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Redundancy in the ecological assessment of lakes: Are phytoplankton, macrophytes and phytobenthos all necessary?

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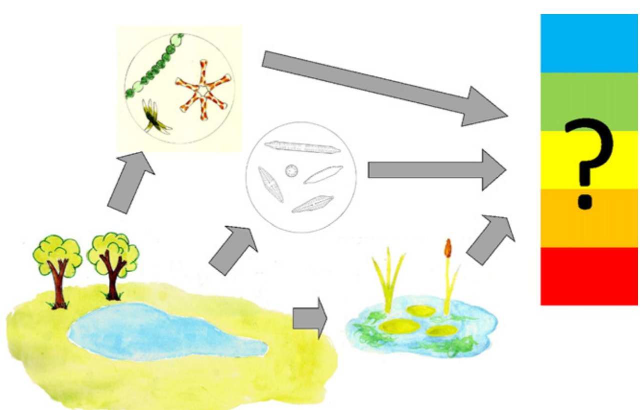
HIGHLIGHTS

- High costs of data mean that the value of different types is evaluated carefully.
- The benefits of phytobenthos to lake classification was examined.
- The added value of phytobenthos as a third indicator was assessed.
- Most impacted lakes were detected using phytoplankton and macrophytes.
- Few additional lakes were detected using phytobenthos in addition to these.
- There are some specific situations where phytobenthos has relevance for lake assessment.

GRAPHICAL ABSTRACT



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ABSTRACT

Although the Water Framework Directive specifies that macrophytes and phytobenthos should be used for the ecological assessment of lakes and rivers, practice varies widely throughout the EU. Most countries have separate methods for macrophytes and phytobenthos in rivers; however, the situation is very different for lakes. Here, 16 countries do not have dedicated phytobenthos methods, some include filamentous algae within macrophyte survey methods whilst others use diatoms as proxies for phytobenthos. The most widely-cited justification for not having a dedicated phytobenthos method is redundancy, i.e. that macrophyte and phytoplankton assessments alone are sufficient to detect nutrient impacts. Evidence from those European Union Member States that have dedicated phytobenthos methods supports this for high level overviews of lake condition and classification;

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however, there are a number of situations where phytobenthos may contribute valuable information for the management of lakes.

1. Introduction

The Water Framework Directive (WFD: European Union, 2000) is based on the principle that healthy ecosystems are the basis for sustainable water resources. The various components that comprise a healthy ecosystem are interconnected (e.g. via food webs) and will, in turn, provide ecosystem services as well as having sufficient intrinsic resilience to counteract short-term impacts. The overall condition of these components for any water body is the "ecological status", a term with a very similar meaning to "ecological health" or "ecological integrity". The definition, as given in the WFD, breaks ecological status down into components reflecting the physical, chemical and biological state of the water body, and each of these is further divided. In the case of biological quality elements (BQEs), particular characteristics ("species composition", "abundance") of named groups of organisms ("phytoplankton", "benthic invertebrates" etc.) that should be assessed are prescribed in Annex V and it is easy to lose sight of the holistic principles behind the legislation amidst all the detail. As the objective of the WFD is to raise all water bodies to at least "good ecological status" (GES), assessment serves not just to determine the condition of the biota with respect to this objective, but also to diagnose reasons for failure to achieve GES. In practice, the widespread nature of common problems in lakes (e.g. eutrophication) means that the role of assessing status and diagnosing causes can overlap, and this in turn suggests a potential for redundancy: if BQE 1 indicates that the lake is eutrophic, then why measure BQE 2, if that, too, is responsive to nutrients? As ecological assessment is an expensive activity, savings made could free up resources for more efficient use elsewhere (Lovett et al., 2007). Yet, at the same time, such savings come at a cost to the holistic insights that should arise from having information from several interconnected components of the ecosystem and may affect confidence in ecological assessments and hence the willingness to take action (Moss, 2008).

The WFD and subsequent European Commission documentation gives countries leeway in deciding national approaches to ecological assessment, representing the guiding principle of "subsidiarity", which underlies all European law (European Union, 2002, Article 5). For example, it is not necessary to use a BQE (or, by inference, part of a BQE) if "… it is not possible to establish reliable type-specific reference conditions … due to high degrees of natural variability in that element, not just as a result of seasonal variations" (WFD: Annex II, 1.3.). Moreover, a key principle of the EU's intercalibration exercise (see Poikane et al., 2015) is that where a BQE consists of two components, "… it may be sufficient to use only one of the two components" (European Commission, 2010). European Commission (2010) go on to say that "It is up to the Member State to decide how it develops its methods. If only one component is used then it must be demonstrated that the impacts of the existing pressures are being sufficiently detected by that component."

The assessment of "macrophytes and phytobenthos" in lakes and rivers represents one particular instance where the issue of a potentially redundant metric occurs. These two very different components of the benthic freshwater flora are generally assessed separately (Kelly et al., 2015; Poikane et al., 2015) but are included as a single BQE in Annex V of the WFD which, in turn, has led some countries to argue that assessment of phytobenthos (i.e. benthic algae, or "periphyton") is "redundant" because their national assessment system for macrophytes is adequate to detect the pressures to which phytobenthos are sensitive (e.g., Pall and Moser, 2009). This is despite a widespread understanding that macrophytes and phytobenthos react at different time and spatial scales, e.g. macrophytes generally react over yearly time scales to changes in pollution whereas phytobenthos can react within days or even hours (Schaumburg et al., 2004; European Commission, 2010). However, in lakes, unlike most rivers, phytoplankton are also assessed and some countries have argued that these provide an adequate proxy for the rapidly-reacting component. Such arguments, however, bypass functional ecology and focus on a superficial value of different biological components as "indicators" (Moss, 2008). It could equally be argued that phytobenthos and macrophytes provide complementary roles in the structure and carbon-flow within river and lake littoral ecosystems, thus rendering phytoplankton redundant, whilst Trobajo et al. (2002); Jones and Sayer (2003); Moss (2010) and others demonstrate how all three components interact with each other and with invertebrates and fish to maintain ecological integrity in shallow lakes. This broadens the debate from simply considering how including or excluding a component influences the high-level classification of water bodies, to thinking about the types of information that a lake manager might need in order to restore a water body to GES.

The current paper, therefore, aims to gather together data from those countries within the EU that have separate macrophyte and phytobenthos assessment systems (the latter based on diatoms as proxies for the whole benthic algal community), in order to test whether redundancy exists. A further source of confusion lies in the inclusion of filamentous macroalgae in some macrophyte-based assessment systems (most of which already include charophytes). A purely legal interpretation of the WFD would suggest that countries which adopt this practice have fulfilled their obligations. Therefore, a further set of analyses looks at the unique contribution that filamentous macroalgae make to one macrophyte assessment system (UK; Willby et al., 2009). Finally, we consider situations where a separate phytobenthos method may provide additional insights over and above a statistically-driven approach to classification of ecological status.

