Defining ecologically relevant water quality targets for lakes in Europe

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Summary

1. The implementation of the Water Framework Directive requires EU member states to establish and harmonize ecological status class boundaries for biological quality elements. In this paper, we describe an approach for defining ecological class boundaries that delineates shifts in lake ecosystem functioning and, therefore, provides ecologically meaningful targets for water policy in Europe.

2. We collected an extensive data set of 810 lake-years from nine Central European countries, and we used phytoplankton chlorophyll *a*, a metric widely used to measure the impact of eutrophication in lakes. Our approach establishes chlorophyll *a* target values in relation to three significant ecological effects of eutrophication: the decline of aquatic macrophytes, the dominance of potentially harmful cyanobacteria and the major functional switch from a clear water to a turbid state.

3. Ranges of threshold chlorophyll *a* concentrations are given for the two most common lake types in lowland Central Europe: for moderately deep lakes (mean depth 3–15 m), the greatest ecological shifts occur in the range 10–12 $\mu g\ L^{-1}$ chlorophyll *a*, and for shallow lakes (< 3 m mean depth), in the range 21–23 $\mu g\ L^{-1}$ chlorophyll *a*.

4. *Synthesis and applications*. Our study provides class boundaries for determining the ecological status of lakes, which have robust ecological consequences for lake functioning and which, therefore, provide strong and objective targets for sustainable water management in Europe. The results have been endorsed by all participant member states and adopted in the European Commission legislation, marking the first attempt in international water policy to move from physico-chemical quality standards to harmonized ecologically based quality targets.

Key-words: chlorophyll, cyanobacteria, ecological threshold, eutrophication, macrophyte, phytoplankton, stable states, water framework directive

Introduction

Lakes world-wide are subject to significant anthropogenic pressures, such as eutrophication, acidification, alien species and climate change (UNEP 2007; EEA 2012). In many cases, these human impacts have resulted in degraded ecological structure (e.g. plant loss) and altered functioning (a shift of primary production from littoral to pelagic food webs). Many of these ecological changes have severe consequences for lake ecosystem services, limiting uses for water supply, recreation and tourism (MEA Millennium Ecosystem Assessment 2005). The European Water Framework Directive (WFD; EC 2000) establishes a framework for managing these challenges. European

member states (MS) are required to develop ecologically based assessment systems for biological communities, defined by a number of mandatory biological quality elements, and to set river basin management plans to achieve at least 'good' ecological status for all waters ideally by 2015, or in later management cycles at 2021 or, ultimately, by 2027.

One of the most critical steps in creating ecological assessment methods is in the definition of the 'good–moderate' status class boundary because achieving 'good' status is a legal obligation for MS. The WFD defines 'good status' in general terms, as a condition that should 'deviate only slightly from undisturbed conditions'. This definition allows a wide scope for interpretation (e.g. Gabriels *et al.* 2010).

There is no commonly accepted guidance on how much deviation should be deemed 'slight' or 'moderate'. Some have argued that degrees of water quality can only be established through expert judgment, as there are 'no absolute meanings of good, moderate, poor and bad' (Moss *et al.* 2003). Others have claimed that ecology-based elements must be included in boundary setting (Birk *et al.* 2012; Brucet *et al.* 2013) and that ecologically meaningful guiding images should be provided (Willby 2011; Kelly 2012). In the present article, we aim to provide ecology-based definitions for 'good' status boundaries and to link the ecological effects of eutrophication to the definition of lake management targets.

In the last decade, a broad consensus has emerged that (i) boundaries for ecological status classes must be based on pressure–response relationships (Davies & Jackson 2006), (ii) ecological thresholds should play a key role in boundary setting (Groffman *et al.* 2006) and (iii) target values must reflect environmental conditions that are socially desirable or acceptable (Smyth, Watzin & Manning 2007). However, in practice, boundary setting does not always follow ecological principles. A purely statistical approach is often used, in which the metric gradient is divided into an appropriate number of equal-width classes, referred to hereafter as the 'equal division' approach (e.g. Gabriels *et al.* 2010). Although this approach allows for a straightforward assessment of ecological status, the boundaries between classes are not based on meaningful biological considerations (Grenier *et al.* 2010) and ecologically meaningful guiding images (Willby 2011). Despite these obvious drawbacks, the equal division approach is the most broadly used boundary-setting approach for the WFD in Europe (Birk *et al.* 2012). If status class boundaries can be defined based on shifts in ecosystem functioning, they will be more relevant to the sustainable provision of ecosystem services and will communicate the condition of aquatic resources more effectively to the public.

Despite improvements in some regions, the eutrophication of freshwater ecosystems is still the most important pan-European pressure and one of the main causes for the less than 'good' ecological status in 44% of European lakes (EEA 2012). Furthermore, eutrophication poses a threat to public health and incurs significant economic costs (Pretty *et al.* 2003). Nevertheless, countries have advocated widely divergent assessment methodologies and criteria that need to be harmonized (Heiskanen *et al.* 2004). A pan-European agreement was at least partly achieved in the EU guidance on Eutrophication (EC 2009), which states that the condition of phytoplankton can be considered 'good' if accelerated algal growth does not result in a significant undesirable disturbance to the aquatic ecosystem. Such undesirable effects could include use-related impacts, such as the development of cyanobacteria blooms or significant ecological changes, such as a loss of macrophyte vegetation.

Hence, in this paper, we propose an ecosystem-based approach to define ecological quality targets for phytoplankton chlorophyll a, a parameter widely used as an indicator of eutrophication. Our approach establishes thresholds of phytoplankton chlorophyll a concentrations that are set in relation to the functional effects of eutrophication that are well-established in ecological theory (Fig. 1). Common phenomena in response to nutrient enrichment include a decline in aquatic macrophytes, dominance by cyanobacteria and a switch from a clear vegetation-dominated to a turbid non-vegetated state (Scheffer 1998; Downing, Watson & McCauley 2001; Søndergaard *et al.* 2010). These effects are widely considered undesirable, with consequent impacts on biodiversity and water use, particularly on recreation and water supply (e.g. Chorus *et al.* 2000). Therefore, these effects represent important ecological thresholds for setting the critical 'good–moderate' status class boundary.

