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OVERVIEW

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Warding off freshwater salinization: Do current criteria measure up?

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Abstract

Salinization is a global threat to freshwater habitats that has been intensified by climate change. Monitoring, assessment and management of salinity is therefore essential. The first step is to set criteria that are sufficiently stringent to protect ecosystem health. However, many countries have not yet defined criteria, and there are substantial differences between criteria. This has been noted in the EU, where salinity is a required "supporting element" for ecological status in inland waters but also for implementation of UN Sustainable Development Goal (SDG) indicator 6.3.2. for "good ambient water quality" where different approaches and widely different threshold values were reported for salinity criteria. Much of this information has not been published and is difficult to access, hindering further efforts to address the problem. We first discuss the implications of salinization for freshwater ecological health. We go on to discuss the principles and guidelines on how salinity criteria to protect ecology should be established. Next, we review salinity criteria submitted as part of implementation of SDG indicator 6.3.2 and the EU Water Framework Directive. Finally, we discuss setting salinity thresholds in an already-warming world and the challenges facing anyone trying to develop salinity criteria to protect freshwater ecosystems.

This article is categorized under:

Water and Life > Stresses and Pressures on Ecosystems

Water and Life > Conservation, Management, and Awareness

Science of Water > Water Quality

KEYWORDS

lakes, rivers, salinization, salinity, water quality criteria

1 | INTRODUCTION

Salinization, 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 (Figure 1) and for the ecosystem services that they provide (Cunillera-Montcusí et al., 2022; Jeppesen et al., 2020; Kaushal et al., 2018). Salinization also influences physical and physico-chemical processes such as thermal stratification and hypolimnetic oxygen dynamics (Ficker et al., 2019; Ladwig et al., 2023; Radosavljevic et al., 2022). It is also a direct pressure defined in Annex V of the Water Framework Directive (EU, 2000) albeit one for which relatively few countries in the European Union have standards designed to protect good ecological status (Kelly, Phillips, Teixeira, Salas Herrero, et al., 2022). 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).

Several pressures can give rise to salinization of inland waters (Bolen, 2020; Prosser et al., 2017; Szklarek et al., 2022; Thorslund et al., 2021), some of which are linked to global warming. Many countries in the Global South are predicted to receive less precipitation in the coming decades, meaning less runoff, potentially leading to salinization 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).

One result of this is an upward trend in salinization (Figure 2; see also Kaushal et al., 2021) 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 (Fritz et al., 1999; Reed, 1998) and, more recently, a growing awareness that salinization of inland waters is a widespread and inevitable outcome of global warming (Cañedo-Argüelles et al., 2013; Cunillera-Montcusí et al., 2022; Kaushal et al., 2021; Short et al., 2016). There are, however, considerable gaps in our knowledge of how this will affect ecosystems and how it interacts with other stressors. Also missing is a consensus on how the impact of salinization should be addressed by regulators to ensure sustainable water resources. Which chemical parameters should be used to set criteria? What are the key thresholds?

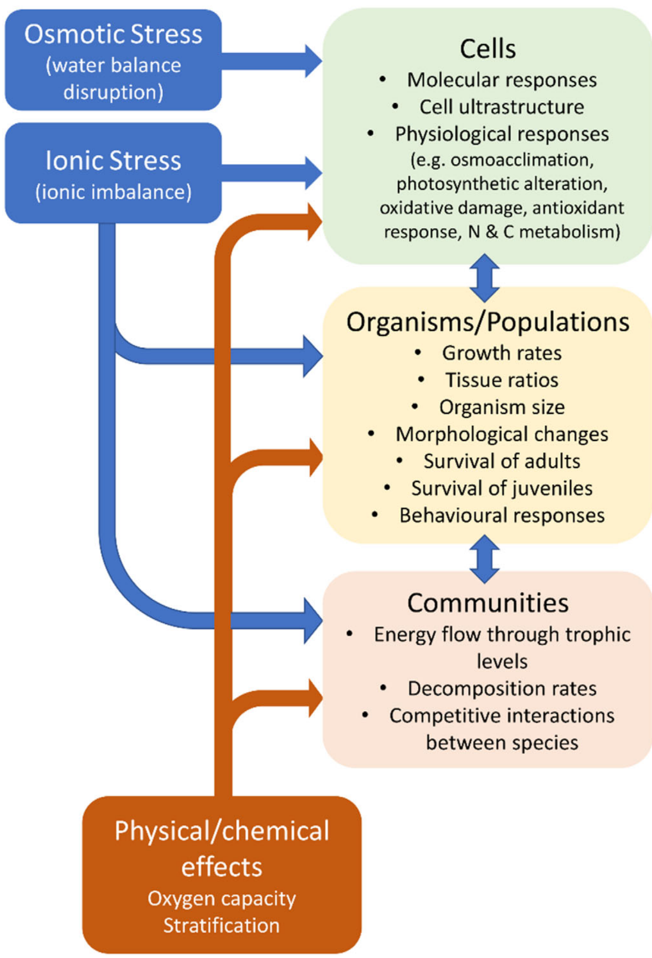

FIGURE 1 Interactions between stresses and biological processes which determine ecological responses to salinization of inland waters. Adapted from Sandoval-Gil et al. (2023).