2. Methods

2.1. Theoretical consideration of redundancy

Several countries claim that phytobenthos analysis in lakes is redundant because it offers no additional information over and above that provided by macrophytes and/or phytoplankton (Pall and Moser, 2009). However, to be objective, this concept needs to be translated into terms relevant to the WFD. If we argue that the purpose of ecological assessment is to detect change due to anthropogenic pressures, then the null hypothesis for these assessments is that such pressures have no more than a slight impact on the biota of a particular water body (i.e. corresponding to the definition of GES). Consequently, "redundancy" can be defined as omission of a BQE (or sub-element) that will have a low risk of a Type 2 error (erroneous retention of null hypothesis); in other words, we are unlikely to wrongly classify an impacted lake as being at GES or High Ecological Status (HES) when following the classification guidance given in the WFD. This stipulates that the final status of a water body is defined by the lowest of the measured BQEs (i.e. the "one out, all out" principle). In practice, "macrophytes and phytobenthos" form a single BQE. The analyses that follow assume that Member States use the most stringent of the two sub-elements to determine the classification; however, a few Member States (e.g. Germany: Schaumburg et al., 2004) prefer to average these sub-elements.

Type 2 errors are assessed by statistical power analysis, which traditionally uses a $P = 0.8$ threshold (e.g. Dalgaard, 2002). Therefore, we have assumed that redundancy exists if 80% of water bodies are classified to the same class using different methods, and 100% are classified to ± 1 class. A simulated dataset was constructed in Excel, using the RAND function in association with a constant to convert two populations each of 100 linearly-arranged data points arranged along a scale 0–1 (corresponding to Ecological Quality Ratios, EQRs) into two populations of points whose relationship to one another could be varied by adjusting the constant. Trial and error was used to adjust the constant until there was 80% agreement between x and y (corresponding to BQE1 and BQE2). Fig. 1a shows what 80% agreement of class looks like, assuming a 1:1 relationship. Assuming status class boundaries occur at EQR = 0.2, 0.4, 0.6 and 0.8 then 81% of sites are assigned to the same class using both BQEs; the coefficient of determination (r^2) for the illustrated relationship is 0.87.

However, this strength of relationship was never encountered in practice (see below) and Fig. 1b shows a relationship that is more typical of the relationships agreement between macrophyte and phytobenthos assessment results (49% agreement to same class; 94% agreement to one class; $r^2 = 0.70$) but retaining slope = 1:1. This lower level of agreement can be explained by interactions with non-pressure variables as well as by different response times of algae and higher plants and sensitivities to other pressures.

In practice, however, few of the relationships examined had slope = 1 and this, too, can contribute to redundancy between metrics. This is illustrated in Fig. 1c, where the data have similar dispersion to that in Fig. 1b but, this time, BQE2 is more stringent than BQE1, and largely determines the final class based on application of the WFDs 'one out all out rule'. Therefore, we could argue that BQE1 is redundant, despite the relatively low correspondence between the two metrics. A further point in support of redundancy is that the area where BQE1 is less precautionary does not correspond with the important good/moderate boundary (EQR = 0.6, assuming normalised axes) so lakes that are at less than good status are unlikely to be misclassified as good status or higher if BQE1 is not used. This situation could represent genuine differences in type or rate of response between the two BQEs (for example, macrophytes are also highly sensitive to hydromorphological alteration), but it could also reflect how the methods were developed (for example, were the same reference concepts and boundary setting procedures used?).

In summary, Fig. 1a represents "reciprocal redundancy", where the two metrics have sufficiently similar pressure-response relationships that the information gain from using both is small, whereas Fig. 1c represents "stringency-based redundancy", where one of the pair of metrics is substantially more sensitive to the pressure(s) under consideration.

These simple models can now be extended to simulate the benefit of adding additional metrics to an assessment, assuming that each has a

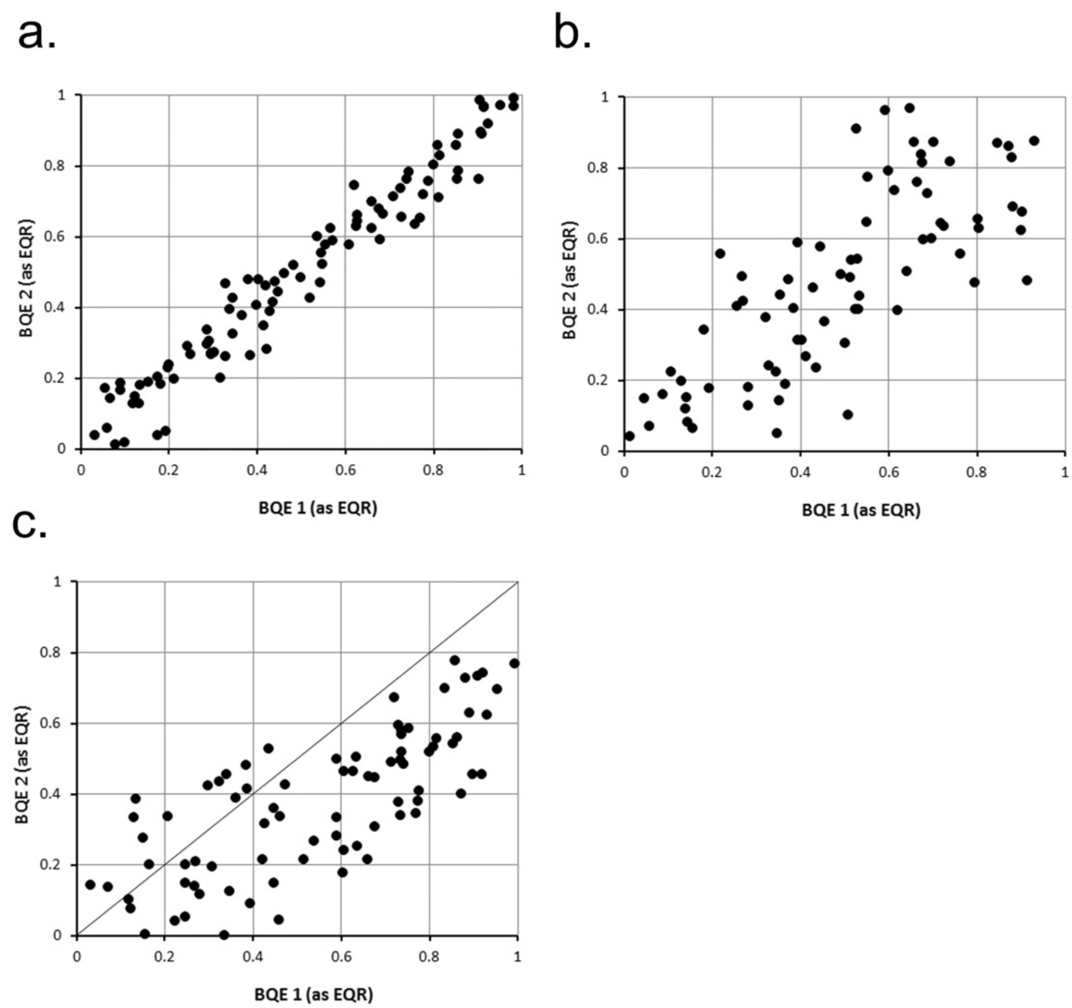

Fig. 1. Scenarios for redundancy between complementary biological quality elements (BQEs) for classifying water bodies. BQEs are expressed as Ecological Quality Ratios (0–1) assuming status class boundaries at 0.2, 0.4, 0.6 and 0.8. a. High redundancy (80% classified to same class, 100% classified to ± 1 class, $r^2 = 0.87$, slope = 1; b. Moderate redundancy, 49% classified to same class, 94% to ± 1 class, $r^2 = 0.70$, slope = 1; c. High redundancy, based on same strength of association as b. but with slope < 1; 44% classified to same class; 82% to ± 1 class.

