REVIEW

Approaches for integrated assessment of ecological and eutrophication status of surface waters in Nordic Countries

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Abstract We review approaches and tools currently used in Nordic countries (Denmark, Finland, Norway and Sweden) for integrated assessment of ‘ecological status’ sensu the EU Water Framework Directive as well as assessment of ‘eutrophication status’ in coastal and marine waters. Integration principles for combining indicators within biological quality elements (BQEs) and combining BQEs into a final-integrated assessment are discussed. Specific focus has been put on combining different types of information into indices, since several methods are currently employed. As a consequence of the variety of methods used, comparisons across both BQEs and water categories (river, lakes and coastal waters) can be difficult. Based on our analyses, we conclude that some principles and methods for integration can be critical and that a harmonised approach should be developed. Further, we conclude that the integration principles applied within BQEs are critical and in need of harmonisation if we want a better understanding of potential transition in ecological status between surface water types, e.g. when riverine water enters a downstream lake or coastal water body.

Keywords Ecological status · Water Framework Directive · Biological quality elements · Coastal eutrophication · Integration · Assessment

INTRODUCTION

The EU Water Framework Directive (Directive 2000/60/EC) represents a paradigm shift in Europe in regard to how

status of aquatic ecosystems is to be assessed (WFD, Anon. 2000). Given that one of the overarching objectives of the Directive is that all surface water bodies should be in at least ‘good ecological status’, a baseline for implementation of measures is a prerequisite. The baseline is in the WFD context called an initial assessment.

The first status classification of European surface water bodies using the indicators for biological quality elements (BQEs) according to the WFD was made in 2008 and revealed several problems and inconsistencies in their design and practical use in individual water bodies. These problems include differences in the appropriateness of developed indicators, use of the precautionary principle, issues related to the sensitivity of overall classifications of ecological status to uncertainties in reference conditions and class boundaries, as well as routines for classification based on expert judgement (Rolff 2009; EEA 2012).

A few things are in principle carved in stone in the context of the WFD: First, the overall structure of an assessment of ecological status is given by the BQEs, which include (1) phytoplankton, (2) submerged aquatic vegetation and periphyton, (3) benthic invertebrates and (4) fish (Annex V, 1.2, Anon. 2000). Second, the method of combination of the quality elements into an integrated assessment of ecological status has in general been to apply the one-out-all-out principle (OO-AO) (Annex V, 1.4.2(i), Anon. 2000).

Only a few status assessment tools based on the use of multiple indicators are currently in use (OSPAR 2003, 2008; HELCOM 2009, Andersen et al. 2010, 2011; Fleming-Lehtinen et al. 2015). Within the EU WFD, however, most adhere to the one-out-all-out principle, although this principle has also been critically debated for potentially inflating the risk of misclassification and being based on simple heuristic rules rather than being rooted in

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Table 1 Quality Elements (groups of indicators) used for assessments according to the WFD, MSFD and regional marine conventions, i.e. HELCOM and OSPAR

Quality Elements	Lakes	Rivers	Coastal	MSFD	HELCOM/OSPAR
Phytoplankton	+	–	+	+	+
Submerged aquatic vegetation	+	+	+	+	+
Benthic invertebrates	+	+	+	–	+
Fish	+	+	–	–	(–)
Supporting indicators	(–)	(–)	(–)	+	+

Supporting indicators include nutrient and oxygen concentrations, as well as Secchi depth

‘+’ indicates use of the QE, ‘–’ indicates no use, and ‘(+)’ indicates an occasional use

ecosystem understanding (Alahuhta et al. 2009, Borja and Rodriguez 2010; Caroni et al. 2013).

This review of integration principles and methods also includes the currently used principles and tools for assessment of eutrophication status in the regional sea conventions for the Northeast Atlantic (OSPAR Common Procedure (OSPAR COMP); OSPAR 2003; Claussen et al. 2009) and the Baltic Sea (HELCOM Eutrophication Assessment Tool (HEAT); HELCOM 2009; Andersen et al. 2010, 2011), since these assessment tools also include coastal waters affected by nutrient enrichment and eutrophication. Specifically, HEAT was developed with an aim of aligning assessments in coastal waters and open basins of the Baltic Sea.

