A revised approach to setting Water Framework Directive phosphorus standards.

UKTAG

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Contents

1. Summary

1. This report presents a proposal for a set of revised phosphorus standards for protecting High and Good status under the Water Framework Directive. It is part of a wider analysis commissioned by the UKTAG into the management of nutrients in freshwater (Willby et al 2012) following on from a review undertaken to inform refinements to UK conservation guidance on the setting of conservation objectives for designated river habitat (Mainstone 2010a, 2010b).

2. The current standards for High Ecological Status and Good Ecological Status, generate a large number of classification mismatches, with biology failing more frequently than phosphorus. Now that revised macrophyte and diatom tools have been developed, their boundaries intercalibrated with those from other Member States, and a combination rule devised, the opportunity has been taken to develop a new set of P standards that are better tuned to biological conditions.

3. In addition to analysis of a large dataset of biological and chemical data from UK rivers, the project also reviewed the scientific literature on eutrophication and its consequences in rivers (p. 4-7) and collated the sparse information available on standards used in other EU countries.

4. An innovative approach to standard setting has been developed. This first predicts the P concentration at reference/near reference conditions, using alkalinity and site altitude as variables, and then calculates an "P EQR" as the ratio between observed and expected P. Finally, a regression equation links the biological and P EQRs, allowing P concentrations associated with the mid point of each biological class to be determined (p. 8-9 and Appendix 1).

5. One benefit of this approach is that it does not rely on a typology, as site-specific nutrient targets are developed. This reduces the variability caused by the large range of conditions covered by the current typology and in particular the significant country effect (Fig 4)

6. As a primary objective of the review of the standards was to reduce the mismatch between the phosphorus and biological classifications, it is proposed that class boundaries are set at a point mid-way between the classes, a phosphorus concentration where there is an equal likelihood of the biology being in adjacent classes (Fig 2). Values produced for both high/good and good/moderate status boundaries are, generally, more stringent than the existing type-specific boundaries, although they are closer to concentrations at which the consequences of eutrophication have been reported in the literature (Tables 3-6; pp. 11-12).

7. The proposed values align better with predicted biology classes than the current standards with 46% of sites classified into identical classes, 85% classified to ± 1 class and a low level of bias (5%) and towards chemical, rather than biological, failures (existing standards show a bias of 28% to biological, rather than chemical failures) (Tables 7 & 8; Figs 6-9, pp. 15-17; Appendix 4).

8. Overall, there will be a 9-13% increase in the number of sites that will fail the proposed good/moderate boundary criterion in comparison with the current standards (see Appendix 4).

9. The site specific standards could be simplified to a type-specific approach, with similar levels of overall performance. However, the wide range of threshold P concentrations for high alkalinity lowland rivers (Type 3n) and the significant country differences between England and other areas of the UK will inevitably lead to more misclassifications and it is proposed that site standards are used.

10. Comparison with standards set by other countries was complicated by the wide variety of river types and approaches to standard setting adopted by these countries (Table 9, p. 19) and no general conclusions can be reached about the relative levels of precaution adopted. The upper end of proposed UK values is more relaxed than standards for some countries (e.g. Ireland) but of a similar magnitude to those in countries such as The Netherlands, allowing for the differences in determinand and statistics used.

11. Overall, the new site-specific standards are recommended as an improvement over existing standards that will yield fewer mismatches between biology and chemistry and greater probabilities of beneficial outcomes when used as part of integrated nutrient management programmes.

12. A comparison of recent biological classifications (2010) with the proposed boundaries will be carried out to provide an assessment of the impact of the changes using a larger independent data set.

13. In carrying out this work a comparison was also made with phosphorus targets developed by the UK conservation agencies for protecting river habitat in rivers with special wildlife designations. This concluded that the modelling approach presented here could be used to inform future agreement over favourable condition targets. However, further work is required before clear recommendations can be made. This will be taken forward, together with a similar review of lake P standards and is not included in this extract report.

2. Introduction

This report outlines a rationale for revising regulatory standards for phosphorus to achieve High Ecological Status (HES) and Good Ecological Status (GES) objective in rivers under the Water Framework Directive. It is an extract of a wider review requested by UK TAG which included a comparison with nutrient targets used for rivers with special designations for wildlife (Willby et al 2012). This report aims to address problems associated with current standards for GES and proposes revised standards derived from a combination of the evidence in the scientific literature and data from UK water bodies which links observed ecological status to water chemistry. Further work is needed to extend the analysis undertaken to provide a broader framework which can guide planners towards the steps necessary for restoration or protection of freshwater habitats, including those with special wildlife designations. The original brief was to review both lake and river standards, but because of delays in the intercalibration process for lakes, it has only been possible to review standards for rivers in this report. These additional issues will be reported separately in a subsequent document.

There is a large body of research demonstrating the importance of nutrients in determining the ecology of rivers (Hilton *et al*., 2006; Mainstone, 2010) plus a general awareness that nutrient concentrations in UK freshwaters are a major cause of failures to achieve legislative goals (Carvalho & Moss, 1995). Apart from well-documented effects on biodiversity, elevated nutrient concentrations also have indirect effects on rivers and, as a result, on ecosystem services, particularly through enhanced productivity, leading, for example, to deoxygenation, fish kills and increased flood risk caused by reduced conveyance.

Standards for phosphorus to support High Ecological Status (HES) and GES in rivers were developed as part of the first round of environmental standard development (UK TAG, 2008). They were developed using limited data and were established before the biological metric boundaries were finalised during the intercalibration process. Standards were based on the distribution of reactive P at sites which achieved HES or GES using diatoms (the macrophyte assessment tool was not available at this stage) (Table 1). The decision of UK TAG to use the 95th percentile of this distribution resulted in mismatches between the chemical standards and diatom results (Barahona, 2009) with approximately twice as many river sites in England and Wales classified as high or good status using phosphorus(65%), compared to using diatoms (35%) (Phillips, 2008), in Scotland 40% and in Northern Ireland 75% of sites were classified at least one class lower by diatoms than by phosphorus.

The availability of a broader range of WFD ecological tools along with more data, a growing awareness of the limitations of the WFD phase 1 standards and a need to reach agreement on standards with the UK conservation agencies led to this topic being revisited by UK TAG using a consistent approach across both lakes and rivers. Furthermore, both diatom and macrophyte methods have developed since the first set of standards were produced. Both have now been successfully intercalibrated and WFD ecological boundaries for these have been finalised.

Note on terminology

Most analyses of P by UK agencies are of molybdate reactive P in unfiltered samples from which large particles have been allowed to settle, referred to here as "reactive P" (RP). In practice, the difference between RP and soluble (= filtrable) reactive P (SRP) is usually minor

Table 1. Current P standards to protect WFD ecological status in rivers (UKTAG, 2008). Values expressed as annual mean Soluble Reactive P / Total Reactive P at a sampling point (µg L^{-1}).

Type	Altitude (m)	Akalinity (mg L^{-1} $CaCO_3$)	SRP concentration			
			High	Good	Moderate	Poor
1n	< 80	< 50	30	50	150	500
2n	> 80	< 50	20	40	150	500
3n	< 80	> 50	50	120	250	1000
4n	> 80	> 50	50	120	250	1000

3. Evidence base

This section summarises evidence available at the time of writing. Readers are referred to Mainstone (2010) for a more thorough review of the subject.

Aquatic ecosystems in their natural state are mostly nutrient-poor (even naturally eutrophic systems), and the specialist photosynthetic organisms that thrive under these conditions have efficient nutrient uptake mechanisms, adaptations to use "bound" phosphorus and, often, slow growth rates (Biggs et al., 1998). Addition of nutrients favours species adapted to exploit enriched conditions which will, as concentrations increase, eventually displace the specialists. This response is central to the establishment of boundaries for ecological status as defined by the WFD. Biological classification tools recognise changes in the quality element as a whole in terms of a continuum of change (species loss or turnover, or change in abundance) in relation to increasing pressure. Such changes are captured through the combination of various metrics which integrate the biological response to pressures. As such, we regard the relationship between pressure and biology as essentially linear, although at progressively finer levels of organisation it is rather obvious that non-linearities must exist.

Many of the plant taxa adapted to high nutrient concentrations are capable of producing large biomass if other conditions are favourable (e.g. light, stable substratum, low frequency/magnitude of hydraulic scour, availability of other potentially-limiting nutrients). This biomass, and associated primary productivity, can have effects on other trophic levels. Shifts in community composition occur in the macroinvertebrate and fish communities as the nature and quantity of food sources change (Harper et al 2009, Graham et al 2009), leading to reduced competitiveness of species with efficient foraging mechanisms (such as many stoneflies and Atlantic salmon) adapted to naturally low levels of productivity. At high levels of enrichment respiration by a high plant biomass can generate significant mortality of invertebrates and fish through deoxygenation. These are examples of the “undesirable disturbances” referred to in the normative definitions of GES in the WFD.

The concentrations at which adverse changes can be observed vary widely in the literature, reflecting the many other factors which can also influence manifestations of eutrophication.

1. Nutrient status varies naturally with the natural productivity of the catchment, and the characteristic communities of the river (algae, macrophytes, macroinvertebrates, and fish) are composed of species adapted to that status as part of the environmental conditions of the river. Natural nutrient status and environmental conditions generally are driven by factors such as geology and altitude, as well as position within the river system (Vannote et al. 1980) in which nutrients spiral downstream from higher energy supply zones to downstream lower energy depositional reaches. Anthropogenic nutrient enrichment is superimposed on this natural ecological template.

2. There is a widespread assumption that phosphorus limits plant growth in freshwaters, although there is increasing evidence that nitrogen plays a key role (Elser et al., 2007) and that co-limitation is common. It is possible that silicon, too, may also limit under some circumstances (Casey et al., 1981).

3. In many rivers, the magnitude and frequency of scouring flows critically dictates the size of the algal standing crop observed at any one time. Other factors, such as management by cutting, grazing by water-birds (most notably swans), or shading by riparian trees will also influence the extent and composition of aquatic vegetation at a given concentration of P.

4. Grazers can limit the amount of plant biomass that can form at a site (Rosemond et al., 1993), such that nutrient enrichment increases the dependency on the grazing community for limiting plant biomass accrual. Other stresses on the grazer community, such as toxic pollution or siltation, may lead to an uncoupling of the relationship between grazers and algae even at relatively low nutrient concentrations, and result in higher plant biomasses (Armitage, 1979).

5. A final consideration is the extent to P species not detected by the routine measurements made by regulators exert an effect on the biota (see Whitton & Neal, 2010).

