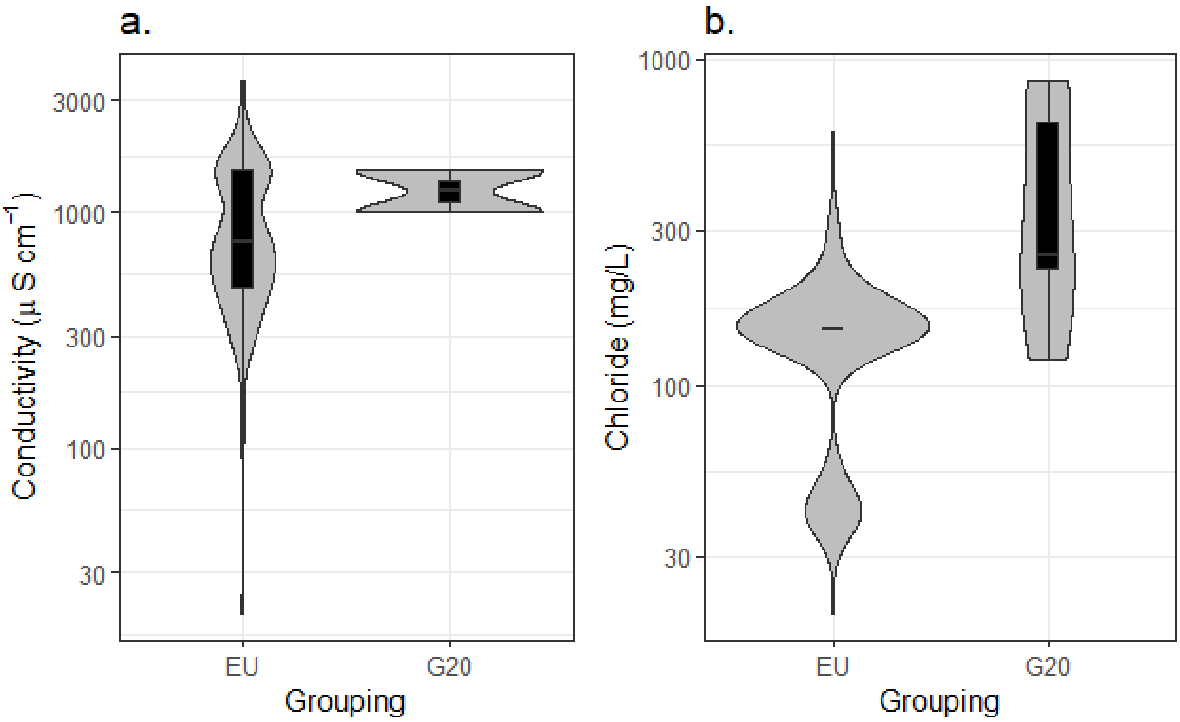



**Figure. 4.8.** Comparison of a) conductivity and b) chloride standards in EU (rivers) and other G20 countries. EU data from Kelly et al. (2022b); G20 data collated for this study. Note log scale. Low conductivity standards in the EU protect against "alkalinisation" in soft water streams (mostly in Spain). They have been retained due to difficulties in determining an unambiguous difference between alkalinisation and salinisation.

Within the EU, where all Member States are subject to the same legislation, there is considerable latitude in how this is interpreted (Table 4.4), with an interquartile range of reported thresholds of 450-1000 µS $cm^{-1}$ for conductivity and 50-200 mg $l^{-1}$ for chloride concentration for rivers. Part of this difference may represent genuine differences in the sensitivity of waterbodies across Europe and part to differences in how criteria are used to manage waterbodies. However, a lesson from comparing nutrient criteria within the EU was that there are also differences in the robustness of the science behind criteria setting (Poikane et al., 2019a) and this is also likely to play a role when setting salinity criteria.

**Table 4.4.** Summary table of conductivity (µS $cm^{-1}$) and chloride thresholds (mg $l^{-1}$) for rivers and lakes in Europe used to implement the Water Framework Directive. AA – annual average, SA – seasonal average, MAC - Maximum allowable concentration, P90 – 90th percentile.

| | | Rivers | | Lakes | |
|---|---|---|---|---|---|
| | Metrics | Conductivity µS $cm^{-1}$ | Chloride mg $L^{-1}$ | Conductivity µS $cm^{-1}$ | Chloride mg $L^{-1}$ |
| | | Single threshold or Range of type-specific thresholds (median value) | | | |
| Austria | AA | - | 150 | 1010[1] | 60[1] and 150 |
| | MAC | - | 600 | - | - |
| Belgium | AA | 800 | 150 | - | - |
| | P90 | 600-1000 | 120-200 | - | - |
| Bulgaria | AA | 750-900 (750) | - | - | - |
| Cyprus | AA | 750 | - | - | - |
| Germany | AA | - | 200 | - | - |
| Hungary | AA | 600 - 1200 (1000) | 20-60 (50) | 20 - 1500 (70) | - |
| Luxembourg | AA | - | 200 | - | - |
| Netherlands | SA | - | 40-300 (150) | - | 40 – 200 (200) |
| Poland | AA | 300 - 850 (480) | - | 100 - 600 (600) | - |
| Romania | P90 | 1500 | - | - | - |
| Spain | AA | 300 - 350 (300)<br>20-3500[2] | 50-500[2] | 350<br>20-700[2] | - |
| | MAC | 300 - 700 (500)<br>100 - 2200[2] | - | 600 - 3600[3] (600) | - |

[1] Threshold for Lake Neusiedl - a unique soda lake, represents a minimum value for conductivity and chlorides

[2] Waterbody specific thresholds set for several types

[3] Highest type-specific values for two karstic calcareous lake types

In coastal environments, faunal communities and seagrasses can be highly vulnerable to localised acute effects of increased salinity, with a wide range of tolerances observed for different organisms (see review by Pistocchi et al., 2020 and references therein). Salinity effects on life-history behaviours are also of concern, with few studies available. There is little guidance on threshold values for brine discharge regulation. Barrio et al. (2021) suggested compliance to a threshold of 38.5 psu in Spain whilst the California State Ocean Plan allows a daily maximum increment of "*2 parts per thousand (ppt) above natural background salinity measured no further than 100 meters horizontally from each discharge point*" to be the limit if local ecosystems are to be protected. Lizaso et al. (2007), for example, used ecotoxicology to set targets to protect seagrass beds from discharges from desalinisation plants. No part of the meadow should exceed 38.5 psu for more than 25% of observations per year; nor exceed 40 psu for more than 5% of observations. This is a good example of a situation where a tailored approach to protect a keystone species of angiosperms is beneficial rather than the use of a broadly-prescribed national threshold to protect the BQE.

## 4.4 Setting salinity thresholds in an already-warming world

It is clear that climate change has already led to observable changes in both inland and TRAC waters and that this is driving much of the research in effects of climate and, by extension, recognition of the need to define protective criteria. In practical terms, this may limit the number of available sites in some regions from which the "acceptable" state may be defined (i.e. the left hand distribution in Fig. 4.3). Without long-term records, however, this may not even be apparent. This means that decisions based on contemporary data – chemical and biological – will be influenced by "shifting baseline syndrome" (Soga & Gaston, 2018; Jones et al., 2020). Palaeoecological investigations have the potential to reveal the extent of changes before contemporary monitoring started but, from a policy perspective, setting criteria that are unachievable (bearing in mind that global objectives are to slow or halt warming, rather than reverse it) is of limited use.

An alternative view is that salinisation is often one ingredient of a cocktail of stressors which interact in different ways. Therefore, management of a stressor such as phosphorus that could, potentially, lead to improved ecology and enhanced ecosystem services, needs to be informed by the scale of effect of other stressors which may be less amenable to management. This was highlighted – albeit for temperature and precipitation rather than salinisation – by Spears et al. (2022). The policy challenge for long-standing legislation such as the EU WFD is that ambition was determined in an era before warming was recognised to be as significant as it is now. Accepting the inevitability of change due to interactions with climate effectively requires an additional effort in terms of measures to reduce salinity pressures (e.g. reduce water abstraction). Otherwise, a lowering of the ambition is needed (e.g. less stringent objectives wherever WFD-compliant justifications can be found). Where ambition is set purely in terms of metrics based on changes in species turnover, there may be little prospect of persuading stakeholders of the benefits. However, where there are direct links to ecosystem services (e.g. frequency of cyanobacterial blooms, quality of fisheries), then knowledge of interactions may lead to more protective thresholds for phosphorus being set in order to offset the impact of the second stressor. Failure to quantify stressors can also lead to unexpected interactions ("ecological surprises": King, 1995; Filbee-Dexter et al., 2017; Birk, 2019): simply having a science-based criterion does, at least, alert managers to the potential for a stressor such as salinity to be in a range where such interactions are possible.

However, others have pointed out that ecological tipping points are difficult to detect from empirical data (Hillebrand et al., 2020; Carrier-Belleau et al., 2022) calling into question the use of criteria for stressors such as salinity. Hillebrand et al. (2020)'s conclusion was that "safe operating spaces" are unlikely to be quantifiable; however, their work generated considerable discussion, with Dudney and Suding (2020) arguing that they had failed to take account of multiple stressor interactions and that a press-pulse framework (Harris et al., 2018) may better explain dynamics. Depending on circumstances, salinity may be both a "press" (i.e., chronic effect on shallow lake communities) and a "pulse" (i.e. short-lived but extreme events associated with road salt). Seen through this lens, salinity criteria - so long as they are heavily caveated - certainly do have a role to play.

A final perspective is that too much emphasis on thresholds may miss the point by focussing too much on the proximity of site-specific data to a value derived from a general understanding of the problem of salinisation. Any monitoring program that detects a trend towards a value of concern is fulfilling a valuable role by providing early warning of likely effects. The principle of "no deterioration" is integral to the WFD, for example, and applies irrespective of whether or not a threshold is crossed. This, in effect, translates the medical ethic of *Primum non nocere* ("first, do no harm") into a salinity criterion that can be applied relatively easily to any place where sufficient monitoring data to derive summary statistics already exists.

## 4.5 Conclusions

- Salinisation is a growing problem, and water managers need clear guidance on both trends and thresholds;
- More work is needed to understand relationships between salinity and BQEs in inland waters, in order to provide a foundation for the derivation of robust boundaries;
- A better understanding of the effects of increased salinity on transitional and coastal ecosystems is also needed;
- Conductivity is a valuable proxy measurement for salinisation but it is important to recognise that ion composition matters and an understanding of local geochemistry is important.
- Conductivity is an adequate proxy measurement for salinity in inland waters so long as the relationship with local geology is well understood. Chloride is a better measure of toxic effects. The two values are generally closely related although there will be regional differences. Likely target values are given in Table 4.1.
- Thresholds need to be region specific and account for the baseline conditions that largely depend on the catchment geology and climate (Le et al., 2019; 2021)
- A thorough characterization of salinisation impacts within a country is necessary before setting criteria. Different parameters, metrics and thresholds will apply depending on water category and the nature of the pressure.
- Sampling frequencies need to be tuned to seasonal patterns in the stressor; more intensive sampling at particular periods of the year may be appropriate (continuous monitoring, where appropriate).
- Analysis of trends in salinity parameters is important, even if values fall well below thresholds. Ideally, this requires a predictive capacity too so that management actions within a catchment that may exacerbate salinisation can be avoided.
- Ecosystem health may require different thresholds to other uses. In the absence of an ecology-based criterion, the drinking water quality criterion offers a useful alternative; but its relevance to a particular region needs to be evaluated before this is applied;
- Thresholds that have been derived and/or tested using locally-generated data are recommended. Where laboratory data are used, this should use local biota, and reflect those organism groups which are likely to be most sensitive;
- Community-level responses are often difficult to interpret due to interactions with other stressors;
- There is a role for broadly-based (national or regional) criteria but also a case for moving towards waterbody-specific criteria;
- Salinity criteria need to be harmonised, especially for transboundary water bodies; and;
- The principle of “no deterioration” offers a further option for a salinity criterion that can be applied anywhere where sufficient monitoring data to derive robust summary statistics is available.

**Table 4.5.** Summary guidance for setting ecological thresholds for salinity in inland waters. See section 1.1 for explanation of use recommendations (× and ✓).

| Level | Threshold | Source | Use? |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | | None | ✗ |
| 2. Threshold based on published literature: general prescription (chloride) | 150 mg $L^{-1}$ Cl (chronic)<br>600 mg $L^{-1}$ Cl (acute) | Wolfram et al. (2014) | ✓ |
| 3. Threshold based on published literature: general prescription (conductivity) | 1000 µS $cm^{-1}$ (chronic)<br>3500 µS $cm^{-1}$ (acute) | See Box 2 | ✓ |
| 4. Threshold based on published literature: targeted | See 1.1 | | ✓ |
| 5. Threshold based on national data | See 1.1 | Unlikely to be able to demonstrate causal relationships in most datasets | ✓ |

# 5 Acidification

## 5.1 Background

Acidification is one of the physico-chemical elements specified in the WFD for inland waters but it is not specifically mentioned in Annex V for coastal and transitional waters or in the Marine Strategy Framework Directive. Nonetheless, Member States have a responsibility to address any factor that precludes a water body attaining good status. Concerns about ocean acidification means that Member States should, at least, consider whether acidification thresholds are necessary for their coastal and transitional waters.