The combination of indicators within a given BQE has in our opinion not yet been addressed, neither by the WFD and its pan-European Common Implementation Strategy nor in the scientific literature. The objective of this study is therefore to: (1) provide an up-to-date analysis of assessment systems developed and applied in Nordic countries (Denmark, Finland, Norway and Sweden), including the monitoring systems on which the assessment are based, (2) review a variety of integration methods applied within the BQE level and (3) outline a way forward, arriving at integration principles enabling fully harmonised assessments of ecological status. The latter could potentially facilitate comparisons across water categories and potentially improve our understanding of what may trigger shifts in ecological status in downstream water bodies.

DATA AND ASSESSMENT METHODS

All of the Nordic countries have well-developed programmes for monitoring of surface waters. Focus has been on inland water and coastal waters, while the spatial and temporal coverage of open marine waters seems to be reduced compared to the coastal waters (e.g. HELCOM 2010). The amount of data generated is much larger than the amount of data used for temporal trend assessments for

individual indicators and for assessment of ecological and/or eutrophication status (See Annex 1).

The information available from all four Nordic countries provides a good overview, not only of data availability for various types of assessments, but also of the tools and integration principles (Table 1). The review of the degree of integration is based on available information for the BQEs phytoplankton, submerged aquatic vegetation and periphyton, and benthic invertebrates as well as supporting indicators, and for the following assessment types: lakes, rivers and coastal water (all WFD related) as well as MSFD (Anon. 2010) and OSPAR/HELCOM. In the following, we will use the broader-term quality elements (QE) for the combination of BQEs and supporting elements, i.e. indicators not specifically referring to a BQE. A summary of the QEs used can be found in Table 1, while a summary of the indices analysed in this review can be found in Table 3.

In addition to the scientific literature, this review is based on: (1) The Water Framework Directive (WFD) and the guidance documents from the Common Implementation Strategy, (2) HELCOM and OSPAR thematic eutrophication assessments and relevant background documents, and (3) published information from Denmark, Finland, Norway and Sweden regarding (1) the WFD and MSFD Initial Assessments and (2) national processes related to monitoring and development of indicator and indices. Information on monitoring of surface waters in Denmark, Finland, Norway and Sweden, including detailed descriptions of the indices assessed, are available as [electronically supplementary material](#).

Marine indicator-based eutrophication assessments are regularly carried out under the auspices of HELCOM (www.helcom.fi) and OSPAR (www.ospar.org). The indicators used are largely the same as those used for assessment of ecological status, e.g. chlorophyll-a concentrations, depth limits of submerged aquatic vegetation and community indices for benthic invertebrates. The indicators are grouped by OSPAR as: (1) causative effects, (2) direct effects, (3) indirect effects and (4) other effects, and by HELCOM along the lines of the WFD: (1)

	EQR	Disturbance	Status
OBSERVED VALUE in relation to REFERENCE VALUE	1.0 = 0.0	No, very minor or slight ----- Moderate, major or severe	= Acceptable ecological status ----- = Impaired ecological status

Fig. 1 Basic principle for classification of ecological status according to the WFD. The status is calculated as an ‘ecological quality ratio’ (EQR), being the ratio between observed value and reference value. The reference value is referred to in the WFD as reference conditions. The boundary between ‘acceptable’ and ‘impaired’ (not acceptable and requiring restoration measures), here represented by the dashed line, is set for individual indicators by Member States and is equivalent to the boundary between slight and moderate deviations from reference conditions. ‘Acceptable ecological status’ is in the context of the WFD equivalent to High or Good status, while the unacceptable ‘impaired ecological status’ is equivalent to Moderate, Poor or Bad status

Indicators	Good/Moderate boundary	Integration	Status
Indicator 1:	[RefCon; Good] [Moderate or lower]	}	High
Indicator 2:	[RefCon; Good] [Moderate or lower]		Good
Indicator 3:	[RefCon; Good] [Moderate or lower]		Moderate
			Poor
			Bad

Fig. 2 Overarching principles for integration of indicators within a BQE leading to a classification in five classes: High (including reference conditions), Good, Moderate, Poor or Bad. Only indicators with a numerically defined Good/Moderate boundary, which is calculated as the ‘reference condition’ $\pm$ an ‘acceptable deviation from reference conditions’, can be used for the assessment. In practice, the G/M boundary will vary from indicator to indicator, but should represent the same degree of deviation from the reference conditions

phytoplankton, (2) submerged aquatic vegetation, (3) benthic invertebrates and (4) supporting indicators (e.g. nutrients, Secchi depth and oxygen). Both HELCOM and OSPAR have developed and applied tools for integrated assessment of eutrophication status, i.e. the HELCOM Eutrophication Assessment Tool (HEAT) and the OSPAR Comprehensive Procedure (COMP).