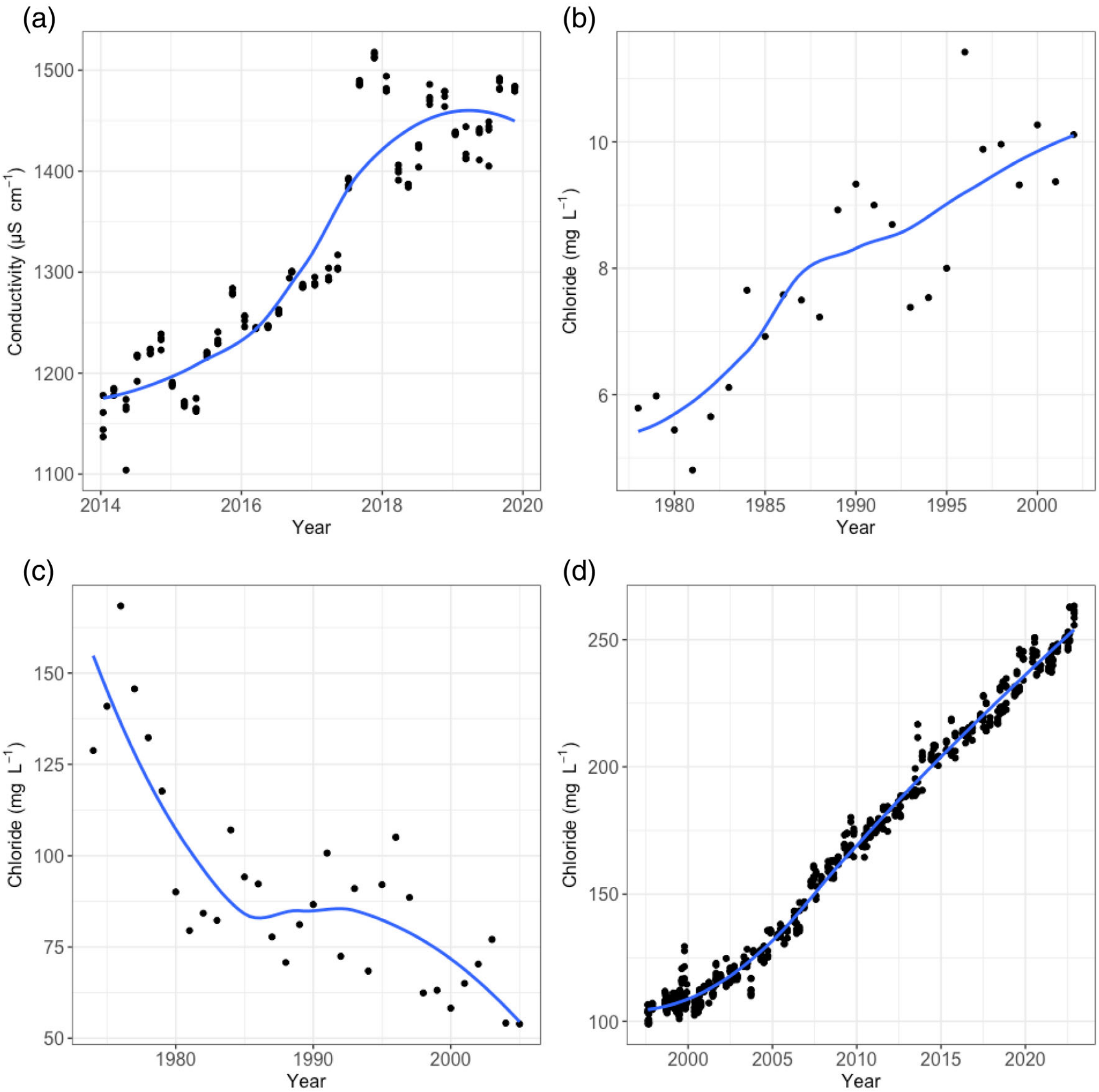

FIGURE 2 Trends in conductivity and chloride concentrations in four contrasting European waterbodies: (a) Lago di Trasimeno, central Italy (data from ARPA, Umbria); (b) River Rhône, Switzerland (GEMstat); (c) River Mersey, UK (GEMstat); (d) groundwater-fed lake near Vienna located close to a busy road (G. Wolfram, unpublished data). The declining trend in the River Mersey reflects a combination of improvements in water treatment and closures and relocation of many of the traditional industries of the region.

How well do criteria established for other purposes (irrigation, drinking water) protect ecology? Do these thresholds need adjustment if other major pressures such as high nutrient concentrations 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). Here we review the current situation with the objective of generating some guiding principles. We have used both peer-reviewed literature and national regulations and guidelines to build a picture of how salinity criteria to protect ecology should be established. We also consider how this is affected by capacity of national organizations to monitor and assess their freshwaters. How much of what a high-income country considers necessary can be stripped away to give a workable approach for low and middle-income countries with greater resource constraints and larger capacity gaps? This is important in light of the United Nation's inclusion of salinity as part of Indicator 6.3.2 for their Sustainable Development Goals (discussed below).