"noisy" relationship both to the underlying pressure gradient(s) and to each other (Fig. 2). Assuming that a risk of a Type 2 error exists, it is useful to know how much this risk will decrease as additional metrics are deployed.

The criterion that is tested here is the ability to detect lakes that are truly below GES and Fig. 2 shows the results of 10 simulations for each of four scenarios (1, 2, 3 and 4 BQEs assessing the same pressure gradient) based on randomly-generated data with slope = 1 and dispersion similar to that shown in Fig. 1b. Where a single metric was used, an average of 58 out of 100 lakes were classified as less than good status (assuming good/moderate status threshold = 0.6). When two metrics were used, a further ten lakes were identified as < GES. However, the number of additional lakes identified declined steeply when a third or fourth metric was deployed (the "Law of Diminishing Marginal Utility"; Lipsey and Chrystal, 1995).

2.2. Analyses of national datasets

Following the theoretical analysis of redundancy, datasets were obtained from countries that used phytoplankton, macrophytes and phytobenthos as part of their classification systems. Data were obtained from routine assessment programmes and there were variations in the ways that data were aggregated. In most cases, all biological components and chemistry were collected in the same year; where multiple samples were collected, these were aggregated to the year of collection. In a few cases, biological and chemical elements came from different years, but were used as the basis for comparisons on the advice of national experts (i.e. knowing that there were no major changes in lake conditions over the period of data collection).

Ten countries provided datasets that were subjected to the following analyses:

1. Exploration of relationship between phytoplankton, macrophytes and phytobenthos metrics. The proportions of sites or samples (depending on the national dataset) assigned to the same class, or to ±1 class, based on intercalibrated boundaries were calculated to give an indication of the redundancy between two metrics. In a few Member States, provisional boundaries were used for metrics that had not been formally intercalibrated.
2. Pressure-response relationships between EQRs for phytoplankton, macrophytes and phytobenthos and TP and TN (or other N fractions). Results assessed as the significance of the regression (as F) and the coefficient of determination (r^2). No phytoplankton EQR is available for Denmark; pressure-response relationships are therefore expressed as the relationship between chlorophyll concentration and nutrients.
3. The marginal utility of phytobenthos, assuming that phytoplankton and macrophytes are the preferred BQEs. For this analysis, lakes were divided into two groups: those deemed to be at high or good status using all three assessments (phytoplankton, macrophytes and phytobenthos), and those for which one or more assessments failed to demonstrate high or good status. The latter group were then filtered to determine the number of lakes where phytobenthos was the only one of the three assessments to demonstrate less than good status. These lakes represent the "false negatives" that would be missed if only phytoplankton and macrophytes assessments were used.
4. A modification of this analysis shows the sum of all the high and good status sites along with those determined to be at less than good status on the basis of phytobenthos analyses alone. This tests the assumption that phytobenthos alone is a reliable means of assessing the ecological status of a lake.

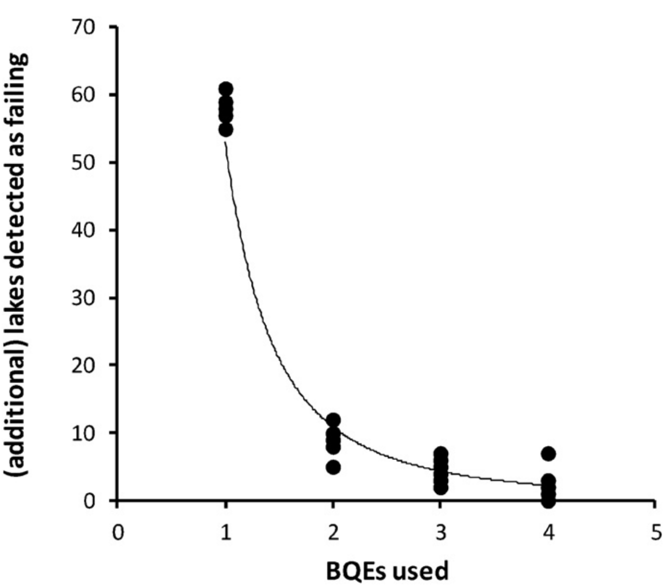

Fig. 2. Diminishing marginal utility of employing extra BQEs in lake assessment: simulations based on data with similar dispersion to that used in Fig. 1b and assuming a 1:1 relationship between each slope and each metric having identical response to stresses.

Full details of analyses of national datasets are given in the Supplementary material.

2.3. Impact of filamentous algae on macrophyte assessments in lakes

The possibility that inclusion of filamentous algae in macrophyte surveys may fulfil a country's obligation to assess phytobenthos was examined by analysing a dataset of 4500 surveys of UK lakes performed using LEAFPACS, the UK's macrophyte assessment tool which comprises five separate metrics, including Lake Macrophyte Nutrient Index (LMNI), an index based on measured TP optima which assesses trophic status based on the composition of macrophytes (including macroalgae) (Willby et al., 2009, UK TAG, 2014). Within LEAFPACS there is a conditioning metric that deals solely with the relative cover of filamentous algae and which can, under some circumstances, lower the overall classification of a site. To assess the influence of algal cover as an additional metric on classifications 4500 surveys covering a range of lake types were assessed with and without consideration of relative algal cover. The null hypothesis was that exclusion of filamentous algae had minimal effect on the final classification of lakes.

3. Results

3.1. Comparison of phytoplankton, macrophytes and phytobenthos assessments

The issue of redundancy was explored by analysis of lake datasets provided by those MS who have both phytobenthos and macrophyte assessment methods. Data for eight countries were included in this exercise (Denmark was excluded as no data were available for their phytoplankton method and Slovenia was excluded as data were only available for two lakes). Overall, median agreement between methods ranged from 40% (phytoplankton v macrophytes) to 54.5% (phytoplankton v phytobenthos; Table 1) but no MS achieved the 80% threshold suggested above to demonstrate redundancy. Macrophytes were the most stringent element (i.e. most likely to cause a downgrade in class) in all cases except for Italy, perhaps reflecting their sensitivity to hydromorphological as well as eutrophication pressures (Table 2).