The WFD defines principles for boundary setting, i.e. WFD Annex 5 (Anon. 2000) and a key assessment principle is the calculation of an ‘ecological quality ratio’, which is outlined in Fig. 1. Neither the WFD nor its annexes define principles or guidance on integration of indicators within a BQE, which could be seen as an injudicious support of the implementation process. Ecological status is defined normatively at the BQE level (see WFD Annex 5 and Fig. 2) and although the WFD does not define five classes at the indicator level, this has in many cases been ignored.

The assessment of eutrophication status in the Baltic Sea, based on the application of the HELCOM eutrophication assessment tool (HEAT), is in principle equivalent to an assessment of ecological status cf. WFD. When HEAT was developed, it was a requirement that the assessments of coastal waters and open-water basins should be fully harmonized and coordinated. Hence, HEAT has

applied a number of principles assumed to be compliant with WFD Annex 5 requirements in regard to the use of indicators, assessing target confidence, nesting in BQEs as well as integration and final classification using the OO-AO principle (see Andersen et al. 2011). It should, however, be noted that the OSPAR COMP differs slightly from WFD and HEAT integration principles, because it applies a ‘one out, all out’ principle between indicators in the same group (OSPAR 2003). This is stricter than what is done in the WFD, where the OO-AO principle is typically applied only among BQEs.

The integration of quality elements is generally straightforward and based on the OO-AO principle. In WFD, HEAT and OSPAR COMP, the most impaired group or BQE/QE (i.e. the group with the lowest status classification) is considered most sensitive to human activities and thus determines the final classification of ecological status.

As a consequence of the lack of joint principles for integration of indicators within QEs, as described above, a specific objective of this review has been to discuss principles for integration within (indicated by the curly bracket in Fig. 2) and between QEs. Based on this we synthesise, the information available and identify potential next steps toward improved and harmonized tools for integrated assessment of surface waters.

COMBINING QUALITY ELEMENTS INTO A WHOLE SYSTEM ASSESSMENT

The combination of various BQEs/QEs into a final assessment of ecological status or eutrophication status should be based on the OO-AO principle (Anon. 2000, HELCOM 2009). Examples of integration at the BQE/QE level as well as discussions of the application of the OO-AO principle and its strengths and weaknesses can be found in Alahuhta et al. (2009), Caroni et al. (2013), Borja et al. (2014), Fleming-Lehtinen et al. (2015) and Andersen et al. (submitted).

In Danish coastal waters, no integration of QEs has taken place in the first round of the WFD initial assessment, because classification was based on a single indicator, either eelgrass depth limit or chlorophyll-*a*. In the second round, more BQEs were available. Hence, the OO-AO principle has been applied in (i) coastal waters where information on two or three BQEs was available; (ii) lakes where more than one BQE was monitored (plankton [chlorophyll-*a* and a species index], submerged aquatic vegetation, and fish); and (iii) those streams where more than one of the following was monitored: submerged aquatic vegetation, invertebrates and fish.

The Finnish guidance documents (Vuori et al. 2009, Aroviita et al. 2012) outline the procedures for integrating QEs and accounting for issues with data quality. Similarly to Denmark, there was no need to integrate indicators in Finnish coastal waters because the classification was based only on the single biological indicators: chlorophyll-*a*, brackish water benthic index (BBI) or the growth limit zone of *Fucus* sp. In lakes and rivers in Finland, the OO-AO principle has not been directly applied among BQEs due to the misclassification risk (Alahuhta et al. 2009; Borja et al. 2009) when monitoring data inadequately represent anthropogenic pressures on the water body (Vuori et al. 2009). In contrast to the coastal assessments, the ecological status of lakes and rivers was determined in a comprehensive way emphasizing the importance of evaluating spatial and temporal representativeness of the monitoring data. First, a mean across normalized ecological quality ratios (EQRs) of BQEs was calculated for each water body (Vuori et al. 2009; Rask et al. 2011; Aroviita et al. 2012). Then the support from indicators of the physico-chemical and hydromorphological quality elements was included by regional experts into the status assessments by applying precautionary principle. When assigning the overall water body status class, the QE in worst status was given more weight, if the data were representative and the degradation of the BQEs was considered to be due to anthropogenic pressures.