The range of soluble P concentrations over which effects have been reported are summarised in Fig. 1, whilst Table 2 provides a range of limits recommended to protect against eutrophication impacts. Increased algal growth and standing crop have been observed at a range of concentrations down to a few micrograms per litre (Bowes et al., 2007; Biggs, 2000; Lohmann et al., 1992), reaching a plateau at concentrations around 100 µg/L (detailed interpretation is difficult due to the variety of methods in use). Effects on macrophytes are often confounded with a range of other driving variables shaping the macrophyte community, including hydraulic scour. O'Hare et al. (2010) suggest a growth response to enrichment from *Ranunculus* that plateaus at around 100 µg/L. Effects on other trophic levels have been observed at low P concentrations: increased leaf decomposition rates at around 10-30ug/L (Gratten & Suberkropp, 2001; Gulis & Suberkropp, 2003) reducing the role of shredders plus a general shift towards grazers and fine particle collectors (Harper et al., 2009). Increased algal biomass, in turn, leads to higher invertebrate biomass which tends to favour predators with less efficient foraging strategies; e.g. trout were found to be favoured over salmon, at >30 µg/L reactive P in Irish streams (Graham et al. 2009). What is less clear is how these observed effect concentrations map onto the range of river types and sizes in the UK, and what concentrations should be set in different river types to regulate these effects to an appropriate degree.

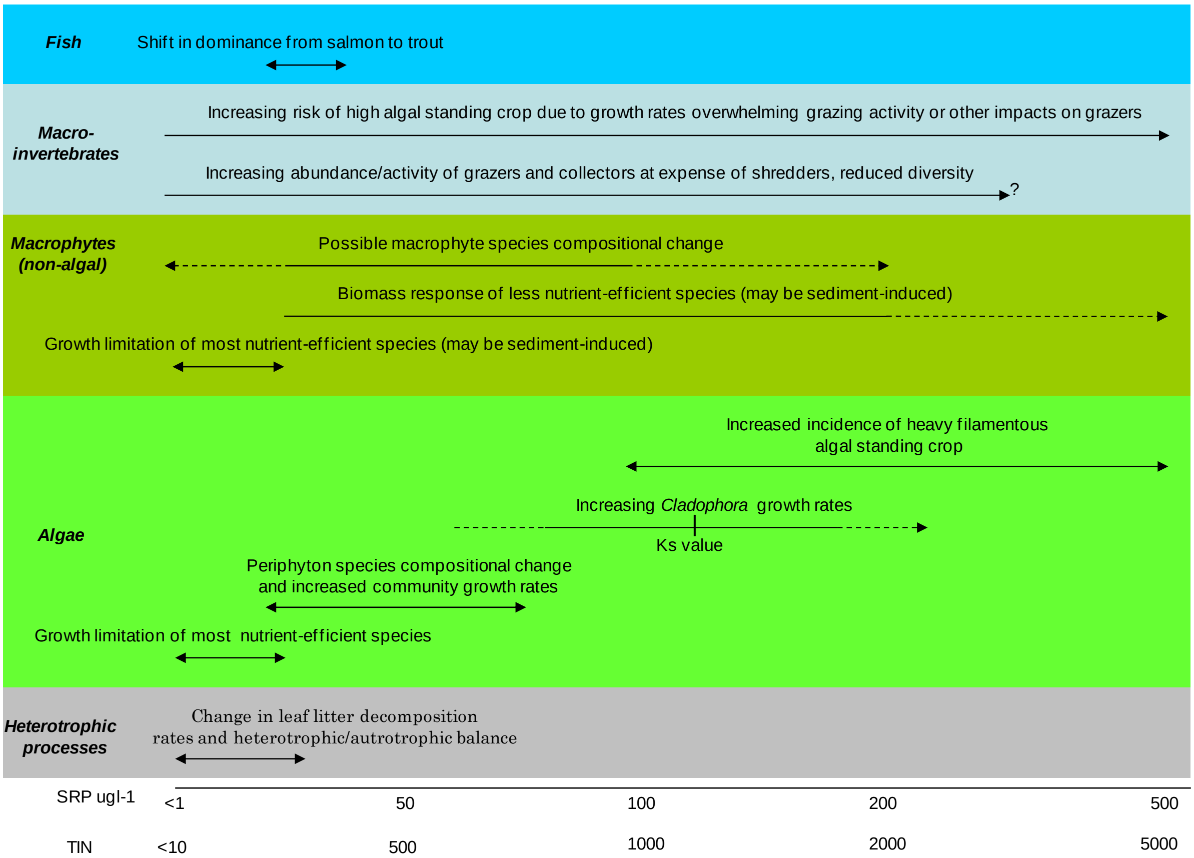

Fig. 1. Synthesis of reported biological changes in streams along a gradient of nutrient availability (from Mainstone, 2010).

Table 2. Some recommended phosphorus limits (μgl^{-1}) for controlling nuisance conditions and ecological degradation caused by riverine algae.

Total P	SRP	Risk	Source	Comments
38-90		Nuisance	Dodds *et al*. (1997)	
20		*Cladophora*	Chetelat *et al*. (1999)	From study of algal standing crop in a range of Canadian streams with conductivities of 60-700 μs/cm
10-20		*Cladophora*	Stevenson, pers. comm. to Welch et al. (2004)	
	60	Eutrophy	Environment Agency (1998)	
	10	Eutrophy	Biggs (2000)	From study of algal biomass accrual in a wide range of streams and rivers in New Zealand, across a range of geologies and land uses
	3	Invertebrates	Nordin (1985)	
	15	Nuisance	Quinn (1991)	
18	6	Nuisance	Sosiak (2002)	From a study of periphyton and macrophytes in the Bow River (Canada), running off the Rocky Mountains
	30	Invertebrates	McGarrigle (2009)	From a large dataset of routine macroinvertebrate data on Irish rivers and streams
	50	Algal growth	Bowes et al. (2007)	From stream mesocosm study of algal biomass accrual in a chalkstream
30		Invertebrates and algae in large rivers	Smith and Tran (2010)	Based on observed shifts in the structure of benthic diatom and macroinvertebrate communities in 40 large rivers in New York State

4. Analysis of UK data sets from routine monitoring

Data sets from spatially matched sites in the UK (i.e. sites where there is contemporaneous data on diatoms, macrophytes and nutrients collected over the same time period usually from the same site but always <500m apart)provide the opportunity to establish empirical relationships between observed phosphorus concentration and biological status as assessed by the WFD biological metrics derived from benthic diatoms and macrophytes. Such relationships allow a more robust derivation of standards applicable to UK rivers. The rationale behind model development is outlined in Appendix 1. The outcome is a model that first predicts an expected 'background' phosphorus concentration at reference or near-reference conditions (equation 1) and then relates the increase in observed phosphorus from background (a measure of anthropogenic pressure) to changes in diatom and macrophyte communities (equation 2). Reference P concentrations are predicted using alkalinity and site altitude, and thus take account of the main sources and controls of natural variation in soluble P concentrations (i.e. rock weathering). No other factors in our dataset could explain a significant amount of additional variation in P concentrations in the population of reference sites. Residual variation in P concentrations in reference sites presumably reflects the influence of differences in processing rates, the importance of biological and sediment sinks, different routes of delivery and the sensitivity of all these processes to climatic and site-based environmental factors. This overall approach mirrors that used for phosphorus standards for lakes (UK TAG, 2008), and the use of an alkalinity-altitude reference P model in rivers is analogous to the well-established use of the morpho-edaphic index (based on alkalinity and depth) in lakes.

$$\text{Expected RP} = 10\text{\^{}}(0.454 \times \log_{10}(\text{alkalinity}) - 0.0018 \times (\text{altitude}) + 0.476) \qquad \text{(equation 1)}$$

$$\text{Boundary RP} = 10\text{\^{}}((1.0497 \times \log_{10}(\text{EQR}) + 1.066) \times (\log_{10}(\text{Expected RP}) - \log_{10}(\text{Upper Anchor})) + \log_{10}(\text{Upper Anchor})) \qquad \text{(equation 2)}$$

where:

RP = reactive P as µg/L annual mean
as Alkalinity = long term annual mean in mg/L $CaCO_3$ (min 2.0, max 250.0)
altitude = altitude above mean sea level as metres (max 355)
EQR = biological EQR for "macrophytes and phytobenthos" (minimum of diatoms and macrophyte assessments, see appendix for values used).
Upper Anchor = maximum value for SRP in training data set (3500 P as µg/l; this is, in essence, an arbitrary value used to invert the P EQR such that a highly impacted site has an EQR approaching 0)

This model and its associated uncertainty allows a prediction of the phosphorus concentrations expected to be found at any point along a biological impact gradient defined by the EQR (Fig 2). The WFD requires 5 classes to be defined, but, due to the wide range of factors that influence the P concentration at any given site and the biological response to this concentration, a relatively wide range of phosphorus concentrations can be expected at any particular biological status. Given this variability it is relatively difficult to identify precise concentrations of phosphorus at which there is a likely to be a clear shift in biological status as defined by plant-based EQRs. However, we propose that an appropriate ecological status boundary condition is defined as the point where the errors of the predicted phosphorus concentrations for adjacent classes overlap (dotted lines in Fig 2). This approach produces standards for classification that minimise the chance of mis-classifications when compared to plant-based EQRs, and values that are neither precautionary nor relaxed when used to guide decisions in water management. In Fig 2 we have chosen for convenience to represent the uncertainty in the modelled standards using the upper and lower 25th quantiles of the model residuals. The choice of the type of estimate of model precision to use is not critical in our approach since we have located the standards in terms of the overlap of the error of adjacent classes and, following this rationale, this value will not be changed by using alternative statistical measures of model precision such as the prediction error or the 95% confidence limits.