Phytobenthos and phytoplankton EQRs tended to have stronger relationships with TP (Table 3), despite the generally more stringent response of macrophytes compared to these BQEs. It is also important to recognise that the phytoplankton is, itself, a significant component of TP and that there is, as a result, some auto-correlation in the phytoplankton-TP relationship. Not all countries had data for nitrogen, but where data were available, phytoplankton showed the strongest relationship in three cases and macrophytes in one (Table 4). No statistically significant relationships were observed for the remaining four instances.

The final set of analyses look at the marginal utility of phytobenthos, over and above phytoplankton and macrophyte analyses. In practice, few lakes were detected as failing GES using phytobenthos alone, once the data had been screened using the other two BQEs (Table 5). Some caution needs to be applied when interpreting these data, particularly where datasets were biased towards lakes at HES or GES (e.g. Sweden).

Based on the definition of redundancy outlined above, it is also possible to turn the argument around and suggest that phytobenthos alone is adequate for detecting impacts on lakes, particularly if the primary concern is eutrophication and the risk from hydromorphological alteration is low. Fig. 3 shows the percentage of impacted sites (i.e. less than good status) that were detected by phytobenthos alone, combined with the percentage of non- or only slightly-impacted sites (high or good status). Note that the sensitivity of this analysis is strongly affected by the distribution of sites between "impacted" and "non-impacted" categories and the possibility that non-nutrient pressures were responsible for some of the differences also needs to be considered. Over 80% of lakes were classified correctly (i.e. based on outcome of phytoplankton, macrophytes and phytobenthos) by phytobenthos alone in four of the nine countries. However, in three countries fewer than 60% of lakes were assigned to the same class by phytobenthos alone compared to when three biological elements were used. Further analyses looked at the effect of lake type within each national dataset, but no systematic trends were apparent.

3.2. Impact of filamentous algae on macrophyte assessments in lakes

A further possibility is that MS which include macroalgae within their macrophyte assessment systems are fully compliant with the WFD normative definitions because they have a single, integrated "macrophyte and phytobenthos" assessment method, albeit often referred to as a macrophyte assessment method. This section examines this claim, using the UK's lake macrophyte assessment system as a test case. This is a multimetric system that includes a "filamentous algae" metric based on the relative abundance of algae identified either to genus (e.g. *Cladophora*, *Hydrodictyon*) or higher categories ("filamentous green algae"). Filamentous algae also contribute to the compositional metric LMNI which is calculated based on the entire flora.

Unless filamentous algae are associated with LMNI species scores substantially different from those of other members of the vegetation at a site, their overall effect on the compositional metric value will be small. Fig. 6 confirms that *in general* there is a close relationship between the value of the compositional metric (LMNI) calculated with or without the inclusion of filamentous algae. Difficulties in field identification mean that different types of filamentous algae are often "lumped" into an indeterminate group (rather than named species or genera) by macrophyte surveyors. The broad ecological amplitude of such filamentous algae aggregates means that their weights in the UK metric are around the mid-point of the global range of species scores. This mid-point will be above the typical site score for a low alkalinity upland lake (low LMNI site scores). Consequently, even a modest relative abundance of filamentous algae will have some effect on the site score in such lakes, especially if the vegetation is relatively species poor. The opposite effect is observed in high alkalinity lowland lakes (high LMNI site scores) because the score associated with filamentous algae tends to be lower than that associated with other common members of the flora.

3.3. Effect of including filamentous algae on classifications in lakes

High cover of filamentous algae in UK lakes is generally uncommon, with an average cover in the UK lake macrophytes database of 3%. This average rises to 13% when surveys in which no filamentous algae were recorded are excluded. High cover of filamentous algae can occur across a wide range of vegetation composition in lakes; however, the highest relative cover of filamentous algae (algal cover/total macrophyte cover), tends, as expected, to occur in communities associated with higher alkalinity, fertile lakes (Fig. 4). Survey timing may have an effect here: extensive filamentous algal cover is often observed early in the season in shallow enriched lakes, particularly those where nutrient reduction has resulted in a spring clear water period, but may then give way to other macrophytes.

High cover of filamentous algae can occur across the ecological status gradient (Fig. 5). Whether higher relative cover of algae should be permissible at high or good ecological status is debatable but the potential for a diverse assemblage of more nutrient sensitive macrophyte taxa

Table 1
Proportions of lakes assigned to the same (or ±1) class for different combinations of BQEs. N = number of lakes in the national dataset.

MS	N	Phytoplankton v macrophytes		Phytoplankton v phytobenthos		Macrophytes v phytobenthos	
		Same class	±1 class	Same class	±1 class	Same class	±1 class
BE (FL)	18	12	82	41	88	53	76
FI	24	38	83	62	88	33	83
FR	47	51	100	51	94	28	98
HU	46	33	46	30	76	43	76
IE	56	45	91	59	96	54	88
IT	19	32	90	58	95	47	90
SE	33	52	94	79	91	54	91
UK	66	42	84	50	86	38	85
Median		40	87	54.5	89.5	45	86.5
Lowest		12	46	30	76	28	76
Highest		52	100	79	96	54	98

Table 2
Relative bias towards different BQEs. The most stringent element is listed first (based on analyses used in this table), followed by remaining elements in order. ">" indicates that the relationship on the left is considerably stronger; "=" indicates both relationships had similar statistical strengths.

MS	Order
BE(FL)	Macrophyte > phytobenthos > phytoplankton
FI	Macrophyte > phytoplankton > phytobenthos
FR	Macrophyte > phytoplankton > phytobenthos
IE	Macrophyte > phytobenthos > phytoplankton
IT	Phytoplankton = phytobenthos > macrophytes
HU	Macrophyte > phytoplankton > phytobenthos
SE	Macrophyte > phytoplankton > phytobenthos
UK	Macrophyte > phytobenthos > phytoplankton

Table 3
Strength of relationships between BQEs and total phosphorus (TP). Bold text indicates a statistically-significant relationship. ">" indicates that the relationship on the left is considerably stronger; "=" indicates both relationships had similar statistical strengths.

MS	Order
BE(FL)	**Phytobenthos** > macrophytes = phytoplankton
DK	**Phytobenthos** > (phytoplankton) = macrophytes
FI	**Phytoplankton** > **phytobenthos** > **macrophytes**
FR	Phytoplankton = macrophytes = phytobenthos
HU	**Phytobenthos** > **macrophytes** > **phytoplankton**
IE	**Phytoplankton** > **macrophytes** > **phytobenthos**
IT	**Phytobenthos** > **phytoplankton** > **macrophytes**
SE	**Phytoplankton** > **phytobenthos** > **macrophytes**
SI	**Phytoplankton** > **macrophytes** > phytobenthos
UK	**Phytobenthos** > **phytoplankton** > **macrophytes**

Table 4
Strength of relationships between BQEs and total nitrogen or similar fractions. Bold text indicates a statistically-significant relationship. ">" indicates that the relationship on the left is considerably stronger; "=" indicates both relationships had similar statistical strengths.