Acidification is defined as a process characterised by increasing concentrations of hydrogen ions in soil or water due to human activity. It is possible for a water body to be naturally acid, with a biota composed of organisms that are tolerant to low pH. However, it is also common for low pH to be the result of human activities, particularly industrial air emissions of $SO_2$ and NOx leading to long-range transboundary acid rain and dry deposition but also plantation forestry and other activities. In such cases, the natural freshwater flora and fauna may be replaced by a biota composed of acid-tolerant species, typically with lower diversity than comparable unacidified sites. Juvenile stages of fish can be particularly vulnerable, with implications for ecosystem services.

The “critical load” is a widely-used measure of sensitivity to acidification. It indicates the amount of acid an ecosystem can tolerate in the long-term without being harmed. However, maps showing critical load exceedance are only available for a few areas of Europe. In other regions, geology can help to identify vulnerable areas.

Ocean acidification is caused by absorption of carbon dioxide from the atmosphere by seawater. Although the fall in pH may be small, this can be enough to cause difficulties for calcifying organisms such as crustaceans, molluscs, corals and some types of plankton.

In addition to setting low pH thresholds to protect against acidification, many countries also set high pH thresholds. Although it is possible for human activities to lead to highly alkaline systems, the main purpose of these higher thresholds seems to be to indicate high primary productivity which, in turn, suggests an increased risk of secondary effects (“undesirable disturbances”) of eutrophication, like the use of oxygen upper thresholds to detect oxygen diel fluctuations (see previous chapters). These high pH standards are not considered any further in this chapter.

## 5.2 Approaches to setting thresholds

### 5.2.1 Starting points

- Juvenile stages of fish are one of the groups most sensitive to acidification in freshwater ecosystems, and thus are suitable for setting thresholds. However, they tend to be relatively slow to respond to ecosystem recovery and expensive to monitor, so other groups, e.g. crustaceans and molluscs and other invertebrates, as well as macrophytes and algae also need to be included in monitoring programs.

- The approach developed by ECOSTAT to define reference conditions does not account for atmospheric deposition. A different type of reference model, then, is needed in softwater areas, taking account of atmospheric deposition: the MAGIC-model (Cosby *et al*., 1985; 2001; Helliwell *et al*., 2014) is one widely-used model, using a baseline of 1860 to estimate natural pH-values. Juggins *et al*. (2016) use a simpler approach based on the relationship between ANC and Ca. These abiotic reference values are then combined with abiotic data from impacted sites and concurrent biological data to set target values.

- Acidity in poorly-buffered regions is highly variable, and biota are particularly sensitive to low pH episodes. The number of low pH events, their duration and the abruptness of change all need to be considered. Some episodic events are related to hydrology and are, thus, predictable to some extent. Others (e.g. as a consequence of forest fires) are not.

- Relatively few countries have metrics specifically calibrated against acidification gradients and only those using invertebrates in N GIG have been intercalibrated. Multimetrics developed for “general degradation” that include a diversity sub-metric may respond to acidification, depending on the combination rule used.

- Ecotoxicological approaches may be useful for determining critical concentrations of $H^{+}$ or $Al^{3+}$ but these then need to be related back to local circumstances (e.g., natural buffering capacity, ANC) in order to derive a threshold compatible with WFD objectives.

- "Critical loads" is an important concept, but this measures the intensity of the stressor, rather than the response of biota. Nonetheless, Member States need to report pressures, and critical loads are a way of confirming whether or not acidification is significant, particularly in regions where alkalinity is low.

### 5.2.2 The influence of types

Acidification is only likely to be a widespread stressor in low alkalinity water bodies. In such regions, a typology may be needed for water bodies with very low alkalinity. However, whereas alkalinity is frequently used to differentiate types when dealing with other stressors, acidification disrupts the natural buffering capacity of inland waters, rendering alkalinity less useful as a type descriptor. Calcium concentration is a more useful parameter to differentiate the most acid-vulnerable types (e.g. Ca < 1 mg $L^{-1}$, Ca 1-4 mg $L^{-1}$, Norwegian Classification Guidance), along with measures of the concentration of humic matter (e.g., total or dissolved organic carbon, transparency). It is also possible to use descriptor variables to build predictive equations, allowing site- rather than type-specific thresholds to be derived.

For other areas, the issue is more likely to be that there are not enough sites showing clear effects of acidification for differentiation of water bodies into types to be particularly meaningful.

### 5.2.3 Monitoring strategies, determinands and parameters

Although pH is the most widely used parameter, this measures "acidity" rather than "acidification", which is why Acid Neutralising Capacity (ANC) is a better measure of acidification. ANC measures the difference between base cations and acid anions and, as such, measures the buffering capacity of a water body. The concentration of labile aluminium is a more direct measure of the likely impact on biota and is widely used for routine assessments in Norway and Sweden. However, this is difficult and time-consuming to measure and not used routinely elsewhere.

Most countries report pH as their preferred determinand for assessing this stressor, with a few (mostly in Scandinavia) also using ANC. Both central tendencies (mean, median) and percentiles are used (Tables 5.1 & 5.2). This in turn, contributes to the variability observed between standards adopted by countries. Geology and land use will also play a role. Those parts of Europe with naturally soft water and extensive coniferous forestry are more vulnerable to "acid rain" and its consequences, probably leading to more rigorous assessments of the capacity for local biota to withstand acidification, with extensive international co-ordination ("ICP-Waters"; Austnes *et al*., 2018).

Levels of acidity (whether measured as pH or ANC) can vary considerably over time, due to changes in hydrology (Kelly-Quinn *et al*. 1996; Feeley *et al*., 1997) with events such as snowmelt causing particularly severe acid "shocks" in some systems (Schaefer *et al*., 1990). Intermittent episodes of low pH are important in determining the condition of the flora and fauna (Feeket *et al*., 2011; Juggins *et al*., 2016) and, as a result, minima rather than means may be more reliable parameters for assessing the risk to biota than measures of central tendency (Jüttner *et al*., 2020). However, this may mean that more frequent chemical sampling is necessary if these episodes are not to be missed. Low frequency sampling is likely to miss acid episodes; however, these should be detected by biology. Therefore, it is important, that the "one out, all out" rule is followed strictly when assessing acidification.

In regions where critical load exceedances are unlikely, pH should be adequate as a coarse screening for the unlikely event of a major impacts. Biology, too, should show impacts of acidification in such circumstances, if biological metrics developed to assess acidification impacts are used for assessment. Metrics tuned to "general degradation" may not pick up acidification (and may even give spurious indications of "good status"). Diversity metrics are more likely to detect toxic influences in such cases.

**Table 5.1:** Overview of parameters used to monitor acidification across all water categories (from Kelly *et al.*, 2022; Teixeira *et al.*, 2022). The numbers in the table indicate the number of countries using the different parameters.

| Supporting element | Water category | | | |
|---|---|---|---|---|
| | River | Lake | Transitional | Coastal |
| ANC | 7 | 4 | 0 | 0 |
| pH | 21 | 15 | 1 | 0 |
| Other determinand (Alkalinity, inorganic aluminium concentration ($Al^{3+}$)) | 5 | 3 | 0 | 0 |

**Table 5.2**: Overview of metrics used to monitor pH in inland waters (from Kelly *et al.*, 2022). Numbers refer to the number of countries reporting standards that use a particular parameter/metric combination. Metrics have been split into those that measure the central tendency (e.g. mean, median) and those measuring a more extreme statistic (e.g. percentiles, maximum and minimum).

| Supporting element | central tendency | | percentile | other |
|---|---|---|---|---|
| | annual | seasonal | | |
| **Lakes** | 9 | 2 | 2 | 2 |
| **Rivers** | 8 | 1 | 15 | 2 |

### 5.2.4 Empirical approaches, expert judgement or ecotoxicology?

Various approaches have been used to set acidification standards, these include ecotoxicology, expert judgement, and empirical approaches. Whilst there is an extensive literature on the response of freshwater organisms to pH in the laboratory, difficulties in relating these to acidification mean that the potential for setting thresholds appropriate for the WFD is limited.

Expert judgement can also be used. Several countries appear to have adopted the thresholds from the Freshwater Fish Directive (pH: 6 - 9), for example. These may well be appropriate, particularly for those countries lacking pH gradients long enough to establish thresholds from first principles, but they have not been validated against WFD criteria for "good ecological status" and should only be used as a last resort.

Extensive empirical data have been used by Fõlster *et al*. (2007), McFarland *et al*. (2010) and Moe *et al*. (2010) to demonstrate relationships between invertebrate assemblages and acidification parameters, all of which have been or could be also be used to derive thresholds. Malcolm *et al*. (2014) describes an alternative regression-based approach which, rather than using a biological community, focusses on parr and fry of *Salmo trutta*, both recognised to be life stages that are particularly sensitive to low pH and elevated Al.

In addition, gradient forests have also been used to derive thresholds (Fõlster *et al*., 2021), using either split density or cumulative importance to identify likely thresholds along ANC gradients. Such methods are independent of WFD metrics although they still require data from along a long gradient in order to be effective.

Where baseline data are not available, integrating ecological and paleobiological approaches may provide a broader temporal dimension and support threshold setting, as demonstrated by Battarbee et al. (2014) for tracking changes in lake diatom assemblages recovering from acidification.

## 5.3 Role of climate change

Climate change is likely to have significant effects on the expression of acidification and its consequences over coming decades. Effects, however, are difficult to predict due to interactions amongst many variables, and there is likely to be variation in the scale of local effects. For this reason, a broad range of variables will need to be monitored in order to understand effects in individual catchments. In transitional waters Van Dam & Wang (2019) showed estuarine pH variation to be driven by the combined effects of global-scale changes in climate, regional-scale changes in precipitation/river discharge, and local-scale changes in estuarine biogeochemistry (e.g., net ecosystem metabolism, stratification patterns).

Several studies have reported increases in DOC in inland waters (Laudon *et al*., 2011; de Wit et al., 2016). Climate is often implicated although it has also been suggested that this is an indirect consequence of reductions in atmospheric deposition (Evans *et al*., 2006; Hruška *et al*., 2009). Drought, too, has been suggested as an explanation for increased DOC (Clark *et al*., 2012) though generalisations are difficult due to variation in the expression of effects between catchments (Worrall & Burt, 2008).

A growing trend to plant trees for carbon capture as part of a broader “green shift” could also, inadvertently, lead to exacerbation of acidification in some areas. Comparing matched acid-sensitive catchments, streams with coniferous forestry plantations were found to have lower taxonomic richness, although recovery was evident downstream (Feeley et al., 2011). Although, an increase in soil pH can also happen depending on the alkalinity in the soil and the type of trees (Hong et al. 2018).

## 5.4 Conclusions

- Acidification criteria are essential in regions where water has a naturally low buffering capacity. It is important to recognise that pH is an important natural stressor, as well as a potentially-significant anthropogenic stressor, and the two need to be disentangled by applying type-specific reference values and analysis of pH against biological metrics developed to assess acidification. This means that it is difficult to make generalisations, and all prescriptions will need to be tested against local monitoring data.

- ANC is a better measure of the extent of acidification, although many empirical and ecotoxicological studies measure responses to pH or labile aluminium ($Al^{3+}$). Lower percentiles are recommended as the most appropriate means for aggregating data, as these will be more effective at protecting against “spikes” in acidity;

- Acidification is unlikely to be a serious concern in well-buffered waters. In such instances, thresholds are likely to be necessary purely as a precautionary measure, to detect episodic events (e.g. industrial discharges). Thresholds set for the Freshwater Fish Directive are likely to be adequate in such areas, unless discharges coincide with particular life stages of the most sensitive species. For this reason, we follow US EPA guidance and recommend a slightly higher threshold of pH 6.5 (Table 5.3). The decision should be based on local knowledge of the fish fauna.

