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Review

A hitchhiker's guide to European lake ecological assessment and intercalibration

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ABSTRACT

The Water Framework Directive is the first international legislation to require European countries to establish comparable ecological assessment schemes for their freshwaters. A key element in harmonising quality classification within and between Europe's river basins is an "Intercalibration" exercise, stipulated by the WFD, to ensure that the good status boundaries in all of the biological assessment methods correspond to similar levels of anthropogenic pressure. In this article, we provide a comprehensive overview of this international comparison, focusing on the assessment schemes developed for freshwater lakes. Out of 82 lake ecological assessment methods reported for the comparison, 62 were successfully intercalibrated and included in the EC Decision on intercalibration, with a high proportion of phytoplankton (18), macrophyte (17) and benthic fauna (13) assessment methods. All the lake assessment methods are reviewed in this article, including the results of intercalibration. Furthermore, the current gaps and way forward to reach consistent management objectives for European lakes are discussed.

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1. Introduction

Many benefits provided by aquatic ecosystems can only be maintained if the ecosystems are protected from deterioration (Millennium Ecosystem Assessment, 2005; de Groot et al., 2010). This aim requires (1) suitable methods to assess anthropogenic impact on aquatic ecosystems and to evaluate ecological integrity, (2) common management objectives across state boundaries and administrative barriers, and (3) concerted action aimed at halting and reversing degradation on the national and international level (Palmer et al., 2005; Hering et al., 2013).

Many countries have adopted legislation to determine the ecological integrity of surface waters including streams, rivers, lakes, estuaries and coastal waters. The purpose of the US Clean Water Act (CWA) is to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters." Also in Australia, a broader, more holistic approach to aquatic ecosystem management is adapted "to maintain and enhance the ecological integrity of freshwater and marine ecosystems" (ANZECC, 2000). Similarly, the South African National Water Act aims at "protecting aquatic and associated ecosystems and their biological diversity". Still, in many cases, these legislation acts have not fulfilled their ambitions (Doremus and Dan Tarlock, 2013; Adler, 2013), mainly due to a lack of clear guidelines for the assessment of biological integrity (Davies and Jackson, 2006), the insufficient development and quality of bioassessment methods (Adler, 2003; Yoder and Barbour, 2009), a lack of consistent management objectives (Davies and Jackson, 2006; Adler, 2013), and poor comparability of biological data (Cao and Hawkins, 2011; Diamond et al., 2012).

In Europe, the Water Framework Directive (EC, 2000; WFD) establishes a framework for the protection and improvement of inland and coastal waters, which aims to achieve 'good' surface water status by 2015 or, at the latest, by 2027. In contrast to other legislations, the WFD provides operational definitions for assessing ecological status, setting management objectives, and harmonising EU Member States' ecological assessment systems. In short, the WFD is based on the following main principles:

- Biological assessment uses numerical measurements of communities of plants and animals (phytoplankton, aquatic flora, benthic invertebrates and fish fauna) as stipulated in the Directive (e.g., biomass, taxonomic composition, diversity, etc.).
- In biological assessment, the observed condition is compared with the reference status with the result given in five classes: 'high' status (no differences to reference conditions), 'good' status (slight differences), 'moderate' status (moderate differences), 'poor' and 'bad' statuses (major differences).
- 'Good' ecological status represents the target value that all surface water bodies must achieve in the near future. These values (expressed as 'good' status class boundaries) are compared and harmonised through the intercalibration exercise, ensuring consistent management objectives across Europe.

Since the adoption of the European Water Framework Directive (WFD) in 2000, huge progress has been made in the ecological assessment of European waters. Many European countries now have a set of assessment tools for indicating the state of Europe's water resources and for monitoring improvements in relation to investments in river basin management, or deterioration in response to future environmental changes (Birk et al., 2012a; Brucet et al., 2013b). These assessment methods are composed of several metrics (see Tables 1–5), and combination rules are applied to calculate the ecological assessment result for the whole system.

In order to harmonise ecological assessment systems and to ensure a consistent level of ambition in the protection and restoration of surface water bodies across the EU, an intercalibration exercise was launched, involving hundreds of experts from all Member States (Nõges et al., 2009). This exercise led to the development of innovative new approaches to accomplishing this highly complex task (Birk et al., 2013). In total, 230 methods from 28 countries were intercalibrated and published in the EC Decision (EC, 2013). This flagship document sets the harmonised boundaries for the Member States' national methods for classifying the ecological quality of their rivers, lakes, coastal waters and estuaries.

In this article, we provide an overview of this international comparison, focusing on the assessment schemes developed for freshwater lakes. More specifically, we (1) briefly review the assessment methods developed for lakes focusing on the metrics included and the pressures addressed; (2) describe the intercalibration exercise performed on lake assessment methods; (3) assess the gaps in the lake assessment methods regarding biological communities, pressures addressed and geographical regions.

2. Intercalibration methodology

A step-by-step methodology for the comparison and harmonisation of ecological assessment methods was developed (EC, 2011; Birk et al., 2013). The assessment methods were first checked for their compliance with the WFD requirements - only methods that

Table 1
Overview of the Member States (MS) lake phytoplankton assessment methods (only intercalibrated methods). For detailed description of metrics see supplementary material S2, Table 1.