MS	Order
BE(FL)	**Phytoplankton** > macrophytes = phytobenthos
DK	(Phytoplankton) = macrophytes = phytobenthos
FI	**Phytoplankton** > **phytobenthos** > **macrophytes**
FR	Phytoplankton = macrophytes = phytobenthos
HU	**Phytoplankton** > macrophytes = phytobenthos
IT	Phytoplankton = macrophytes = phytobenthos
SE	**Macrophytes** > **phytoplankton** > **phytobenthos**
SI	Phytoplankton = macrophytes = phytobenthos

to coexist with high algal cover might suggest that high algal cover in lakes is often ephemeral and therefore has limited long term impact on rooted macrophytes.

The overall effect on classifications of integrating relative algal cover into assessments is very small. The average EQR is lowered by about 0.008, or about 1% of a class, rising to 0.02 if sites with no recorded algal cover are excluded. There is no evidence that the bias associated with including filamentous algae differs between lake types. The size of change in EQR was large enough in only 0.3% of cases for the class to be altered by the inclusion of algae. Of course, assigning a higher a priori weight to filamentous algae when determining an overall classification would change this picture, with the inclusion or exclusion of filamentous algae then becoming more influential. However, the strength of the relationship between filamentous algae and lake TP was sufficiently weak that a higher weighting in favour of filamentous algae was difficult to justify in the UK method.

4. Discussion

The question of the complementarity of different types of biological assessment has been addressed before (Resh, 2008; Rimet et al., 2015; Schneider et al., 2012; Eigemann et al., 2016) but not from a perspective that permits a critical analysis of the marginal benefits of adding additional measurements. The current study differs from earlier work in that it recognises the cost of ecological assessment and, therefore, the need to be rigorous in determining the unique information that each component adds. In particular, we see that information gathering to support lake management is an iterative process, and recognise that different stages of this process may require different types of information (DeNicola and Kelly, 2014). In particular, we separate the process of classification (i.e. placing each water body into an appropriate status class) from the diagnosis of problems and developing a programme of measures to restore an impacted lake to GES.

Table 5
Number of lakes (by MS) that are either at high or good status (HES or GES) or where the class is determined by phytobenthos (i.e. where phytoplankton and macrophytes both indicate HES or GES). The Annexes include breakdowns of these data by lake type, but as no clear patterns emerged, these have been omitted from this table for the sake of clarity.

MS	N	HES or GES	Class determined by phytobenthos
BE (FL)	17[a]	7	0
FI	24	9	0
FR	47	30	3
HU	46	3	3
IE	56	33	0
IT	19	17	
SE	33	27	0
SI	6	3	0
UK	66	23	0

[a] One lake (Galgenweel) omitted due to slight brackish influences which had a disproportionate effect on phytoplankton and macrophyte classifications.

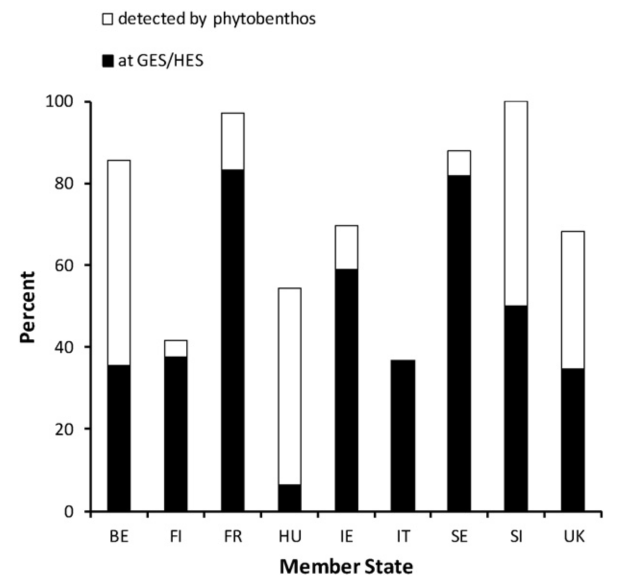

Fig. 3. The percentage of sites in each national dataset that are either at GES or HES (based on all BQEs) or are detected as being < GES using phytobenthos alone. The solid part of the bars indicates the proportion of sites unambiguously classified as high or good status by phytoplankton, macrophytes and phytobenthos; the open part of the bar shows the proportion of the remaining sites that were detected as being less than good status by phytobenthos.

The theoretical results (Figs. 1 and 2) appear to justify the use of at least two primary producer components for classification, as approximately 20% more lakes below GES were identified when a second BQE was deployed, compared to the situation when just one was used. These results, however, assume a 1:1 relationship between metrics and the effects would become increasingly pronounced as the relationship deviated from this ratio (as is the case for macrophytes compared with both phytoplankton and phytobenthos). A further possibility is that deviations from 1:1 represent lakes in non-equilibrium states, or reflecting a time lag as macrophytes (longer life span, persistent propagules and dependent upon sediment nutrients) respond more slowly than phytoplankton (Eigemann et al., 2016) or phytobenthos. In addition, there is growing evidence that nitrogen has a significant structuring effect on macrophyte assemblages (González Sagrario et al., 2005; James et al., 2005; Barker et al., 2008). There is no a priori reason why phytobenthos has to be the third choice; however, this seems to be

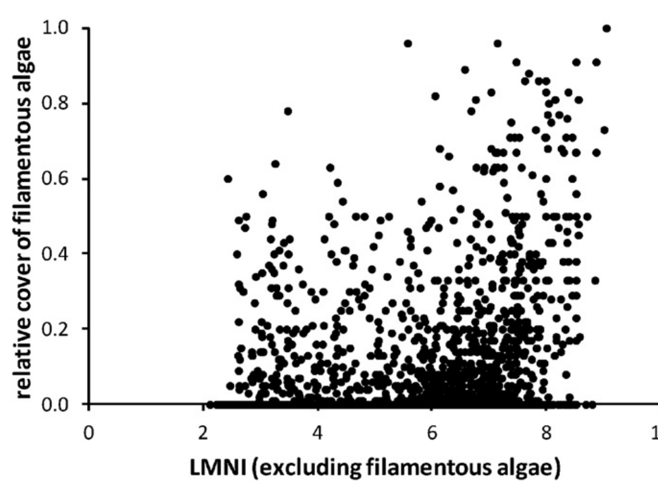

Fig. 4. Relative cover of filamentous algae (% algal cover/total macrophytes cover) in relation to values of the Lake Macrophyte Nutrient Index (LMNI) in 4500 surveys of UK lakes.