We define a lake in 'good' status as having a low probability of occurrence of cyanobacteria blooms, a low probability of a significant decline in macrophyte-colonized depth and (for shallow lakes) an ecosystem in a clear water macrophyte-dominated state.

The aims addressed in this paper are as follows:
1. To define a methodology for setting ecologically meaningful boundaries for phytoplankton abundance for Central European lakes;
2. To determine the 'good–moderate' status class boundaries for phytoplankton chlorophyll *a* for the main lake types in Central Europe based on response relationships between phytoplankton abundance and the ecological effects of eutrophication.

Materials and methods

DATA

Data were collected from more than 400 lakes and 810 lake-years from nine countries (Table 1). The data set contained general location and descriptive data (altitude, surface area, mean depth), physico-chemical data (alkalinity, nutrients, chlorophyll *a*, Secchi depth) and biological data (macrophyte and phytoplankton composition). Total phosphorus and chlorophyll *a* values were
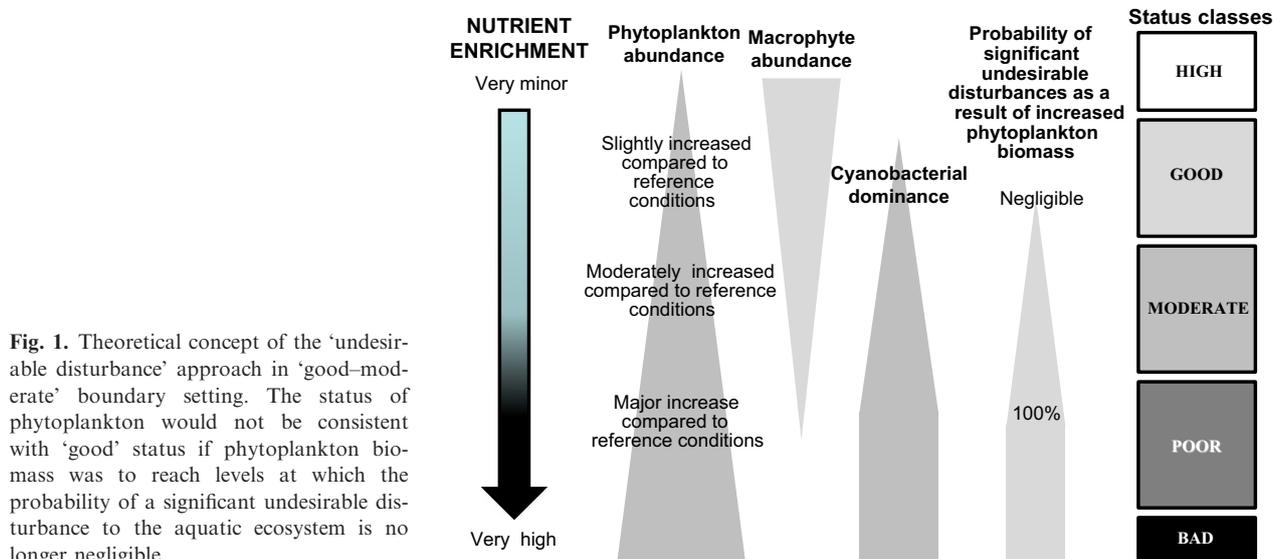
Fig. 1. Theoretical concept of the 'undesirable disturbance' approach in 'good–moderate' boundary setting. The status of phytoplankton would not be consistent with 'good' status if phytoplankton biomass was to reach levels at which the probability of a significant undesirable disturbance to the aquatic ecosystem is no longer negligible.

Table 1. Descriptive statistics of deep ($n = 539$) and shallow ($n = 271$) lakes. Number of lake-years by countries: Belgium 4, Germany 61, Denmark 98, Estonia 11, Lithuania 47, Latvia 215, the Netherlands 203, Poland 147, United Kingdom 67. Deep lakes with mean depth 3–15 m, shallow lakes with mean depth <3 m

Lake type	Lake characteristics	Min	25‰	Median	75‰	Max
Deep lakes	Lake area (km^2)	0·02	0·49	1·04	2·61	113·4
	Lake depth (m)	3·1	4·6	6·0	9·0	15·0
	Altitude (m a.s.l.)	0·8	49·9	116·1	133·8	200
	Alkalinity (meq L^{-1})	1·0	1·9	2·3	2·7	7·7
	TP (mg L^{-1})	7	23	34	55	680
	Chl-a (μg L^{-1})	0·5	5·0	8·3	16·0	198·0
Shallow lakes	Lake area (km^2)	0·01	0·25	0·50	2·00	40·80
	Lake depth (m)	0·3	1·4	1·9	2·5	3·0
	Altitude (m a.s.l.)	0·1	15	58	103	160
	Alkalinity (meq L^{-1})	1·3	1·9	2·3	2·8	5·0
	TP (mg L^{-1})	6	30	50	92	948
	Chl-a (μg L^{-1})	1·0	10·6	23·5	54·4	220·0

averaged over the growing season, which in most cases is April (May)–September (October). The number of samples during one vegetation season in one lake varied between 2 and 28 samples, typically 3–4 times per year (see Appendix S1, Supporting information). A standard spectrophotometric method (ISO 1992) was used for chlorophyll *a* analysis; inverted microscopy following the Utermöhl technique (CEN 2006) was used for the enumeration of phytoplankton. Macrophyte data were collected using a transect-based method (CEN 2003) and reported at the species (or higher taxonomic group) level using the ECOFRAME abundance scale (Moss *et al.* 2003).