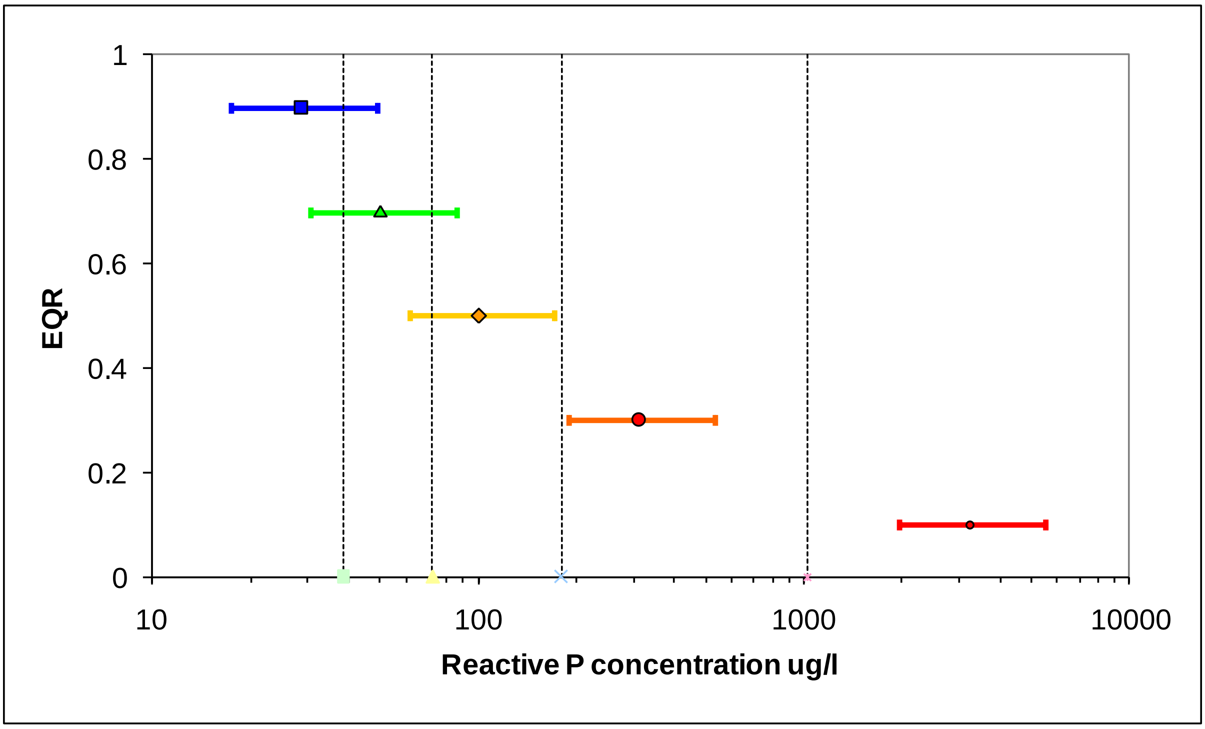

Fig 2 Predicted range of phosphorus concentrations at the mid point of High, Good, Moderate, Poor and Bad biological status for a typical lowland high alkalinity river (Alkalinity = 130 mgCaCO3/L, Altitude = 28m). Horizontal lines represent predicted reactive phosphorus ± 25th percentile of model residuals (50% of predicted results), dotted lines mark proposed reactive phosphorus boundaries at the mid point of adjacent classes.

We recognise that the error structure of our model is complex and that by focussing on the statistical uncertainty of the model itself we are effectively ignoring several in-built sources of error associated with measurement of model terms (e.g. P concentration and biological EQR) and the prediction of reference P. It would be instructive in the future and with less pressing time scales to quantify these sources of error and explore alternative statistical approaches for defining boundaries that may capture additional elements of complexity in this system. We note that several alternative models (not presented here in the interests of brevity) using the same predictors or different formulations of the P EQR generated very similar boundary values but with higher uncertainty or a non-linear form. We also used a bespoke dataset on macrophytes linked to a large suite of environmental data collected at the same sites but were unable to explain additional variability in the pressure-response signal using factors such as accrual time and tree shading.

The model provides predicted phosphorus values for any site, given site altitude and alkalinity. This allows site-specific standards to be produced. Fig 3 shows the range of site-specific reference and boundary values for each of the current river types. To further illustrate the magnitude of the proposed standards, Tables 3-5 show example concentrations generated for an indicative range of alkalinity and altitude for reference/near-reference conditions and the high/good and good/moderate status boundaries. Site-specific standards provide the most appropriate standards for each site, as they take into account the significant variation in conditions (principally alkalinity) that occur within the current river typology and between countries (Fig 4). However, indicative type standards have also been calculated (Table 6) from the median site boundary values derived from the site standards applied to all river sites monitored by the UK environment agencies (c6900 sites).

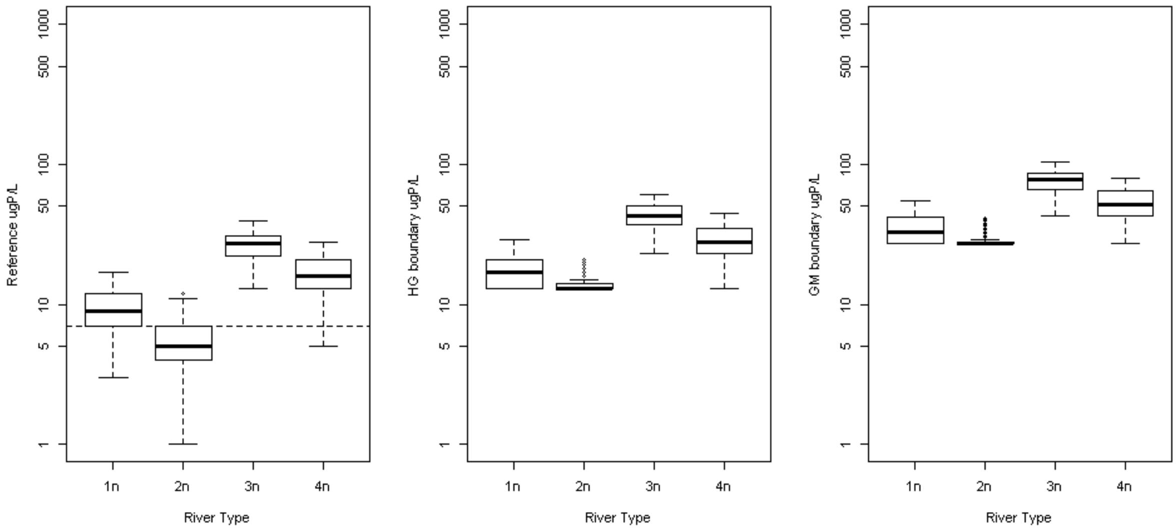

Fig 3 Range of site specific reference, high/good and good/moderate boundary values for UK rivers derived from proposed model. (note that due to analytical detection limits a minimum value for reference RP of 7µg/l is used, represented as a dotted line). The “box” shows the 25th and 75th quartiles along with a bold line indicating the median. The "whiskers” show the largest and smallest observations within a distance of 1.5 times the box size from the nearest hinge.

These numbers are generally lower than the current standards, but are closer to values reported in the literature, with the maximum G/M boundary always less than 100 µg/l. This is to be expected as the approach used in Phase 1 effectively sought to establish the highest P values associated with all rivers of a given type at a given status; within a river type such values will always be found at the highest alkalinity and lowest altitude.

Type standards have the advantage of simplicity, but as the current river typology covers a relatively wide range of conditions, which vary between countries within the UK, they will provide a standard that may not be appropriate at the site-level. Boundary effects in a simple typology are considerable, resulting in large step changes in standards along individual rivers (see 4.3: Fig 10 for example). Country-level variation within individual river types also means that, for example, a type-specific median boundary value of 77 ug/L for lowland high alkalinity rivers (type 3n) would be too stringent on average for this river type in England yet not sufficiently protective for the rest of the UK. This is because in England sites have significantly higher predicted boundary values than similar types from other parts of the UK, due to English examples being skewed (through a combination of geological factors and the disposition of the monitoring network) towards the lower altitude and higher alkalinity extremes of the 3n river type (Fig 4).

This within-type variation in expected phosphorus (particularly acute for type 3n) also contributes to uncertainty in standards derived from the distribution of observed phosphorus concentrations within each of the river types (Fig 5). The use of site standards overcomes these issues and, while potentially more difficult to illustrate (see fig A4 for example presentation) they are simple to calculate and administer. They are therefore the recommended approach.

Table 3. Mean reactive P concentrations (µg L^{-1}) for rivers at reference/near-reference conditions, predicted from equation 1 (with min value of 7 applied). Cell colours indicate the typology used for phase 1 standards: yellow: type 1 (low alkalinity, low altitude); orange type 2: low alkalinity, high altitude; blue: types 3 (high alkalinity low altitude) and green 4: high alkalinity, high altitude.

Ref		Alkalinity (mg/l $CaCO_3$)									
		5	10	20	40	50	75	100	150	200	250
Altitude (m)	0	7	9	12	16	18	21	24	29	33	37
	20	7	8	11	15	16	19	22	27	30	33
	40	7	7	10	13	15	18	20	24	28	31
	60	7	7	9	12	13	16	18	22	25	28
	80	7	7	8	11	12	15	17	20	23	25
	100	7	7	7	10	11	13	15	18	21	23
	200	7	7	7	7	7	8	10	12	13	15
	300	7	7	7	7	7	7	7	7	8	9
	350	7	7	7	7	7	7	7	7	7	7

Table 4. Mean reactive P concentrations (µg L^{-1}) for rivers at the high/good status boundary, predicted from equations 1 and 2. See Table 1 for guide to cell colours.

HG		Alkalinity (mg/l $CaCO_3$)									
	0.72	5	10	20	40	50	75	100	150	200	250
Altitude (m)	0	13	16	21	27	30	35	40	47	53	58
	20	13	14	19	25	28	33	37	43	49	53
	40	13	13	17	23	25	30	34	40	45	49
	60	13	13	16	21	23	28	31	37	41	45
	80	13	13	15	20	22	25	29	34	38	42
	100	13	13	14	18	20	23	26	31	35	38
	200	13	13	13	13	13	15	17	21	23	25
	300	13	13	13	13	13	13	13	14	15	17
	350	13	13	13	13	13	13	13	13	13	14

Table 5. Mean reactive P concentrations (µg L^{-1}) for rivers at the good/moderate status boundary, predicted from equations 1 and 2. See Table 1 for guide to cell colours.