In Norway, the continued update of the legislation (Vannforskriften 2006) and the most recent guidance

document (Veileder 02:2013 2013) on updated indices and integration methods of quality elements into integrated assessment form the basis for coastal water quality assessment.

The Swedish legislative texts (HVMFS 2013: 19) and guidance documents (Naturvårdsverket 2007) on the integration of quality elements into an overall assessment of ecological status outline a procedure which attempts to incorporate biological and supporting quality elements and to some degree account for uncertainties due to missing data. For the BQEs, the OO-AO principle is applied. Physico-chemical indicators are allowed to influence assessments by changing status from “good” to “moderate”, while hydromorphological indicators can only change assessments from “high” to “good” (HVMFS 2013:19). Although guidance documents developed for the first WFD-cycle include routines for quality and uncertainty assessment (e.g. Naturvårdsverket 2007), experiences during the first and second cycles have shown that issues related to procedures for expert judgment and missing data for certain quality elements have a large impact on how rules for integrated assessment are applied in practice. Improving and harmonising such routines among river basin district authorities is on-going and documented in an extended set of guidance documents (available from www.vattenmyndigheterna.se).

In contrast to the WFD initial assessments, the eutrophication status assessments made by Denmark, Finland, Norway and Sweden have all been based on multiple indicators and QEs, resulting in integrated assessments (OSPAR 2003, 2008; HELCOM 2009, 2014). Indicators and their target values are combined at the QE level. Assessments of eutrophication status are made at the QE level and are subsequently combined into an integrated assessment based on the OO-AO principle. Several tools are available and widely used, i.e. the OSPAR COMP and the HELCOM eutrophication assessment tool.

In addition to availability, the tools are transparent and documented by scientific literature. A key difference between the application of these eutrophication assessment tools and the WFD assessments carried out so far is the extensive use of multiple indicators per BQE/QE.

A country-wise overview of BQEs (phytoplankton, submerged aquatic vegetation, benthic invertebrates and fish) and QEs (supporting indicators, e.g. nutrient levels) used in the various types of assessments can be found in Table 2.

AGGREGATION AND INTEGRATION WITHIN QUALITY ELEMENTS: COMBINING INDICATORS

When it comes to aggregation and integration of indicators within BQEs/QEs, several principles and methods are in

Table 2 Country-wise overview of Quality Elements used in the various types of assessments

Assessment type	Phyto	SAV/PB	BI	Fish	Supp	Level of integration
Denmark						
Lakes (WFD)	+	—	—	—	—	No integration (based on single indicator)
Rivers (WFD)	—	+/-	+	—	—	Integration within BQE
Coastal (WFD)	+	+	—	—	—	A mix of 1, 2 or 3 indicators/BQEs
MSFD	+	+	+	—	—	Integrated assessment
OSPAR/HELCOM	+	+	+	—	+	Integrated assessment
Finland						
Lakes (WFD)	+	+/+	+ ¹	+	+	Integrated assessment
Rivers (WFD)	—	-/+	+	+	+	Integrated assessment
Coastal (WFD)	+/-	+*-	+*-	—	+	Integrated assessment
MSFD	+	—	+	—	+	Integrated assessment
HELCOM	+	—	+	—	+	Integrated assessment
Norway						
Lakes (WFD)	+	+	+	+	—	Integrated assessment
Rivers (WFD)	—	-/+	+	+	+	Integrated assessment
Coastal (WFD)	+	+/-	+	—	+	Integrated assessment
OSPAR	+	+	+	—	+	Integrated assessment
Sweden						
Lakes (WFD)	+	+	+	+	—	Integrated assessment
Rivers (WFD)	—	-/+	+	+	+	Integrated assessment
Coastal (WFD)	+	+	+	—	+	Integration within BQE
MSFD	+	—	+	—	+	Integrated assessment
OSPAR/HELCOM	+	+	+	—	+	Integrated assessment

A plus (+) indicates use of the QE, while a minus (—) indicates that the QE is not currently used. An asterisk indicates that multiple indicators or indices are being applied—no asterisk indicates that only a single indicator has been applied per group or BQE

WFD Water Framework Directive, MSFD Marine Strategy Framework Directive, *Phyto* phytoplankton, *SAV* submerged aquatic vegetation, *PB* phyto-benthos, *BI* benthic invertebrates, *Supp* supporting indicators, e.g. nutrients, oxygen and Secchi depth

¹ Both littoral and profundal

use (Borja et al. 2014). In essence, the methods applied include: (1) OO-AO, (2) TO-AO, also called the two out—all-out principle, (3) averaging (arithmetic or weighted mean), (4) scoring or decision tree approaches, (5) probabilistic methods, and (6) multi-metric or multivariate methods (please see Borja et al. 2014 for details).