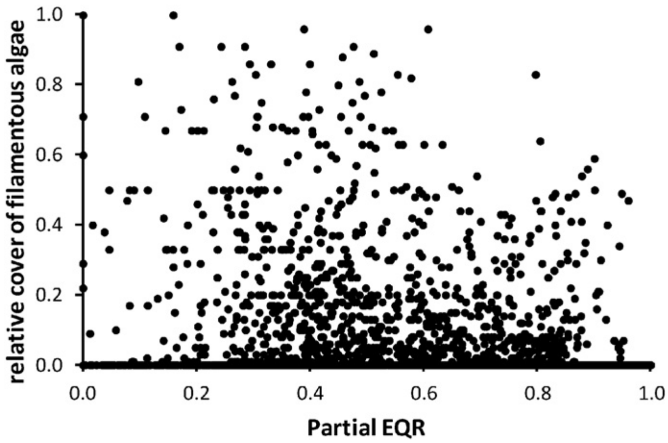

Fig. 5. Distribution of lake surveys by EQR excluding filamentous algae from the assessment ("Partial EQR") at different levels of relative filamentous algal cover. Class boundaries at 0.2 unit intervals.

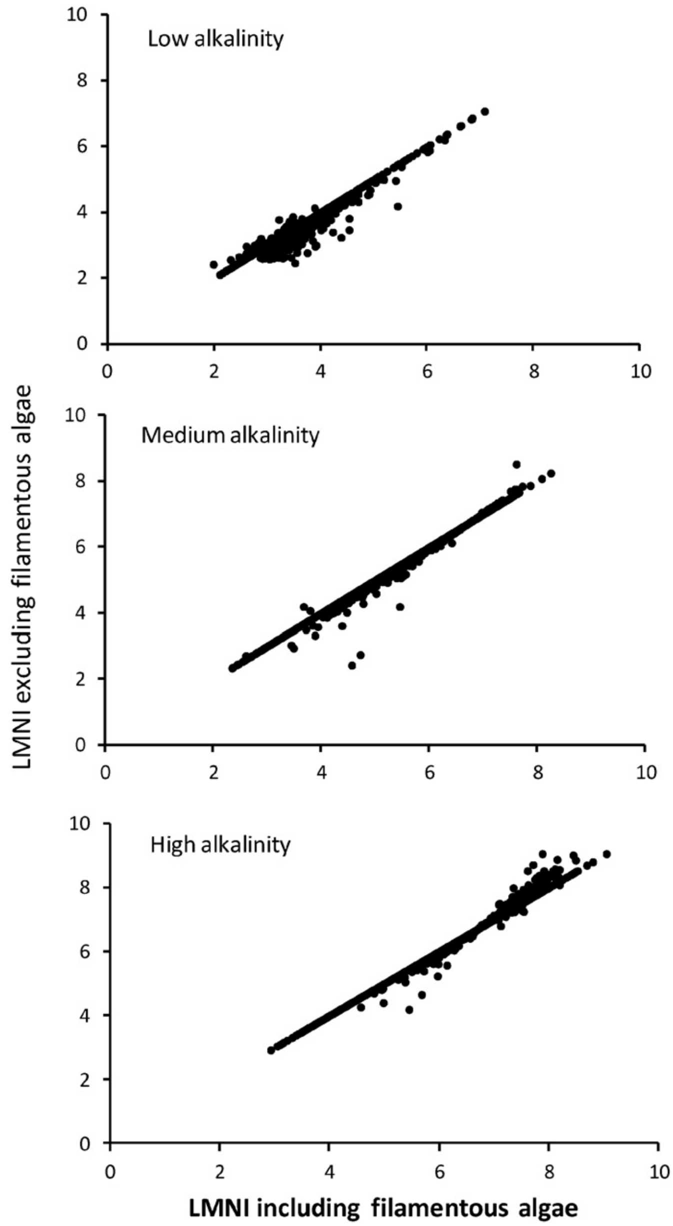

Fig. 6. Relationship between the UK Lake Macrophyte Nutrient Index (LMNI) calculated with (x: LMNI_with_alg) and without (y: LMNI_excl_alg) the inclusion of filamentous algae. Lakes are divided into three types, based on alkalinity: low (<10 mg/L $CaCO_3$); moderate (≥10; <50 mg/L $CaCO_3$) and high (>50 mg/L $CaCO_3$).

the practice that has evolved throughout much of Europe and therefore has to be addressed here.

The situation can, therefore, be generalised as "what is the marginal utility of phytobenthos, given typical ecological assessment scenarios in Europe?" In rivers, although phytoplankton is specified in the WFD normative definitions, it is only used routinely by a few countries for large rivers (Mischke et al., 2011), so phytobenthos is rarely lower than "second choice" for assessing the photosynthetic biota. It is, furthermore, perceived to be the "fast responding" complement to macrophytes and, as a result, phytobenthos methods have been adopted by 23 out of 28 MS for use in rivers (Kelly et al., 2014, 2015). In lakes, on the other hand, the role of "fast responder" is taken by phytoplankton (which, in addition, links more directly to undesirable disturbances such as bloom frequency and decline of submerged vegetation; Carvalho et al., 2013; Poikane et al., 2014). Consequently, phytobenthos generally drops to "third choice" and the "marginal utility" of the phytobenthos drops to a point where MS feel justified in arguing that the sub-element is redundant.

In practice, the situation is more nuanced than a straightforward consideration of multiple indicators of a single pressure. Phytoplankton assessment, for example, considers both the level of the pressure and the risk of secondary effects (Cyanobacterial blooms: Carvalho et al., 2013); macrophytes are also sensitive to hydromorphological changes (Mjelde et al., 2013) and provide habitat for invertebrates and fish (Jeppesen et al., 1998) whilst diatoms, the phytobenthos component most widely used for ecological assessment, are highly sensitive to acidification (Battarbee et al., 2014). Local or regional considerations, therefore, may influence decisions about the appropriate combination of assessment methods.

Such considerations are not restricted to purely scientific arguments. In many cases, WFD assessment methods build on existing experience which, in turn, provides continuity with historical data and the skills necessary to use a method (Kelly et al., 2015). There are, in addition, resource implications, although it is difficult to make generalisations between countries, each of which has different strategies for temporal and spatial replication within water bodies, which complicates a straightforward comparison of the costs of a single analysis or survey.

4.1. Effect of including filamentous algae on classifications

Based on the analysis of the UK dataset, composition metrics based on non-algal macrophytes should reflect the overall composition of the vegetation except in a few rare cases where species richness is very low and algal cover very high. However, this scenario seems unlikely outside very small or shallow lakes, or perhaps where there is extensive hard substrate in shallow water to support the growth of benthic algae.

A limitation of the use of filamentous algae in LEAFPACS is that there is very limited taxonomic discrimination within the category. This means that a potential nuisance alga such as *Cladophora glomerata* will not be differentiated from alga such as *Ulothrix* which might cover large areas of the lake littoral zone for short periods irrespective of ecological status. Nonetheless, it would seem that, in the vast majority of cases, classifications based on lake macrophytes provide a reasonable assessment of status without the additional information provided by filamentous algae.