MS	Metrics included in national systems			Pressures addressed and pressure indicators
	Abundance	Taxonomic composition	Sensitivity/tolerance	
Austria	CHL-A, TBV		Brettum index	EUTR: TP
Belgium	CHL-A	%CYA		EUTR: TP, TN
Cyprus	CHL-A, TBV	BV-CYA	IGA index	EUTR: TP, land-use, PD
Denmark	CHL-A	%CYA, %CHRY	Sensitivity Index	EUTR: TP, TN
Estonia	CHL-A_{surf}, CHL-A_{tot}	Phytoplankton community description	PCQ index	EUTR: TP, TN
Finland	CHL-A, TBV	%CYA (impact taxa)	TPI index	EUTR: TP
Germany	CHL-A, CHL-A_{max}, TBV	Algal class metrics	PTSI index	EUTR: TP, TN
Ireland	CHL-A		IPI index	EUTR: TP, TN
Italy-lakes	CHL-A, TBV		PTI_{OT} index	EUTR: TP
Italy-reservoirs	CHL-A, TBV	BV-CYA	MedPTI index	EUTR: TP, land-use, PD
Netherlands	CHL-A		Bloom index	EUTR: TP, TN
Norway	CHL-A, TBV	BV-CYA_{max}	PTI_{NO} index	EUTR: TP
Poland	CHL-A, TBV	BV-CYA		EUTR: TP, TN
Portugal	CHL-A, TBV	BV-CYA	IGA index	EUTR: TP, land-use, PD
Slovenia	CHL-A, TBV		Brettum index	EUTR: TP
Spain	CHL-A, TBV	%CYA	IGA index	EUTR: TP, land-use, PD
Sweden	CHL-A, TBV	%CYA	TPI index	EUTR: TP
UK	CHL-A, TBV	BV-CYA	PTI_{UK} index	EUTR: TP, TN

CHL-a, chlorophyll-a concentration; TBV, total biovolume; BV-CYA, biovolume of Cyanobacteria; %CYA, percentage of Cyanobacteria of total biovolume; CHRY, Chrysophyta; EUTR, eutrophication; TP, total phosphorus; TN, total nitrogen; PD, population density.

Table 2
Overview of the Member States (MS) lake macrophyte assessment methods (only intercalibrated methods). For detailed description of metrics see supplementary material S2, Table 3.

MS	Metrics included in national systems				Pressures addressed and pressure indicators
	Abundance	Composition	Sensitivity/tolerance	Functional and richness/diversity	
Austria	Colonisation depth Vegetation density	Species composition index	Trophic index	Type-specific zonation	EUTR and GD: TP, CHL-a, SD
Belgium	Abundance of submerged vegetation	Type-specific species composition index	Disturbance index	Macrophyte growth forms	EUTR and others: TP, TN, CHL-a
Denmark	Colonisation depth Total coverage		Presence of indicator species		EUTR: TP, TN, CHL-a
Estonia	Colonisation depth (deep lakes)	Abundance of different taxonomic groups	Abundance of sensitive/tolerant taxa		EUTR and HM: TP, TN, CHL-a
Finland		PMA index	Trophic index PTST index		EUTR and HM: TP
France			IBML index		EUTR and GD: TP, chl-a, SD
Germany	Colonisation depth	Dominance of selected taxa	Reference Index		EUTR and GD: TP, TN, CHL-a, SD
Ireland	Colonisation depth Average depth of presence	% RF Chara % RF elodeids	Plant trophic score % RF tolerant taxa		EUTR: TP, TN, CHL-a
Italy	Colonisation depth	Dissimilarity index, Invasive species	Trophic score		EUTR and GD: TP, chl-a, SD
Latvia	Colonisation depth	Abundance of different taxonomic groups	Abundance of sensitive/tolerant taxa		EUTR: TP, TN, CHL-a
Lithuania	Colonisation depth	Dominance of selected taxa	Reference Index		EUTR: TP, TN, CHL-a
Netherlands			Indicator species metrics	Growth form metrics	EUTR and HM: TP, TN, CHL-a
Norway			Trophic index		EUTR: TP
Poland	Colonisation index			Pielou's index (syntax level)	EUTR and others: TP, TN, CHL-a
Slovenia	Colonisation depth	Depth limit of charophytes	Trophic index		EUTR and GD: TP, CHL-a, SD
Sweden			Trophic index		EUTR: TP
UK	Mean cover	Relative cover of filamentous algae	LMNI index	Number of functional groups Number of taxa	EUTR: TP, TN, CHL-a

EUTR, eutrophication; HM, hydromorphological modifications; GD, general degradation; RF, relative frequency; TP, total phosphorus; TN, total nitrogen; CHL-a, chlorophyll-a concentration; SD, Secchi depth.

met these criteria could be intercalibrated. For example, assessment schemes must establish "biological reference conditions" from which the degree of human impact is measured using an Ecological Quality Ratio (EQR) – the ratio of the observed assessment value to the expected value under reference conditions. Additionally, the assessment method must include all the biological attributes included in the Directive, e.g., for phytoplankton: average phytoplankton biomass, composition and abundance of planktonic taxa, frequency and intensity of planktonic blooms.

Secondly, the assessment methods were checked to ensure that intercalibration was feasible, with analyses restricted to methods that address similar water body types and anthropogenic pressure and which are based on similar concepts. For example, some assessment schemes for lake benthic invertebrates were

Table 3
Overview of the Member States (MS) lake phytobenthos assessment methods (only intercalibrated methods). For detailed description of metrics see supplementary material S2, Table 2.

MS	Sensitivity/tolerance metrics	Pressures addressed and pressure indicators
Belgium	PISIAD index	EUTR: TP, chl-a
Germany	Trophic-index TI_{Nord} Quotient of reference species RAQ	EUTR: TP
Finland	IPS index	EUTR: TP
Hungary	IBD index; EPI-D index; TDIL index	EUTR: TP
Ireland	Lake Trophic Diatom Index	EUTR: TP
Poland	Trophic index TJ Index of reference species GR_J	EUTR: TP
Slovenia	Trophic index (Lake – TI)	EUTR: TP
Sweden	IPS index	EUTR: TP
UK	Diatom Assessment Of Lake Ecological Quality (DARLEQ)	EUTR: TP

EUTR, eutrophication; TP, total phosphorus; CHL-a, chlorophyll-a concentration.