GM		Alkalinity (mg/l $CaCO_3$)									
	0.543	5	10	20	40	50	75	100	150	200	250
Altitude (m)	0	27	31	40	51	56	64	71	82	91	99
	20	27	29	37	48	52	60	66	77	85	92
	40	27	27	35	45	48	56	62	71	79	85
	60	27	27	32	41	45	52	57	66	73	79
	80	27	27	30	39	42	48	53	62	68	74
	100	27	27	28	36	39	45	50	57	64	69
	200	27	27	27	27	27	31	35	40	44	48
	300	27	27	27	27	27	27	27	28	31	33
	350	27	27	27	27	27	27	27	27	27	28

Table 6. Median reactive P concentrations (µg L^{-1}) at reference conditions and status class boundaries (further information on moderate/poor and poor/bad status is in Appendix 3) for the four river types used for current standards.

	reference	High/Good	Good/Mod	Mod/Poor	Poor/Bad
Lowland Low alkalinity Type 1n	8	17	33	90	592
Upland low alkalinity Type 2n	7	13	27	76	548
Lowland high alkalinity Type 3n	21	43	77	173	816
Upland high alkalinity Type 4n	13	28	52	129	706

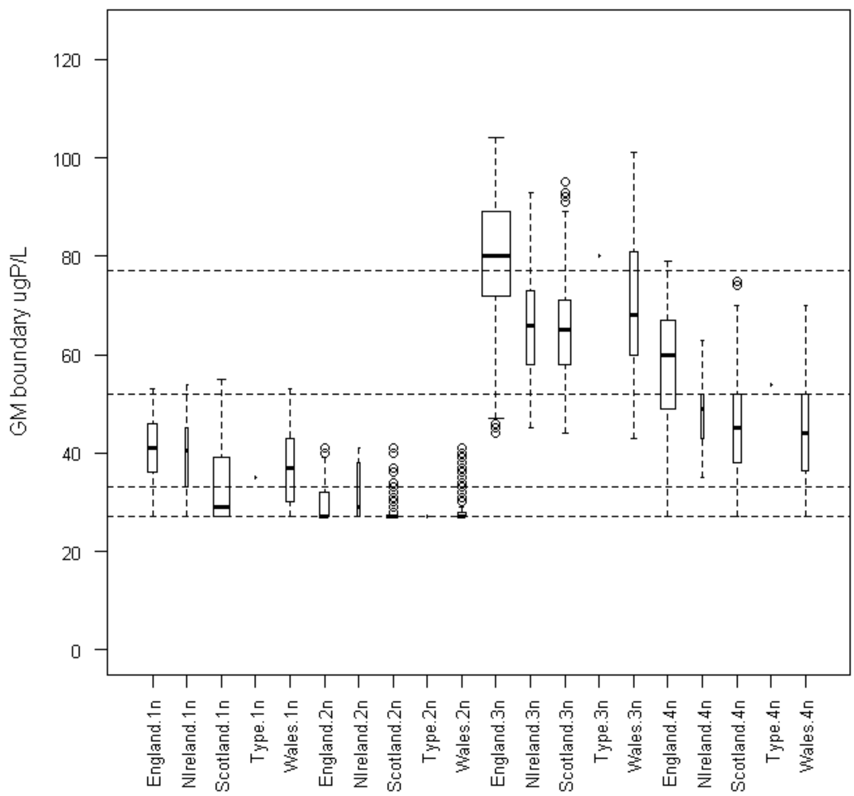

Fig 4 Range of site-specific predictions for good/moderate boundary values split by river type and country. Horizontal lines mark type-specific good /moderate boundary values defined as the median of site-specific values from all countries (ranked from 2n, 1n, 4n, 3n upwards).

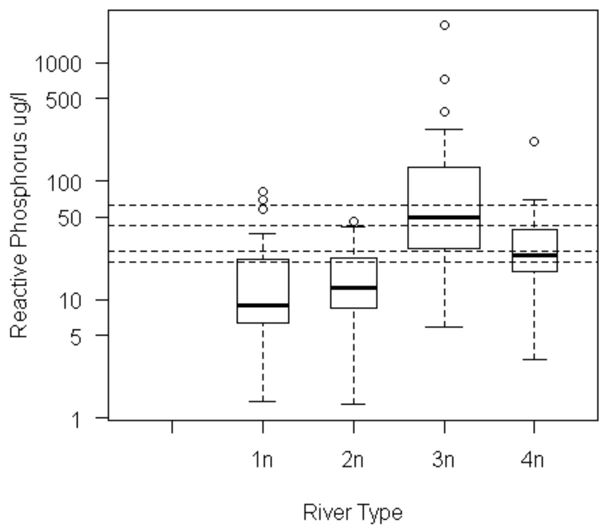

Fig 5 Range of measured mean phosphorus concentration in sites classified at Good status by the worst of either diatoms or macrophytes. Horizontal lines show type-specific good/moderate boundaries.

5. Consequences of implementing revised P standards in the UK

5.1 Compliance with WFD criteria:

Unconstrained, the models produce reference phosphorus concentrations and high good boundary values for some rivers that are close to the limits of detection for routine analysis. In addition the effectiveness of both diatom and macrophyte classification tools prove limited at very low phosphorus concentrations and the combination of these effects initially produced a significant misclassification when biological and P classifications were compared for the low alkalinity river types. To overcome this, a lower threshold concentration of 7 µg/l has been imposed on the reference model. While this truncates the distribution of site-specific boundary values for the low alkalinity rivers, particularly Type 2n, this is justified in the short term to avoid failures due to analytical issues. In the longer term, the issue of lower detection limits (e.g. 1 µg/L) for routine water chemistry needs to be addressed and standardised between countries. Consideration should also be given to the contribution made by other phosphorus species under such circumstances.

Having constrained the model the proposed standards align better with predicted biology classes than the current standards. Amendments to both the macrophyte and diatom tools means that agreement between the biological classification (minimum of macrophytes and diatoms) and the current standards is already improved over the situation described in Phillips (2008), with 46% of sites being classified identically, and 84% of sites classified to $\pm$ 1 class (Table 7) although there is still a bias (28%) towards biology giving a more stringent classification than chemistry. The proposed site standards, on the other hand, have a very similar overall agreement (46% of sites classified into identical classes and 85% classified to $\pm$ 1 class; Table 8); however, the bias is now much lower (5%) and towards chemical, rather than biological, failures.

The highest levels of agreement between biology and phosphorus are for types 1n and 2n (65 and 67% respectively: Figs 6 & 7), slightly lower (48%) for type 4n (Fig 9), but only 31% for Type 3n (Fig. 8). In types 1n and 4n, the proposed site-specific phosphorus boundary values result in slightly more sites downgraded by biology rather than by phosphorus whilst in types 2n and 3n the opposite is true. Across all types, however, the mismatch is lower than with the current standards.

Tighter standards will inevitably result in more sites failing the phosphorus criteria: overall, there is a 9-13% increase in the number of sites that will fail the proposed good/moderate boundary criterion in comparison with the current standards (see Appendix 4). However, since about half of these additional sites would fail anyway on the basis of their biology when a one-out, all-out approach is implemented the net increase in failure is significantly smaller.

The type-specific standards appear to perform relatively well when compared with the site-specific approach in terms of agreement with the biology. The overall classification had a similar number of exact matches (46%) as the site model along with a slightly lower bias (0.9%) towards chemical failures. As the differences are relatively small, ***on average*** type-specific standards could perform reasonably well. However, the wide range of site boundaries for high alkalinity lowland rivers (Type 3n) and the significant country differences between England and other areas of the UK will inevitably lead to more misclassifications. Thus, despite the evidence from the test data set it is still proposed that site standards are used.

Table 7 Agreement between classifications based on biology (minimum of macrophytes and diatoms) and current P standards; based on UK wide test dataset (575 sites)

GLOBAL		Biology					
		High	Good	Moderate	Poor	Bad	
Phase 1 P	High	157	83	52	6	0	298
	Good	14	48	54	7	0	123
	Moderate	6	26	47	9	1	89
	Poor	3	11	20	14	1	49
	Bad	0	1	7	7	1	16
		180	169	180	43	3	575

Table 8. Agreement between classifications based on biology (minimum of macrophytes and diatoms) and proposed site P standards; based on UK wide test dataset (575 sites)

GLOBAL		Biology					
		High	Good	Moderate	Poor	Bad	
Proposed P	High	138	55	31	4	0	228
	Good	27	51	43	6	0	127
	Moderate	11	40	53	6	0	110
	Poor	4	21	44	19	1	89
	Bad	0	2	9	8	2	21
		180	169	180	43	3	575

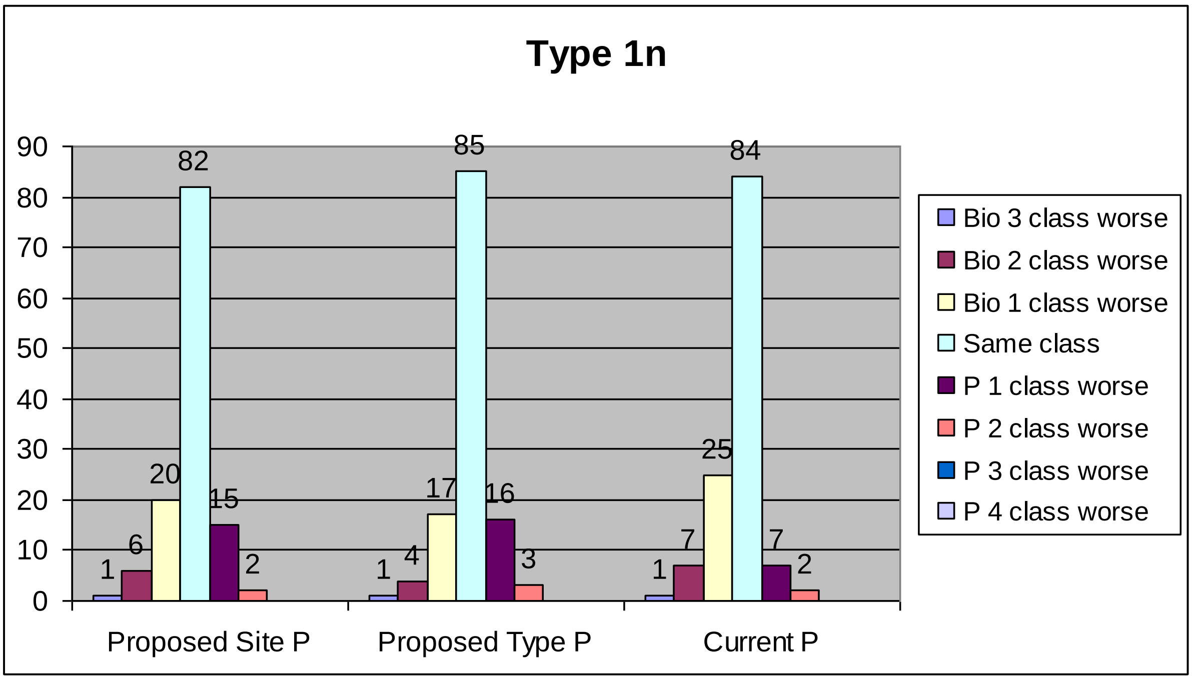

Fig. 6. Comparison between biological and chemical classifications of lowland, low alkalinity UK rivers (Type 1n) achieved using the proposed site-specific P standard, the alternative type-specific P standard or the current P standard.