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. For instance, the United Nations Sustainable Development Goal 6 (SDG 6) aims "to ensure availability and sustainable management of water and sanitation for all" including global access to safe drinking water (SDG target 6.1) and good ambient water quality (SDG target 6.3). However, "safe drinking water" and "good ambient water quality" need to be expressed in tangible and practical terms. Each criterion combines an appropriate parameter (e.g., "conductivity"), metric (e.g., "annual mean") and threshold (e.g., "1000 μS cm^{-1}"; Poikane, Kelly, et al., 2019). 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, prioritize those with the greatest needs, design restoration strategies and measure progress toward 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 fall into two categories:

1. Use-related, 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 (Figure 3).
2. Ecology-related, intended to protect biodiversity and aquatic life. In this case, criteria are based on the ecosystem response to increased salinity. Different biological communities have different sensitivities, and the criteria are usually derived to protect the most sensitive components (Hart et al., 1991).

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 (e.g., South African Water Quality Guidelines on Industrial Use) or at regional or international levels (WHO Guidelines for Drinking Water Quality). The World Health Organization, for example, has established international water quality guidelines for

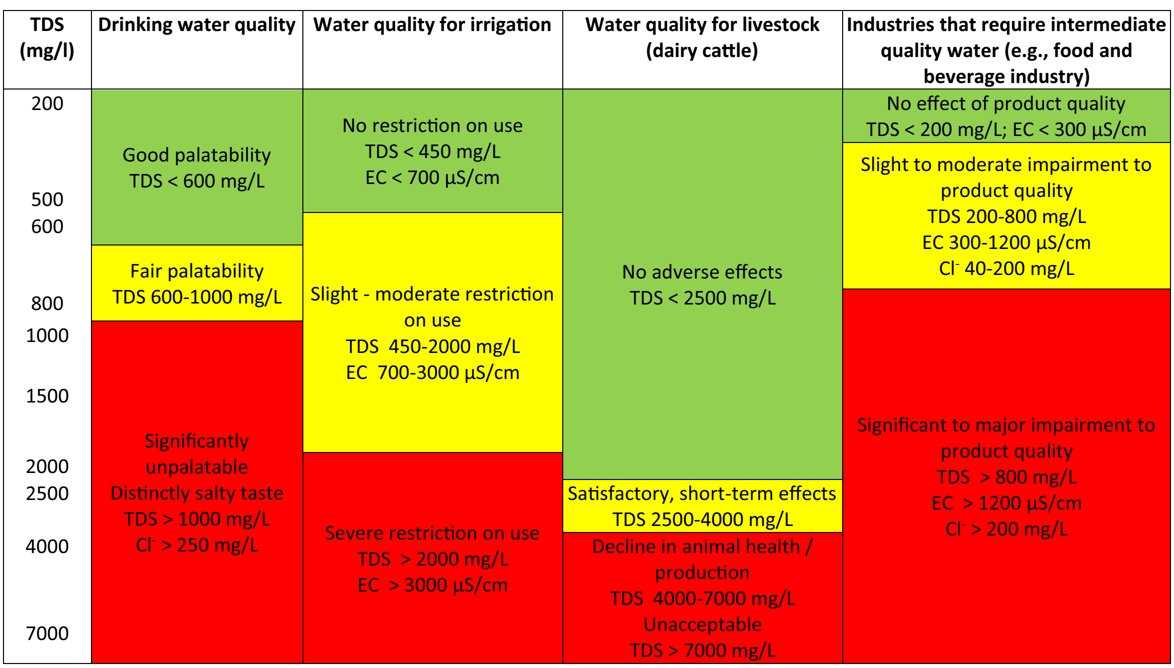

FIGURE 3 Overview of the effect of salinity on water uses. Green = conditions will support the use; yellow = some effects may be observed; red = likely to be detrimental to use. EC = electric conductivity, TDS = total dissolved solids, Cl^- = chloride concentration. Based on data from WHO (2022; drinking water quality), Ayers and Westcot (1985); irrigation), DWAF (1996); industrial water use), ANZECC (2000); water for livestock).

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 (Figure 3). Similarly, the authoritative international water quality guidelines for agricultural uses were developed for the Food and Agricultural Organization (Ayers & Westcot, 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:

1. Ecosystem complexity, leading to limited understanding of their functioning and difficulties in establishing causal relationships between increasing salinization and ecological effects;
2. Other pressures (nutrients, hydromorphology, invasive species) are often seen as a greater priority for policymakers (Poikane et al., 2020); and,
3. 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 (Carvalho et al., 2019; Milhorance et al., 2021; Pittock et al., 2013) 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. Even where criteria are published, there are disagreements about their effectiveness. 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 salinization on inland waters continue to be reported (Table 1), in some cases deserving to be considered as ecological disasters (Free et al., 2023). This emphasizes the need for "safe operating limits" for salinity to be defined (or reassessed) in order to protect ecosystems from human-induced salinization.

3 | HOW CAN SALINITY CRITERIA BE ESTABLISHED?

3.1 | Starting point: The art of the possible

Salinization of inland waters is a worldwide problem requiring solutions that can be applied across the whole spectrum of geographic and socio-economic circumstances. A lack of awareness around salinization issues is also compounded in many countries by limited capacity to monitor and assess water quality.