Table 4
Overview of the Member State (MS) lake benthic invertebrate assessment methods (only intercalibrated methods). For detailed description of metrics see supplementary material S2, Table 4.

Metrics included in the national assessment systems					Pressures addressed and pressure indicators
MS	Composition metrics	Sensitivity/tolerance metrics	Richness/diversity metrics	Functional metrics	
Belgium		Number of sensitive taxa Mean tolerance score	EPT taxa richness Total taxa richness Shannon-Wiener diversity		EUTR, HM: not tested
Estonia		ASPT index Swedish Acidity index	EPT and total taxa richness Shannon-Wiener diversity		EUTR: TP, land use
Finland		BQI_{FI} index			EUTR: TP, TN, CHL-a, SD HM: Morphoindex
Germany-ALP[a]	RA of Odonata	Fauna index	Shannon-Wiener diversity	RA of gatherers *r*/*k* ratio	HM: Morphological Index
Germany-CB[a]	RA of Odonata	Fauna index	ETO taxa richness	RA of habitat lithal	EUTR, HM: Combined stressor index
Lithuania	RA of COP	ASPT index	CEP taxa richness Hill's number		EUTR: TP
Netherlands		DN%, DP%, KM%	KM% (taxa)		HM: shore characteristics
Norway		AWIC index; Acidity index	Ephemeroptera taxa richness Gastropoda taxa richness		ACID: pH, ANC, Lal
Slovenia		Littoral fauna index	Total taxa richness Margalef diversity		HYMO: Lake Modification Index
Sweden-BQI[b]		BQI_{SE} index			EUTR: TP
Sweden-MILA[b]	RA Ephemeroptera RA Diptera	AWIC index	Ephemeroptera taxa richness Gastropoda taxa richness	RA of predators	ACID: pH
UK-CPET[c]		CPET index			EUTR: TP
UK-LAMM[c]		LAMM index			ACID: pH, ANC

RA, relative abundance; EUTR, eutrophication; HM, hydromorphological modifications; ACID, acidification; TP, total phosphorus; TN, total nitrogen; CHL-a, chlorophyll-a concentration; SD, Secchi depth; ANC, acid neutralising capacity; LAI, labile aluminium.
[a] Germany intercalibrated two benthic fauna assessment methods: for Alpine and for Central Baltic lake types.
[b] Sweden intercalibrated two methods: MILA for acidification am BQI for eutrophication.
[c] UK intercalibrated two methods: LAMM for acidification am CPET for eutrophication.

developed to measure the impact from acidification (McFarland et al., 2010), whereas other national schemes measured the impact of eutrophication (Ruse, 2010). These two types of schemes are not comparable. Similarly, it was not possible to compare methods based on littoral and profundal benthic invertebrate communities, even if they address the same eutrophication pressure (Sandin et al., 2014). In several cases, assessment systems developed for deep lakes include specific metrics (e.g. macrophyte colonisation depth) and cannot be applied to shallow lakes (Søndergaard et al., 2010).

Thirdly, the biological quality element (BQE) and pressure data (e.g., nutrient levels for eutrophication, pH or acid neutralising capacity (ANC) levels for acidification) were collected from the countries involved and options for intercalibration were evaluated.

Intercalibration can be carried out in three ways, depending on the availability and similarity of national assessment methods (Fig. 1):

- If Member States have common sampling methods and thus similar kinds of data (e.g., the number of individuals of all taxa at identical determination level), then national methods were compared directly by applying each national method to the data of other countries (direct comparison; Sandin et al., 2014; Gassner et al., 2014);
- If Member States did not have common sampling methods, the results of national assessment methods were compared using common metrics, a metric developed that was sufficiently comparable with other countries to enable a comparison to be made (indirect comparison; Lyche-Solheim et al., 2013; Kelly et al., 2014);
- If assessment methods had not yet been developed, the Member State could choose to set boundaries based on a common database using the same assessment metrics (common boundary setting; Wolfram et al., 2009; Poikane et al., 2010).

Table 5
Overview of the Member States (MS) lake fish fauna assessment methods (only intercalibrated methods). For detailed description of metrics see supplementary material S2, Table 5.

Metrics included in the national assessment systems					Pressures addressed and pressure indicators
MS	Abundance/composition	Sensitivity/tolerance	Functional metrics Richness/diversity	Age structure metrics	
Austria	Fish biomass (hydroacoustics) Abundance index of alien species	Abundance index of type specific species/sensitive species	Abundance index of migrating spawners/of spawning guilds/of small-bodied species	Length frequency of sentinel species	EUTR, HM, fisheries, alien species: index based on expert judgement
Finland	Total biomass of fish Total number of fish Biomass proportion of cyprinid fish	Indicator species			EUTR: TP, landuse
Germany		Sentinel/type/side species number Sentinel species abundance	Habitat/Spawning preferences Abundance of habitat/spawning preferences	Reproduction of potentially stocked species	EUTR, HM, fisheries, alien species: index based on expert judgement
Ireland	Total biomass Native fish biomass Perch/roach biomass Rel. abundance of bream/rudd Rel. biomass of cyprinid species/non-native species		Relative abundance of rheophilic/lithophilic/phytophilic species Species evenness/dominance index	Maximum length of dominant species (based on BPUE)	EUTR: TP, CHL-a
Italy		NPUE of the guiding species Reduction % of guiding + accompanying species	Relative richness of alien species	Population structure of the guiding species Reproductive success % of guiding + accompanying species	EUTR, HM, fisheries, alien species: index based on expert judgement

EUTR, eutrophication; HM, hydromorphological modifications; TP, total phosphorus; CHL-a, chlorophyll-a concentration.