However, collecting data on filamentous algae in lakes is quick and easy and represents no additional cost in a macrophyte survey. High filamentous algal cover is commonly regarded as an indicator of undesirable disturbance and in lakes provides some evidence of lake health in terms of grazer abundance. It is, therefore, difficult to argue that filamentous algae should be ignored as part of the assessment of ecological status in lakes using macrophytes as such information is likely to be of value for interpretive purposes, even if in most cases it would not alter classifications.

4.2. Potential roles for phytobenthos assessment in lakes

These analyses all suggest that the benefit of a separate phytobenthos assessment to a country that already uses macrophytes and phytoplankton to assess lakes is limited. Inclusion of macroalgae in macrophyte assessment methods provides de facto compliance with the WFD normative definitions, but countries that exclude them are probably not missing many impacted lakes as a result. However, these are high-level analyses, concerned only with the classification of lakes. The purpose of the WFD extends beyond collecting data to demonstrate compliance with legislation; it is primarily concerned with the sustainable management of water resources and results presented here should not be interpreted as evidence that phytobenthos has no role. Moreover, the analyses here follow the logic of the justifications provided by EU Member States who cite redundancy as the principal reason for not using phytobenthos. Fig. 3 shows that similar arguments could be constructed to demonstrate the value of phytobenthos over other BQEs. The case for or against a particular approach to assessment is rarely based on scientific grounds alone (Kelly et al., 2015).

As a general rule, we would argue that as aquatic ecosystems are highly interconnected complex systems (Moss, 2010), effective management should be based on information collected from as many different compartments as possible. This reflects Wimsatt's (1994) premise that robust decision-making in the face of complexity should rest on multiple strands of evidence. However, there are also particular circumstances where we believe phytobenthos assessment does offer genuine advantages to lake managers (DeNicola and Kelly, 2014). In particular, phytobenthos may be a better indicator than macrophytes of nutrient effects on littoral zones where pronounced hydrologic pressures (e.g., boat traffic, lake-level alterations) suppress the macrophyte assemblage (Ostendorp et al., 2009; Schmieder et al., 2004). Moreover, nuisance benthic algal blooms (e.g., *Cladophora*) can be a problem for lake users (Parker and Maberly, 2000; Higgins et al., 2005). Because phytobenthos reacts faster to environmental changes than macrophytes due to higher turnover rates (Schneider et al., 2012) it is often a good indicator of changing conditions (Jones et al., 1989; Denys, 2006; Battarbee et al., 2014). In particular, the proven track record of littoral diatoms as indicators of acidification suggests that phytobenthos may have a particular role to play in such circumstances. Finally, phytobenthos can provide a way to detect local hot spots created by stressors around the perimeter of lakes (Lambert et al., 2008; Spitale et al., 2014; Rimet et al., 2016). For example, phytobenthos is affected more quickly than plankton by watershed stressors, such as fire, timber harvesting, and shoreline development, and may be better early indicators than plankton communities (Planas et al., 2000; Lambert et al., 2008; Rosenberger et al., 2008).

These factors all point to a potential role for phytobenthos in operational and investigative monitoring, even in circumstances where their role in classification and surveillance monitoring may be limited. Rimet et al. (2015) also point out how both phytoplankton and phytobenthos can contribute complementary information about the state of a lake. All of these arguments highlight the need to understand a lake ecosystem if it is to be managed sensitively.

5. Conclusions

The importance of phytobenthos as a component of a healthy lake ecosystem is not in doubt. However, this does not necessarily mean that failing to include phytobenthos in routine lake ecological assessment will prejudice the objectives of the WFD. The analyses of redundancy between different metrics suggest that this is probably not the case, so long as both phytoplankton and macrophytes are assessed (Table 5). There are, however, legitimate concerns about whether the process of restoring failing lakes to GES (and, indeed, preventing deterioration in all lakes) can be achieved without information about phytobenthos. More specifically, we believe that countries that do not have phytobenthos assessment systems may be limiting their capability to manage lakes in a sustainable manner, particularly where the macrophyte community is challenged by other pressures, and where there are threats from diffuse pollution around the lake margin.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2016.02.024.

References

Barker, T., Hatton, K., O'Connor, M., Connor, L., Moss, B., 2008. Effects of nitrate load on submerged plant biomass and species richness: results of a mesocosm experiment. Fundam. Appl. Limnol. 173, 89–100.
Battarbee, R.W., Simpson, G.L., Shilland, E.M., Flower, R.J., Kreiser, A., Yang, H., Clarke, G., 2014. Recovery of UK lakes from acidification: an assessment using combined palaeoecological and contemporary diatom assemblage data. Ecol. Indic. 37, 365–380.
Carvalho, L., Poikane, S., Lyche Solheim, A., Phillips, G., Borics, G., Catalan, J., De Hoyos, C., Drakare, S., Dudley, B.J., Järvinen, M., Laplace-Treyture, C., Maileht, K., McDonald, C., Mischke, U., Moe, J., Morabito, G., Nõges, P., Nõges, T., Ott, L., Pasztaleniec, A., Skjelbred, B., Thackeray, S.J., 2013. Strength and uncertainty of phytoplankton metrics for assessing eutrophication impacts in lakes. Hydrobiologia 704, 127–140.
Dalgaard, P., 2002. Introductory Statistics With R. Springer, New York.
DeNicola, D., Kelly, M.G., 2014. The role of benthic algal communities in the ecological assessment of lakes. Freshwater Science 33, 619–638.
Denys, L., 2006. Calibration of littoral diatoms to water chemistry in standing fresh waters (Flanders, lower Belgium): inference models for historical sediment assemblages. J. Paleolimnol. 35, 763–787.
Eigemann, F., Mischke, U., Hupfer, M., Schaumburg, J., Hilt, S., 2016. Biological indicators track differential responses of pelagic and littoral areas to nutrient load reductions in German lakes. Ecol. Indic. 91, 905–910.
European Commission, 2010. Common Implementation Strategy for the Water Framework Directive (2000/60/EC) Guidance Document No. 14: Guidance Document on the Intercalibration Process 2008–2011. Office for Official publications of the European Communities, Luxembourg.
European Union, 2002. Treaty on European Union (consolidated version). Official Journal of the European Communities C 325, 5.
González Sagrario, M.A., Jeppersen, E., Gomà, J., Søndergaard, M., Peder Jensen, J., Lauridsen, T., Landkildhaus, F., 2005: 50, 27–41.

Higgins, S.N., Howell, E.T., Hecky, R.E., Guildford, S.J., Smith, R.E., 2005. The wall of green: the status of *Cladophora glomerata* on the northern shores of Lake Erie's eastern basin, 1995–2002. J. Great Lakes Res. 31, 547–563.

James, C., Fisher, J., Russell, V., Collings, S., Moss, B., 2005. Nitrate availability and hydrophyte species richness in shallow lakes. Freshw. Biol. 50, 1049–1063.