**Table 5.3.** Summary guidance for setting ecological thresholds for pH in rivers. in well-buffered water with very low risk of acidic episodes. See section 1.1 for explanation of use recommendations (✗ and ✓).

| Level | Threshold | Source | Use? |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | 6 – 9 | Indicative values from Freshwater Fish Directive (78/659/EEC) | ✓ |
| 2. Threshold based on published literature: general prescription | 6.5 – 9.0 | US EPA (1986) concluded that pH 6.0 – 6.5 is «unlikely to be harmful unless free carbon dioxide is present in excess of 100 ppm». Some evidence, too, of impairment of early life stages between pH 6.0 and 6.5. | ✓ |
| 3. Threshold based on published targeted literature: | See 1.1 | | ✓ |
| 4. Threshold based on national data | | | ✗ |

**Table 5.4.** Summary guidance for setting ecological thresholds for pH n poorly-buffered water; moderate or high risk of acidic episodes. See section 1.1 for explanation of use recommendations (✗ and ✓).

| Level | Threshold | Source | Use? |
|---|---|---|---|
| 1. Thresholds from existing Directive or guideline | 6 – 9 | Indicative values from Freshwater Fish Directive (78/659/EEC) | ✗ |
| 2. Threshold based on published literature: general prescription | | | ✗ |
| 3. Threshold based on published targeted literature: | See 1.1 | | ✓ |
| 4. Threshold based on national data | See 1.1 | | ✓ |

# 6 Overall Conclusions

The initial objective of this project was to examine variations in **thresholds** for physico-chemical supporting elements across Europe. However, this work quickly revealed major differences not just in thresholds, but also in approaches to measurement and aggregation. We have, therefore, broadened the approach to consider **criteria**, of which the threshold is one part, along with the choice of determinand ("parameters") and the means of data aggregation ("metrics", Poikane et al., 2019). For some supporting elements and types, there seems to be widespread agreement on appropriate criteria and many thresholds already appear to support GES. However, for other supporting elements, particularly oxygenation, thermal conditions and salinity, there are concerns about whether some Member States' approach to the criterion as a whole is fit-for-purpose. Four areas of particular concern have been identified:

1. **"Legacy standards":** thresholds set in the past may no longer provide adequate protection for conditions encountered today. Many standards for inland waters, in particular, seem to be based on thresholds set in the Freshwater Fish Directive (78/659/EEC) and/or reflect a time when chronic effects of heterotrophic oxygen consumption were major issues. The state of Europe's waters has changed substantially over the past 40 years, leading to a need to revisit many criteria in light of modern challenges, including climate change.

2. **Data gathering:** many national sampling programs are still based on quarterly or monthly sampling programs. This may be adequate to detect long-term chronic effects of pollutants but will need to be reconsidered for some supporting elements where the risks are from short-term acute events. A particular concern is detection of hypoxia, where sporadic sampling during "working hours" may miss the most acute manifestations. We envisage greater use of continuous monitoring in the future. Even if not deployed at every site on a monitoring network, it would provide more granular information on variability which would, at least, offer insights into the reliability of results from traditional sampling approaches.

3. **Climate change:** all regions and all water categories are experiencing effects of climate change. We have identified several instances where criteria need to be revised, particularly to protect against short-term acute "shocks" to ecosystems. The recent problem in the River Oder is a good example of climate interacting with a number of pre-existing conditions to create an ecological disaster (Free et al., 2022). However, there is also a need for a wider consideration of interactions amongst stressors as a result of climate change, and the implications of this for threshold setting.

4. **Multiple stressors:** this report has considered setting thresholds for individual stressors whereas, in practice, they often occur in combination. Interactions amongst stressors have profound effects on the aquatic biota. The presence of a second stressor can alter ecosystem responses to the stressor of interest, sometimes in unexpected ways (King, 1995; Filbee-Dexter et al., 2017; Birk, 2019). This is beyond the scope of this report. but it is important to realise that interrelationships amongst variables may lead to erroneous conclusions when setting thresholds for individual stressors in isolation (Phillips et al., 2019). We emphasise the importance of identifying causal relationships before attempting to derive thresholds.

Approaches to setting thresholds described in Phillips et al. (2018) and Kelly et al. (2021) can be adapted for use with other supporting elements (see Phillips et al., 2023, for updated recommendations). Examples of their application can be found in 2.4 (Transparency) and 3.3 (Oxygenation) as well as in Teixeira et al. (2023). There is no reason why these approaches could not be applied to salinity and acidification parameters too, so long as suitable data (adequate stressor gradient, few confounding stressors, stressor-specific biological metrics) are also available. Dealing with uncertainty in estimates of thresholds was recognised as a key challenge for nutrients and this will also be relevant to other supporting elements. Constraints identified in the work on nutrients (e.g. small datasets and short gradients) will also apply to other supporting elements. The best statistical approach for estimating thresholds can only ever give an accurate reflection of the datasets from which they were derived. It is, therefore, important to ensure that datasets give an honest characterisation of conditions. For some types of water body, and for smaller countries, it may not be possible to obtain an adequate dataset so there is a strong case for regional collaboration to develop robust thresholds.

We also recognise that there are alternatives to deriving thresholds from monitoring data. Both Phillips et al. (2018) and this report (4.2.5) have shown how historical data can be used. This depends, of course, on long time-series of data collected to a consistent standardised sampling method, criteria that will not always be fulfilled. However, when this is available, it is an immensely valuable resource that circumvents "shifting baseline syndrome" (i.e. the gradual change in accepted norms for the condition of the natural environment due to a lack of past information, Soga & Gaston, 2018).

This report has also explored the value of ecotoxicological approaches. These have proved to be particularly effective when strong stressor gradients are not revealed by routine monitoring data (e.g. salinity: 4.2.4.2) or where sensitive life stages may not be captured by assessment tools (e.g. juvenile stages of salmonids: Fig. 4.7). There are, however, issues with using ecotoxicology: test conditions and taxa need to be relevant to the region of interest and ecotoxicology often considers highly simplified systems a long way removed from real world situations (Fig. 6.1). Mismatches between ecotoxicology and field monitoring results are to be expected and the insights of an experienced biologist will be essential when interpreting test results.

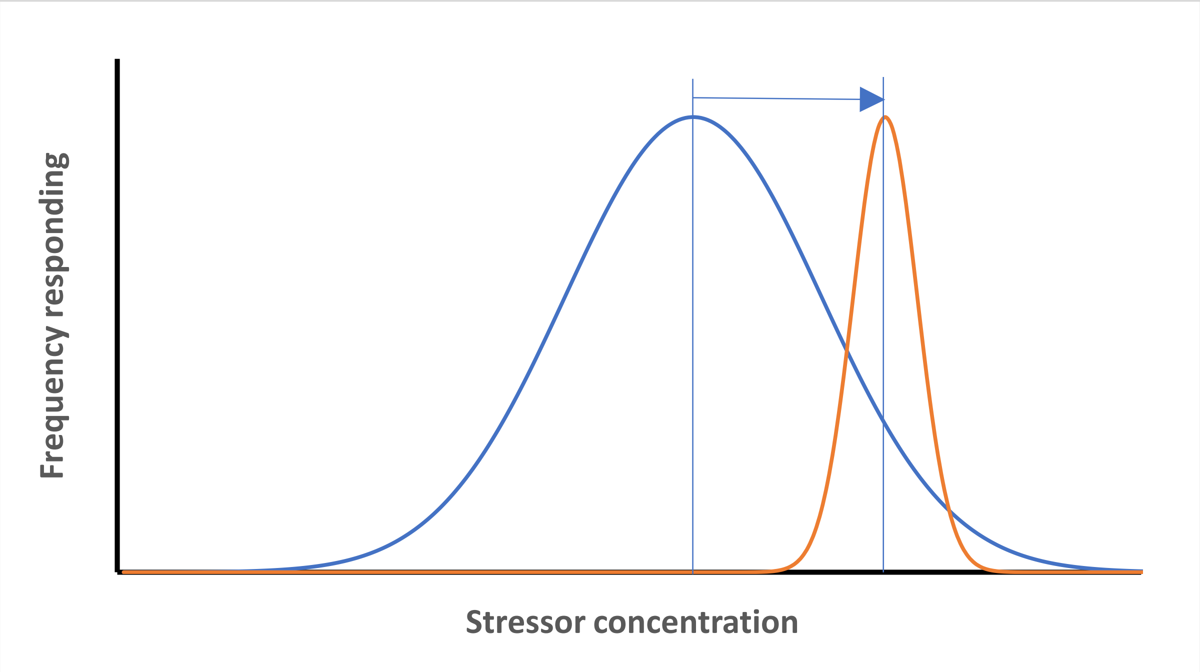



**Figure. 6.1.** The predicted shift in response to toxic chemical stress which might be observed within a population during the transfer from field (blue line) to laboratory conditions (orange line) (=selection bottleneck). Note that both the optimum concentration and the variance differ between the two conditions (after Baird, 1992).

A broader problem when reviewing thresholds and limits from the literature and websites of non-EU countries is that thresholds need to be very closely attuned to local conditions and set to protect ecology to standards equivalent to good ecological status. The limited data that we have been able to gather (Fig. 4.9; Fig. 5.7) suggests that thresholds set in the EU are no less stringent than those in use elsewhere. The challenges faced when comparing thresholds within one continent are, however, magnified fivefold when comparing thresholds across the globe.

Experience from setting nutrient thresholds to protect good ecological status has revealed many challenges (Kelly et al., 2021). Parallel work for ECOSTAT (Phillips et al., 2023) has focussed on the statistical aspects whilst this report has expanded the focus to place results in a broader ecological and geochemical context. We close by stressing that the process of threshold setting should never be considered in isolation from other aspects of the water management process. Thresholds have implications for classification and, in turn, for regulation (e.g. licensing discharges) and, as such, are an essential component of river basin management plans. They act, in effect, as expressions of the endeavour that will be required if good ecological status is to be achieved and, as such, are a key part of a broader political process through which stakeholders and the public can express their opinions (Article 14: Public information and consultation). The temptation to set (or retain) lenient thresholds which are inconsistent with good ecological status will, ultimately, undermine the bold ambition set out in the Water Framework Directive.

## References


Aarup, T. (2002). Transparency of the North Sea and Baltic Sea-a Secchi depth data mining study. Oceanologia 44: 323-327.

Altieri, A. H. & Gedan, K. B. (2015). Climate change and dead zones. Global Change Biology 21: 1395-1406. https://doi.org/10.1111/gcb.12754

Arle, J. & Wagner, F. (2013). Effects of anthropogenic salinisation on the ecological status of macroinvertebrate assemblages in the Werra River (Thuringia, Germany). Hydrobiologia 701: 129-148.https://doi.org/10.1007/s10750-012-1265-z

Altieri, A. H., Harrison, S. B., Seemann, J., Collin, R., Diaz, R. J., Knowlton, N. (2017). Tropical dead zones and mass mortalities on coral reefs. Proceedings of the National Academy of Sciences 114: 3660-3665. https://doi.org/10.1073/pnas.1621517114

Arts, G. (2002). Deterioration of Atlantic Soft Water Macrophyte Communities by Acidification, Eutrophication and Alkalinisation. Aquatic Botany 73: 373–393. https://doi.org/10.1016/S0304-3770(02)00031-1.

Austnes, K., Aherne, J., Arle, J., Čičendajeva, M., Couture, S., Fölster, J., ..., de Wit, H. (2018). Regional assessment of the current extent of acidification of surface waters in Europe and North America. NIVA-rapport. https://edepot.wur.nl/462869

ANZECC (2000). Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Environment and Conservation Council. Agriculture and Resource Management Council of Australia and New Zealand, Canberra, Australia.

Arnott, S.E., Celis-Salgado, M.P., Valleau, R.E., DeSellas A.M., Paterson, A.M., Yan, N.D., Smol, J.P. & Rusak J.A. (2020). Road salt impacts freshwater zooplankton at concentrations below current water quality guidelines. Environmental Science & Technology 54: 9398–9407.

Ayers, R.S. & Westcot, D.W. (1985). Water Quality for Agriculture. FAO Irrigation and Drainage Paper 29 (rev. 1). Food and Agriculture Organization, Rome.

Baird, D. J. (1992). Predicting population response to pollutants: in praise of clones. A comment on Forbes & Depledge. Functional ecology 6: 616-617.

Barbier, L., Suaire, R., Durickovic, I., Laurent, J. & Simonnot, M. O. (2018). Is a road stormwater retention pond able to intercept deicing Salt? Water Air Soil Pollut. 229. https://doi. org/10.1007/s11270-018-3908-9.

Barrio, R. N., Sola, I., Blanco-Murillo, F., del-Pilar-Ruso, Y., Fernández-Torquemada, Y., Sánchez-Lizaso, J. L. (2021). Application of salinity thresholds in Spanish brine discharge regulations: Energetic and environmental implications. Desalination 501: 114901. https://doi.org/10.1016/j.desal.2020.114901

Battarbee, R. W., Shilland, E. M., Kernan, M., Monteith, D. T., & Curtis, C. J. (2014). Recovery of acidified surface waters from acidification in the United Kingdom after twenty years of chemical and biological monitoring (1988–2008). Ecological Indicators 37: 267-273.

Batten, S. D., Ostle, C., Hélaouët, P., & Walne, A. W. (2022). Responses of Gulf of Alaska plankton communities to a marine heat wave. Deep Sea Research Part II: Topical Studies in Oceanography 195: 105002. https://doi.org/10.1016/j.dsr2.2021.105002

Belanger, S., Barron, M., Craig, P., Dyer, S., Galay-Burgos, M., Hamer, M., Marshall, S., Posthuma, L., Raimondo, S., & Whitehouse, P. (2017). Future needs and recommendations in the development of species sensitivity distributions: Estimating toxicity thresholds for aquatic ecological communities and assessing impacts of chemical exposures. Integrated Environmental Assessment and Management 13: 664-674. https://doi.org/10.1002/ieam.1841

Best, M. A., Wither, A. W., & Coates, S. (2007). Dissolved oxygen as a physico-chemical supporting element in the Water Framework Directive. Marine pollution bulletin 55: 53-64. https://doi.org/10.1016/j.marpolbul.2006.08.037

Birk, S. (2019). Detecting and quantifying the impact of multiple stress on river ecosystems. pp. 235-253. In: Multiple Stressors in River Ecosystems (edited by S. Sabater, A. Elosegi & R. Ludwig). Elsevier, Amsterdam. https://doi.org/10.1016/B978-0-12-811713-2.00014-5

Blindow, I. (1992) Declines of charophytes during eutrophication: comparison with angiosperms. Freshwater Biology 28, 9-14.