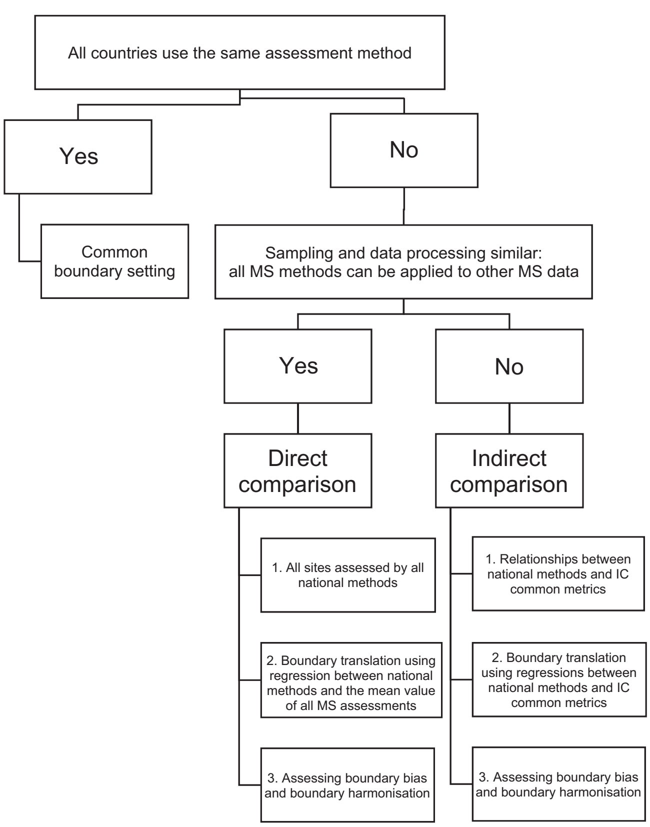

Fig. 1. Flow chart of the main steps of the intercalibration process.

Due to biogeographical and typological reasons, as well as differences in data acquisition, the biological data of different countries cannot be compared without concern (Cao and Hawkins, 2011; Birk et al., 2013). As an example, the number of benthic invertebrate taxa might be generally higher in one country than in others, because the sampling covers much more area per site (Böhmer et al., 2014). The richness and diversity of fish species are related mainly to geographical and climatic variables (Brucet et al., 2013a). Hence, a "benchmarking" procedure was applied with the aim of correcting any biogeographical and methodological differences within a common dataset that can cause incomparability. Three different approaches were applied: "reference benchmarking" based on near-natural reference sites (Pardo et al., 2012), "alternative benchmarking" using sites at similar impairment levels (Birk et al., 2012b), and "continuous benchmarking" using pressure–response gradients (Kelly et al., 2014; for a detailed description of this procedure see Birk et al., 2013).

The final objective of intercalibration was to compare and harmonise national boundaries. This was done using a standardised analytical procedure and harmonised comparability criteria. The main criteria used for evaluating comparability was *boundary bias*: the deviation of a class boundary relative to the common view of the Member States (defined by the common metrics or by the global mean of all of the methods).

The most that any national boundary could deviate from the global mean view of all countries was ±0.25 classes and therefore the most widely divergent national methods could not differ from each other by more than 0.5 classes. National methods that did not comply with these criteria were required to adjust their boundaries until acceptable boundary bias and class agreement were achieved. It is important to stress that intercalibration checks whether the results are comparable, irrespective of method.

For definition of the main terms used in the intercalibration of ecological assessment methods see supplementary material S3.

3. Lake assessment methods

3.1. Lake assessment methods: phytoplankton

The use of phytoplankton for water quality assessment of lakes has a long history, and considerable knowledge of this subject has accumulated over the past century (Willén, 2000). Phytoplankton has traditionally been one of the dominant elements in lake assessment and thus has played an important role in the development of WFD-compliant assessment systems. Nearly all of the Member States have developed phytoplankton assessment methods. Out of 24 methods participating in the intercalibration exercise, 18 methods were included in the final intercalibration results. All of the methods address eutrophication pressure and follow similar assessment principles (Table 1):

- All include measures of phytoplankton abundance – all national methods consider chlorophyll-a, and most also consider total phytoplankton biovolume (Carvalho et al., 2013b);
- Most include measures of cyanobacteria (as proxy measures of the risk of toxic algal blooms) as cyanobacteria biovolume or cyanobacteria as a percentage of the total phytoplankton biovolume (Carvalho et al., 2013b);
- All include metrics based on the trophic preferences of species or algal groups, e.g., Brettum index (Austria, Slovenia; Brettum, 1989) and PTI index (Finland, Sweden; Willén, 2007). However, functional and diversity indices are not used (except Estonia where Pielou's evenness index is used).

Despite some variation between countries in sampling methods and sampling periods (Nõges et al., 2010), all of the intercalibrated methods exhibited significant relationships with total phosphorus (TP), and some of the methods also exhibited significant relationships with total nitrogen (TN) and catchment characteristics. However, it has been difficult to establish relationships for Eastern Continental lakes and Mediterranean reservoirs due to both a lack of data (Padisák et al., 2006) and peculiarities of the water bodies. For example, Mediterranean reservoirs are affected by extreme changes in their water level due to seasonal climate variation (Marchetto et al., 2009) while Eastern Continental lakes are considered naturally eutrophic with nutrient concentrations that are considerably higher than limiting thresholds (Borics et al., 2013).

3.2. Lake assessment methods: macrophytes

Macrophytes have been used in lake assessment for many years, and the macrophyte community is generally regarded as a key indicator of ecological status both in deep (Pall and Moser, 2009) and in shallow lakes (Søndergaard et al., 2010). However, in contrast to phytoplankton assessment systems, existing macrophyte assessment was confined to relatively small geographic regions, mainly in Scandinavia and Central Europe, and was based primarily on indicator species analyses (Melzer, 1999), while the WFD requires lake assessment based on both macrophyte abundance and composition. Most European countries have undertaken significant efforts to develop WFD-compliant macrophyte assessment tools. Twenty systems have been developed recently, and 17 of these have been intercalibrated (Table 2).