Any guidelines for setting suitable criteria to protect inland water quality should acknowledge the limited capacity of many low-income countries, and any proposals to develop criteria must incorporate suitable flexibility with various entry points based on a nation's capacity. Solutions that work in a country that routinely collects large volumes of data at high spatial and temporal scales for numerous parameters might be impracticable in countries that struggle to routinely collect data for a limited list of parameters for key water bodies only. Additionally, many low-income countries have limited or inconsistent historical water quality datasets, meaning that efforts to determine baseline conditions and trends will be much more difficult than in those that can draw upon long-term and robust datasets.

Circumstances may also differ amongst countries with an advanced monitoring and assessment capacity due to geographical differences: a sampling frequency that is practicable in a small country such as Denmark or Ireland might represent an unaffordable logistical challenge in some regions of a large country such as Canada or Australia.

A final point is that salinity-related problems do not respect national boundaries, with the recent disaster on the River Oder serving as a case in point (Free et al., 2023). The lower part of this river forms the border between Poland and Germany and management of the problem required co-operation between the two countries, which includes having a shared understanding of the ecological thresholds of concern.

3.2 | Parameters

Salinity is the sum of the concentrations of all major ions; however, the principal ions driving salinization depend on local sources. Typically, when derived from seawater encroachment or salt spreading on roads as well as in coal mining

TABLE 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 506–5220 μS cm^{-1} Cl^- 39–1500 mg L^{-1} SO_4^{2-} 61–343 mg L^{-1}	Arle & Wagner, 2013 Braukman & 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} EC 277–3422 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} TDS 100–21,100 mg L^{-1} Cl^- 8–7292 mg L^{-1}	Sowa et al., 2020
Oder river, Germany /Poland	Phytoplankton	Massive blooms of *Prymnesium parvum*, massive fish kills	EC 470–7290 μS cm^{-1}	Free et al., 2023; Kolada, 2022
Rio Grande, USA/Mexico	Fish fauna	Functional and taxonomic homogenization of fish fauna	EC 1160–3440 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 Degradation of benthic community	EC 766–5439 μS cm^{-1} Cl^- 46–1490 mg L^{-1} 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 >3134 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 1400–132,400 μS cm^{-1}	Ladrera et al., 2017

Abbreviations: EC, electric conductivity; Cl^-, chloride; SO_4^{2-}, sulphate concentration; TDS, total dissolved solids.

wastewaters where pyrite oxidation occurs in the presence of halite (Rinder et al., 2020) sodium and chloride are the dominant ions. However, there are also situations where other ions are important (e.g., coal mining regions of Appalachia, USA: Kunz et al., 2013). 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, Hawkins, et al., 2016; Schuler et al., 2019). However, measuring all ions on a routine basis across all sites in a monitoring network may be impractical, especially when budgets are tight, so 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.

1. *Total dissolved solids* (TDS) is the most direct measure of salinization. Ideally, TDS is measured by gravimetry; however, it is sufficiently closely related to conductivity that it is often inferred from a conversion factor (Flanagan, 1990). This, however, raises questions about why a straightforward conductivity measurement cannot be used.
2. *Conductivity* is an easy measurement to make, with robust and relatively cheap instruments available. It is, as a result, widely used and is the parameter recommended for SDG Indicator 6.3.2. The principal issue with conductivity is that measurements also reflect the influence 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.
3. In many respects, *chloride concentration* is a better measure of salinization 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 given either a general (Herbert et al., 2015) or a locally specific (Schulz & Cañedo-Argüelles, 2019) understanding of their relationship. Chloride concentration is also unsuitable as a salinity measure where other ions predominate (e.g., Buchwalter et al., 2019; Soucek & Kennedy, 2005) and there are situations where these other ions should be measured (Cañedo-Argüelles, Hawkins, et al., 2016; Schuler et al., 2019).
4. *Practical Salinity Units* (PSU) are derived from the practical salinity scale developed by oceanographers and are 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 recognizes 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 (Clements & Kotalik, 2016; Kunz et al., 2013). 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 is unsuitable in inland waters affected by other salts. Besides, it is unlikely that benefits of using chloride concentration as a salinity measure outweigh advantages of moving from historical use of conductivity.

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 rather than central tendencies. An annual mean based on two or three samples—as is often only possible in countries with limited financial resources—is of limited value but may still provide valuable information 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 roads, and thus provides an opportunity to use upper percentiles for the assessment of acute toxicity due to short-term exposure.

A comparison of salinity criteria used in different countries gives a very heterogeneous picture (Table 4, S1 and S2), 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 (Arle & Wagner, 2013; CCME, 2011; Vosylienė et al., 2006). Some countries use both annual mean and maximum values (Austria for rivers), others have included only the 90th percentile (Lithuania) or seasonal averages (the Netherlands) in their regulations. Canada and Austria differentiate between chronic and acute exposure. The ratio of acute and chronic exposure derived from bioassays varies between roughly two and eight (Elphick et al., 2011; Hassell et al., 2006; Paradise, 2009).

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 (Marcé et al., 2016) may provide better protection for aquatic life than isolated 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).