Bonacina, L., Fasano, F., Mezzanotte, V., & Fornaroli, R. (2023). Effects of water temperature on freshwater macroinvertebrates: a systematic review. Biological Reviews 98: 191-221. https://doi.org/10.1111/brv.12903

Borja, Á., Muxika, I., Franco, J. (2006). Long-term recovery of soft-bottom benthos following urban and industrial sewage treatment in the Nervión estuary (southern Bay of Biscay). Marine Ecology Progress Series 313: 43-55. DOI:10.3354/meps313

Buchwalter, D., Scheibener, S., Chou, H., Soucek, D. & Elphick, J. (2019). Are sulfate effects in the mayfly Neocloeon triangulifer driven by the cost of ion regulation? Philosophical Transactions of the Royal Society B: Biological Sciences, 374(1764), 20180013.

Burton, L. R. (2003). The Mersey Basin: an historical assessment of water quality from an anecdotal perspective. Science of the Total Environment 314: 53-66. https://doi.org/10.1016/S0048-9697(03)00094-9

Braukman, U., & Böhme, D. (2011). Salt pollution of the middle and lower sections of the river Werra (Germany) and its impact on benthic macroinvertebrates. Limnologica, 41, 113-124.

Cañedo-Argüelles, M., Kefford, B.J., Piscart, C., Prat, N., Schäfer, R.B., & Schulz, C.J. (2013). Salinisation of rivers: an urgent ecological issue. Environmental pollution, 173, 157-167.

Cañedo-Argüelles, M., Hawkins, C.P., Kefford, B.J., Schäfer, R.B., Dyack, B.J., Brucet, S., … & Timpano, A.J. (2016a). Saving freshwater from salts. Science, 351(6276), 914-916.

Cañedo-Argüelles, M., Sala, M., Peixoto, G., Prat, N., Faria, M., Soares, A. M., Barata, C., & Kefford, B. (2016b). Can salinity trigger cascade effects on streams? A mesocosm approach. Science of the Total Environment, 540, 3–10.

Capuzzo, E., Stephens, D., Silva, T., Barry, J., & Forster, R. M. (2015). Decrease in water clarity of the southern and central North Sea during the 20th century. Global change biology 21: 2206-2214.

Cardoso, A. C., Free, G., Nõges, P., Kaste, Ø., Poikane, S., & Solheim, A. L. (2009). Lake Management, Criteria. In G. E. Likens (Ed.), Encyclopedia of Inland Waters (pp. 310–331). Academic Press. https://doi.org/10.1016/B978-012370626-3.00244-1

Carrier-Belleau, C., Pascal, L., Nozais, C., & Archambault, P. (2022). Tipping points and multiple drivers in changing aquatic ecosystems: A review of experimental studies. Limnology and Oceanography 67: S312-S330. https://doi.org/10.1002/lno.11978.

Carstensen, J. (2010). Censored data regression: Statistical methods for analyzing Secchi transparency in shallow systems. Limnology and Oceanography: Methods 8: 376-385. https://doi.org/10.4319/lom.2010.8.376

Carstensen, J., Dahl, K., Henriksen, P., Hjorth, M., Josefson, A., Krause-Jensen, D. (2011). Coastal Monitoring Programs. Pp. 175-206. In: Treatise on estuarine and coastal science (eited by E. Wolanski & D. McLusky), Elsevier, Amsterdam.

Carvalho, L., Mackay, E. B., Cardoso, A. C., Baattrup-Pedersen, A., Birk, S., Blackstock, K. L., Borics, G., Borja, A., Feld, C. K.,Ferreira, M. T., Globevnik, L., Grizzetti, B., Hendry, S., Hering, D., Kelly, M., Langaas, S., Meissner, K., Panagopoulos, Y., Penning, E., …Solheim, A. L. (2019). Protecting and restoring Europe's waters: An analysis of the future development needs of the water framework directive. Science of the Total Environment, 658, 1228–1238.

CCME (2011). Canadian Water Quality Guidelines for the Protection of Aquatic Life: Chloride. Canadian Council of Ministers of the Environment, Winnipeg.

Chen, Q., Wang, S., Ni, Z., Guo, Y., Liu, X., Wang, G., & Li, H. (2021). Non-linear dynamics of lake ecosystem in responding to changes of nutrient regimes and climate factors: Case study on Dianchi and Erhai lakes, China. Science of the Total Environment 781: 146761. https://doi.org/10.1016/j.scitotenv.2021.146761

Clark, J. M., Heinemeyer, A., Martin, P., Bottrell, S. H. (2012). Processes controlling DOC in pore water during simulated drought cycles in six different UK peats. Biogeochemistry 109: 253-270. https://doi.org/10.1007/s10533-011-9624-9

Clements, W.H., Cadmus, P. & Brinkman S.F. (2013). Responses of aquatic insects to Cu and Zn in stream microcosms: understanding differences between single species tests and field responses. Environmental Science and Technology 47, 7506-7513.

Cosby, B. J., Hornberger, G. M., Galloway, J. N., Wright, R. F. (1985), Modeling the effects of acid deposition: assessment of a lumped parameter model of soil water and streamwater chemistry. Water Resources Research 21: 51– 63. doi:10.1029/WR021i001p00051.

Cosby, B. J., Ferrier, R. C., Jenkins, A., Wright, R. F. (2001). Modelling the effects of acid deposition: refinements, adjustments and inclusion of nitrogen dynamics in the MAGIC model, Hydrology and Earth System Sciences 5: 499–518. https://doi.org/10.5194/hess-5-499-2001

Conroy, E., Turner, J. N., Rymszewicz, A., O'Sullivan, J. J., Bruen, M., Lawler, D., Lally, H., & Kelly-Quinn, M. (2016). The impact of cattle access on ecological water quality in streams: Examples from agricultural catchments within Ireland. Science of The Total Environment, 547, 17–29. https://doi.org/10.1016/j.scitotenv.2015.12.120

Council of the European Communities (2000). 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. L327, 72.

Council of the European Communities (2013). Commission Decision of 20 September 2013 establishing pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise and repealing Decision 2008/915/EC. Official Journal of the European Communities 480, 1–47.

Cunha, M.E., H. Quental-Ferreira, A. Parejo, S. Gamito, L. Ribeiro, M. Moreira, I. Monteiro, F. Soares, P. Pousão-Ferreira (2019). Methodology for assessing the individual role of fish, oyster, phytoplankton and macroalgae in the ecology of integrated production in earthen ponds. MethodsX 6: 2570-2576. DOI: 10.1016/j.mex.2019.10.016.

Cunillera-Montcusí, D., Beklioğlu, M., Cañedo-Argüelles, M., Jeppesen, E., Ptacnik, R., Amorim, C. A., ... & Matias, M. (2022). Freshwater salinisation: a research agenda for a saltier world. Trends in Ecology & Evolution 37: 440-453. https://doi.org/10.1016/j.tree.2021.12.005

Cuttle, S. P., & James, A. R. (1995). Leaching of lime and fertilisers from a reseeded upland pasture on a stagnogley soil in mid-Wales. Agricultural water management 28: 95-112. https://doi.org/10.1016/0378-3774(95)01174-H.

Dell'Uomo, A. (1998). Use of algae for monitoring rivers in Italy: current situation and perspectives. In J. Prygiel, B.A. Whitton & J. Bukowska (Ed.) Use of Algae to Monitor Rivers III, Douai: Agence de l'Eau Artois-Picardie. pp.17-25.

Devlin, M. J., Barry, J., Mills, D. K., Gowen, R. J., Foden, J., Sivyer, D., & Tett, P. (2008). Relationships between suspended particulate material, light attenuation and Secchi depth in UK marine waters. Estuarine, Coastal and Shelf Science, 79: 429-439.

de Wit, H. A., Valinia, S., Weyhenmeyer, G. A., Futter, M. N., Kortelainen, P., Austnes, K., . Vuorenmaa, J. (2016). Current Browning of Surface Waters Will Be Further Promoted by Wetter Climate. Environmental Science and Technology Letters, 3, 430-435.

Dickey, J. W., Cuthbert, R. N., Lugo, S. C. M., Casties, I., Dick, J. T., Steffen, G. T., & Briski, E. (2021). The stars are out: Predicting the effect of seawater freshening on the ecological impact of a sea star keystone predator. Ecological Indicators 132: 108293. https://doi.org/10.1016/j.ecolind.2021.108293.

Direktoratsguppen vanndirektivet (2018). Klassifisering av miljøtilstand i vann. Veileder 02:2018. Norwegian Classification Guidance, 2018. Klassifiseringsveileder 02:2018 Klassifisering av miljøtilstand i vann. Økologisk og kjemisk klassifiseringssystem for kystvann, grunnvann, innsjøer og elver. Direktoratsgruppen for gjennomføringen av vannforskriften. Link: https://www.vannportalen.no/veiledere/klassifiseringsveileder/

Dudney, J., & Suding, K. N. (2020). The elusive search for tipping points. Nature Ecology & Evolution 4: 1449-1450. https://doi.org/10.1038/s41559-020-1273-8.

Dupont, N., & Aksnes, D. L. (2013). Centennial changes in water clarity of the Baltic Sea and the North Sea. Estuarine, Coastal and Shelf Science 131: 282-289.

DWAF (1996). South African Water Quality Guidelines. 2nd Edition, Volume 7: Aquatic Ecosystems. Department of Water Affairs and Forestry, Pretoria.

Edmundson, J. A. & Koenings, J. P. (1986). The Effects of Glacial Silt On Primary Production, Through Altered Light Regimes and Phosphorus Levels, In Alaska Lakes. FRED report #68, Alaska Department of Fish and Game, Fairbanks, 26pp.

Elphick, J.R.F., Bergh, K.D., Bailey, H.C. (2011). Chronic toxicity of chloride to freshwater species: effects of hardness and implications for water quality guidelines. Environmental Toxicology and Chemistry 30: 239-246. https://doi.org/10.1002/etc.365

Eriksen, T.E., Lindholm, M., Kile Røst, M., Lyche Solheim, A., Friberg, N., (2015). Vurdering av kunnskapsgrunnlag for leirpåvirkede elver. NIVA Rep. 6792. In Norwegian.

Evans, C.D., Chapman, P.J., Clark, J.M., Monteith, D.T., Cresser, M.S. (2006). Alternative explanations for rising dissolved organic carbon export from organic soils. Global Change Biology 12: 2044-2053. https://doi.org/10.1111/j.1365-2486.2006.01241.x

Feeley, H.B., Bruen, M., Blacklocke, S., Kelly-Quinn, M. (1997). A regional examination of episodic acidification response to reduced acidic deposition and the influence of plantation forests in Irish headwater streams. Science of The Total Environment 443: 173-183. https://doi.org/10.1016/j.scitotenv.2012.10.074

Feeley, H.B., Kerrigan, C., Fanning, P. et al. (2011). Longitudinal extent of acidification effects of plantation forest on benthic macroinvertebrate communities in soft water streams: evidence for localised impact and temporal ecological recovery. Hydrobiologia 671, 217–226 https://doi.org/10.1007/s10750-011-0719-z

Feio, M. J., Aguiar, F. C., Almeida, S. F. P., Ferreira, J., Ferreira, M. T., Elias, C., ... & Vieira, C. (2014). Least disturbed condition for European Mediterranean rivers. Science of the Total Environment 476: 745-756. https://doi.org/10.1016/j.scitotenv.2013.05.056

Feld, C.K., Lorenz, A.W., Peise, M., Fink, M. & Schulz, C.J., (2023). Direct and indirect effects of salinisation on riverine biota: a case study from river Wipper, Germany. Hydrobiologia, 850, 3043-3059.

Ficker, H., Luger, M., Gassner, H. (2017). From dimictic to monomictic: Empirical evidence of thermal regime transitions in three deep alpine lakes in Austria induced by climate change. Freshwater Biology 62: 1335-1345. https://doi.org/10.1111/fwb.12946

Filbee-Dexter K., Pittman J., Haig H. A., Alexander S. M., Symons C. C., & Burke M. J. (2017). Ecological surprise: concept, synthesis, and social dimensions. Ecosphere 8: e02005. https://doi.org/10.1002/ecs2.2005.