In general, the aggregation of indicators and indices within each BQE/QE in Nordic countries is based on arithmetic means or weighted means. A few exceptions from this general rule have been identified, e.g. in Sweden different indicators are combined when using phytoplankton for assessing lakes. When indicators are weighted together, the parameters total biomass, trophic plankton index (TPI) and proportion of cyanobacteria, form the basis for the classification of the lake's status as regards nutrients. The use of TPI cannot be used unless four species in a sample have been assigned an indicator number. In some lakes the classification will be based solely on total volumes and cyanobacterial proportion. For lakes with high concentrations of *Gonyostomum semen*, the total biomass

parameter may be unsuitable, particularly if the biomass is very large. As such blooms are not necessarily a sign of eutrophication, and *Gonyostomum*-lakes are therefore quality-classed using the TPI value and cyanobacterial proportion instead of using total biomass. Parameters are weighted as follows: First, the weighting must be based on the status for total biomass, cyanobacterial proportion and TPI. The status classes are given a numerical value in accordance with Naturvårdsverket (2007). Second, the mean value for the numeric classes (Nclass) of the three parameters is calculated, which becomes the weighted classification of phytoplankton. The status classification is determined by the mean value for the numerical classification in accordance with Naturvårdsverket (2007).

It should be noted that the OSPAR COMP applies a simple OO-AO principle for indicators within a QE (OSPAR 2003), while the eutrophication assessments in the Baltic Sea (HELCOM 2009; Andersen et al. 2010, 2011) and the eastern North Sea (Andersen and Murray, in press) combine G/M boundaries by averaging, either

simple or weighted. Weights can be set by experts based on information on sampling frequencies, annual variations and uncertainty, or if the same weighting score is applied to all indicators, then the weighting is neutral.

AGGREGATION AND INTEGRATION WITHIN QUALITY ELEMENTS: MULTI-METRIC INDICES

Whereas aggregation of BQEs/QEs into a final-integrated assessment and aggregation of indicators within BQEs/QEs has been scrutinised and debated extensively (Borja et al. 2009; Borja and Rodriguez 2010; Caroni et al. 2013), the principles and methods for integration of indicators into indices have not yet been addressed in a comprehensive way. This review therefore represents a first attempt not only to have a cross-cutting look into principles for building indices for assessment of surface water status, but also for discussion of some unforeseen consequences of the approaches currently applied.

In total, we present and discuss 44 indices. The division across quality elements is 4 for phytoplankton, 15 for submerged aquatic vegetation and 25 for benthic invertebrates (Table 3). A detailed analysis of fish indices has been omitted since this BQE is not monitored and assessed in all surface water categories. Detailed information about the 44 indices can be found in annex S2. Due to methodological overlap between some indices, we have evaluated only 33. Annex S2 includes information about the clustering of indices.

Phytoplankton

Phytoplankton is generally not a relevant BQE in Nordic rivers and not included in the assessment systems. In lakes, Denmark, Finland and Sweden have applied phytoplankton indices, i.e. FPI (Søndergaard et al. 2013) and TPI (Willén 2007; Naturvårdsverket 2007; Löfgren et al. 2009). In addition, in Sweden and Finland two other indices are used: total biomass and % cyanobacteria (Naturvårdsverket 2007; Aroviita et al. 2012). In Norway, a multi-metric assessment method is used comprised combining the biomass metrics chlorophyll-*a* and total biovolume, the taxonomic composition metric PTINO and the bloom intensity metric maximum cyanobacteria biovolume (Miljødirektoratet, 2013). Information on phytoplankton indicators for coastal waters used in Nordic countries can be found in Högländer et al. (2013). No multi-metric phytoplankton indices have, to our knowledge, been used in Danish, Finnish, Norwegian and Swedish coastal waters. However, a new multi-metric phytoplankton index has recently been

tested for assessment of ecological status in Finnish coastal waters (Lugoli et al. 2012).