Analysis was performed for two major lake types (Table 1): lowland calcareous shallow lakes with a mean depth of <3 m (from here on called 'shallow lakes') and lowland calcareous moderately deep lakes with a mean depth of 3–15 m (from here

on called 'deep lakes'). Our aim was to broadly differentiate polymictic shallow systems from deeper, seasonally stratified systems due to well-established differences in their structure and functioning (Scheffer 1998). We used the 3-m depth limit as: (i) it ensures that all shallow lakes in our analyses were unstratified and that most of the deep lakes were temporarily or permanently stratified during summer (Søndergaard *et al.* 2005); (ii) it is widely used both in scientific studies and national typologies (e.g. Scheffer 1998; Moss *et al.* 2003; Søndergaard *et al.* 2005). Additionally, these depth types were identified by a range of European experts during the WFD intercalibration exercise and, although the precise split at 3 m is arbitrary, it represents the best available agreed typology that is likely to minimize natural biological variation in European lakes (e.g. Phillips *et al.* 2008).

CHLOROPHYLL *A* BOUNDARIES BASED ON CHANGES IN SUBMERGED MACROPHYTE ABUNDANCE

Boundary setting was based on a direct relationship between chlorophyll *a* concentration and submerged macrophyte abundance calculated for individual lake-years in classes ranging from 0 to 5 (see Appendix S2, Supporting information).

We established response curves plotting the fraction of lakes with three levels of macrophyte abundances ($\geq 3·5$, $\geq 2·5$, $\geq 1·5$) in relation to chlorophyll *a*. The macrophyte target abundances can be interpreted as follows: submerged macrophyte abundance $\geq 1·5$ – submerged macrophytes are present, at least in low-to-moderate amounts; $\geq 2·5$ – lakes in a macrophyte-dominated state; and $\geq 3·5$ – a high abundance of submerged macrophytes. Lakes with low amount or no macrophytes were not included in the analyses as they show high variability of chlorophyll *a* values ranging from 10 to 400 μg L^{-1}.

The relationships between chlorophyll *a* and macrophyte abundance were developed as follows: the lake-years were sorted by increasing order of chlorophyll *a* concentration and the fraction of lake-years complying with three macrophyte abundance levels ($\geq 1·5$, $\geq 2·5$, $\geq 3·5$) was calculated as the moving average of the 30 nearest data points. We used data smoothing (moving averages) as it is a simple and efficient technique to reveal the underlying

trend in data and to detect thresholds (e.g. Downing, Watson & McCauley 2001).

The aim was to determine the threshold of phytoplankton biomass increase at which the abundance of macrophytes decreased sharply, indicating a shift from a clear vegetation-dominated state to a turbid phytoplankton-dominated state (Scheffer 1998). The thresholds were set using visual inspection of the relationship between chlorophyll and macrophyte abundance where there was a steep reduction in the fraction of lakes with abundant macrophytes. To validate the results of this boundary setting method, we ran a classification tree analysis (see Appendix S3, Supporting information) where lakes with high and low macrophyte abundances were separated using several possible explanatory variables.

CHLOROPHYLL *A* BOUNDARIES BASED ON CHANGES TO THE MAXIMUM COLONIZATION DEPTH OF SUBMERGED MACROPHYTES (C_{MAX})

The relationship between mean growing season chlorophyll *a* and C_{max} was determined by linear regression after square root transformation of C_{max} and log transformation of chlorophyll *a*. Linear regression was used because both the effect and response variables are continuous while data transformation was used to ensure their homogeneity of variance, as is common practice for data of this type (Phillips *et al.* 2008).

For the boundary-setting procedure, we first determined reference type-specific C_{max} values using established reference chlorophyll *a* values (Poikāne *et al.* 2010). The 'good–moderate' class boundary was then set as a point at which there was a low probability ($P < 0.05$) of being at 'poor' status, defined as an undesirable change in C_{max} to ≤1·5 m for deep lakes and C_{max} ≤1·0 m for shallow lakes. Several studies (e.g. Moss *et al.* 2003; Søndergaard *et al.* 2005) have established Secchi depth values for 'poor 'status in the range of 0·5–0·9 m (shallow lakes) and 1·0–1·3 m (deep lakes), corresponding to a C_{max} of 1 m for shallow lakes and 1·5 m for deep lakes (Middelboe & Markager 1997). Additionally, these values correspond to the breaking point where C_{max} becomes independent of the light climate (Middelboe & Markager 1997) as macrophytes compensate for light limitation by establishing their biomass just below the water surface (Søndergaard *et al.* 2010).

CHLOROPHYLL *A* BOUNDARIES BASED ON CHANGES IN THE DOMINANCE OF CYANOBACTERIA

Data on total phytoplankton biovolume and the biovolume of cyanobacteria in samples from the mid to late summer period (July–September) were collated. Most Chroococcales genera were excluded as typical for oligotrophic waters with the exception of the genera *Microcystis* and *Woronichinia* (Ptacnik *et al.* 2008). Logistic regression was used to describe the relationship between chlorophyll *a* and the probability of cyanobacteria blooms. This is an appropriate technique to model relationships if the response variable is a proportion or frequency. The aim of the analysis was to set a chlorophyll *a* threshold where there was a low probability of a significant cyanobacterial bloom. Proportions of 50% (for deep lakes) and 75% (for shallow lakes) of relative abundance of cyanobacteria were chosen as bloom criteria. Phytoplankton composed of >50% cyanobacteria biomass is the most widely used indicator of cyanobacteria-dominant algal blooms

(Downing, Watson & McCauley 2001); thus, this level was chosen as the bloom criterion for deep lakes. However, it is well-established that cyanobacteria are more favoured in shallow lakes because the production of algal biomass in deeper lakes is limited by the poor light supply (Nixdorf & Deneke 1997); thus, a higher proportion of cyanobacteria can be expected in shallow lakes (Scheffer 1998; Scheffer & van Nes 2007). Several thresholds have been proposed, for example >95% (Moss *et al.* 1993), but we used 75% because even in heavily dominated lakes, the proportion of cyanobacteria typically ranges from 75 to 90% (Nixdorf & Deneke 1997; Downing, Watson & McCauley 2001).