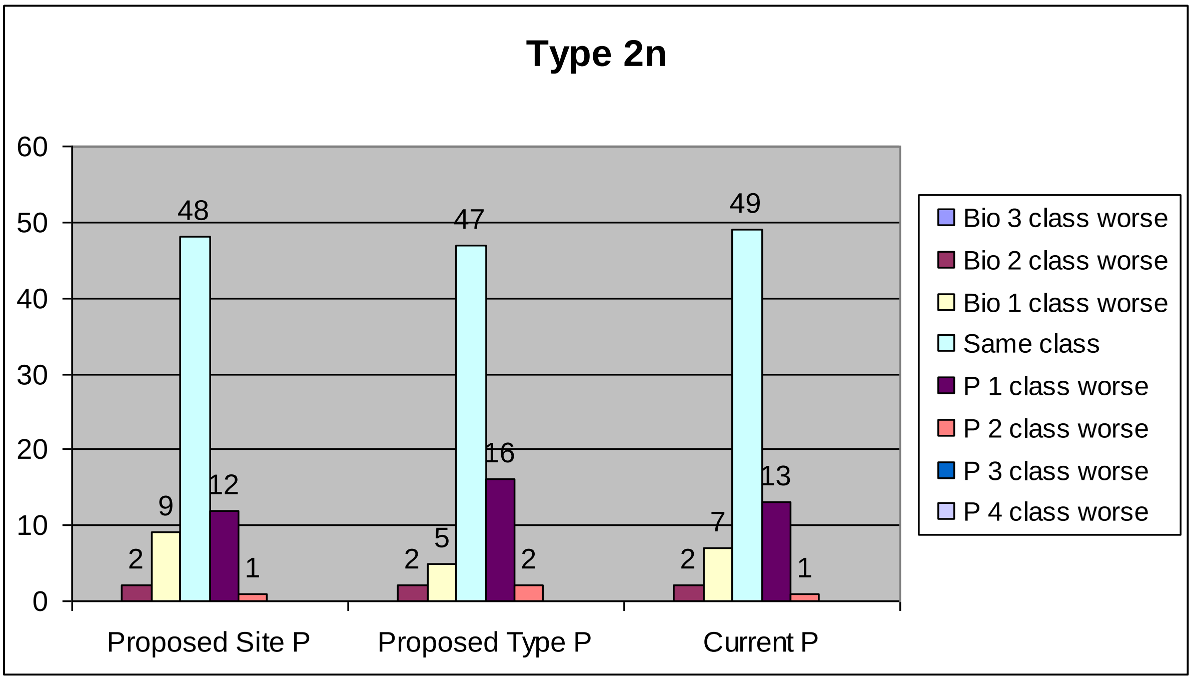

Fig. 7. Comparison between biological and chemical classifications of upland, low alkalinity UK rivers (Type 2n) achieved using the proposed site-specific P standard, the alternative type-specific P standard or the current P standard.

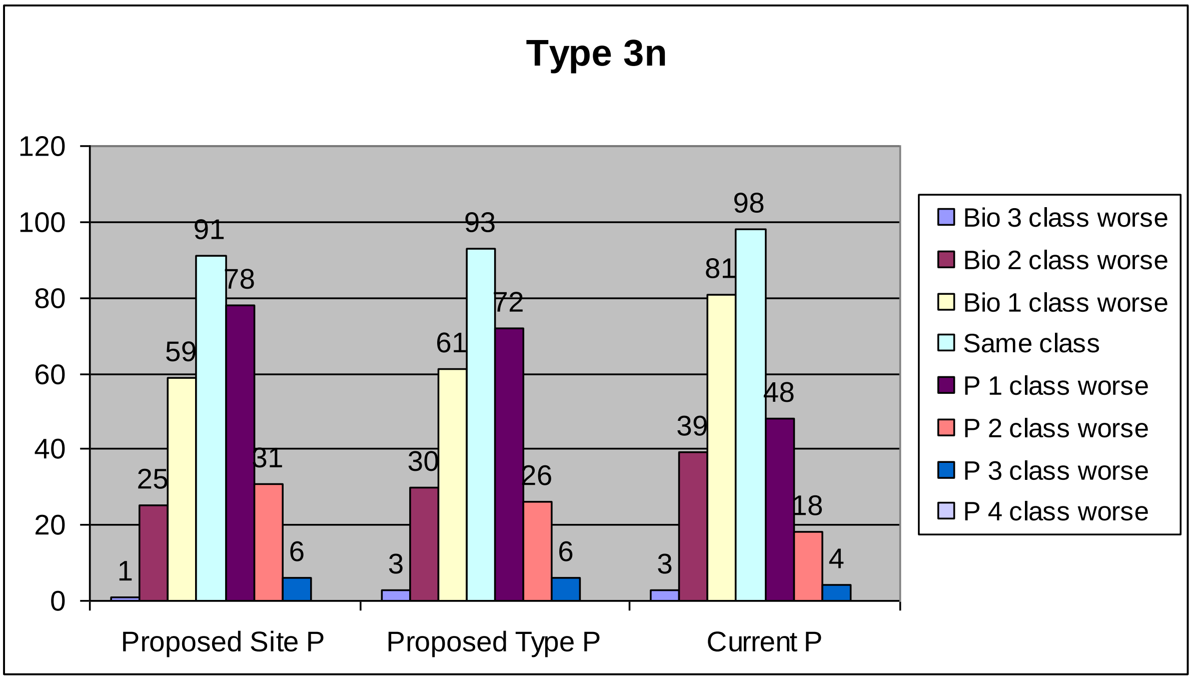

Fig. 8. Comparison between biological and chemical classifications of lowland, high alkalinity UK rivers (Type 3n) achieved using the proposed site-specific P standard, the alternative type-specific P standard or the current P standard.

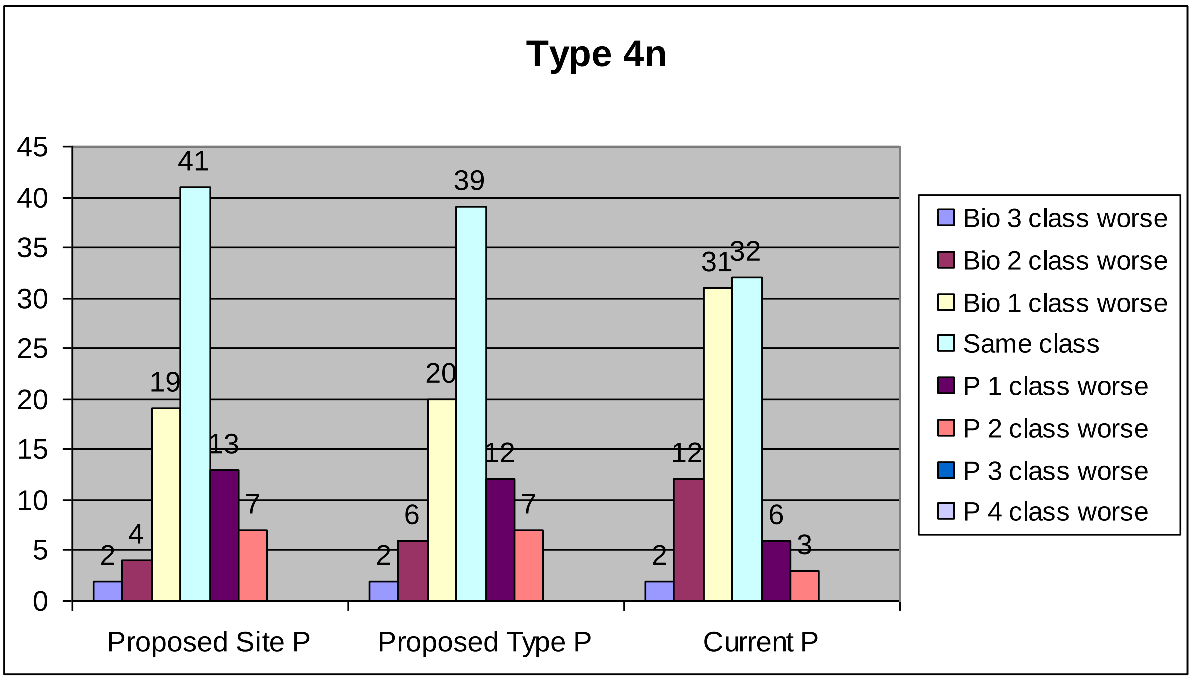

Fig. 9. Comparison between biological and chemical classifications of upland, high alkalinity UK rivers (Type 4n) achieved using the proposed site-specific P standard, the alternative type-specific P standard or the current P standard.

5.2 comparison with standards elsewhere in Europe

Arle et al. (2011) present an overview of the situation throughout the EU; however, comparisons are complicated by the use of different determinands (TP versus reactive P), statistical conventions (means versus percentiles) and typologies. The UK standards are erroneously reported in this document as TP rather than SRP, and comparisons here are, therefore, based on direct contact with others involved in ECOSTAT and intercalibration (Table 9). The type-specific nature of the UK standards means that they span a greater range than for most countries, although the low standards will only apply at high altitude low alkalinity rivers. The upper end of proposed UK values is more relaxed than standards for some countries (e.g. Ireland) but of a similar magnitude to those in countries such as The Netherlands (allowing for the differences in determinand and statistics used). Based on this very limited information it is impossible to determine relative levels of precaution. It should also be noted that the standards themselves are not necessarily an indicator of relative approaches to nutrient management. Information provide by Denmark, for example, suggests that they while they do not have a phosphorus standard for rivers, they apply general binding rules with a very stringent phosphorus standard (0.3 mg/l) to all point source discharges.

Table 9. Comparison between proposed UK standards and P standards currently in force elsewhere in Europe

MS	H/G				G/M			
	SRP		TP		SRP		TP	
	Mean	90th %ile (3)	Mean	90th %ile (3)	Mean	90th %ile (3)	Mean	90th %ile (3)
AT		7-70				15-200		
IE	25	45			35	75		
NL(1)				50-60				120-140
SE (2)		12.5						
PT							100-130	
EE			50				80	
BE-F (1)					70-140		140	
NO			5-20				8-29	
CZ							150	
UK	**13-58**				**27-99**			

(1) Growing season mean (BE-F: TP as growing season mean; SRP as annual mean)
(2) Limits are based on an EQR, with “expected" TP calculated using an equation; upper threshold for H/G is 12.5 µg/L
(3) Based on UK data annual mean is approximately half of the 90th percentile value

5.3 Case studies

Figs. 10 and 11 show how these proposed standards compare to current values for two contrasting rivers. In both cases, the proposed site standards show a smooth transition along the length of the river, and, while they are more stringent than current standards, a benefit of the revised approach that is immediately obvious from Fig 10 is that errors associated with being close to a type boundary are much reduced, compared to the phase 1 standard (note the step change in current standards at km 110).