All macrophyte assessment systems (except that in Poland) include sensitivity/tolerance metrics expressed as indices that are based on species indicator values, e.g., a trophic index (Pall and Moser, 2009) or relative abundance of sensitive versus tolerant taxa (Schaumburg et al., 2004; Free et al., 2006). Most macrophyte assessment systems include an assessment of abundance (except the methods of Norway, Sweden and Finland). The most widely used abundance measures are the macrophyte colonisation depth and the abundance of submerged macrophytes.

Most macrophyte assessment systems claim to address not only eutrophication but also other pressures, such as general degradation and hydromorphological changes. However, significant pressure response relationships have only been shown for eutrophication parameters (nutrient and chlorophyll-a concentrations and water transparency) (Table 2). Even in these cases there remain some difficulties with the response of macrophyte metrics to pressure, especially the delayed response to both increasing and decreasing eutrophication (Pall and Moser, 2009) and the non-linear reaction to eutrophication following the alternative stable state theory (Penning et al., 2008). Further challenges are establishing pressure–response relationships for pressures that are themselves quite hard to quantify (hydromorphological modifications, i.e. shoreline developments and water level fluctuations), as well as for multiple stressors.

3.3. Lake assessment methods: phytobenthos

Although phytobenthos is only one part of the BQE "macrophytes and phytobenthos" (Annex V, WFD), most Member States have developed separate assessment methods for macrophytes and phytobenthos. Moreover, only 11 of the 27 Member States of the EU took part in the intercalibration of phytobenthos methods (Tables 3 and 7). Nevertheless, the collective view of the phytobenthos expert group (Kelly et al., 2014) was that a Member state cannot be considered to be fully compliant with the WFD requirements if it possesses only a macrophyte (or only phytobenthos) method. There are situations (e.g. where the lake is subject to hydromorphological stress, navigation, etc.) where macrophytes have completely disappeared and will not give any reliable indication of the impact of nutrients on littoral flora, as well as situations when the two elements react at different rates to changes in their environment (Schaumburg et al., 2004; Pall and Moser, 2009).

All national phytobenthos methods assess the composition and relative abundance of diatoms, assumed to be proxies for the phytobenthos community as a whole (Table 3). All the methods address eutrophication pressure and have demonstrated significant relationships to total phosphorus concentrations with the exception of some methods in low alkalinity lakes, where a combination of short gradients and the confounding effects of low pH led to very weak relationships with TP.

3.4. Lake assessment methods: benthic invertebrates

The lack of WFD-compliant macroinvertebrate assessment tools was identified as one of the major knowledge gaps impeding the full assessment of the ecological quality of lakes (Solimini et al., 2006). Macroinvertebrates have been recognised as one of the most difficult biological groups to consider in the assessment of lake quality for three main reasons: their complex biotic structure, their high spatial and temporal variability (Solimini et al., 2006; Free et al., 2006; Solimini and Sandin, 2012). In spite of these factors, 20 systems have been developed, of which 13 have been successfully intercalibrated (Table 4). Of the 44 metrics, almost half (42%) are sensitivity/tolerance metrics included in all national systems. Some countries have used traditional indices, such as the ASPT index (Lithuania and Estonia; Armitage et al., 1983), Benthic Quality Index (Sweden and Finland; Wiederholm, 1980) and Acidity index (Norway and Estonia; Henrikson and Medin, 1986). Most of the Member States have developed new sensitivity indices, such as the Littoral Fauna Index (Slovenia; Urbanic, 2014), Mean Tolerance Score (Belgium; Gabriels et al., 2010) and the LAMM index (United Kingdom; McFarland et al., 2010). Eight methods also contain richness/diversity methods, the most frequently used being metrics of total taxa richness, EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa richness (Lenat, 1988) and Shannon–Wiener diversity. Only four methods contain composition metrics, while functional metrics are rarely used.

Most of the current benthic fauna assessment systems are based on significant pressure–response relationships. Nonetheless, the variation explained by the models was found to be low in many cases and was considered unsatisfactory (Free et al., 2006). The low explanatory power of the models has several explanations, i.e. complex biotic structure, impact of various environmental factors and multiple pressures, and habitat impact (Solimini et al., 2006; Free et al., 2009). In short, while the development of benthic assessment methods can be considered a partial success story, there is still a need to further understand the structure of lake benthic macroinvertebrate communities and their response to anthropogenic pressure to reduce the uncertainty of the metrics developed. Recently, a new harmonized multimetric assessment system has been proposed for Western, Northern, Central and Southern Europe (including natural Mediterranean lakes; Miler et al., 2013) and its implementation in national assessment systems may represent the way forward to overcoming those limitations.

3.5. Lake assessment methods: fish fauna

It is well established that fish are sensitive indicators of environmental degradation and offer the major advantage of integrating the direct and indirect effects of stress over large scales of space and time (Minns et al., 1994). Fish exhibit reactions to eutrophication (Mehner et al., 2007), habitat destruction, shoreline degradation, lake use intensity (Belpaire et al., 2000), hydromorphological degradation, connectivity (Degerman et al., 2001), acidification (Henriksen et al., 1989) and combined degradation (Whittier, 1999). Nevertheless, the fish community is often an overlooked and neglected aspect of lake monitoring. So far, only eight Member States have finalised fish assessment systems (Kelly et al., 2012; Olin et al., 2013), and only five of these have shown significant pressure–response relationships (Table 5).