Flanagan, P.J. (1990). Parameters of water quality. Environmental Research Unit, Dublin, 160 pp. ISBN 1-85053-095-5

Fleming-Lehtinen, V., & Laamanen, M. (2012). Long-term changes in Secchi depth and the role of phytoplankton in explaining light attenuation in the Baltic Sea. Estuarine, Coastal and Shelf Science 102: 1-10.

Fölster, J., Andrén, C., Bishop, K., Buffam, I., Cory, N., Goedkoop, W., ... & Wilander, A. (2007). A novel environmental quality criterion for acidification in Swedish lakes—an application of studies on the relationship between biota and water chemistry. Water, Air and Soil Pollution Focus 7: 331-338. https://doi.org/10.1007/978-1-4020-5885-1_37

Fölster J., Garmo Ø.A., Carlson P., Johnson R., Velle G., Austnes K., Hallstan S., Holmgren K., Schartau A.K., Moldan F., Aroviita J. (2021). Acidified or not? : a comparison of Nordic systems for classification of physicochemical acidification status and suggestions towards a harmonised system. Institutionen för vatten och miljö, Sveriges lantbruksuniversitet, Uppsala, Sweden, Rapport 2021: 1 https://pub.epsilon.slu.se/22175/

Forsberg, C. (1992). Will an increased greenhouse impact in Fennoscandia give rise to more humic and coloured lakes?. Hydrobiologia 229: 51-58. https://doi.org/10.1007/978-94-011-2474-4_4

Free, G., Tierney, D., Little, R., Kelly, F.L., Kennedy, B., Plant, C., Trodd, W., Wynne, C., Caroni, R. Byrne, C. (2016). Lake ecological assessment metrics in Ireland: relationships with phosphorus and typology parameters and the implications for setting nutrient standards. In Biology and Environment: Proceedings of the Royal Irish Academy 116: 191-204.

Free, G., Van De Bund, W., Gawlik, B., Van Wijk, L., Wood, M., Guagnini, E., Koutelos, K., Annunziato, A., Grizzetti, B., Vigiak, O., Gnecchi, M., Poikane, S., Christiansen, T., Whalley, C., Antognazza, F., Zerger, B., Hoeve, R. and Stielstra, H. (2023). An EU analysis of the ecological disaster in the Oder River of 2022, EUR 31418 EN, Publications Office of the European Union, Luxembourg, 2023, ISBN 978-92-76-99314-8, doi:10.2760/067386, JRC132271.

Friberg, N., Skriver, J., LARSEN, S. E., Pedersen, M. L., & Buffagni, A. (2010). Stream macroinvertebrate occurrence along gradients in organic pollution and eutrophication. Freshwater Biology 55: 1405-1419.https://doi.org/10.1111/j.1365-2427.2009.02373.x

Fritz, S.C., Cumming, B.F., Gasse, F. & Laird, K.R. (1999). Diatoms as indicators of hydrologic and climate change in saline lakes. pp. 41-72. In: The Diatoms: Applications for the Environmental and Earth Sciences. E.F. Stoermer & J.P. Smol (Ed.). Cambridge University Press, Cambridge.

Froend, R.H., Halse, S.A. & Storey, A.W. (1997). Planning for the recovery of Lake Toolibin, Western Australia. Wetlands Ecology and Management, 5, 73-85.

Gacia, E., Invers, O., Manzanera, M., Ballesteros, E., & Romero, J. (2007). Impact of the brine from a desalination plant on a shallow seagrass (Posidonia oceanica) meadow. Estuarine, Coastal and Shelf Science 72: 579-590. https://doi.org/10.1016/j.ecss.2006.11.021

Gaiser, E. E., Deyrup, N. D., Bachmann, R. W., Battoe, L. E., Swain, H. M. (2009). Multidecadal climate oscillations detected in a transparency record from a subtropical Florida lake. Limnology and Oceanography 54: 2228-2232. https://doi.org/10.4319/lo.2009.54.6.2228

Garrote-Moreno, A., Fernández-Torquemada, Y., & Sánchez-Lizaso, J. L. (2014). Salinity fluctuation of the brine discharge affects growth and survival of the seagrass Cymodocea nodosa. Marine Pollution Bulletin 81: 61-68. https://doi.org/10.1016/j.marpolbul.2014.02.019

Gibbs, R.J. (1970). Mechanisms controlling world water chemistry. Science (New York) 170: 1088-1090.DOI: 10.1126/science.170.3962.108

Griffith, M.B. (2014). Natural variation and current reference for specific conductivity and major ions in wadeable streams of the conterminous USA. Freshwater Science, 33(1), 1-17.

Halle, M. & Müller, A. (2014). Korrelationen zwischen biologischen Qualitätskomponenten und allgemeinen chemischen und physikalisch-chemischen Parametern in Fließgewässern. Final report. Projekt O 3.12 des Länderfinanzierungsprogramms „Wasser, Boden und Abfall“ 2012. Umweltbüro Essen & Chromgruen, Essen & Velbert. 190 pp.

Halle, M. & Müller, A. (2017). Ergänzende Arbeiten zur Korrelation zwischen biologischen Qualitätskomponenten und allgemeinen physikalisch-chemischen Parametern in Fließgewässern. Final report. Projekt O 3.15 des Länderfinanzierungsprogramms „Wasser, Boden und Abfall“ 2015. Umweltbüro Essen & Chromgruen, Essen & Velbert. 227 pp.

Harris, R.M., Beaumont, L.J., Vance, T.R., Tozer, C.R., Remenyi, T.A., Perkins-Kirkpatrick, S.E., Mitchell, P.J., Nicotra, A.B., McGregor, S., Andrew, N.R., Letnic, M. (2018). Biological responses to the press and pulse of climate trends and extreme events. Nature Climate Change 8: 579-587. https://doi.org/10.1038/s41558-018-0187-9

Hart, B.T., Bailey, P., Edwards, R., Hortle, K., James, K., McMahon, A., Meredith, C. & Swadling, K. (1991). A review of the salt sensitivity of the Australian freshwater biota. Hydrobiologia, 210(1-2), 105-144.

Hartman, K.J.; Wellman, D.I., Jr.; Kingsbury, J.W.; Cincotta, D.A.; Clayton, J.L.; Eliason, K.M.; Jernejcic, F.A.; Owens, N.V.; Smith, D.M. A Case Study of a Prymnesium parvum Harmful Algae Bloom in the Ohio River Drainage: Impact, Recovery and Potential for Future Invasions/Range Expansion. Water 13: 3233. https://doi.org/10.3390/w13223233

Hassell, K. L., Kefford, B. J., Nugegoda, D. (2006). Sub-lethal and chronic lethal salinity tolerance of three freshwater insects: Cloeon sp. and Centroptilum sp. (Ephemeroptera: Baetidae) and Chironomus sp. (Diptera: Chironomidae). Journal of Experimental Biology 209: 4024-4032. doi: 10.1242/jeb.02457.

Hayashida, H., Matear, R. J., & Strutton, P. G. (2020). Background nutrient concentration determines phytoplankton bloom response to marine heatwaves. Global Change Biology 26: 4800-4811. https://doi.org/10.1111/gcb.15255

HELCOM (2018). Oxygen debt. HELCOM core indicator report. HELCOM core indicator report [date viewed: 06/04/2023]

Helliwell, R. C., Richard F. Wright, Leah A. Jackson-Blake, Robert C. Ferrier, Julian Aherne, Bernard J. Cosby, Christopher D. Evans, Martin Forsius, Jakub Hruska, Alan Jenkins, Pavel Kram, Jiri Kopáček, Vladimir Majer, Filip

Moldan, Maximilian Posch, Jacqueline M. Potts, Michela Rogora, Wolfgang Schöpp (2014). Assessing Recovery from Acidification of European Surface Waters in the Year 2010: Evaluation of Projections Made with the MAGIC Model in 1995. Environmental Science & Technology 48: 13280-13288. https://doi.org/10.1021/es502533c

Herbert, E. R., P. Boon, A. J. Burgin, S. C. Neubauer, R. B. Franklin, M. Ardón, K. N. Hopfensperger, L. P. M. Lamers & Gell, P. (2015). A global perspective on wetland salinization: ecological consequences of a growing threat to freshwater wetlands. Ecosphere 6: art206 doi:https://doi.org/10.1890/ES14-00534.1.

Hillebrand, H., Donohue, I., Harpole, W. S., Hodapp, D., Kucera, M., Lewandowska, A. M., ... & Freund, J. A. (2020). Thresholds for ecological responses to global change do not emerge from empirical data. Nature Ecology & Evolution 4: 1502-1509. https://doi.org/10.1038/s41559-020-1256-9

Hintz, W.D., Arnott, S.E., Symons, C.C., Greco, D.A., McClymont, A., Brentrup, J.A., ... & Weyhenmeyer, G.A. (2022). Current water quality guidelines across North America and Europe do not protect lakes from salinization. Proceedings of the National Academy of Sciences, 119(9), e2115033119

Holland, D.G. & Harding, J.P.C.H. (1984) Mersey. pp. 113-144. In: Ecology of European Rivers (edited by B.A. Whitton). Blackwell, Oxford.

Hoover, Z., Ferrari, M.C. & Chivers, D.P. (2013). The effects of sub-lethal salinity concentrations on the anti-predator responses of fathead minnows. Chemosphere, 90(3), 1047-1052.

Hong, S., Piao, S., Chen, A. et al. (2018). Afforestation neutralizes soil pH. Nature Communications 9, 520 (2018). https://doi.org/10.1038/s41467-018-02970-1.Horne, A. J., & Goldman, C. R. (1994). Limnology (2nd ed.). McGraw-Hill.

Horrigan, N., Choy, S., Marshall, J., & Recknagel, F. (2005). Response of stream macroinvertebrates to changes in salinity and the development of a salinity index. Marine and Freshwater Research 56: 825-833. https://doi.org/10.1071/MF04237.

Horrigan, N., Dunlop J.E., Kefford, B.J. & Zavahir, E. (2007). Acute toxicity largely reflects the salinity sensitivity of stream macroinvertebrates derived using field distributions. Marine and Freshwater Research 58: 178-186. https://doi.org/10.1071/MF05241.

Hruška, J., Pavel Krám, William H. McDowell, Filip Oulehle, (2009). Increased Dissolved Organic Carbon (DOC) in Central European Streams is Driven by Reductions in Ionic Strength Rather than Climate Change or Decreasing Acidity. Environmental Science & Technology 43: 4320-4326. DOI: 10.1021/es803645w

Huo, S., He, Z., Ma, C., Zhang, H., Xi, B., Xia, X., ... & Wu, F. (2019). Stricter nutrient criteria are required to mitigate the impact of climate change on harmful cyanobacterial blooms. Journal of Hydrology 569: 698-704. https://doi.org/10.1016/j.jhydrol.2018.12.030

James K., Cant B., & Ryan T. (2003). Responses of freshwater biota to rising salinity levels and implications for saline water management: a review. Australian Journal of Botany, 51, 703-713. https://doi.org/10.1071/BT02110

Järvinen, M., Drakare, S., Free, G., Lyche-Solheim, A., Phillips, G., Skjelbred, B., Mischke, U., Ott, I., Poikane, S., & Søndergaard, M. (2013). Phytoplankton indicator taxa for reference conditions in Northern and Central European lowland lakes. Hydrobiologia 704: 97–113. https://doi.org/10.1007/s10750-012-1315-6

Jeppesen, E., Søndergaard, M., & Liu, Z. (2017). Lake restoration and management in a climate change perspective: an introduction. Water 9: 122. https://doi.org/10.3390/w9020122

Jeppesen, E., Beklioğlu, M., Özkan, K., & Akyürek, Z. (2020). Salinization increase due to climate change will have substantial negative effects on inland waters: a call for multifaceted research at the local and global scale. The Innovation 1: 100030. https://doi.org/10.1016/j.xinn.2020.100030

Jeppesen, E., J. Audet, T. A. Davidson, et al. (2021). Nutrient Loading, Temperature and Heat Wave Effects on Nutrients, Oxygen and Metabolism in Shallow Lake Mesocosms Pre-Adapted for 11 Years. Water 13: 127. doi:10.3390/w13020127

Jeremias, G., Barbosa, J., Marques, S. M., De Schamphelaere, K.A.C., Van Nieuwerburgh, F., Deforce, D., Gonçalves, F.J.M., Pereira, J.L. & Asselman, J. (2018). Transgenerational inheritance of DNA hypomethylation in Daphnia magna in response to salinity stress. Environmental Science & Technology, 52(17), 10114-10123

Jones, J. I., Davy-Bowker, J., Murphy, J., Keller, V., Williams, R., & Davies, C. (2008). Review of the evidence for organic pollution thresholds to protect rivers with special designations for wildlife. CEH Report Number: C03400/2008/NERC/Centre for Ecology & Hydrology, report to English Nature, 213pp.

Jones, L. P., Turvey, S. T., Massimino, D., & Papworth, S. K. (2020). Investigating the implications of shifting baseline syndrome on conservation. People and Nature 2: 1131-1144. https://doi.org/10.1002/pan3.10140.