Boundary setting followed a three-step procedure based on a low probability of occurrence of undesirable effects of eutrophication:

1. Reference conditions for cyanobacteria were established based on reference chlorophyll *a* values (Poikāne *et al.* 2010);

2. 'Poor' status was then defined as the presence of undesirable effects of eutrophication, that is, 50% of summer phytoplankton samples cyanobacteria-dominated;

3. Finally, the 'good–moderate' boundary was defined as conditions under which undesirable effects are unlikely to occur. However, deviation from reference conditions should be taken into account to ensure that 'good' status does not coincide with reference conditions. There is no accepted guidance on what an acceptable deviation from the reference condition is. Different sources report values from 15 to 50% (e.g. Andersen, Conley & Hedal 2004). As a general rule, we used 25% as an acceptable deviation; thus, if a reference condition is a 10% bloom frequency, the 'good' status boundary will be a 12·5% bloom frequency.

Results

CHLOROPHYLL *A* BOUNDARIES BASED ON CHANGES IN SUBMERGED MACROPHYTE ABUNDANCE

A response curve analysis revealed the sharp decrease in submerged macrophyte abundance with increasing chlorophyll *a* (Fig. 2). The boundary-setting procedure was based on the threshold points at which steep reductions in macrophytes occur in relation to chlorophyll *a*. Among shallow lakes (Fig. 2a) with low chlorophyll *a* concentrations, *c.* 60–70% are vegetation-dominated (macrophyte abundance ≥2·5). This fraction declines steeply between 20 and 30 μg L^{-1} chlorophyll *a*, with only 20% of lakes at 30 μg L^{-1} chlorophyll *a* dominated by macrophytes and <10% above 70 μg L^{-1} chlorophyll *a*. The 'good–moderate' boundary was defined at the threshold at which the proportion of macrophyte-dominated lakes falls below 0·5, corresponding to a chlorophyll *a* value of 21 μg L^{-1}. For deep lakes (Fig. 2b), the most pronounced transition occurred at a chlorophyll *a* concentration of 11 μg L^{-1}, where the fraction of lakes with macrophyte abundance ≥1·5 decreased sharply from *c.* 0·8 to below 0·5. For shallow and deep lakes, different macrophyte abundance curves were used (≥2·5 and ≥1·5, respectively), as in shallow lakes most of the lake bottom can be potentially covered by vegetation, while in deep lakes the surface where macrophytes can grow is light limited.
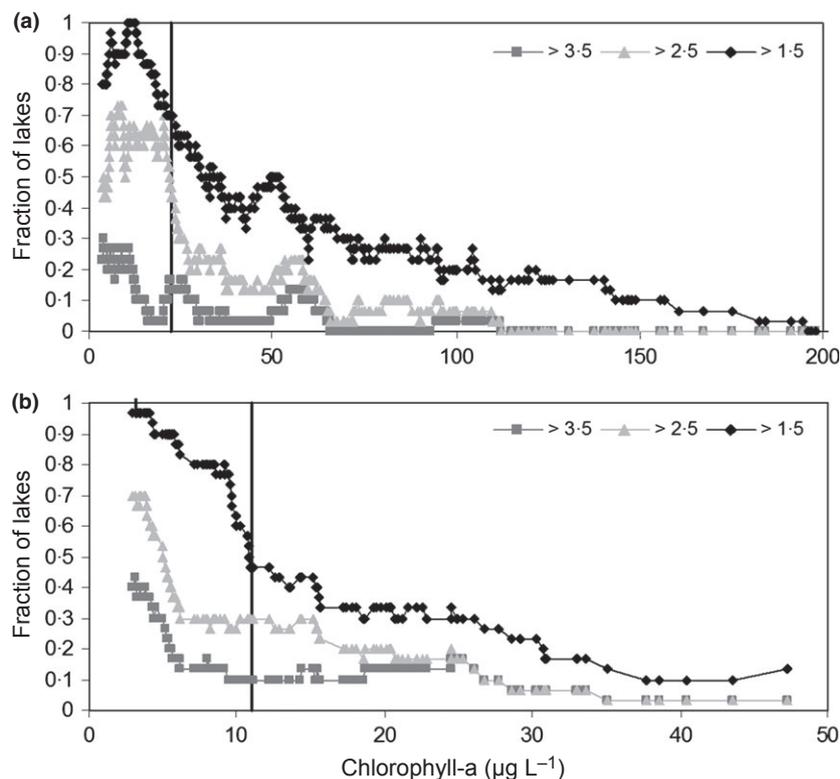
Fig. 2. Fraction of lakes with a macrophyte abundances of >1·5, >2·5 and >3·5 in relation to chlorophyll *a* for (a) shallow and (b) deep lakes calculated as the moving average of the 30 nearest data points. Proposed chlorophyll *a* boundary values are shown with vertical lines (see text for explanation).

DERIVATION OF CHLOROPHYLL *A* BOUNDARIES BASED ON CHANGES TO THE MAXIMUM COLONIZATION DEPTH OF SUBMERGED MACROPHYTES

A linear model between C_{max} (square root transformed) and log chlorophyll *a* was highly significant ($R^2 = 0.539$, $P < 0.001$; Eqn 1) and provided uniform residuals across the range of chlorophyll *a* values:

$$C_{max} = (2·489(\pm 0·044) - 0·732(\pm 0·035) \log \mathrm{Chl} - a)^2$$

(eqn 1)

The back-transformed fitted model, together with confidence intervals, is shown in Fig. 3. The model was used to establish boundaries for chlorophyll *a*. Boundaries were based on the deviation from reference conditions and probability of undesirable impacts of C_{max} being ≤1·5 m for deep lakes and ≤1·0 m for shallow lakes (defined as 'poor' status).