The use of fish communities as indicators of environmental quality is potentially challenging (Kelly et al., 2012), with several problems: (i) a wide variety of sampling methods are used by the Member States, including multi-mesh gill nets, electro-fishing, trawling and hydro-acoustics; (ii) the activities of fishing, stocking and the introduction of exotic species, all of which can have a large impact on the natural fish fauna; (iii) lakes are subjected to multiple pressures (shoreline degradation, eutrophication and water level regulation), and fish, at the topmost level of the trophic cascade, indirectly integrate the effects of these on lower trophic levels; (iv) high natural variability in fish metrics, which may be related to lake size, depth and water chemistry; (v) fish are mobile and can avoid areas of environmental stress, resulting in this BQE being less sensitive to pressures than others. Taking all these factors together, it is not surprising that there are few significant relationships between fish metrics and specific pressure indicators (Olin et al., 2013). Fish are, however, important indicators: they are at the top of the food chain, have significant economic and social importance and their assessment is an important part of an integrated approach to water management. One solution to this problem is to base assessment of pressure on expert judgement (e.g., Aubry and Elliott, 2006); Austria, Germany, and Italy have demonstrated a response of their overall fish assessment result to a combined pressure index (all common pressures scored and summed up to create an overall pressure index).

4. Lake intercalibration

4.1. Intercalibration groups

Fourteen Lake Intercalibration groups were formed, each focusing on a specific geographical region/BQE/anthropogenic pressure combination (Table 7, only finalised results included):

- For phytoplankton, macrophytes and phytobenthos, all geographic groups just addressed eutrophication pressure;
- For phytobenthos, all methods were intercalibrated within a single pan-European group due to the low number of methods available;
- For benthic invertebrates, the Alpine group addressed hydromorphological alterations whilst the Central Baltic group addressed combined pressures, including hydromorphological alteration and eutrophication. In the Northern region, two separate groups were created, one that addressed eutrophication using profundal communities whilst another addressed acidification using littoral communities.
- For fish fauna, the Alpine group addressed diverse pressures, but the Northern group only addressed eutrophication.

In several cases, it was not possible to intercalibrate the methods:

- Due to the low number of methods in the group (the Mediterranean benthic fauna group and the Central Baltic fish group);
- Due to different pressure/assessment concepts on which the methods were based (e.g. the Norwegian fish method which addressed acidification could not be included in the Northern fish group dealing with eutrophication).

4.2. Common metrics and benchmarking

In the next step, datasets were collected and compared. In most cases (nine groups), the assessment methods and data were sufficiently similar to enable direct comparison, where each national method was applied to all data in a common dataset. For five groups, specific common metrics were developed for the intercalibration exercise (Table 7, for details see Lyche-Solheim et al., 2013; Phillips et al., 2013). For the lake phytobenthos intercalibration exercise, the trophic index (TI: Rott et al., 1999), a widely used phytobenthos metric, was used as the common metric of intercalibration.

Half of the groups applied reference-site benchmarking. This was true primarily in the Northern and Alpine regions, where near-natural reference sites were available in sufficient numbers. Six Intercalibration groups applied continuous benchmarking, which uses the available pressure gradient (e.g. from good to bad status) to identify country-specific differences. This approach was used in the Central Baltic region, where reference sites are rare, or no longer exist, and the range of pressures was high across the group but not always adequately represented within each country. It was also used in several cases in the Northern and Alpine regions, as this approach was independent of national views of reference and is, therefore, more robust.

4.3. Boundary harmonisation

In the final step, national boundaries on the EQR scale were compared and, if necessary, adjusted if the agreed comparability criteria were exceeded.

Only three groups (Alpine phytoplankton, Alpine benthic fauna, and Northern Benthic fauna acidification methods) found that no boundary adjustment was needed, as the comparability analysis

Table 6
Overview of the Member States (MS) lake assessment methods.

MS	Phytoplankton	Macrophytes	Phytobenthos	Benthic fauna	Fish fauna
Austria	Intercalibrated	Intercalibrated	–	–	Intercalibrated
Belgium	Intercalibrated	Intercalibrated	Intercalibrated	Intercalibrated	–
Bulgaria	–	Submitted	–	–	–
Cyprus	Intercalibrated	n.a.	n.a.	n.a.	n.a.
Denmark	Intercalibrated	Intercalibrated	–	–	–
Estonia	Intercalibrated	Intercalibrated	–	Intercalibrated	–
Finland	Intercalibrated	Intercalibrated	Intercalibrated	Intercalibrated	Intercalibrated
France	Submitted	Intercalibrated	Submitted	Submitted	Submitted
Germany	Intercalibrated	Intercalibrated	Intercalibrated	Intercalibrated (2) Submitted	Intercalibrated
Greece	–	–	–	–	–
Hungary	Submitted	Submitted	Intercalibrated	Submitted	
Ireland	Intercalibrated	Intercalibrated	Intercalibrated		Intercalibrated
Italy	Intercalibrated (2)	Intercalibrated	Submitted	Submitted	Intercalibrated
Latvia	Submitted	Intercalibrated	–	–	–
Lithuania	Submitted	Intercalibrated	–	Intercalibrated	–
Netherlands	Intercalibrated	Intercalibrated	–	Intercalibrated	Submitted
Norway	Intercalibrated	Intercalibrated	–	Intercalibrated	Submitted
Poland	Intercalibrated	Intercalibrated	Intercalibrated	–	–
Portugal	Intercalibrated	n.a.	n.a.	n.a.	n.a.
Romania	Submitted	Submitted	–	Submitted	–
Slovenia	Intercalibrated	Intercalibrated	Intercalibrated	Intercalibrated	–
Spain	Intercalibrated	Submitted	–	Submitted	–
Sweden	Intercalibrated	Intercalibrated	Intercalibrated	Intercalibrated (2) Submitted	Submitted
UK	Intercalibrated	Intercalibrated	Intercalibrated	Intercalibrated (2)	–
In total	18 intercalibrated 5 submitted	17 intercalibrated 3 submitted	9 intercalibrated 2 submitted	13 intercalibrated 6 submitted	5 intercalibrated 4 submitted

Intercalibrated, method intercalibrated and included in the final results; Submitted, method participated in the intercalibration but not included in the final results; n.a., not applicable, method development and intercalibration not feasible due to lack of natural lakes. (2), MS has intercalibrated 2 assessment methods (e.g., for different human impacts).

showed that the methods yielded very similar assessments (in agreement with the comparability criteria defined in the Intercalibration Guidance). These were groups with just a few (2–4) similar assessment methods (for example, Alpine phytoplankton methods all include biomass metrics and composition metrics each with harmonised boundary values agreed upon during development (Wolfram et al., 2009).