Submerged aquatic vegetation

In rivers and streams, only Denmark monitors submerged macrophytes (DVPI) at present, whereas in Finland, Norway and Sweden periphyton is currently used and macrophyte-based assessments are being developed. For streams in Sweden, the IPS index (Indice de Polluo-sensibilité Spécifique) is used to assess eutrophication and the ACID index to assess acidification (Naturvårdsverket 2007). In Finland, IPS was used in a preliminary system but currently river (and lake littoral) periphyton is assessed using two indices: occurrence of type-specific taxa (TT, Aroviita et al. 2008) and the percent model affinity (PMA, Novak and Bode 1992). The Norwegian periphyton PIT method focuses on non-diatom species and is sensitive to eutrophication, in particular phosphorous concentrations (Schneider and Lindstrøm 2011).

Lake macrophytes are assessed in Denmark, Finland, Sweden and Norway. A Danish lake macrophyte index has been developed (MFI; Søndergaard et al. 2013). In Finnish lakes, assessments of macrophytes are based on the Reference Index (RI, Penning et al. 2008), proportion of type-specific taxa (PTT, Vuori et al. 2009) and percent model affinity index (PMA, Novak and Bode 1992). In Swedish lakes, a macrophyte index has been applied (TMI; Naturvårdsverket 2007). The Norwegian national method includes a species composition index (TIC). The method has been designed to detect the impact from eutrophication on aquatic macrophytes and can be applied to boreal and lowland freshwater lakes in Norway (Miljødirektoratet 2013).

Assessments related to submerged aquatic vegetation in coastal waters are generally based on depth limits of specific species (Blomqvist et al. 2012), often eelgrass (Krause-Jensen et al. 2005) or *Fucus* (Bäck and Ruuskanen 2000; Torn et al. 2006). Denmark has developed and tested a multi-metric index for the total cover of macroalgae (TCI; Carstensen et al. 2008). In Finland, there were no multi-metric indices available for submerged aquatic vegetation in coastal waters, but a multi-metric index integrating three to five indicator macrophyte species has been finalized and is ready for the next status assessment (Ruuskanen 2014). In Norway, a multi-metric macroalgae index based on the species composition in the coastal zone has been applied. In Sweden, the multi-species maximum depth index (MSMDI) is currently used (HVMFS 2013:19, Blomqvist et al. 2012). This index is based on the mechanistic coupling between eutrophication and light penetration, but other approaches and mechanisms are currently being studied (e.g. Blomqvist et al. 2012).

Table 3 Overview of indices included in this review

No	Water category	Acronym	Country
Phytoplankton			
1	Coastal	ISS	F
2	Lakes	FPI	D
3	Lakes	TPI	F + S
4	Lakes	TPI _{NO}	N
Submerged aquatic vegetation or periphyton			
5	Coastal	TCI	D
6	Coastal	RSLA	N
7	Coastal	MSMDI	N, S
8	Lakes	MFI	D
9	Lakes	TMI	S
10	Lakes	PTT	F
11	Lakes	RI	F
12	Lakes	TT	F
13	Lakes	PMA	F
14	Lakes	Tlc	N
15	Rivers	IPS	S
16	Rivers	DVPI	D
17	Rivers	TT	F
18	Rivers	PMA	F
19	Rivers	PIT/AIP	N
Benthic invertebrates			
20	Coastal	DKI	D
21	Coastal	BBI	F
22	Coastal	NQII	N
23	Coastal	H'	N
24	Coastal	ES ₁₀₀	N
25	Coastal	ISI	N
26	Coastal	NSI	N
27	Coastal	DI	N
28	Coastal	BQI	S
29	Lakes	PICM	F
30	Lakes	PMA	F
31	Lakes	TT (lit.)	F
32	Lakes	PMA (lit.)	F
33	Lakes	BQI	S
34	Lakes	ASPT	S
35	Lakes	MILA	S
36	Rivers	DSFI	D
37	Rivers	TT	F
38	Rivers	TEPTF	F
39	Rivers	PMA	F
40	Rivers	ASPT	N
41	Rivers	RAMI	N
42	Rivers	ASPT	S
43	Rivers	DJ	S