Thus, the boundary-setting procedure for deep lakes (Fig. 3a) was carried out as follows:

1. Type-specific reference C_{max} values were determined using reference chlorophyll *a* values established from a reference site analysis (Poikāne *et al.* 2010). The reference chlorophyll *a* value for deep lakes was 3·1 µg L^{-1}, giving a modelled reference C_{max} value of 4·6 m.

2. The C_{max} value for 'poor' status (defined as an undesirable change in C_{max}) was represented by a C_{max} of 1·5 m, giving a modelled chlorophyll *a* of 53 µg L^{-1}.

3. The 'good–moderate' class boundary was then set at a point at which there is a low probability ($P < 0·05$) of being at 'poor' status and only a slight change from reference conditions. Projecting along the *x*-axis for C_{max} 1·5 m (the mid-point of 'poor' status) to intersect with the $P = 0·05$ confidence limit (a low confidence of being at 'poor' status) determined that the 'good–moderate' boundary for chlorophyll *a* should be set at 10 µg L^{-1}.

The same procedure was used to set the boundaries for shallow lakes (Fig. 3b), and the results are summarized in Table 2.

DERIVATION OF CHLOROPHYLL *A* BOUNDARIES BASED ON CHANGES IN THE DOMINANCE OF CYANOBACTERIA

The frequency of dense blooms (cyanobacteria >50% of total phytoplankton biovolume) was less than 10% for the reference chlorophyll *a* concentration range (0–6 µg L^{-1}) and increased to more than 80% at chlorophyll *a* levels above 90 µg L^{-1} (Fig. 4).

The logistic analyses showed significant relationships ($P < 0·001$) between chlorophyll *a* concentration and two cyanobacteria abundance classes, >50% and >75% (Fig. 5). The boundary setting of chlorophyll *a* was based on the change in proportion of cyanobacteria (Table 3). Based on a reference value of chlorophyll *a* concentration of 3·1 µg L^{-1} for deep lakes, the model predicted a probability of *c.* 8·9–10·4% that a single sample taken during the summer would have cyanobacteria dominating more than 50% of the total phytoplankton biovolume. For shallow lakes, a reference chlorophyll *a* concentration of 5·8 µg L^{-1} predicted a probability of *c.* 6·6–7·6% for more than 75% cyanobacteria.
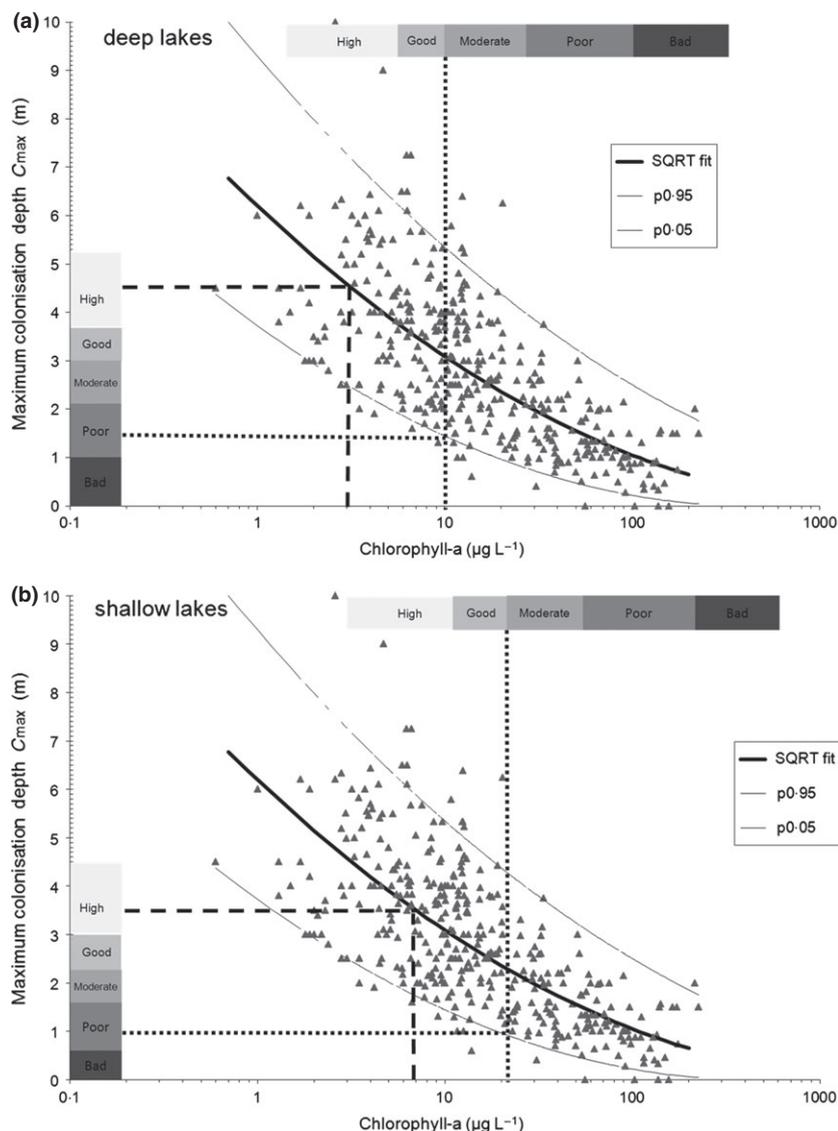
Fig. 3. Regression model of macrophyte colonization depth (C_{max}) versus log-transformed chlorophyll *a* together with confidence intervals ($P = 0.05$ and 0.95). 'Good–moderate' class boundary set at a point with a low probability ($P = 0.05$) of C_{max} being at poor status, shown by dotted line (⋯⋯). Reference conditions set using chlorophyll *a* values from reference sites (Poikāne *et al*. 2010), shown by dashed line (- - - -).