In all other groups, boundary adjustments were needed. In some cases, only the class boundaries of a few methods were adjusted. In several cases, the assessment methods were so dissimilar that more profound changes were needed, e.g., in the values associated with reference status or in the way data was used and combined. For example, reference values of the Belgian benthic fauna system were revised, and the combination rules of the specific metrics used in the Norwegian and British phytoplankton systems were adjusted.

For detailed description of one intercalibration exercise see supplementary material S3.

5. Intercalibration gaps

5.1. Intercalibration gaps: fish fauna, phytobenthos and benthic invertebrates

The situation with respect to different BQEs is highly variable. Nearly all Member States have developed and intercalibrated assessment methods for phytoplankton and macrophytes. The only exception is Greece (which has not developed methods for either BQE) and Bulgaria (which has not developed a method for phytoplankton). In contrast, only five countries have intercalibrated lake fish-based assessment methods, nine have phytobenthos methods and 10 have benthic fauna assessment methods.

5.2. Eastern Continental and Mediterranean regions

There have been particular difficulties in the development and intercalibration of ecological assessment methods in the Eastern Continental and Mediterranean regions (Table 6). Due to high evaporation/precipitation ratios and low geographic relief, lakes in the Eastern Continental region are often endorheic and naturally eutrophic. In this region, there are relatively few examples of lakes with catchments in a near natural state, and as the least impacted also have relatively high nutrient concentration, e.g. TP > 100 μg/l, pressure–response models show asymptotic behaviour and high variation (Borics et al., 2013). Thus, there have been considerable difficulties in establishing sound pressure–response relationships and setting meaningful ecological class boundaries (Poikane et al., 2014). Further research is, therefore, needed to develop and intercalibrate ecological assessment tools for these lakes.

The main difficulties encountered in the Mediterranean region are the small number and high diversity of natural lakes. In the entire Mediterranean region, only 257 lakes are reported, mostly (84%) in Spain, which includes many lakes with water surface areas of less than 50 ha. Lakes are highly diverse: for example, in the Mediterranean part of Italy, there are 13 lakes, including the paleosaline Lago di Pergusa – a unique saltwater lake in Europe, the Lago di Trasimeno – a very large shallow lake, and several volcanic lakes of different depths. Therefore, in spite of a common effort within the Mediterranean intercalibration group, it has not been possible to intercalibrate natural Mediterranean lakes, due to the lack of a sufficient number of lakes within common types. Nevertheless, Mediterranean countries must develop tools for evaluating the ecological quality of their lakes, even though this development is hindered by the limited amount of data available and the high inter- and intra-annual variability of the biological communities and water characteristics (Boix et al., 2005). In both regions, the

Table 7
Overview of the Member States (MS) lake intercalibration exercises (only finalised exercises).

BQE	Region	Pressure addressed	Intercalibration option	Common metrics used	Benchmarking applied	MS methods intercalibrated	MS participated but not intercalibrated
Phytoplankton	ALP	EUTR	Direct comparison	Average of each national methods EQRs	Reference sites	AT, DE, IT_{lakes}, SI	FR
	CB	EUTR	Comparison via common metrics	Multimetrics of Chlorophyll-a and PTI EQR (Phillips et al., 2013)	Continuous	BE, DE, DK, EE, IE, NL, PL, UK	LV, LT
	MED	EUTR	Direct comparison	Average of each national methods EQRs	Reference sites + continuous	CY, ES, $IT_{reservoirs}$, PT	RO, FR
	NOR	EUTR	Direct comparison Comparison via common metrics	Multimetrics of Chlorophyll-a and PTI EQR (Phillips et al., 2013)	Continuous	FI, IE, NO, SE, UK	
Macro phytes	ALP	EUTR	Direct comparison	Average of each national methods EQRs	Reference sites	AT, DE, FR, IT, SI	
	CB	EUTR	Direct comparison	Average of each national methods EQRs	Continuous	BE, DE, DK, EE, LT, LV, NL, PL, UK	
	NOR	EUTR	Comparison via common metrics	Lake Macrophyte Intercalibration common metrics (Hellsten et al., 2014)	Reference sites	FI, IE, NO, SE, UK	
Phytobenthos	All	EUTR	Comparison via common metrics	Trophic index (Rott, 1999)	Continuous	BE, FI, DE, HU, IE, PL, SI, SE, UK	IT, FR
Benthic invertebrates	ALP	HM	Comparison via common metrics	Multimetrics of Fauna index, number of taxa, *r*/*k* ratio and RA of feeding type gatherer (Böhmer et al., 2014)	Continuous	$DE_{ALP\text{-}eulit}$, SI	$DE_{ALP\text{-}sublit}$, FR, IT
	CB	HM and EUTR	Comparison via common metrics	Multimetrics of EPTCBO taxa number, ASPT index, relative abundance of ETO and microhabitat type lithal (Böhmer et al., 2014)	Continuous	BE, DE, EE, LT, NL, UK	
	NOR	EUTR	Direct comparison	Average of each national methods EQRs	Reference sites	FI, SE_{MILA}	SE_{ASPT}, UK_{CPET}
	NOR	ACID	Direct comparison	Average of each national methods EQRs	Reference sites	NO, SE, UK_{LAMM}	
Fish fauna	ALP	Diverse impacts	Direct comparison	Global mean of all the methods	Site-specific references	AT, DE, IT	FR
	NOR	EUTR	Direct comparison	Global mean of all the methods	Reference sites	FI, IE	SE, NO

EUTR, eutrophication; HM, hydromorphological modifications; ACID, acidification; BQE, biological quality element; ALP, Alpine; CB, Central Baltic; MED, Mediterranean; NOR, Northern.

use of palaeolimnological data (e.g. Bennion et al., 2004) may be more suitable for defining site-specific reference conditions.