Table 3 continued

No	Water category	Acronym	Country
44	Rivers	MISA	S

Country abbreviations are *D* Denmark, *F* Finland, *N* Norway, *S* Sweden. Each indicator is summarised in Annex 2 (see [electronic supplementary material](#)). For detailed descriptions of the individual indices, including overlap between indices, please confer with Annex 2. Lit. = littoral

Benthic invertebrates

The status of benthic invertebrates in rivers is assessed using a variety of indices. Denmark uses the DVFI (Skriver et al. 2000), Finland the TT, the PMA and occurrence of type-specific EPT-families (TEPTF; Vuori et al. 2009, Aroviita et al. 2012), Norway uses ASPT for organic pollution and general degradation and RAMI (sometimes AcidIndex 2) for acidity (Miljødirektoratet 2013) and Sweden uses the ASPT index for general ecological quality, DJ index (Dahl and Johnson 2004) for eutrophication and MILA for acidity (Naturvårdsverket 2007).

Benthic invertebrates in lakes in Finland are currently assessed applying a profundal invertebrate community metric (PICM; Jyväsjärvi et al. 2014) and in Sweden using a benthic quality index (BQI; Wiederholm 1980). The PICM is a derivative of the BQI with an extension that takes into account the whole assemblage instead of only chironomids. In Finland, the status of lake littoral invertebrate assemblages is assessed with two indices: the occurrence of type-specific taxa (TT; Aroviita et al. 2008) and the percent model affinity (PMA; Novak & Bode 1992). In Sweden, benthic invertebrate assemblages from lake littoral regions are used to assess general ecological status using ASPT (Average Score Per Taxon; Armitage et al. 1983) and a multi-metric index (MILA) for lake acidity (Naturvårdsverket 2007; Johnson et al. 2007).

Benthic invertebrates are not assessed in Danish and Norwegian lakes, neither in the littoral nor in the profundal zone in the context of the WFD. Furthermore, at a regional level, monitoring of benthic invertebrates has occurred in lake littorals to some degree using a “littoral Index” dating back to the early 1980s (Dall 1983). Recent re-analyses of these regional data have not resulted in the establishment of WFD compliant metrics (Wiberg-Larsen; pers. comm.), although a previous study of 39 lakes indicated that a metric sensitive to trophic status could be developed for Danish littoral zones (Brodersen et al. 1998).

All Nordic countries have national multi-metric indices for coastal soft-bottom invertebrate communities, i.e. BBI

in Finland (Perus et al. 2007), BQI in Sweden (Rosenberg et al. 2004; Leonardsson et al. 2009), DKI in Denmark and NQI in Norway (Josefson et al. 2009). The four indices are all multi-metric with several similar elements reflecting the status of invertebrate communities from a combination of species diversity and contribution from tolerant and sensitive species. All indices can be calculated from community data, but use different diversity indices and classification systems for weighting sensitive and tolerant species in the community. Several studies have described high correlation among several of the Nordic multi-metric methods (Perus et al. 2007; Josefsson et al. 2009; Carletti and Heiskanen 2009).

Brackish water benthic index (BBI) was developed for classification of invertebrate assemblages for the Finnish low-saline and species poor coastal waters in the Baltic Sea (Perus et al. 2007). The naturally low diversity of the Baltic Sea ecoregion is compensated by the maximum BQI value for a specific coastal type and depth range (BQI_{max}) and the maximum Shannon–Weaver index for a specific coastal type and depth range (H'_{max}). The benthic index has proven to be a useful tool for classification of the invertebrate community.

Benthic quality index (BQI) was developed for classification of coastal waters along the salinity gradient from the low-saline and species-poor Bothnian Bay in the Baltic Sea to the full saline and species-rich Skagerrak, using different sensitivity values in Baltic Sea compared to Kattegat and Skagerrak (Rosenberg 2004). BQI classification has been less useful in species-poor areas of the Baltic, where species are classified into defined sensitivity classes compared to continuous values used in the Kattegat and Skagerrak areas.

Danish quality index (DKI) has had problems to adjust for communities in areas of reduced salinity and diversity but has recently been modified with a salinity component to enhance the method performance in Baltic Sea areas (Josefsson et al. 2009).