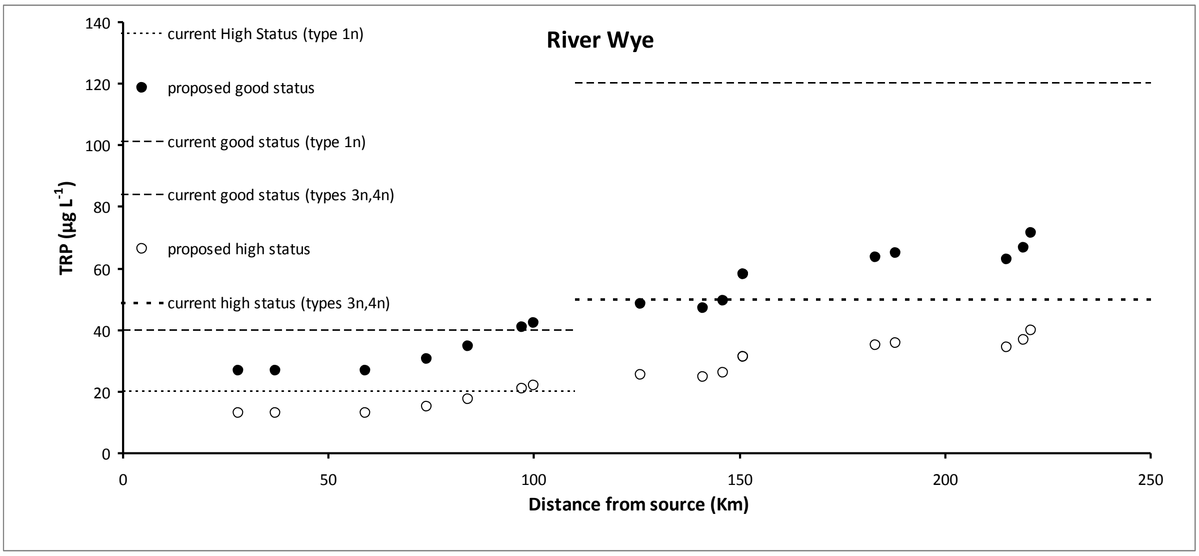

Fig. 10. Comparison between existing and proposed P standards for the River Wye. Values for proposed standards assume EQR at the class boundary (H/G for high status; G/M for good status). TRP = total reactive P (≈ RP)

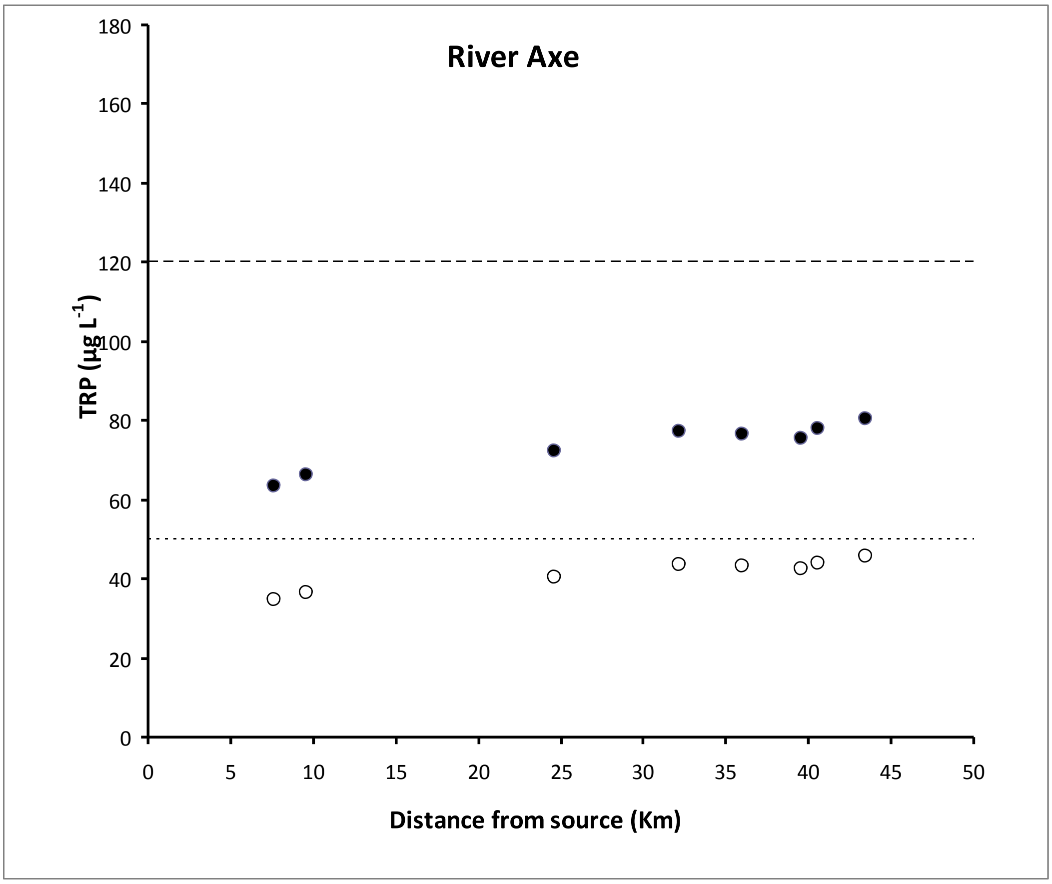

Fig. 11. Comparison between existing and proposed P standards for the River Axe. See Fig 10 for legend.

6. Conclusions / Recommendations

We recommend that site-specific standards, derived from the model relating WFD status assessed using the worst of either diatom or macrophyte EQRs (Kelly et al., 2011) to reactive P are used to replace the current phase 1 standards for phosphorus. Although these are more stringent than the current standards there will be fewer classification mismatches with biology than is the case at present and the overall proportion of sites failing on either biology or P is lower using the new standards than is the case using the current standards (Fig. 12). The values are also closer to concentrations at which detrimental effects of phosphorus are reported in the literature (Table2) and will provide more appropriate protection of river ecosystems. This evaluation assumes a simplistic dose-response relationship between phosphorus and ecology and the number of mismatches should fall further if other confounding variables (e.g. nitrogen) could be accounted for. However, many factors (environmental, biotic, and inertia) will intervene in the match between biology and nutrients and consequently this relationship will be inherently noisy. It would also be wrong to assume that the relationship between nutrients and biology is one directional. Equally we have used spatially discrete data to develop the models we have used and this cannot be assumed to represent changes that will take place at individual sites over time. Thus our models offer a pragmatic approach to deriving values of RP in a format that can be used for regulation and which will, on average, equate to a defined level of ecological status, assuming that are no other constraints. However, they should not be interpreted to mean that 'doing x will cause y to happen'.

Taken together, the analyses and literature cover both the "structure" and "function" of the photosynthetic biota (as per the WFD's definition of "ecological status"): the diatom and macrophyte tools provide a view of the structure that, through intercalibration, is consistent with views held by other member states, whilst the literature provides evidence for the disruption of function at concentrations similar to, or even lower than, the values for the good/moderate boundary presented here. Many of these alterations to function will increase the likelihood of undesirable disturbances (e.g. night-time deoxygenation), with concomitant impacts on ecological services.

Overall, the proportion of failures due to biology that are not supported by a P failure is reduced, whilst there is only a slight increase in the proportion of failures caused by P that are not supported by biology (Fig. 13). Site- and type-specific standards had a similar level of performance when applied to the training dataset; however, initial evaluation of national data sets confirms that the site standards produce the lowest level of misclassification as site standards should result in values that are better tuned to conditions within individual catchments, particularly where these span gradients of altitude and/or alkalinity and large differences between countries exist. Whilst a site-specific standard may prove initially harder to apply than a type standard, the principle of a standard as a continuous rather than a categorical variable is better aligned with ecological theory (e.g. Vannotte, 1980) and with biological metrics.

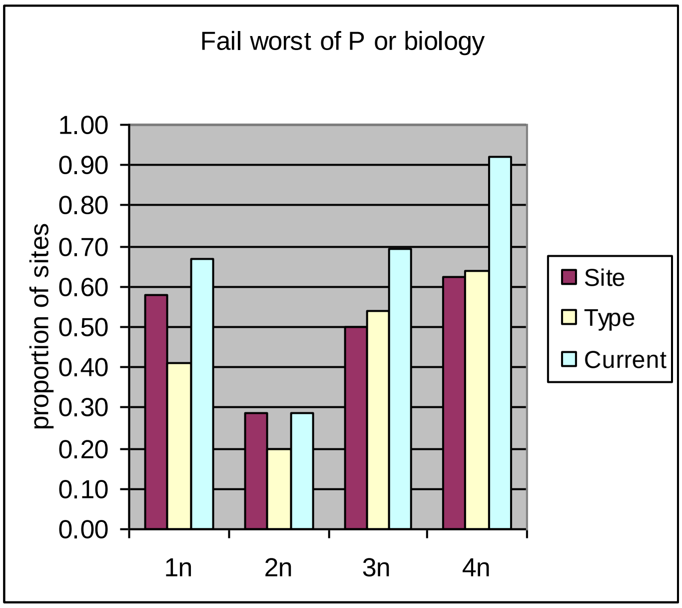

Fig. 12. Proportion of failures on either phosphorus or biology for UK sites included in the MADpacs database (Kelly et al., 2011)

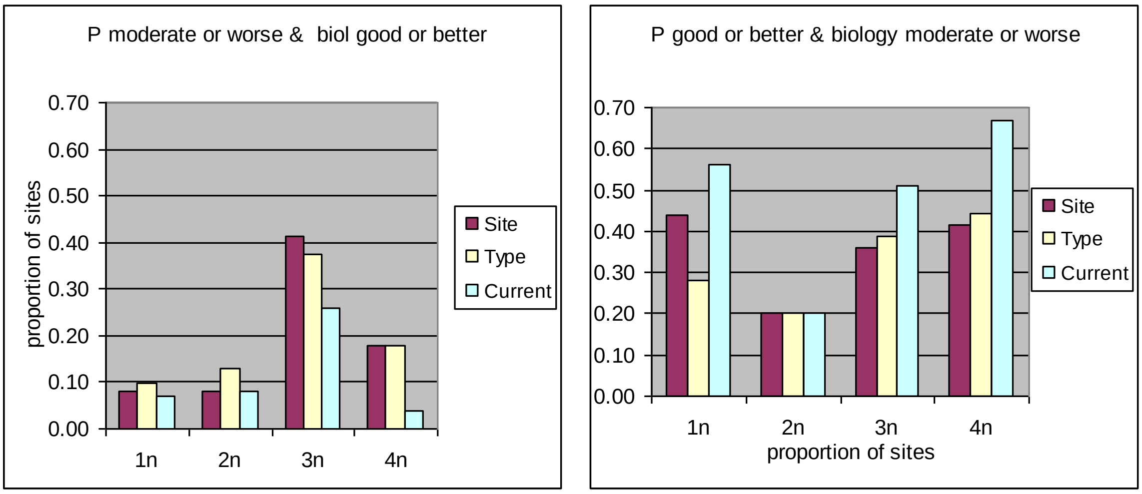

Fig. 13 Proportion of mis classifications where sites failing for a) phosphorus and not biology and b) biology and not phosphorus.