Juggins, S., Kelly, M., Allott, T., Kelly-Quinn, M., Monteith, D. (2016). A Water Framework Directive-compatible metric for assessing acidification in UK and Irish rivers using diatoms. Science of the Total Environment 568: 671-678.. https://doi.org/10.1016/j.scitotenv.2016.02.163

Jüttner, I., Kelly, M. G., Evans, S., Probert, H., Orange, A., Ector, L., Marsh-Smith, S. (2020). Assessing the impact of land use and liming on stream quality, diatom assemblages and juvenile salmon in Wales, United Kingdom. Ecological Indicators 121: 107057. https://doi.org/10.1016/j.ecolind.2020.107057

Kakouei, K., Kraemer, B. M., Anneville, O., Carvalho, L., Feuchtmayr, H., Graham, J. L., ... & Adrian, R. (2021). Phytoplankton and cyanobacteria abundances in mid-21st century lakes depend strongly on future land use and climate projections. Global Change Biology 27: 6409-6422. https://doi.org/10.1111/gcb.15866

Karatayev, A.Y., Burlakova, L.E., Padilla, D.K. (2002). Impacts of Zebra Mussels on Aquatic Communities and their Role as Ecosystem Engineers. pp. 443-446. In: Leppäkoski, E., Gollasch, S., Olenin, S. (eds) Invasive Aquatic Species of Europe. Distribution, Impacts and Management. Springer, Dordrecht. https://doi.org/10.1007/978-94-015-9956-6_43

Kaushal, S. S., Mayer, P. M., Vidon, P. G., Smith, R. M., Pennino, M. J., Newcomer, T. A., ... & Belt, K. T. (2014). Land use and climate variability amplify carbon, nutrient, and contaminant pulses: a review with management implications. JAWRA Journal of the American Water Resources Association 50: 585-614.

Kaushal, S. S., Likens, G. E., Pace, M. L., Utz, R. M., Haq, S., Gorman, J., & Grese, M. (2018). Freshwater salinization syndrome on a continental scale. Proceedings of the National Academy of Sciences 115: E574-E583. https://doi.org/10.1073/pnas.1711234115.

Kaushal, S. S., Likens, G. E., Pace, M. L., Reimer, J. E., Maas, C. M., Galella, J. G., ... & Woglo, S. A. (2021). Freshwater salinization syndrome: From emerging global problem to managing risks. Biogeochemistry 154: 255-292. https://doi.org/10.1007/s10533-021-00784-w.

Kaushal, S. S., Likens, G. E., Utz, R. M., Pace, M. L., Grese, M., & Yepsen, M. (2013). Increased river alkalinization in the Eastern US. Environmental science & technology 47: 10302-10311. https://doi.org/10.1021/es401046s.

Kefford, B.J., Nugegoda, D., Zalizniak, L., Fields, E.J., Hassall, K.L. (2007). The salnity tolerance of freshwater macroinvertebrate eggs and hatchlings in comparison to their older life-stages: a diversity of responses. Aquatic Ecology 41: 335-348. https://doi.org/10.1007/s10452-006-9066-y

Kelly, M., Phillips, G., Teixeira, H., Salas Herrero, F., Várbíró, G., Kolada A., Lyche Solheim, A., Poikane, S. (2022). Physico-chemical supporting elements in inland waters under the Water Framework Directive: a review of national standards to support good ecological status. EUR 31040 EN, Publications Office of the European Union, Luxembourg, 2022. ISBN 978-92-76-51133-5, doi:10.2760/470539, JRC127875.

Kelly, M.G., Juggins, S., Moschandreou, K., Kemitzoglou, D., Tsiaoussi, V. (2023). Development of novel diatom metrics to assess ecological status of phytobenthos in Greek lakes. Ecologcial Indicators 147: 109974. https://doi.org/10.1016/j.ecolind.2023.109974

Kelly, M., Phillips, G., Teixeira, H., Várbíró, G., Salas-Herrero, F., Willby, N. J., Poikane, S. (2022b). Establishing ecologically-relevant nutrient thresholds: A tool-kit with guidance on its use. Science of The Total Environment 807: 150977. https://doi.org/10.1016/j.scitotenv.2021.150977

Kelly-Quinn, M., Tierney, D., Coyle, C., Bracken, J. (1996), Factors affecting the susceptibility of Irish soft-water streams to forest-mediated acidification. Fisheries Management and Ecology 3: 287-301. https://doi.org/10.1046/j.1365-2400.1996.d01-153.x

King, A. (1995). Avoiding ecological surprises: Lesson from long-standing communities. The Academy of Management Review 20: 961-985. https://doi.org/10.2307/258962.

Klose, A. K., Karle, V., Winkelmann, R., Donges, J. F. (2020). Emergence of cascading dynamics in interacting tipping elements of ecology and climate. Royal Society Open Science 7: 200599. https://doi.org/10.1098/rsos.200599

Koenings, J. P. & Edmundson, J. A. (1991). Secchi disk and photometer estimates of light regimes in Alaskan lakes: Effects of yellow color and turbidity. Limnology and Oceanography 36: 91-105. https://doi.org/10.4319/lo.1991.36.1.0091

Kristensen, P., & Hansen, H. O. (1994). European Rivers and Lakes. Assessment of Their Environmental State. Copenhagen: European Environment Agency, 122pp.

Ladrera, R., Caņedo-Argüelles, M., & Prat, N. (2017). Impact of potash mining in streams: the Llobregat basin (northeast Spain) as a case study. Journal of Limnology, 76, 343-354.

Laudon, H., Buttle, J., Carey, S. K., McDonnell, J., McGuire, K., Seibert, J., Shanley, J., Soulsby, C., Tetzlaff, D. (2012), Cross-regional prediction of long-term trajectory of stream water DOC response to climate change, Geophysical Research Letters 39: L18404. doi:10.1029/2012GL053033.

Lennox, R. J., Pulg, U., Malley, B., Gabrielsen, S.-E., Hanssen, E. M., Cooke, S. J., Birnie-Gauvin, K., Barlaup, B. T., Vollset, K. W. (2021). The various ways that anadromous salmonids use lake habitats to complete their life history. Canadian Journal of Fisheries and Aquatic Sciences 78: 90-100. https://doi.org/10.1139/cjfas-2020-0225

Leite, T., Branco, P., Ferreira, M. T., & Santos, J. M. (2022). Activity, boldness and schooling in freshwater fish are affected by river salinization. Science of the Total Environment, 819, 153046.

Lewis Jr, W. M. (1996). Tropical lakes: how latitude makes a difference. Perspectives in Tropical Limnology 4364: 43-64.

Lewis Jr, W. M. (2011). Global primary production of lakes: 19th Baldi Memorial Lecture. Inland Waters 1: 1-28. https://doi.org/10.5268/IW-1.1.384

Lorenzen, M. W. (1978). Phosphorus models and eutrophication. pp. 31-50. In Mitchell, R. (ed.), Water Pollution Microbiology (Volume 2). John Wiley & Sons, New York.

Lorenzen, M. W. (1980.) Use of chlorophyll-Secchi disk relationships. Limnology and Oceanography 25: 371-372.

Lottig, N.R., Wagner, T., Henry, E.N., Cheruvelil, K.S., Webster, K.E., Downing, J.A., Stow, C.A. (2014). Long-term citizen-collected data reveal geographical patterns and temporal trends in lake water clarity." PLOS ONE 9: e95769. https://doi.org/10.1371/journal.pone.0095769.

Ludovisi, A., & Gaino, E. (2010). Meteorological and water quality changes in Lake Trasimeno (Umbria, Italy) during the last fifty years. Journal of Limnology 69: 174. https://doi.org/10.4081/jlimnol.2010.174

Luger, M., Kammerlander, B., Blatterer, H., Gassner, H. (2021). Von der Eutrophierung in die Klimaerwärmung–45 Jahre limnologisches Monitoring Mondsee. Österreichische Wasser-und Abfallwirtschaft 73: 418-425.

Lyche Solheim, A., Schartau, A.K., Bongard, T., Bækkelie, K.A.E., Dokk, J.G., Edvardsen H., Fosholt Moe, T., Gjelland, K.Ø., Hobæk, A., Håvardstun, J., Jensen, T.C., Mjelde, M., Persson, J., Sandlund, O.T., Skjelbred, B., Walseng, B. 2017. ØKOSTOR 2017: Basisovervåking av store innsjøer (2017). Utprøving av metodikk for overvåking og klassifisering av økologisk tilstand iht vannforskriften. Surveillance monitoring of large lakes 2017. Testing of methodology for monitoring and classification of ecological status according to the WFD. Miljødirektoratet-rapport 1086/2018, NIVA-rapport 7287: 193 s. m1086.pdf (miljodirektoratet.no)

Lyche Solheim, A., Globevnik, L., Austnes, K., Kristensen, P., Moe, S. J., Persson, J., ... & Birk, S. (2019). A new broad typology for rivers and lakes in Europe: Development and application for large-scale environmental assessments. Science of the Total Environment 697: 134043. https://doi.org/10.1016/j.scitotenv.2019.134043

Lyche Solheim, A., J. Moe, J.-E. Thrane, L. Globevnik, L. Snoj, J. Völker, S. Korpinen, J. Attila, L. Hoikkala, J. Franco, A. Borja, D. Vaughan, S. Birk. (2021). Aquatic biological indicators: Feasibility of detecting trends and linking them to multiple pressures and measures. Scoping paper including case studies. Technical Report. European Environment Agency. 90 pp.

Machova, J., Faina, R., Randak, T., Valentova, O., Steinbach, C., Kroupova, H. K., Svobodova, Z. (2017). Fish death caused by gas bubble disease: a case report. Veterinární medicína 62: 231-237. DOI: 10.17221/153/2016-VETMED

Malcolm, I. A., Bacon, P. J., Middlemas, S. J., Fryer, R. J., Shilland, E. M., Collen, P. (2014). Relationships between hydrochemistry and the presence of juvenile brown trout (Salmo trutta) in headwater streams recovering from acidification. Ecological Indicators 37: 351-364. https://doi.org/10.1016/j.ecolind.2012.02.029

McFarland, B., Carse, F. Sandin, L. (2010). Littoral macroinvertebrates as indicators of lake acidification within the UK. Aquatic Conservation: Marine and Freshwater Research 20: 105-116.

McPherson, M. L., Finger, D. J., Houskeeper, H. F., Bell, T. W., Carr, M. H., Rogers-Bennett, L., & Kudela, R. M. (2021). Large-scale shift in the structure of a kelp forest ecosystem co-occurs with an epizootic and marine heatwave. Communications biology 4: 298. https://doi.org/10.1038/s42003-021-01827-6

Milhorance, C., Le Coq, J. F., & Sabourin, E. (2021). Dealing with cross-sectoral policy problems: An advocacy coalition approach to climate and water policy integration in Northeast Brazil. Policy Sciences, 54(3), 557-578.

Miyazono, S., Reynaldo P. & Taylor, C.M. (2015). Desertification, salinization, and biotic homogenization in a dryland river ecosystem. Science of the Total Environment 511, 444-453.

Mladenović-Ranisavljević, I. I., Takić, L., & Nikolić, Đ. (2018). Water quality assessment based on combined multi-criteria decision-making method with index method. Water Resources Management 32: 2261-2276. https://doi.org/10.1007/s11269-018-1927-3

Mo, Y., Peng, F., Gao, X., Xiao, P., Logares, R., Jeppesen, E., Ren, K., Xue,Y. & Yang, J. (2021). Low shifts in salinity determined assembly processes and network stability of microeukaryotic plankton communities in a subtropical urban reservoir. Microbiome 9: 128. https://doi.org/10.1186/s40168-021-01079-w.

Moe, S.J., Schartau, S., Bækken, A.K., Mcfarland, B. (2010), Assessing macroinvertebrate metrics for classifying acidified rivers across northern Europe. Freshwater Biology 55: 1382-1404. https://doi.org/10.1111/j.1365-2427.2010.02413.x

Monteith, D. T., Henrys, P. A., Hruška, J., de Wit, H. A., Krám, P., Moldan, F., ... & Evans, C. D. (2023). Long-term rise in riverine dissolved organic carbon concentration is predicted by electrolyte solubility theory. Science Advances, 9(3), eade3491. DOI: 10.1126/sciadv.ade3491

Moss, B., Kosten, S., Meerhoff, M., Battarbee, R. W., Jeppesen, E., Mazzeo, N., ..., Scheffer, M. (2011). Allied attack: climate change and eutrophication. Inland waters 1: 101-105. DOI: 10.5268/IW-1.2.359

Moyano Salcedo, A.J., Estévez, E., Salvadó, H., Barquín, J., & Cañedo-Argüelles, M. (2022). Human activities disrupt the temporal dynamics of salinity in Spanish rivers. Hydrobiologia, 1-16.

OECD (1982). Eutrophication of Waters. Monitoring, Assessment and Control. Organisation for Economic Cooperation and Development, Paris.