Norwegian quality index (NQI) was developed for classification of coastal waters in the species-rich North Sea and has been intercalibrated with a wide variety of European indices in all ecoregions, where comparable water types exist. Class boundaries for ecological status for each of the WFD indices have been developed for the Norwegian coast and intercalibrated between countries with similar water types (Carletti and Heiskanen 2009). Several additional indices (H' , ES_{100} , ISI, NSI and DI; see Veileder 02:2013 2013) have been described in the new Norwegian guidance document, including the multi-metric method NQII to assess status classification, the use single indices to facilitate interpretation along gradients of different stressors and approaches for averaging indices (Veileder 02:2013 2013). H' and ES_{100} are standard

diversity indices and increase with relative increasing species richness compared to abundance. ISI uses a Norwegian list of sensitivity values without abundance weighting, whereas NSI uses a Norwegian list of continuous sensitivity values and abundance weighting calculation similar to AMBI (Rygg and Norling 2013).

CONCLUSIONS AND SUGGESTED NEXT STEPS

An observation emerging from this study is that application of the OO-AO principle between BQEs is not being used as conservatively as stipulated in the WFD. So far, this principle has been considered as carved in stone, but a closer reading of the WFD seems to reveal a number of potentially contradictory elements in the directive.

For operational monitoring, WFD states: “... *In order to assess the impact of these pressures, Member States shall monitor as relevant: parameters indicative of the BQE, or elements, most sensitive to the pressures to which the water bodies are subject*” (Anon 2000, Annex V, 1.3.2). An interpretation of this could be that in impaired water bodies where the good ecological status target is not being met, other elements than the BQEs should be monitored if regarded as sensitive to pressures and indicative of the BQEs, e.g. nutrient enrichment. A practical implication would be that nutrient concentrations should always be monitored in waters sensitive to or affected by eutrophication.

Further, the WFD states: “*For surface water categories, the ecological status classification for the body of water shall be represented by the lower of the values for the biological and physico-chemical monitoring results for the relevant quality elements classified in accordance ... with table ...*” (Anon 2000, Annex V, 1.4.2). An interpretation of this may imply that the lowest classification of either BQEs or supporting QE such as nutrient concentrations should determine the final classification. In practice, this could mean that nutrient concentrations could be decisive for the final classification if classified lower than any of the BQEs. Furthermore, the interpretation indicates that the OO-AO should not necessarily be applied across the BQEs, but between biota and physico-chemistry.

The WFD was adopted in 2000 and now 15 years after its adoption, numerous indicators and indices have been developed and applied for classification of ecological status (Birk et al. 2012). Despite a pan-European Common Implementation Strategy (CIS) providing comprehensive guidance and two rounds of intercalibration of indicators, apparently no tools for assessment of ecological status of surface waters have been subject to a joint and harmonized pan-European development.

The integration of BQEs/QEs using the OO-AO principle has been considered simple and transparent. In

contrast, integration of indicators within BQEs/QEs uses methods from simple averages to complex aggregations, with substantial variation (e.g. Borja et al. 2014). The absence of a harmonised approach complicates comparisons between different groups of indicators, not only within water bodies but also across water bodies and water categories. Considering specific indices, the variety of integration principles and methods is considerable (see Annex S2), probably because indices have been developed through a bottom-up process. An observation worth mentioning is that a large number of the indices assessed can be used to give a classification in one of five classes. A WFD target value, i.e. value representing the G/M boundary, is in principle defined per indicator, while the five quality classes (H, G, M, P, and B) are defined per BQE/QE.

Aiming for harmonisation and coordination at the index level at this stage is probably a lost cause. However, aiming for harmonisation at the BQE/QE level and also for harmonisation of how BQEs/QEs are combined into a whole system assessment is something worth pursuing. This could potentially be attained through the development and application of harmonised multi-metric indicator-based assessment tools (MIBATs) such as HEAT. The benefits of using MIBATs are likely to include: (1) an ability to make cross-BQE/QE comparisons, (2) an improved capability of making cross-water category comparisons and (3) potentially also an improved understanding of downstream changes in ecological status as well as their upstream causes.

Development of WFD and MSFD-specific multi-metric indicator-based assessment tools alike the tools currently used by HELCOM and OSPAR for assessment of eutrophication status could potentially represent a step forward in regard to assessment of ecological status, especially if the tools are not water category specific. We do not envisage a pan-European process leading to the next generation of MIBATs but rather national and regional testing and application of MIBATs. Ultimately, Member States and Regional Marine Conventions (e.g. HELCOM and OSPAR) have an interest in using tools that allow not only for comparisons between water categories or sub-basins but also allow for comparisons between downstream and upstream waters bodies.

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