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Appendix 1: Model development

The dataset used for this work was developed for an earlier project by the same team and consists of biological and environmental data from 620 sites across the UK (256 in England and Wales, 221 in Northern Ireland and 143 in Scotland). Diatoms were evaluated using the DARLEQ methodology and mean values of 3-6 samples per site are used here. Macrophytes were evaluated using LEAFPACS at the same sites as the diatoms and values here are averages of one or two surveys per site (Kelly et al., 2011). EQRs were calculated using the latest intercalibrated methods for each. Predictive environmental data (alkalinity, slope, altitude etc) are available for all sites along with spatially matched chemistry for 575 sites. The metrics focus on taxonomic composition and, as such, will detect shifts from the reference assemblage and towards dominance by species adapted to enriched conditions. There is also one metric which evaluates the cover of "nuisance" algae (e.g. *Cladophora, Vaucheria*), which provides a proxy measure for the WFD's requirement for an assessment of abundance.

FTT008 recommends approaches for combining results of assessments for diatoms and macrophytes in order to produce an integrated assessment for the "macrophytes and phytobenthos" Biological Quality Element (BQE), as required for the WFD. Various approaches to combining diatoms and macrophytes were considered, with the minimum of each individual assessment being proposed, not only did this give the strongest relationship with the pressure gradient but it also prevents the response of any individual metric being "dampened" by other metrics). FTT008 also identifies situations where either macrophytes or phytobenthos alone may give a reliable assessment of the whole BQE.

The model was developed in two stages: first, the "expected" SRP concentration for any site was derived by stepwise regression of the observed SRP in a subset of 116 sites which either met intercalibration criteria for reference sites (Pardo et al., 2012) or which were only minimally impacted, using environmental predictors. Two environmental variables explained a significant portion of the variation: log alkalinity and site altitude (Table A1; Fig. A1).

Table A1. Model parameters for the prediction of reference SRP.

1. Parameter estimates

Model		Unstandardized Coefficients		Standardized Coefficients		
		B	Std. Error	Beta	t	Sig.
1	(Constant)	.341	.093		3.672	<.0001
	log_alk	.450	.060	.574	7.446	<.0001
2	(Constant)	.476	.088		5.382	<.0001
	log_alk	.454	.055	.579	8.287	<.0001
	site_altitude_m	-.002	.000	-.353	-5.048	<.0001

2. ANOVA[c]

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	5.013	1	5.013	55.444	<.0001[a]
	Residual	10.216	113	.090		
	Total	15.228	114			
2	Regression	6.906	2	3.453	46.472	<.0001[b]
	Residual	8.322	112	.074		
	Total	15.228	114			

a. Predictors: (Constant), log_alk

b. Predictors: (Constant), log_alk, site_altitude_m

c. Dependent Variable: log_SRP

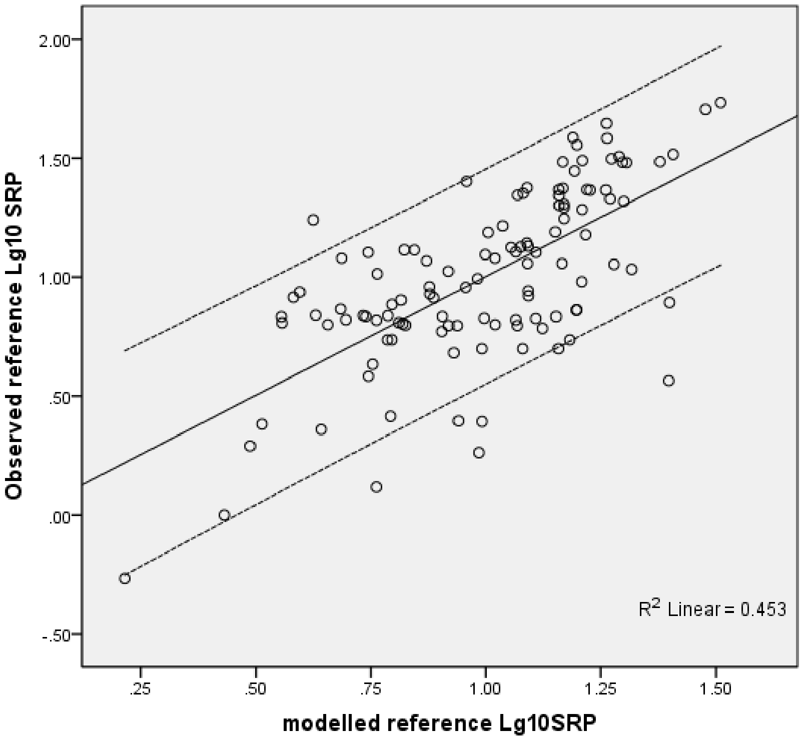

Fig. A1. Relationship between log annual mean SRP and a function derived from log alkalinity and site altitude for a subset of reference and minimally disturbed sites.

The resulting “expected” SRP concentration (eSRP) was then used to calculate an EQR for SRP as:

SRP-EQR = (log (Boundary SRP) – log(Upper Anchor))/(log(eSRP) – log(Upper Anchor))

where Upper Anchor was taken as 3500 P µg/l. Model parameters are given in Table A2.

This was then regressed against the biological EQR in order to predict the SRP concentration expected at different levels of ecological status (Fig. A2).

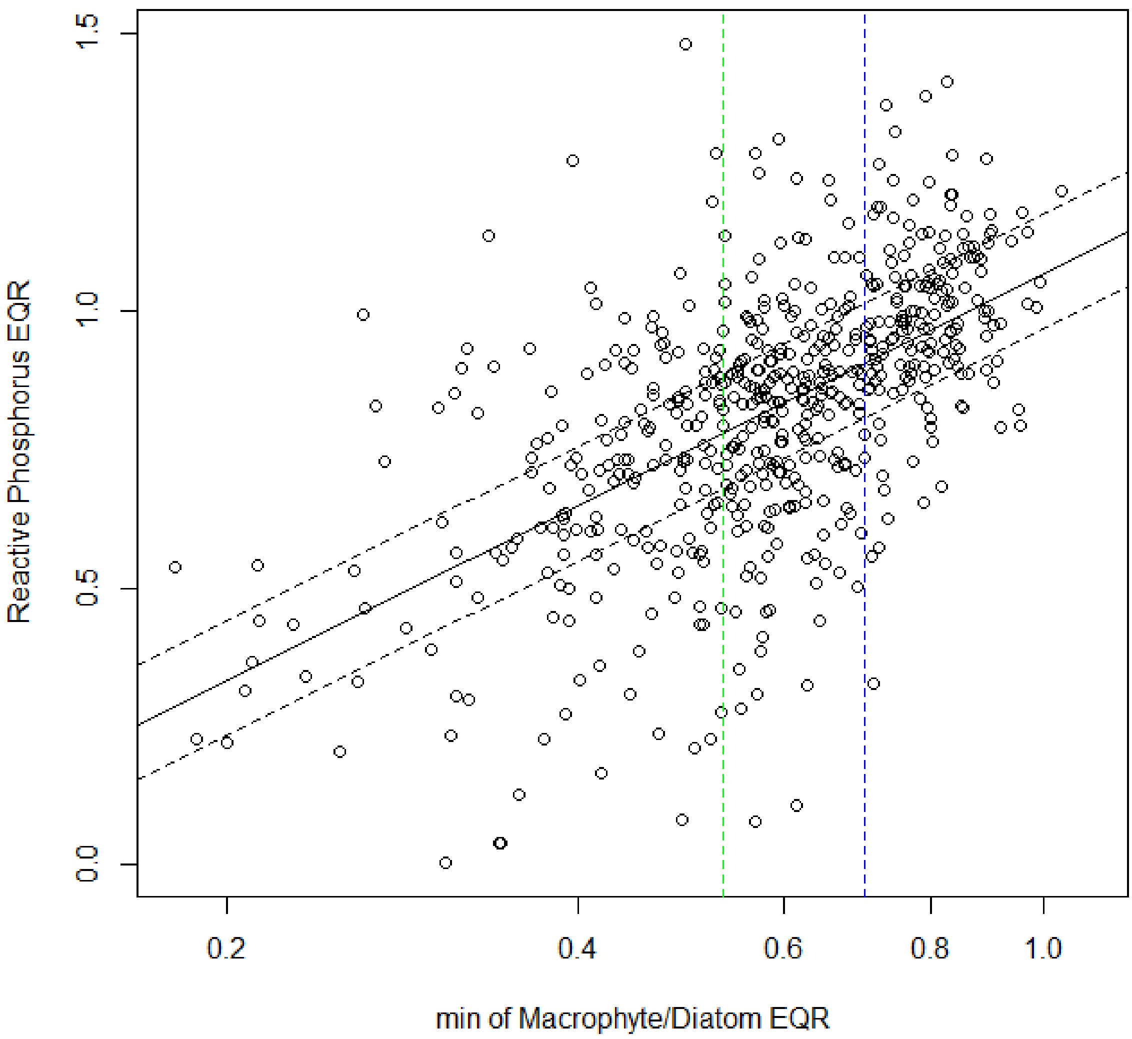

Fig. A2. Relationship between combined “macrophytes and phytobenthos” assessments and SRPEQR. Blue line represents reference HG boundary and green GM boundary, dotted lines show upper and lower 25th quantiles of residuals.

Table A2: Model parameters for the prediction of target P concentrations for a given EQR, using the reference P model.

Parameter	B	Std. Error	t	Sig.
Intercept	1.066	.0168	63.35	.000
logbiolEQR	1.04973	.0622	16.88	.000

2. Tests of Between-Subjects Effects

Dependent Variable:SRP_EQR

Source	Sum of Squares	df	Mean Square	F	Sig.
logbiolEQR	11.039	1	11.039	284.93	<.0001
Error	22.160	572	.0387		

a. R Squared = .333 (Adjusted R Squared = .331)

N=573

Biological EQR values were first adjusted to account for the use of the combined macrophyte and phytobenthos tool, based on the minimum of macrophyte and diatom models. The model based on both elements is slightly more precautionary than either element used alone; however, this is offset by the substantially better fit of the combined model compared to either used individually. In practice, using a single element may be justified in some situations, but only as a means of predicting status based on the combined model (Kelly et al., 2011). Class centres were then predicted from these adjusted boundary values (0.80, 0.625, 0.46, 0.28, 0.095 for the HG, GM, MP and PB boundaries respectively) and these were then matched to the mid-point where the error bars of the predicted values at mid class of adjacent classes overlapped (see fig 2), yielding final boundaries of 0.702, 0.532, 0.356 and 0.166 for HG, GM, MP and PB respectively).