5.3. Anthropogenic pressures addressed by lake assessment methods

Lakes in Europe are subject to manifold anthropogenic pressures (eutrophication, acidification, hydromorphological alterations, alien species and climate change). A recent analysis showed that approximately 30% of European lakes are impacted by hydromorphological modifications and 20% are impacted by acidification (EEA, 2012). Despite this, most of the methods developed focus on eutrophication impacts, while only three methods address acidification (benthic invertebrate methods of Norway, Sweden and the UK) and three – hydromorphological alterations (benthic invertebrate methods of Germany, the Netherlands and Slovenia). Some of the methods claim to address multiple pressures (mainly eutrophication and hydromorphological alterations together), but not all the necessary pressure–response relationships have been established, or they have been established only for eutrophication parameters. These gaps in lake assessment highlight the need for further research, particularly on the impact of combined pressures (e.g. eutrophication and hydrological pressures, such as changes to natural water level regimes or flushing rates).

5.4. Heavily modified and artificial water bodies

Many lakes in Europe are either artificial (e.g. fish ponds) or heavily modified (e.g. reservoirs for hydropower or water storage). For such water bodies the environmental objective is the good ecological potential instead of the good ecological status. Overall, 15.8% of the lake water bodies are designated by the Member States as either heavily modified water bodies or artificial water bodies (EEA, 2012). For ecological potential the same biological quality elements are used as for the ecological status, with a crucial difference in how reference conditions are used. Ecological status always has the undisturbed type-specific reference conditions as a starting point; ecological potential refers to a situation where the negative effects of a physical modification (e.g. a dam) are mitigated as much as possible, taking into account the costs and benefits of these measures (Borja and Elliott, 2007).

The WFD requires intercalibration of biological methods for all water categories including heavily modified water bodies, but so far the focus has been almost exclusively on natural water bodies. Only recently the intercalibration of ecological potential methods has been started, with a focus that is more on comparing and harmonising the ways different countries apply mitigation measures for different water uses, and how existing biological methods are used in their classification.

5.5. Detection of cyanobacteria blooms

Despite successful European policies to reduce nutrient emissions (EEA, 2012), Cyanobacterial blooms are still one of the most widespread effects of eutrophication in Europe (Dolman et al., 2012; Carvalho et al., 2013b) which are further aggravated by climate change (Paerl and Paul, 2012). MS have to measure plankton bloom intensity and frequency and to ensure that persistent summer blooms do not occur. Bloom definition has been developed (Mischke et al., 2011) and a sufficiently strong metric for phytoplankton blooms, based on cyanobacteria biovolume has been demonstrated (Carvalho et al., 2013a). All phytoplankton assessment methods reflect bloom intensity as they either include a cyanobacterial bloom metric or show a strong correlation with a bloom metric (Lyche-Solheim et al., 2014). Bloom frequency is, however, still not tackled in an appropriate way, as the current sampling frequencies are not adequate to capture the temporal dynamics of phytoplankton over days and weeks, especially short-lived blooms (Dubelaar et al., 2004; Søndergaard et al., 2011). With the development of satellite technology in the near future, high frequency and high resolution satellite imagery may enable improved temporal representation of the open water of lakes for parameters such as phytoplankton chlorophyll-a and cyanobacteria biovolume (Hunter et al., 2010), allowing missing metrics such as bloom frequency to be addressed.

5.6. Uncertainty

The intercalibration exercise was concerned solely with harmonizing the position of the high/good and good/moderate boundaries for individual BQEs. In practice, ecological status assessments will be based on the simultaneous assessment of several BQEs and, as these estimates will each differ in their precision, it would be interesting to compare the sensitivity of each method (taking into account uncertainties associated with sampling regimes and analytical procedures) and, indeed, to compare national capabilities for classifying sites when using all BQEs simultaneously. Progress has been made with understanding uncertainty associated with individual BQEs (Kelly et al., 2009; Thackeray et al., 2013), though there is still much to learn about classification procedures under the rules prescribed in the WFD (Caroni et al., 2013; Moe et al., 2015).

6. Conclusions

In total, 82 lake assessment methods were reported for the intercalibration, and, following final adjustments, 62 were successfully intercalibrated. The results include a high proportion of phytoplankton (18), macrophyte (17) and benthic invertebrate assessment methods (13), but few phytobenthos (9) and fish assessment methods (5). Most of the methods were developed to detect the impact of eutrophication (50 methods), although in addition, some methods have demonstrated a significant response to hydromorphological pressures (3), acidification (3) or combined pressures (5). Most of the Central and Northern European countries have developed assessment methods while significant gaps exist for Eastern Continental and Mediterranean countries. The intercalibrated methods are now included in the EC Decision (EC, 2013) setting a legal obligation on Member States to use these harmonised boundaries to assess the ecological quality of their lakes. This is a major step forward in setting consistent management objectives for European water bodies but much remains to be done, mainly developing assessment systems (i) for phytobenthos and fish fauna, (ii) for other pressures and pressure combination, except eutrophication, and (iii) for Eastern Continental and Mediterranean lakes.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2015.01.005.

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