3.4 | Approaches to determining thresholds

3.4.1 | Use of field data

A well-established protocol exists for deriving ecological metrics from community turnover 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., Kelly, Phillips, Teixeira, Várbíró, et al., 2022; Poikane, Phillips, et al., 2019).

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). However, when salinity thresholds for rivers in Europe were compared with actual monitoring data, biological impacts appeared to occur at relatively low chloride concentrations (Figure 4) probably because the metrics used in these assessments were evaluating the overall condition of the biota and salinity parameters were very rarely the dominant stressor in these rivers.

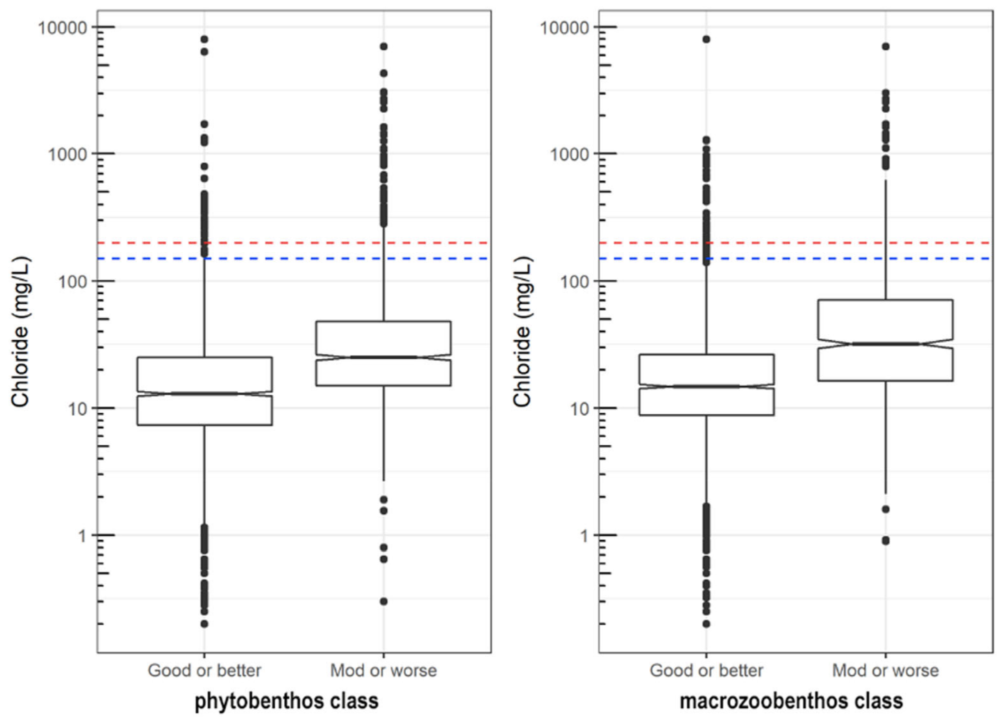

FIGURE 4 River chloride standards used in the EU (dotted lines) overlain on box plots showing the range of chloride concentrations of sites classified by phytobenthos and macro-invertebrates. (90th percentile = red, median = blue). Mod = moderate. From Kelly, Phillips, Teixeira, Salas Herrero, et al. (2022).

Similarly, Sundermann et al. (2015) detected a critical threshold of chlorides at 25 mg L^{-1} even though chlorides were unlikely to have direct effect on benthic invertebrates at these concentrations and, most likely, the deterioration of the community was caused by other stressors acting simultaneously. Such interactions are a major impediment when using biological data to set thresholds.

Most of the indices proposed for salinity assessment focus on species turnover and diversity. Low 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 (Figure 5: 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 pressure to 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. Challenges when assessing the biological effects of salinity are discussed in more detail by Ziemann and Schulz (2011).

3.4.2 | Use of ecotoxicology

The challenges involved in assessing the scale of salinity impacts from field data mean that these 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 toward using information from many species (e.g., Kefford et al., 2023). In particular, broadly applicable thresholds can be set using species sensitivity distributions (SSDs: Figure 6). 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, behavioral changes (Hassell et al., 2006; Hoover et al., 2013; Leite et al., 2022) otherwise the SSD approach may underestimate the whole range of effects of salinization on the ecosystem (Cañedo-Argüelles, Sala, et al., 2016). 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).