Jeppesen, E., Søndergaard, M., Søndergaard, M., Cristoffersen, K. (Eds.), 1998. Structuring Role of Submerged Macrophytes in Lakes. Springer-Verlag, New York.

Jones, J.I., Sayer, C.D., 2003. Does the fish-invertebrate-periphyton cascade precipitate plant loss in shallow lakes? Ecology 84, 2155–2167.

Jones, V.J., Stevenson, A.C., Battarbee, R.W., 1989. Acidification of lakes in southwest Scotland: a diatom and pollen study of the post-glacial history of the Round Loch of Glenhead. J. Ecol. 77, 1–23.

Kelly, M., Urbanic, G., Acs, E., Bennion, H., Bertrin, V., Burgess, A., Denys, L., Gottschalk, S., Kahlert, M., Karjalainen, S.M., Kennedy, B., Kosi, G., Marchetto, A., Morin, S., Picinska-Fałtynowicz, J., Poikane, S., Rosebery, J., Schoenfelder, I., Schoenfelder, J., Varbiro, G., 2014. Comparing aspirations: intercalibration of ecological status concepts across European lakes for littoral diatoms. Hydrobiologia 734, 125–141.

Kelly, M.G., Schneider, S.C., King, L., 2015. Customs, habits and traditions: the role of non-scientific factors in the development of ecological assessment methods. WIRES Water 2, 159–165.

Lambert, D., Cattaneo, A., Carignan, R., 2008. Periphyton as an early indicator of perturbation in recreational lakes. Can. J. Fish. Aquat. Sci. 65, 258–265.

Lipsey, R.G., Chrystal, K.A., 1995. Principles of Economics. nineth ed. Oxford University Press, Oxford.

Lovett, G.M., Burns, D.A., Driscoll, C.T., Jenkins, J.C., Mitchell, M.J., Rustad, L., Shanley, J.B., Likens, G.E., Haeuber, R., 2007. Who needs environmental monitoring? Front. Ecol. Environ 5 (5), 253–260.

Mischke, U., Venohr, M., Behrendt, H., 2011. Using phytoplankton to assess the trophic status of German rivers. Int. Rev. Hydrobiol. 96, 578–598.

Mjelde, M., Hellsten, S., Ecke, F., 2013. A water level drawdown index for aquatic macrophytes in Nordic lakes. Hydrobiologia 704 (1), 141–151.

Moss, B., 2008. The Water Framework Directive: total environment or political compromise? Sci. Total Environ. 400, 32–41.

Moss, B., 2010. Ecology of Freshwaters: A View for the Twenty-first Century. fourth ed. Wiley-Blackwall, Chichester.

Ostendorp, W., Gretler, T., Mainberger, M., Peintinger, M., Schmieder, K., 2009. Effects of mooring management on submerged vegetation, sediments and macro-invertebrates in Lake Constance, Germany. Wetl. Ecol. Manag. 17, 525–541.

Pall, K., Moser, V., 2009. Austrian index macrophytes (AIM-Module 1) for lakes: a Water Framework Directive compliant assessment system for lakes using aquatic macrophytes. Hydrobiologia 633 (1), 83–104.

Parker, J.E., Maberly, S.C., 2000. Biological response to lake remediation by phosphate stripping: control of *Cladophora*. Freshw. Biol. 44, 303–309.

Planas, D., Desrosiers, M., Groulx, S.R., Paquet, S., Carignan, R., 2000. Pelagic and benthic algal responses in eastern Canadian Boreal Shield lakes following harvesting and wildfires. Can. J. Fish. Aquat. Sci. 57, 136–145.

Poikane, S., Portielje, R., Berg, M., Phillips, G., Brucet, S., Carvalho, L., Mischke, U., Ott, I., Soszka, H., Van Wichelen, J., 2014. Defining ecologically relevant water quality targets for lakes in Europe. J. Appl. Ecol. 51 (3), 592–602.

Poikane, S., Birk, S., Carvalho, L., Böhmer, J., de Hoyos, C., Gassner, H., Hellsten, S., Kelly, M., Lyche Solheim, A., Olin, M., Pall, K., Phillips, G., Portielje, R., Ritterbusch, D., Sandin, L., Schartau, A.K., Solimini, A.G., van den Berg, M., Wolfram, G., van de Bund, W., 2015. A hitchhiker's guide to European lake ecological assessment and intercalibration. Ecol. Indic. 52, 533–544.

Resh, V.H., 2008. Which group is best? Attributes of different biological assemblages used in freshwater biomonitoring programs. Environ. Monit. Assess. 138, 131–138.

Rimet, F., Bouchez, A., Montuelle, B., 2015. Benthic diatoms and phytoplankton to assess nutrients in a large lake: complementarity of their use in Lake Geneva (France-Switzerland). Ecol. Indic. 53, 231–239.

Rimet, F., Bouchez, A., Tapolczai, K., 2016. Spatial heterogeneity of littoral benthic diatoms in a large lake: monitoring implications. Hydrobiologia http://dx.doi.org/10.1007/s10750-015-2629-y (in press).

Rosenberger, E.E., Hampton, S.E., Fradkin, S.C., Kennedy, B.P., 2008. Effects of shoreline development on the nearshore environment in large deep oligotrophic lakes. Freshw. Biol. 53, 1673–1691.

Schaumburg, J., Schranz, C., Hofmann, G., Stelzer, D., Schneider, S., Schmedtje, U., 2004. Macrophytes and phytobenthos as indicators of ecological status in German lakes—a contribution to the implementation of the Water Framework Directive. Limnologica 34, 302–314.

Schmieder, K., Dienst, M., Ostendorp, W., Jöhnk, K., 2004. Effects of water level variations on the dynamics of the reed belts of Lake Constance. Ecohydrol. Hydrobiol. 4, 469–480.

Schneider, S.C., Lawniczak, A.E., Picińska-Faltynowicz, J., Szoszkiewicz, K., 2012. Do macrophytes, diatoms and nondiatom benthic algae give redundant information? Results from a case study in Poland. Limnologica 42, 204–211.

Spitale, D., Scalfi, A., Cantonati, M., 2014. Urbanization effects on shoreline phytobenthos: a multiscale approach at lake extent. Aquat. Sci. 76, 17–28.

Trobajo, R., Quintana, X.D., Moreno-Amich, R., 2002. Model of alternative predominance of phytoplankton-periphyton-macrophytes in lentic waters of Mediterranean coastal wetlands. Arch. Hydrobiol. 154, 19–40.

UK TAG, 2014. Lake assessment method: macrophytes and phytobenthos: macrophytes (lake LEAFPACS 2). http://www.wfduk.org/resources/lakes-macrophytes [accessed 27/11/2015].

Willby, N., Pitt, J.-A., Phillips, G., 2009. The ecological classification of UK lakes using aquatic macrophytes. Science Report SC010080/SR2. Environment Agency, Bristol, UK.

Wimsatt, W.C., 1994. The ontology of complex systems: levels of organization, perspectives and causal thickets. Can. J. Philos. 24, 207–274.