Next, 'poor' status was defined as the point at which half of the samples taken during high summer were cyanobacteria-dominated. This level corresponds to 56 µg L^{-1} (deep lakes) and 94 µg L^{-1} chlorophyll *a* (shallow lakes).

'Good' status was defined as the point at which the probability of 'poor' status occurring is low and only a slight deviation (*c.* 25%) from reference occurs. For deep lakes, 12·5% cyanobacteria-dominated samples (compared with 9–10% in the reference condition) corresponded to 12 µg L^{-1} chlorophyll *a*, while for shallow lakes, 10% cyanobacteria-dominated samples (compared with 6–8% in the reference condition) corresponded to 21 µg L^{-1} chlorophyll *a*.

Discussion

SETTING OF ECOLOGICAL QUALITY BOUNDARIES BASED ON A DECREASE IN MACROPHYTES

The depth of colonization of submerged macrophytes declines as lakes become more eutrophic due to an increase in phytoplankton and a concomitant decrease in water transparency (Søndergaard *et al.* 2010). Our findings are in a close accordance with other studies of deep lakes. In Danish lakes, the most pronounced changes in C_{max} were observed in the range from 10 to 20 µg L^{-1} chlorophyll *a* (Søndergaard *et al.* 2010), while Free *et al.* (2006) identified breakpoints at 8–9 µg L^{-1} chlorophyll *a* for deep Irish lakes.

For shallow lakes, C_{max} becomes less relevant, as the whole lake can potentially be covered with macrophytes, and hence, for these lakes, overall macrophyte abundance is a better indicator. A wide range of evidence strongly supports the concepts that (i) the pristine state of the majority of shallow lakes is clear water with rich aquatic vegetation (Scheffer 1998); (ii) with increasing nutrient loading, lakes tend to shift from a clear state to a turbid state (Scheffer 1998) and (iii) restoration of non-vegetated turbid lakes is notoriously difficult (Moss 2007). The phenomenon of alternative clear water or turbid regimes was first attributed to non-stratified lakes (Scheffer 1998);

Table 2. Summary of the chlorophyll *a* (chl-*a*) boundary-setting approach based on the decrease in maximum colonization depth of submerged macrophytes (C_{max})

| Boundary	Deep lakes		Shallow lakes		Explanation
	Chl-*a* ($\mu g\ L^{-1}$)	C_{max} (m)	Chl-*a* ($\mu g\ L^{-1}$)	C_{max} (m)	
Reference value	3·1	4·5	6·8	3·5	Chl-*a* from reference site analysis (median value)
High–good	5·8	3·7	10·8	3·0	Chl-*a* from reference site analysis (75‰)
Good–moderate	10·0	3·0	21·0	2·3	$P = 0.05$ that C_{max} = poor status
Moderate–poor	26·0	2·1	52·0	1·6	$P = 0.25$ that C_{max} = poor status
Mid Poor	53·0	1·5	100·0	1·0	Defined as undesirable effect of eutrophication (see text)
Poor–bad	104·0	1·0	215·0	0·6	$P = 0.75$ that C_{max} = poor status

however, later studies showed that this theory might be valid for deeper lakes as well (Hilt *et al.* 2010). In the light of these facts, it is important to develop lake quality targets that ensure the maintenance of sufficiently high macrophyte abundance associated with the clear water state.

We found pronounced nonlinear relationships between chlorophyll *a* concentrations and macrophyte cover. The fraction of macrophyte-dominated shallow lakes declines steeply between 20 and 30 $\mu g\ L^{-1}$ chlorophyll *a* and becomes negligible above 70 $\mu g\ L^{-1}$. However, at low chlorophyll concentrations near the reference value, *c.* 10% of lakes fall into the abundance class <1·5, indicating that at reference chlorophyll *a* conditions, low macrophyte abundance can occur. There are clear reasons for the absence of macrophytes at high chlorophyll concentrations; the possible reasons for an absence of macrophytes at low chlorophyll concentrations might include high inorganic turbidity, sediment composition (e.g. sediment with a high sand content or very loose, peaty sediment) and wave action facilitating the uprooting of plants or uprooting by birds and fish (Meijer *et al.* 1999). Also, in this group, are lakes recovering from eutrophication that are not always immediately (re)colonized by macrophytes due to other limiting factors, such as grazing by fish or birds or a lack of propagules (Meijer *et al.* 1999). In conclusion, lakes where macrophytes were absent were not included in the boundary setting analyses, because of the high and

unexplained variability of chlorophyll values in these lakes.

It is well known that in the early stages of eutrophication, macrophyte abundance can increase along with changes in species composition, whereas subsequent stages are characterized by macrophyte decline (Hough *et al.* 1989; Moss *et al.* 2003). However, we found no evidence of unimodal relationships between macrophyte abundance and eutrophication pressure; macrophyte abundance showed a clear decrease along the eutrophication gradient. The main reasons for this pattern are due to the macrophyte metrics we used. First, in our metrics, we only considered submerged rooted plants and charophytes, which are expected to decline along the eutrophication gradient, in contrast to non-rooted and floating plants (Hough *et al.* 1989). Secondly, eutrophication can cause an increase in macrophyte biomass (Chambers & Kalff 1987) as well as in species richness (Rørslett 1991). In contrast, our data mostly deal with submerged macrophyte cover (%) and colonization depth, which decrease due to eutrophication (Søndergaard *et al.* 2005; Søndergaard *et al.* 2010).