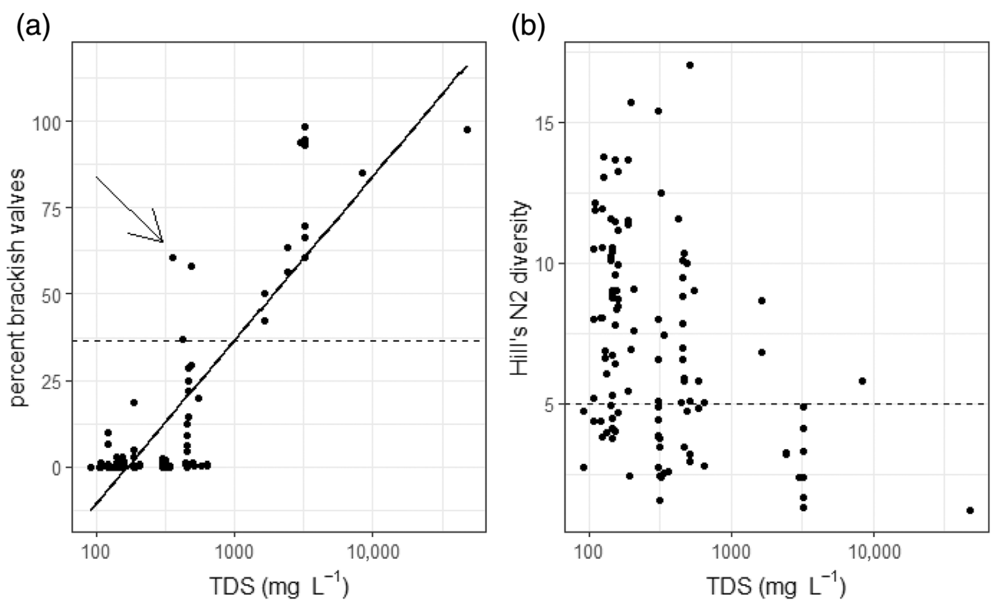

FIGURE 5 Relationship between (a) percentage of diatom taxa sensitive to brackish conditions and TDS in Greek lakes; dashed line = 36% brackish taxa (corresponding to 1000 mg L^{-1} TDS); arrow points to two outliers caused by fluctuations in salinity levels; (b) Hill's N2 diversity and TDS; dashed line shows N2 = 5, corresponding to the 10th percentile of N2 diversity for lakes at "good status." From Kelly et al. (2023).

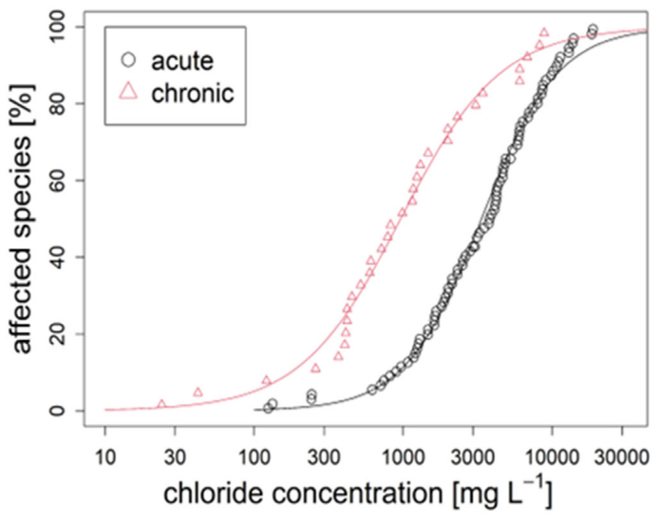

FIGURE 6 A species sensitivity distribution plot summarizing results of single-species tests for acute (n = 83) and chronic (n = 32) toxicity, based on data in Wolfram et al. (2014). The Austrian Water Quality Guideline was set at concentrations corresponding to the intersection with the 5th percentile of test data.

While SSDs 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. Hintz et al. (2022), for example, demonstrated that chloride thresholds established in Canada (120 mg L^{-1}) and the United States (230 mg L^{-1}) do not adequately protect zooplankton communities and, hence, the entire food web is at risk. Similar conclusions were reached by Arnott et al. (2020) based on laboratory experiments on *Daphnia* and from examining cladoceran assemblages of lakes, and by Clements and Kotalik (2016) based on stream mesocosm experiments quantifying the effects of major ions on benthic macroinvertebrates. 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 (Jeremias et al., 2018; Sala et al., 2016).

Belanger et al. (2017) emphasize the importance of 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 emphasize 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 salinization 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 evaluated with great care, taking account of water body types, geological background, and geographical differences of the taxa used.

4 | ECOLOGY-RELATED SALINITY CRITERIA

Information on how countries set salinity criteria is difficult to find, but "expert judgment" is widely used for other stressors (Poikane, Kelly, et al., 2019) 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 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 (Arnott et al., 2020; Clements et al., 2013; Hintz et al., 2022). However, field studies also have disadvantages, associated with confounding variables and multiple stressors (see above).

TABLE 2 Approaches for setting salinity criteria intended to protect aquatic life.

Country	Thresholds	Method	Reference
Australia	Low-risk trigger values EC 30–5000 μS cm^{-1}[a]	Field data: 80th percentile of the reference systems distribution (unmodified or slightly modified ecosystems)	ANZECC, 2000
Canada	Long-term threshold Cl^- 120 mg L^{-1}	Aquatic toxicity tests: SSD method using 28 species (invertebrates, fish, aquatic plants and algae)	CCME, 2011
	Short-term threshold Cl^- 640 mg L^{-1}	Aquatic toxicity tests: SSD method using 51 species invertebrates, fish)	
China	Long term threshold 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 EC 300–850 μS cm^{-1}[a]	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) 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) 860 mg Cl^- L^{-1}	Acute value divided by the acute-to-chronic ratio	

Note: Different regions/states/provinces may have different approaches within a single country.
[a]For different water body types.

Neither field-based methods nor ecotoxicology are perfect and one result is a wide range of thresholds (Table 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, emphasize 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.