OSPAR, 2005. Revised common procedure for the identification of the eutrophication status of the OSPAR Maritime Area. Ref. Numb. 2005-3. OSPAR Commission. 248026.pdf

Palmer, M. A., Reidy Liermann, C. A., Nilsson, C., Flörke, M., Alcamo, J., Lake, P. S., & Bond, N. (2008). Climate change and the world's river basins: anticipating management options. Frontiers in Ecology and the Environment, 6(2), 81-89. https://doi.org/10.1890/060148

Paradise, T. A. (2009). The sublethal salinity tolerance of selected freshwater macroinvertebrate species. Master thesis, RMIT University, Melbourne, Australia.

Pardo, I., Gómez-Rodríguez, C., Wasson, J. G., Owen, R., van de Bund, W., Kelly, M., ... & Ofenböeck, G. (2012). The European reference condition concept: a scientific and technical approach to identify minimally-impacted river ecosystems. Science of the Total Environment 420: 33-42. https://doi.org/10.1016/j.scitotenv.2012.01.026

Pareeth, S., Bresciani, M., Buzzi, F., Leoni, B., Lepori, F., Ludovisi, A., Morabito, G., Adrian, R., Neteler, M., & Salmaso, N. (2017). Warming trends of perialpine lakes from homogenised time series of historical satellite and in-situ data. Science of The Total Environment: 578: 417–426. https://doi.org/10.1016/j.scitotenv.2016.10.199

Phillips, G., 2013. Analysis of Norwegian lake data to support the development of Water Framework Directive boundary criteria for phosphorus, nitrogen and water transparency. Marine Science Support draft report 14th April 2013. 37 pp.

Phillips, G., 2023. Establishing supporting element standards, A revised approach. Report to ECOSTAT, 19pp.

Phillips, G., Kelly, M., Teixeira, H., Salas, F., Free, G., Leujak, W., Pitt, J.-A., Lyche Solheim, A., Varbiro, G., Poikane, S. (2018), Best practice for establishing nutrient concentrations to support good ecologicalstatus, EUR 29329

EN, Publications Office of the European Union, Luxembourg, 2018, ISBN 978-92-79-92906-9, doi:10.2760/84425, JRC112667.

Phillips, G., Teixeira, H., Poikane, S., Herrero, F. S., & Kelly, M. G. (2019). Establishing nutrient thresholds in the face of uncertainty and multiple stressors: a comparison of approaches using simulated datasets. Science of the Total Environment 684: 425-433. doi: 10.1016/j.scitotenv.2019.05.343.

Phillips, P., Free, G., Kelly, M.G., Varbiro, G. & Teixeira, H. (2023). Establishing supporting element standards; a revised approach. Report to ECOSTAT by the Task Group on Supporting Physico-Chemcial Elements.

Piscart, C., Moreteau, J.C. & Beisel, J.N., (2005). Biodiversity and structure of macroinvertebrate communities along a small permanent salinity gradient (Meurthe River, France). Hydrobiologia, 551, pp.227-236.

Pistocchi, A., Bleninger, T., Dorati, C. (2020). Screening the hurdles to sea disposal of desalination brine around the Mediterranean. Desalinisation 491: 114570. https://doi.org/10.1016/j.desal.2020.114570

Piatt, J. F., Parrish, J. K., Renner, H. M., Schoen, S. K., Jones, T. T., Arimitsu, M. L., ... & Sydeman, W. J. (2020). Extreme mortality and reproductive failure of common murres resulting from the northeast Pacific marine heatwave of 2014-2016. PloS one 15: e0226087.DOI: 10.1371/journal.pone.0226087

Pittock, J., Hussey, K. & McGlennon, S. (2013). Australian climate, energy and water policies: conflicts and synergies. Australian Geographer, 44(1), 3-22.

Poikane, S., Kelly, M. G., Herrero, F. S., Pitt, J. A., Jarvie, H. P., Claussen, U., ... & Phillips, G. (2019a). Nutrient criteria for surface waters under the European Water Framework Directive: Current state-of-the-art, challenges and future outlook. Science of the Total Environment 695: 133888.https://doi.org/10.1016/j.scitotenv.2019.133888

Poikane, S., Phillips, G., Birk, S., Free, G., Kelly, M. G., & Willby, N. J. (2019b). Deriving nutrient criteria to support 'good' ecological status in European lakes: An empirically based approach to linking ecology and management. Science of the Total Environment, 650, 2074–2084.

Poikane, S., Salas Herrero, F., Kelly, M.G., Borja, A., Birk, S. & van de Bund, W. (2020). European aquatic ecological assessment methods: A critical review of their sensitivity to key pressures. Science of the Total Environment, 740, 140075.

Posthuma, L., van Gils, J., Zijp, M. C., van de Meent, D. & de Zwart, D. (2019). Species sensitivity distributions for use in environmental protection, assessment, and management of aquatic ecosystems for 12 386 chemicals. Environmental Toxicology and Chemistry 38: 905-917. https://doi.org/10.1002/etc.4373.

Preisendorfer, R. W. (1986). Secchi disk science: visual optics of natural waters 1. Limnology and Oceanography 31: 909-926.

Premazzi, G., Dalmiglio, A., Cardoso, A. C., & Chiaudani, G. (2003). Lake management in Italy: The implications of the Water Framework Directive. Lakes & Reservoirs: Research & Management 8: 41–59. DOI:10.1046/j.1440-1770.2003.00210.x

Prygiel, J. & Coste, M. (1998). Progress in the use of diatoms for monitoring rivers in France. pp.165-179In J. Prygiel, B.A. Whitton & J. Bukowska (Ed.) Use of Algae to Monitor Rivers III (). Douai: Agence de l'Eau Artois-Picardie.

Quevauviller, Q. (2011). Adapting to climate change: reducing water-related risks in Europe – EU policy and research considerations. Environmental Science and Policy 14: 722-729. https://doi.org/10.1016/j.envsci.2011.02.008

Reed, J.M. (1998). A diatom-conductivity transfer function for Spanish salt lakes. Journal of Paleolimnology, 19(4), 399-416.

Regione del Veneto. (2018). Stastical Report Veneto. Regione del Veneto. http://statistica.regione.veneto.it/ENG/Pubblicazioni/RapportoStatistico2018/Statistical_Report_2018.pdf [accessed: 06/04/2023]

Remy, M., Hillebrand, H., & Flöder, S. (2017). Stability of marine phytoplankton communities facing stress related to global change: Interactive effects of heat waves and turbidity. Journal of Experimental Marine Biology and Ecology 497: 19-229. https://doi.org/10.1016/j.jembe.2017.10.002

Reynolds, S. A. & Aldridge, D. C. (2021). Global impacts of invasive species on the tipping points of shallow lakes. Global Change Biology 27: 6129-6138. https://doi.org/10.1111/gcb.15893

Richardson, J., Feuchtmayr, H., Miller, C., Hunter, P. D., Maberly, S. C., & Carvalho, L. (2019). Response of cyanobacteria and phytoplankton abundance to warming, extreme rainfall events and nutrient enrichment. Global Change Biology 25: 3365-3380. https://doi.org/10.1111/gcb.14701

Roberts, S. D., Van Ruth, P. D., Wilkinson, C., Bastianello, S. S., & Bansemer, M. S. (2019). Marine heatwave, harmful algae blooms and an extensive fish kill event during 2013 in South Australia. Frontiers in Marine Science 6: 610. https://doi.org/10.3389/fmars.2019.00610

Robins, P. E., Skov, M. W., Lewis, M. J., Giménez, L., Davies, A. G., Malham, S. K., ..., Jago, C. F. (2016). Impact of climate change on UK estuaries: A review of past trends and potential projections. Estuarine, Coastal and Shelf Science 169: 119-135. https://doi.org/10.1016/j.ecss.2015.12.016

Rogora, M., Buzzi, F., Dresti, C., Leoni, B., Lepori, F., Mosello, R., Patelli, M., & Salmaso, N. (2018). Climatic effects on vertical mixing and deep-water oxygen content in the subalpine lakes in Italy. Hydrobiologia 824: 33–50. https://doi.org/10.1007/s10750-018-3623-y

Rossaro, B., Marziali, L., Cardoso, A. C., Solimini, A., Free, G., & Giacchini, R. (2007). A biotic index using benthic macroinvertebrates for Italian lakes. Ecological Indicators 7: 412–429. https://doi.org/10.1016/j.ecolind.2006.04.002

Rotter, S., Heilmeier, H., Altenburger, R. & Schmitt, M. (2013). Multiple stressors in periphyton – comparison of observed and predicted tolerance responses to high ionic loads and herbicide exposure. Journal of Applied Ecology 50: 1459-1468. https://doi.org/10.1111/1365-2664.12146

Salmaso, N., Boscaini, A., Capelli, C., & Cerasino, L. (2018). Ongoing ecological shifts in a large lake are driven by climate change and eutrophication: Evidences from a three-decade study in Lake Garda. Hydrobiologia 824: 177–195. https://doi.org/10.1007/s10750-017-3402-1

Sala, M., Faria, M., Sarasúa, I., Barata, C., Bonada, N., Brucet, S., Llenas, L., Ponsá, S., Prat, N., Soares, A.M & Cañedo-Arguelles, M. (2016). Chloride and sulphate toxicity to Hydropsyche exocellata (Trichoptera, Hydropsychidae): exploring intraspecific variation and sub-lethal endpoints. Science of the Total Environment, 566, 1032-1041.

Sánchez-Lizaso, J. L., Romero, J., Ruiz, J., Gacia, E., Buceta, J. L., Invers, O., ... Manzanera, M. (2008). Salinity tolerance of the Mediterranean seagrass Posidonia oceanica: recommendations to minimize the impact of brine discharges from desalination plants. Desalination 221: 602–607. https://doi.org/10.1016/J.DESAL.2007.01.119

Sandén, P., & Håkansson, B. (1996). Long-term trends in Secchi depth in the Baltic Sea. Limnology and Oceanography 41: 346-351.

Sandoval-Gil, J. M., Ruiz, J. M., & Marín-Guirao, L. (2023). Advances in understanding multilevel responses of seagrasses to hypersalinity. Marine Environmental Research, 183, 105809. https://doi.org/10.1016/j.marenvres.2022.105809

Schaefer, D. A., Driscoll, C. T., Van Dreason, R., Yatsko, C. P. (1990). The episodic acidification of Adirondack Lakes during snowmelt. Water Resources Research 26: 1639– 1647. doi:10.1029/WR026i007p01639.

Schallenberg, M., Hall, C.J. & Burns, C.W. (2003). Consequences of climate-induced salinity increases on zooplankton abundance and diversity in coastal lakes. Marine Ecology Progress Series, 251, 181-189.

Schneider, S. C., & Skarbøvik, E. (2022). Ecological status assessment of clay rivers with naturally enhanced water phosphorus concentrations. Environmental Advances, 9, 100279. https://doi.org/10.1016/j.envadv.2022.100279

Schiel, D. R., Lilley, S. A., South, P. M. (2018). Ecological tipping points for an invasive kelp in rocky reef algal communities. Marine Ecology Progress Series 587: 93-104. DOI:10.3354/meps12429

Schmidt S., Diallo Il., Derriennic H., Fallou H., Lepage M. (2019). Exploring the Susceptibility of Turbid Estuaries to Hypoxia as a Prerequisite to Designing a Pertinent Monitoring Strategy of Dissolved Oxygen. Frontiers in Marine Science 6: 352. doi: 10.3389/fmars.2019.00352

Schuler, M.S., Cañedo-Argüelles, M., Hintz, W.D., Dyack, B., Birk, S. & Relyea, R.A. (2019). Regulations are needed to protect freshwater ecosystems from salinization. Philosophical Transactions of the Royal Society B, 374(1764), 20180019.

Schulz, C.-J. & Cañedo-Argüelles, M. (2019). Lost in translation: the German literature on freshwater salinization. Philosophical Transactions of the Royal Society B: Biological Sciences, 374, 1764. https://doi.org/10.1098/rstb.2018.0007.

Schröder, M., Sondermann, M., Sures, B. & Hering, D. (2015). Effects of salinity gradients on benthic invertebrate and diatom communities in a German lowland river. Ecological indicators, 57, 236-248.

Short, F.T., Kosten, S., Morgan, P.A., Malone, S. & Moore G.E. (2016). Impacts of Climate Change on Submerged and Emergent Wetland Plants. Aquatic Botany, 135, 3–17.

Siegfried, C., Auer, N.A. & Effler, S.W. (1996) Changes in the zooplankton of Onondaga Lake: causes and implications. Lake and Reservoir Management 12, 59–71.

Smith, D.G., Croker, G.F., McFarlane, K. (1995). Human perception of water appearance, 1. Clarity and colour for bathing and aesthetics. New Zealand Journal of Marine and Freshwater Research 29: 29-43. https://doi.org/10.1080/00288330.1995.9516637

Soga, M., & Gaston, K. J. (2018). Shifting baseline syndrome: causes, consequences, and implications. Frontiers in Ecology and the Environment 16: 222-230. https://doi.org/10.1002/fee.1794.