Macrophyte composition changes can also be used in setting class boundaries either by using a shift in community composition from sensitive to tolerant taxa or using a shift from low-growing macrophytes, such as Charophyta, to canopy-forming species (Moss *et al.* 2003; Free *et al.* 2006). Several difficulties impeded the
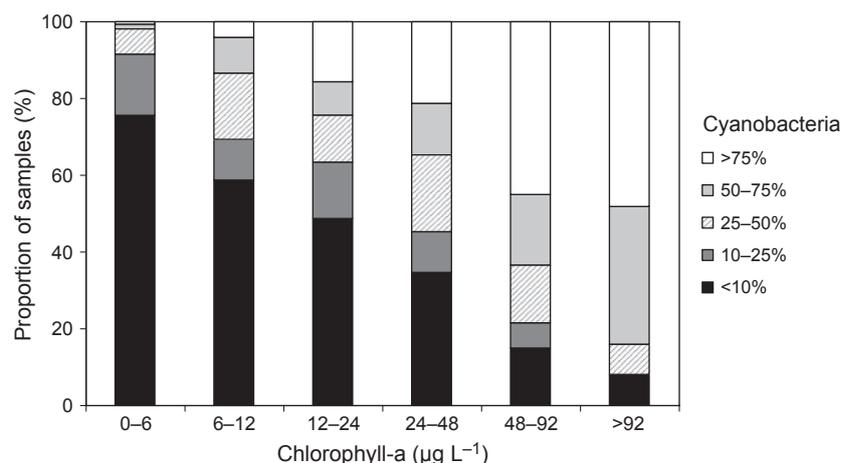
Fig. 4. Proportions of samples with different percentages of cyanobacteria in relation to chlorophyll *a* based on late summer samples.
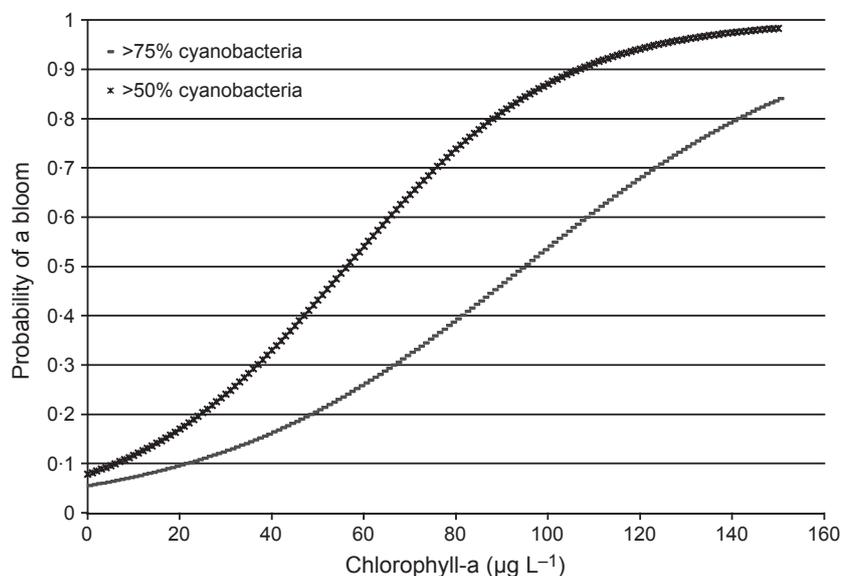
Fig. 5. Modelled probability of a bloom of cyanobacteria in relation to chlorophyll *a* concentration according to bloom definitions based on two different levels of cyanobacteria abundance: >50% and >75% of total phytoplankton biovolume.

Table 3. Summary of the chlorophyll *a* (chl-*a*) boundary-setting approach based on changes in the dominance of cyanobacteria. Values derived from the modelled curve in Figure 5

	Deep lakes	Shallow lakes
Bloom definition	% of samples with >50% Cyanobacteria	% of samples with >75% Cyanobacteria
Reference Chl-*a* (µg L^{-1})	3·1%	5·8%
Modelled bloom frequency at reference Chl-*a*	8·9–9·4%	6·6–7·6%
Definition of bloom frequency at 'poor' status	50%	50%
Definition of bloom frequency 'good–moderate' boundary	12·5%	10%
Chl-*a* value for 'good–moderate' boundary (µg L^{-1})	10	21

application of these approaches to our data set: (i) there is no agreed sensitive/tolerant indicator taxa list across the European region (Penning *et al.* 2008); (ii) there is high variability due to different monitoring practices and biogeographical differences among countries (Penning *et al.* 2008); (iii) it is well known that a number of other factors, in addition to eutrophication pressure, influence macrophyte species composition, such as lake size, depth and climate (Rørslett 1991; Scheffer & van Nes 2007). The use of composition data could be explored in the future provided a more harmonized data set becomes available.

Several studies have found a high variability in macrophyte abundance metrics with no well-defined thresholds (Søndergaard *et al.* 2010) and questioned their use in lake assessment (Penning *et al.* 2008). Our study was, however, successful in finding clear thresholds in macrophyte abundance to eutrophication pressure. The reasons for this difference include (i) in this study, macrophyte abundance data were collected using a harmonized abundance scale (Moss *et al.* 2003) and a harmonized species list; (ii) smoothing techniques were applied to reveal more clearly the underlying trend in data while reducing the effect due to random variation; (iii) in our final analyses, only rooted submerged macrophytes and charophytes were included (excluding emergent, floating, non-rooted macrophytes and macroalgae). We believe the latter reason is a key factor for establishing strong relationships as different growth forms react differently to eutrophication (Hough *et al.* 1989; Moss *et al.* 2003).