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 as it is recommended in SDG 6.3.2, the one for which most data are available (for a complete overview of salinity criteria reported for SDG indicator 6.3.2 see Table S1). 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, 10 had adopted 500 μS cm^{-1} value (UNEP, 2021; see also Table S1).

Reported SDG national thresholds, however, vary widely around this threshold and, unfortunately, supporting literature justifying these is often hard to find. We have analyzed those that are available using per capita GDP (Figure 7) and geographic region (Figure 8) but no consistent patterns emerge. It is also clear from Table 3 that ecological change has been detected at several points along conductivity and chloride gradients, which means that placement of ecological thresholds will always require close consideration of the precise wording of legislation.

Even within the EU, where all Member States are subject to the same legislation, there is considerable latitude in how this is interpreted (Table 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, Kelly, et al., 2019) and this is also likely to play a role when setting salinity criteria.

TABLE 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} 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} 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} Cl^- 40–90 mg L^{-1}	Halle & Müller, 2013
Streams in Queensland, Australia	Most dramatic shift in benthic invertebrate composition	EC 800–1000 μS m^{-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} 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 Decline in total species richness of invertebrate fauna	TDS 2600 mg L^{-1} TDS 4100 mg L^{-1}	Pinder et al., 2005

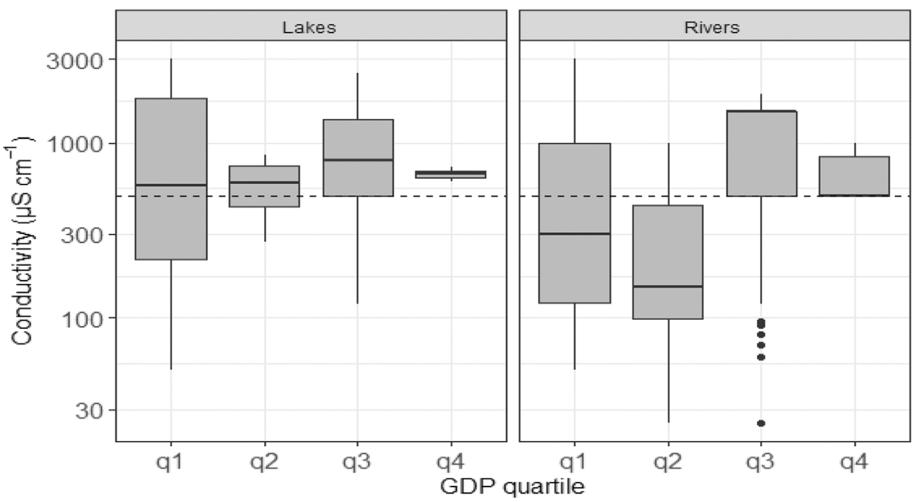

FIGURE 7 Range of salinity thresholds reported to the UN (SDG 6.3.2), arranged by per capita GDP (divided into four quartiles, from Q1 (lowest per capita GDP) to Q4 (highest per capita GDP). See supplementary text S1 for more information.

5 | SETTING SALINITY THRESHOLDS IN AN ALREADY-WARMING WORLD

It is clear that climate change has already led to observable changes in inland waters and that this is driving much of the research on effects of climate and, by extension, recognition of the need to define protective criteria. In practical terms, this may limit the number of sites in some regions from which the "acceptable" state may be defined.

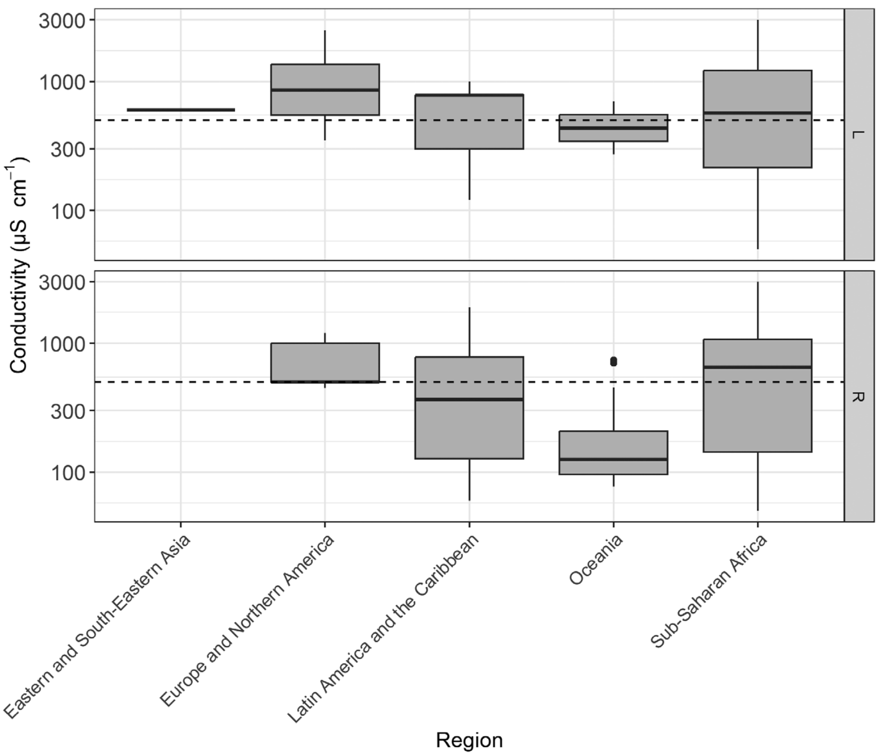

FIGURE 8 Range of salinity thresholds reported to the UN (SDG 6.3.2), arranged by geographic region. See supplementary text S1 for more information.