Søndergaard, M., Johansson, L.S., Levi, E., Olesen, A., & Davidson, T. (2019). Anvendelsen af fysisk-kemiske kvalitetselementer til understøttelse af økologisk tilstandsvurdering i søer. Aarhus Universitet, DCE – Nationalt Center for Miljø og Energi, 74 s. Scientific Report nr. 330 from DCE – Danish Centre for Environment and Energy http://dce2.au.dk/pub/SR330.pdf

Soucek, D.J. & Kennedy, A.J. (2005). Effects of hardness, chloride, and acclimation on the acute toxicity of sulfate to freshwater invertebrates. Environmental Toxicology and Chemistry, 24(5), 1204-1210.

Sowa, A., Krodkiewska, M. & Halabowski, D. (2020). How does mining salinisation gradient affect the structure and functioning of macroinvertebrate communities? Water, Air, & Soil Pollution, 231, 1-19.

Spears, B. M., Chapman, D., Carvalho, L., Rankinen, K., Stefanidis, K., Ives, S., Vuorio K., Birk, S. (2022). Assessing multiple stressor effects to inform climate change management responses in three European catchments. Inland Waters 12: 94-106. doi: 10.1080/20442041.2020.1827891

Steckbauer, A., Duarte, C. M., Carstensen, J., Vaquer-Sunyer, R., Conley, D. J. (2011). Ecosystem impacts of hypoxia: thresholds of hypoxia and pathways to recovery. Environmental Research Letters 6: 025003. DOI 10.1088/1748-9326/6/2/025003

Steinberg, C. E. (2003). Ecology of Humic Substances in Freshwaters. Springer, Berlin, Heidelberg.

Stoicescu, S. T., Lips, U., & Liblik, T. (2019). Assessment of eutrophication status based on sub-surface oxygen conditions in the Gulf of Finland (Baltic Sea). Frontiers in Marine Science, 6, 54. doi: 10.3389/fmars.2019.00054

Szklarek, S., Górecka, A., & Wojtal-Frankiewicz, A. (2022). The effects of road salt on freshwater ecosystems and solutions for mitigating chloride pollution - A review. Science of the Total Environment 805: 150289. https://doi.org/10.1016/j.scitotenv.2021.150289 .

Teixeira, H., Salas Herrero, M.F., Kelly, M., Philips, G., Solheim, A.L., Poikane, S. (2022). Physico-chemical supporting elements: transitional and coastal waters: a review of national standards to support good ecological status, EUR 30958 EN, Publications Office of the European Union, Luxembourg, 2022, ISBN 978-92-76-46489-1, doi:10.2760/07826, JRC128071.

Teixeira, H., et al. 2023. Report on supporting element thresholds in TRAC waters (draft report to ECOSTAT full citation to follow)

Tierney, D., Kelly-Quinn, M., Bracken, J.J. (1998) The faunal communities of upland streams in the eastern region of Ireland with reference to afforestation impacts. Hydrobiologia 389: 115-30. https://doi.org/10.1023/A:1003510016292

Trolle, D., Hamilton, D. P., Pilditch, C. A., Duggan, I. C. & Jeppesen, E. (2011). Predicting the effects of climate change on trophic status of three morphologically varying lakes: Implications for lake restoration and management. Environmental Modelling & Software 26: 354-370. https://doi.org/10.1016/j.envsoft.2010.08.009

UNEP (2020). UNEP GEMS/Water Capacity Development Centre (2020) SDG Indicator 6.3.2 Technical Guidance Document No. 2 Target Values. Available at: https://communities.unep.org/display/sdg632/Documents+and+Materials?preview=/32407814/38306400/CDC_GEMI2_TechDoc2_Targetvalues_20200508.pdf

Uriarte, A. & Borja, A. (2009). Assessing fish quality status in transitional waters, within the European Water Framework Directive: Setting boundary classes and responding to anthropogenic pressures. Estuarine, Coastal and Shelf Science 82: 214-224. https://doi.org/10.1016/j.ecss.2009.01.008

Velasco, J., Gutiérrez-Cánovas, C., Botella-Cruz, M., Sánchez-Fernández, D., Arribas, P., Carbonell, J. A., ... & Pallarés, S. (2019). Effects of salinity changes on aquatic organisms in a multiple stressor context. Philosophical Transactions of the Royal Society B, 374(1764), 20180011. https://doi.org/10.1098/rstb.2018.0011.

Vigiak, O., Grizzetti, B., Udias-Moinelo, A., Zanni, M., Dorati, C., Bouraoui, F., & Pistocchi, A. (2019). Predicting biochemical oxygen demand in European freshwater bodies. Science of the Total Environment 666: 1089-1105. https://doi.org/10.1016/j.scitotenv.2019.02.252

Vinca, A., Riahi, K., Rowe, A., & Djilali, N. (2021). Climate-land-energy-water nexus models across scales: progress, gaps and best accessibility practices. Frontiers in Environmental Science, 9, 691523.

Vosylienė, M. Z., Baltrėnas, P., & Kazlauskienė, A. (2006). Toxicity of road maintenance salts to rainbow trout Oncorhynchus mykiss. Ekologija 2: 15-20.

Weitkamp, D. E., Sullivan, R. D., Swant, T., DosSantos, J. (2003). Gas bubble disease in resident fish of the lower Clark Fork River. Transactions of the American Fisheries Society 132: 865-876. https://doi.org/10.1577/T02-026

Wernberg, T., Bennett, S., Babcock, R. C., De Bettignies, T., Cure, K., Depczynski, M., ..., Wilson, S. (2016). Climate-driven regime shift of a temperate marine ecosystem. Science 353: 169-172. DOI: 10.1126/science.aad8745

Wilby, R.L., Orr, H.G., Hedger, M., Forrow, D., Blackmore, M. (2006). Risks posed by climate change to the delivery of Water Framework Directive objectives in the UK. Environment International 32: 1043-1055. https://doi.org/10.1016/j.envint.2006.06.017

Wolfram, G., Argillier, C., de Bortoli, J., Buzzi, F., Dalmiglio, A., Dokulil, M. T., Hoehn, E., Marchetto, A., Martinez, P.-J., Morabito, G., Reichmann, M., Remec-Rekar, Š., Riedmüller, U., Rioury, C., Schaumburg, J., Schulz, L., & Urbanič, G. (2009). Reference conditions and WFD compliant class boundaries for phytoplankton biomass and chlorophyll-a in Alpine lakes. Hydrobiologia 633: 45–58. https://doi.org/10.1007/s10750-009-9875-9

Wolfram, G., Römer J., Hörl, C., Stockinger, W., Ruzicska, K. & Munteanu, A. (2014). Chloride Report. Impact on aquatic flora and fauna, with special consideration of the four biological quality elements according to the EU WFD [Chlorid-Studie. Auswirkungen auf die aquatische Flora und Fauna, mit besonderer Berücksichtigung der vier biologischen Qualitätselemente gemäß EU-WRRL]. Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, Wien (https://info.bml.gv.at/themen/wasser/wasserqualitaet/chloridstudie.html).

Woolway, R. I., Jennings, E., Carrea, L. (2020). Impact of the 2018 European heatwave on lake surface water temperature. Inland Waters 10: 322-332. DOI: 10.1080/20442041.2020.1712180

Woolway, R. I., E. Jennings, T. Shatwell, M. Golub, D. C. Pierson, Maberly, S. C. (2021). Lake heatwaves under climate change. Nature 589: 402–407. doi:10.1038/s41586-020-03119-1

Woolway, R. I., Weyhenmeyer, G. A., Schmid, M., Dokulil, M. T., de Eyto, E., Maberly, S. C., May, L., & Merchant, C. J. (2019). Substantial increase in minimum lake surface temperatures under climate change. Climatic Change 155: 81–94. https://doi.org/10.1007/s10584-019-02465-y

World Health Organization (2022). Guidelines for Drinking-Water Quality (4th ed., incorporating the first and second addenda). Geneva: World Health Organization.

Worrall, F. & T.P. Burt (2008). The effect of severe drought on the dissolved organic carbon (DOC) concentration and flux from British rivers. Journal of Hydrology 361: 262-274. https://doi.org/10.1016/j.jhydrol.2008.07.051

Yan, N. D. (1983). Effects of changes in pH on transparency and thermal regimes of Lohi Lake, near Sudbury, Ontario. Canadian Journal of Fisheries and Aquatic Sciences 40: 621-626. https://doi.org/10.1139/f83-081

Zhao, Q., Zhang, Y., Guo, F., Leigh, C. & Jia, X. (2021). Increasing anthropogenic salinisation leads to declines in community diversity, functional diversity and trophic links in mountain streams. Chemosphere, 263, p.127994.

Ziemann, H. (1999). Salzgehalt. Bestimmung des Halobienindex. pp. 309–313In: Tümpling, W., Friedrich, G. (Eds.), Biologische Gewässeruntersuchung. Jena.

Ziemann, H. & Schulz, C.-J. (2011). Methods for biological assessment of salt-loaded running waters – fundamentals, current positions and perspectives. Limnologica 41: 90-95. https://doi.org/10.1016/j.limno.2010.09.005

## List of abbreviations and definitions

| | |
|---|---|
| %sat | Percent oxygen saturation” |
| AA | Annual Average |
| AUC | Area Under Curve |
| ANC | Acid Neutralising Capacity |
| BOD | Biochemical Oxygen Demand |
| BQE | Biological Quality Element |
| CCC | Criteria Continuous Concentration |
| CDOM | Coloured Dissolved Organic Matter |
| Chla | chlorophyll a |
| CIS | Common Implementation Strategy |
| CMC | Criteria Maximum Concentration |
| CW | Coastal Waters |
| DO | Dissolved Oxygen |
| DOC | Dissolved Organic Carbon |
| EA | East Atlantic |
| ECOSTAT | A working group dedicated to the ecological status of surface water bodies within implementation of the Water Framework Directive that was set up in November 2002. |
| EEA | European Environment Agency |
| EQR | Ecological Quality Ratio |
| EU | European Union |
| FNU | Formazin Nephelometric Units |
| G20 | An intergovernmental forum comprising 19 sovereign countries, the European Union and the African Union |
| GES | Good Ecological Status |
| GIG | Geographical Intercalibration Group |
| G/M | Good status/Moderate status boundary |
| Helcom | Helsinki Commission (Baltic marine environment protection commission) |
| H/G | High status/Good status boundary |
| IC | Intercalibration |
| lcl | Lower Confidence Interval |
| ICP | Waters International Cooperative Programme on assessment and monitoring of the effects of air pollution on rivers and lakes. |
| MAC | Maximum Allowable Concentration |
| M/P | Moderate status/Poor status boundary |
| MSFD | Marine Strategy Framework Directive |
| NAO | North Atlantic Oscillation |
| NEA | North-East Atlantic |
| NTU | Nephelometric Turbidity Units |
| nEQR | Normalised EQR |

| | |
|---|---|
| OSPAR | Oslo-Paris Convention (Convention for the Protection of the Marine Environment of the North East Atlantic) |
| P/B | Poor status/Bad status boundary |
| PAR | Photosynthetically Available Radiation |
| PSU | Practical Salinity Units |
| SA | Seasonal Average |
| ucl | upper confidence interval |
| SE | Supporting Elements |
| SoE | State of Environment |
| SS | Suspended Solids |
| SSD | Species Sensitivity Distributions |
| TDS | Total Dissolved Solids |
| TN | Total Nitrogen |
| TP | Total Phosphorus |
| TRAC | Transitional And Coastal Waters |
| WFD | Water Framework Directive |
| WHO | World Health Organisation |
| WISE | Water Information System for Europe (WFD database) |

## List of boxes

## List of figures

## List of tables

## GETTING IN TOUCH WITH THE EU

**In person**

All over the European Union there are hundreds of Europe Direct centres. You can find the address of the centre nearest you online (european-union.europa.eu/contact-eu/meet-us_en).

**On the phone or in writing**

Europe Direct is a service that answers your questions about the European Union. You can contact this service:

— by freephone: 00 800 6 7 8 9 10 11 (certain operators may charge for these calls),

— at the following standard number: +32 22999696,

— via the following form: european-union.europa.eu/contact-eu/write-us_en.

## FINDING INFORMATION ABOUT THE EU

**Online**

Information about the European Union in all the official languages of the EU is available on the Europa website (european-union.europa.eu).

**EU publications**

You can view or order EU publications at op.europa.eu/en/publications. Multiple copies of free publications can be obtained by contacting Europe Direct or your local documentation centre (european-union.europa.eu/contact-eu/meet-us_en).

**EU law and related documents**

For access to legal information from the EU, including all EU law since 1951 in all the official language versions, go to EUR-Lex (eur-lex.europa.eu).

**Open data from the EU**

The portal data.europa.eu provides access to open datasets from the EU institutions, bodies and agencies. These can be downloaded and reused for free, for both commercial and non-commercial purposes. The portal also provides access to a wealth of datasets from European countries.

Science for policy

The Joint Research Centre (JRC) provides independent, evidence-based knowledge and science, supporting EU policies to positively impact society

**EU Science Hub**
joint-research-centre.ec.europa.eu

Publications Office
of the European Union