Defining thresholds describing the shift between the vegetated clear water state and the non-vegetated turbid state is complicated by the fact that there is no single critical nutrient or chlorophyll *a* level for maintaining a clear state, as factors such as lake size, depth distribution, climate and biological impacts such as grazing by benthivorous fish (e.g. Scheffer & van Nes 2007) affect the threshold. Macrophytes have a stabilizing effect and may maintain the clear water state even at elevated nutrient concentrations (Scheffer 1998). This effect may explain the surprisingly broad range of TP thresholds for the switch between macrophyte and phytoplankton dominance, with values given in the literature ranging from 50 to 700 µg L^{-1} (e.g. Mjelde & Faafeng 1997). Studies of chlorophyll *a* thresholds are more consistent, indicating that a summer concentration of 20 µg L^{-1} chlorophyll *a* represents a threshold at which a switch to the turbid phytoplankton state occurs (e.g. Søndergaard *et al.* 2010). These results concur with our findings as well as with empirical studies (e.g. Liboriussen & Jeppesen 2003).

SETTING OF ECOLOGICAL QUALITY BOUNDARIES BASED ON AN INCREASE IN CYANOBACTERIA

Cyanobacteria dominance is one of the most important and detrimental ecological effects of eutrophication (Chorus *et al.* 2000), as many cyanobacteria taxa can become a nuisance for water quality management and have severe negative effects on ecosystem services delivered by freshwaters (Carvalho *et al.* 2013). Our analysis revealed a clear relationship between chlorophyll *a* and the relative abundance of cyanobacteria. Our chlorophyll *a* thresholds for the prevention of cyanobacteria bloom formation were in agreement with thresholds identified in other lakes in Europe and around the world. The range of TP values defined as thresholds for cyanobacteria dominance suggested in the literature is quite wide, ranging from 10 to 100 µg L^{-1} (e.g. Downing, Watson & McCauley 2001) due to other factors influencing cyanobacteria dominance such as turbulence, water temperature, pH and light availability (Dokulil & Teubner 2000). Few studies define a threshold for chlorophyll *a* concentrations; however, one study defined a threshold of 10 µg L^{-1} chlorophyll *a* for a 10% risk of cyanobacteria dominance (Downing, Watson & McCauley 2001). These data correspond to our findings of 12 µg L^{-1} chlorophyll *a* for deep lakes and 21 µg L^{-1} chlorophyll *a* for shallow lakes.

APPROACH TO SETTING ECOLOGICALLY RELEVANT STATUS BOUNDARIES

Limnologists have developed a sound fundamental understanding of the eutrophication process, its causes and associated effects (Smith, Joye & Howarth 2006). However, this knowledge has rarely been used to set quality targets for lake management and restoration (Birk *et al.* 2012). We have demonstrated an ecosystem-based approach to define ecological quality targets for phytoplankton abundance in lakes and have established chlorophyll *a* values for the 'good–moderate' class boundary based on three ecological effects of eutrophication-enhanced phytoplankton abundance: a decline of aquatic macrophytes, an increasing dominance of cyanobacteria and a switch from a vegetation-dominated to a phytoplankton-dominated state. The relevance of setting ecologically relevant boundaries that are consistent between biological quality elements becomes even more important because macrophytes provide a structural element for benthic invertebrates and fish (other biological quality elements mandatory under the WFD).

LIMITATIONS OF THE STUDY

Many studies have shown that latitude, residence time, temperature, sediment composition, water level fluctuation and biotic factors can affect phytoplankton and macrophyte composition or abundance (e.g. Scheffer & van Nes 2007). Some of these factors may only have a minor effect on the chlorophyll *a* boundaries because they are restricted by focusing on lakes in the Central European region (i.e. they have similar latitudes and climate related variables). However, more data are needed to support this assumption.

Sampling methodologies and analyses have differed among MS. For instance, different sampling depths may cause slightly different chlorophyll *a* values because of vertical distribution patterns of phytoplankton (Nõges *et al.* 2010). We expect that more comparable data would reduce the statistical noise of the relationships developed in the present data sets. Finally, we have only considered the effects of eutrophication on primary producers, namely, phytoplankton and submerged macrophytes. Effects on consumers (benthic invertebrates, fish and birds) have not yet been considered.

A further limitation of our study is that the results cannot be applied to all lake types, a particular example being naturally eutrophic lakes of continental lowlands (Borics *et al.* 2013) that were not included in our analysis. Due to the high evaporation/precipitation ratio and low geographical relief, these water bodies have long water residence times and are practically endorheic. As a result, the nutrient content of these lakes, even in a relatively natural state, can be high, that is, TP > 100 µg L^{-1} and no significant relationships may be found between phosphorus, chlorophyll and macrophyte coverage (Krasznai *et al.* 2010; Borics *et al.* 2013). As well as different targets, these lakes may require different management measures as a reduction in nutrient loading may not result in an expected response. Identifying functionally different lake types and setting eutrophication targets for these lakes is an important area that requires further research.

TURNING SCIENCE INTO POLICY

Successful lake management requires close cooperation between scientists and decision makers. This work was performed within the framework of the WFD Common Implementation Strategy Intercalibration exercise as a result of intensive cooperation among the scientific and policy communities of all involved MS.

The results have been endorsed by all participant MS, adopted in the European Commission legislation (EC 2008) and in the national classification systems, providing a sound scientific basis for informed policy at the national and international levels. Guidance for the transposition of agreed values to national types is provided in Appendix S4 (Supporting information).

CONCLUSIONS

In conclusion, we defined chlorophyll *a* targets in relation to ecologically relevant shifts in ecosystem structure and functioning. We defined the 'good-moderate' status boundaries in the range of 8-10 µg L^{-1} chlorophyll *a* for moderately deep lakes and 21-23 µg L^{-1} for shallow lakes

in Central Europe. Our approach is based on ecologically meaningful guiding concepts (Willby 2011) and evaluation of risks of human impact. We recommend that this approach of ecological target setting be more widely used for other water body types and for biomonitoring schemes in other types of ecosystems.

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Appendix S1. Sampling and analytical methods of phytoplankton.

Appendix S2. Description of submerged macrophyte abundance scale.

Appendix S3. Classification tree analysis.

Appendix S4. Guidance on the transposition of common boundaries into national assessment systems.