TABLE 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. Single threshold or Range of type-specific thresholds (median value in brackets).

		Rivers		Lakes	
		Conductivity μS cm^{-1}	Chloride mg L^{-1}	Conductivity μS cm^{-1}	Chloride mg L^{-1}
	Metrics	Single threshold or Range of type-specific thresholds (median value)			
Austria	AA	-	150	1010[a]	60[a] 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) 20–3500[b]	50–500[b]	350 20–700[b]	-
	MAC	300–700 (500) 100–2200[b]	-	600–3600[c] (600)	-

Abbreviations: AA, annual average; MAC, maximum allowable concentration; P90, 90th percentile; SA, seasonal average.

[a]Threshold for Lake Neusiedl—a unique soda lake, represents a *minimum* value for conductivity and chlorides.

[b]Waterbody specific thresholds set for several types.

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

Without long-term records, this may not even be apparent meaning that decisions based on contemporary data—chemical and biological—will be influenced by "shifting baseline syndrome" (Jones et al., 2020; Soga & Gaston, 2018). 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 salinization is often one ingredient of a cocktail of stressors, which interact in different ways (Kaushal et al., 2019). 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 salinization—by Spears et al. (2022). The policy challenge for long-standing legislation such as the EU WFD and US Clean Water Act is that ambition was determined in an era before warming was recognized to be as significant as it is now, and accepting the inevitability of change due to interactions with climate effectively involves a lowering of this ambition. Where ambition is set purely in terms of metrics based on species turnover, there may be little prospect of persuading stakeholders of the benefits (Poikane et al., 2016). However, where there are direct links to ecosystem services (e.g., fish kills due to blooms of *Prymnesium*: Free et al., 2023; toxic cyanobacteria blooms: Carvalho et al., 2013; Poikane et al., 2014), then knowledge of interactions amongst stressors may lead to more realistic thresholds. 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 the range where such interactions are possible.

However, others have pointed out that ecological tipping points are difficult to detect from empirical data (Carrier-Belleau et al., 2022; Hillebrand et al., 2020) 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 salinization. Any monitoring program that detects a trend toward a value of concern is fulfilling a valuable role by providing early warning of likely effects. The principle of "no deterioration" is integral to legislation such 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.

6 | CONCLUSIONS

Our review highlights a tension between a practical role for criteria in policy implementation, enabling threatened water bodies to be identified and prioritized and measuring progress toward these objectives, and the major scientific challenge of defining realistic boundaries in the face of considerable scientific uncertainty. Forty years ago, a nationally-applicable threshold derived from single species tests with an appropriate safety factor was considered to be sufficient. Since then, criteria have developed to reflect variations in the natural environment and responses of wider ranges of taxa. Those who criticize the whole idea of criteria for salinity and other stressors are not claiming that there is not a limit to the amount that an ecosystem can tolerate but, rather, pointing out that there is a need to evaluate thresholds on a case-by-case basis and in real time, as other factors in a water body also change. We would argue that the role of criteria is as much about communicating messages about the importance of stressors to wide audiences—including politicians, catchment managers and the public—as it is about the ecological interactions. These are important and, as each generation uncovers more layers of complexity, so more nuanced criteria will need to be developed.

Key messages, and guiding principles, from this work are:

1. Salinization is a growing problem, and water managers need clear guidance on both trends and thresholds;

2. 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;
3. 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)
4. 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;
5. Community-level responses are often difficult to interpret due to interactions with other stressors;
6. There is a role for broadly-based (national or regional) criteria but also a case for moving toward waterbody-specific criteria;
7. Salinity criteria need to be harmonized, especially for transboundary water bodies; and;
8. Conductivity is a valuable proxy measurement for salinization but it is important to recognize that ion composition matters and an understanding of local geochemistry is important.
9. 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.

AUTHOR CONTRIBUTIONS

Martyn Kelly: Conceptualization (equal); visualization (equal); writing – original draft (equal); writing – review and editing (equal). **Gary Free:** Conceptualization (equal); visualization (equal); writing – original draft (equal); writing – review and editing (equal). **Agnieszka Kolada:** Conceptualization (equal); writing – review and editing (equal). **Geoff Phillips:** Conceptualization (equal); data curation (equal); formal analysis (equal); visualization (equal); writing – review and editing (equal). **Stuart Warner:** Conceptualization (equal); writing – review and editing (equal). **Georg Wolfram:** Conceptualization (equal); writing – original draft (equal); writing – review and editing (equal). **Sandra Poikane:** Conceptualization (equal); visualization (equal); writing – original draft (equal); writing – review and editing (equal).

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CONFLICT OF INTEREST STATEMENT

The authors have declared no conflicts of interest for this article.

DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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