Error bars were derived using the upper and lower 25th quantiles of the residuals of the regression between biological EQR and phosphorus EQR (eqn 2). Intersections were calculated using log transformed values for phosphorus concentration.

Values were +0.11 and -0.10

References:

See main reference list

Appendix 2: Comparison between annual means and growing season means

Growing season (April to September inclusive) and annual means were calculated for each of about 5800 sites monitored by EA. Data were available for 2007-2010 for most sites and samples were collected monthly or occasionally bimonthly. Where phosphorus concentrations were below detection limits (1 µg/L) a concentration of 1 µg/L was arbitrarily employed. In analysing the relationship between growing season and annual means data were treated as unique site x year combinations (n = 16750).

On the whole growing season mean SRP is closely related to annual mean SRP when both are >150 µg/L (Fig. A3); however, at lower concentrations, especially <50 µg/L, annual mean tends to be higher, presumably because the effect of biological uptake at low P exceeds the effect of reduced dilution on point source loading.

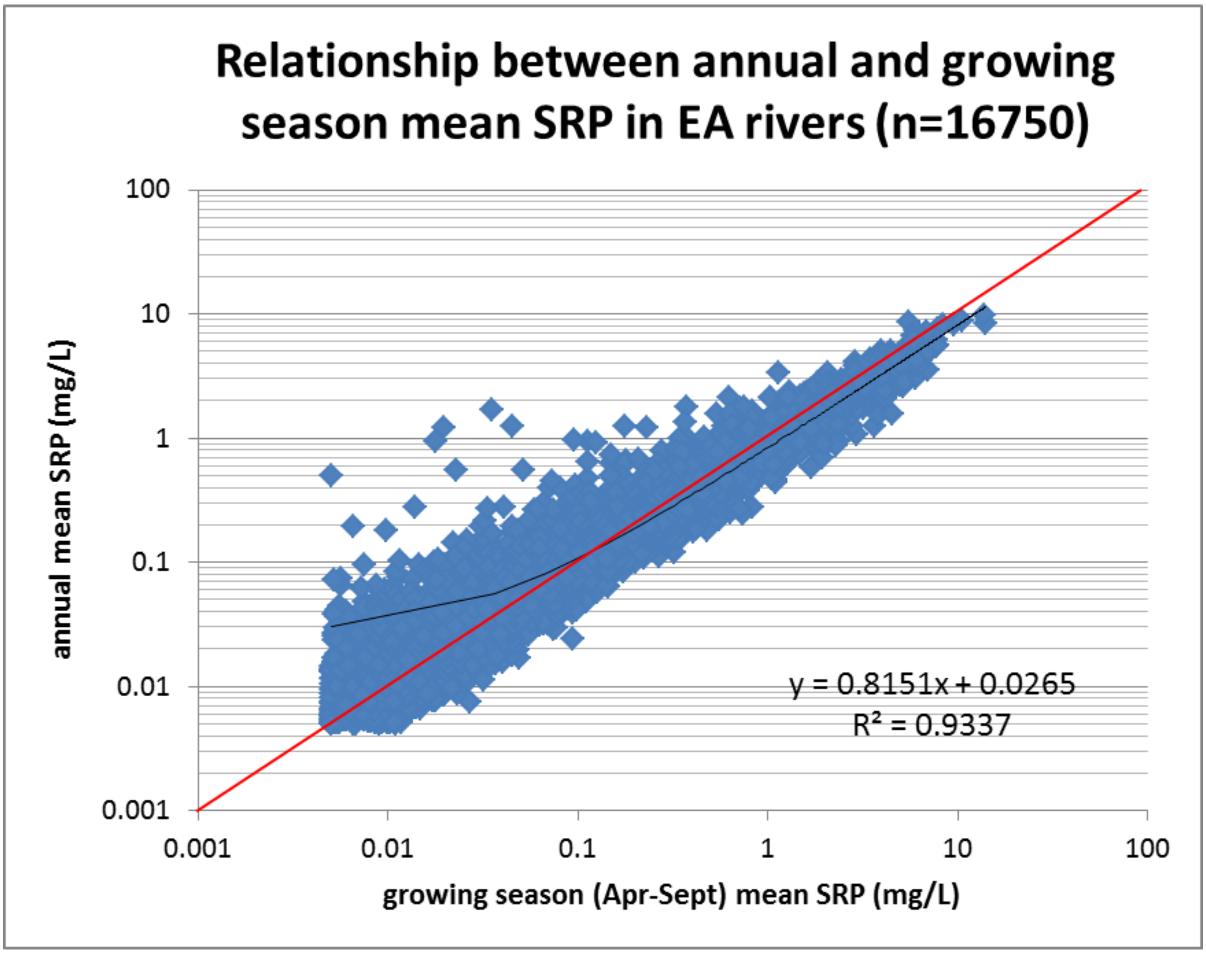

Fig. A3. Log-log relationship between growing season and annual means for 16750 river/years in England and Wales. Data from 2007-2010. Red line shows 1:1 ratio, black is fitted linear model.

Appendix 3: Tables showing P concentrations associated with Moderate/Poor and Poor/Bad status.

Predicted annual mean SRP concentrations associated with rivers at reference, high/good and good/moderate status are presented in chapter 3. Tables A2 and A3 show equivalent calculations for the moderate/poor and poor/bad status class boundaries.

Table A3. Threshold mean reactive P concentrations for rivers at moderate status, predicted from equations 1 and 2. Cell colours indicate the typology used for phase 1 standards: yellow: type 1 (low alkalinity, low altitude); orange type 2: low alkalinity, high altitude; blue: types 3 (high alkalinity low altitude) and green 4: high alkalinity, high altitude

	MP	Alkalinity (mg/l $CaCO_3$)									
	0.36	5	10	20	40	50	75	100	150	200	250
Altitude (m)	0	78	88	106	129	137	154	166	186	202	215
	20	78	83	101	122	130	145	157	176	191	203
	40	78	78	95	115	123	137	149	166	180	192
	60	78	78	90	109	116	130	141	157	170	181
	80	78	78	85	103	110	123	133	149	161	171
	100	78	78	80	97	104	116	126	141	152	162
	200	78	78	78	78	78	87	95	106	115	122
	300	78	78	78	78	78	78	78	80	87	92
	350	78	78	78	78	78	78	78	78	78	80

Table A4. Threshold mean reactive P concentrations for rivers at poor status, predicted from equations 1 and 2. See Table A3 for legend.

	PB	Alkalinity (mg/l $CaCO_3$)									
	0.169	5	10	20	40	50	75	100	150	200	250
Altitude (m)	0	548	580	638	700	722	763	793	838	871	898
	20	548	565	620	681	702	742	772	815	848	874
	40	548	549	604	663	683	722	751	793	825	850
	60	548	548	587	645	665	702	730	772	802	827
	80	548	548	571	628	647	683	710	751	780	804
	100	548	548	556	610	629	665	691	730	759	783
	200	548	548	548	548	548	579	602	636	662	682
	300	548	548	548	548	548	548	548	555	577	594
	350	548	548	548	548	548	548	548	548	548	555

Appendix 4: Comparison between outcomes of current and proposed P standards by type within each administration

Tables A5-A8 compare the outcome of classifications based on the current and proposed P standards for each administration. Overall, approximately 18.5% of sites will be reclassified into a lower class than at present, with the effects most pronounced in lowland hardwater areas. The new standards are generally more stringent than those presently in force, reflecting the cumulative effect of a number of known problems with the original standards:

– the use of a high percentile to set the standards coupled with a small dataset made original calculations vulnerable to outliers;

– the typology was coarse, with the threshold between low and high alkalinity set very low relative to the range of alkalinities encountered in UK rivers;

– The lack of high alkalinity upland sites in the original database led to this type being merged with high alkalinity lowland type;

– The analysis considered only diatoms whereas the full BQE is “macrophytes and phytobenthos”.

Part of the mismatch between current and proposed standards is a consequence of the new approach producing standards that are close to analytical detection limits. Fig A4 highlights this issue by plotting the predicted good/moderate boundary value against actual P values for sites in each administration. Points above the line are at moderate or worse status, points below the line are good or better. Issues may occur when mean P is < 10 µg/L, particularly in Scotland and Northern Ireland.

Table A5: comparison between complicance using current standards and proposed site and type standards for England

		Site Standards					Type Standards				
		Original Class					Original Class				
		B	P	M	G	H	B	P	M	G	H
New Class	B	220	88	0	0	0	220	89	0	0	0
	P	0	885	439	2	0	0	884	519	0	0
	M	0	0	443	413	7	0	0	362	472	4
	G	0	0	1	276	174	0	0	2	224	185
	H	0	0	0	8	811	0	0	0	3	803

Table A6 comparison between complicance using current standards and proposed site-specific standards for Wales

		Site Standards					Type Standards				
		Original Class					Original Class				
New Class		B	P	M	G	H	B	P	M	G	H
	B	1	5	0	0	0	1	4	0	0	0
	P	0	17	17	2	0	0	18	17	0	0
	M	0	0	40	79	6	0	0	39	80	3
	G	0	0	1	56	85	0	0	1	58	76
	H	0	0	0	1	527	0	0	1	0	539

Table A7: comparison between complicance using current standards and proposed site-specific standards for Scotland

		Site Standards					Type Standards				
		Original Class					Original Class				
New Class		B	P	M	G	H	B	P	M	G	H
	B	0	0	0	0	0	0	0	0	0	0
	P	0	12	28	1	0	0	12	25	0	0
	M	0	5	66	161	16	0	5	64	164	1
	G	0	0	3	143	244	0	0	8	143	206
	H	0	0	0	12	1329	0	0	0	10	1382

Table A8: comparison between complicance using current standards and proposed site-specific standards for Northern Ireland

		Site Standards					Type Standards				
		Original Class					Original Class				
New Class		B	P	M	G	H	B	P	M	G	H
	B	0	0	0	0	0	0	0	0	0	0
	P	0	6	10	0	0	0	6	8	0	0
	M	0	0	27	40	0	0	0	29	35	0
	G	0	0	0	34	56	0	0	0	40	36
	H	0	0	0	1	138	0	0	0	0	158

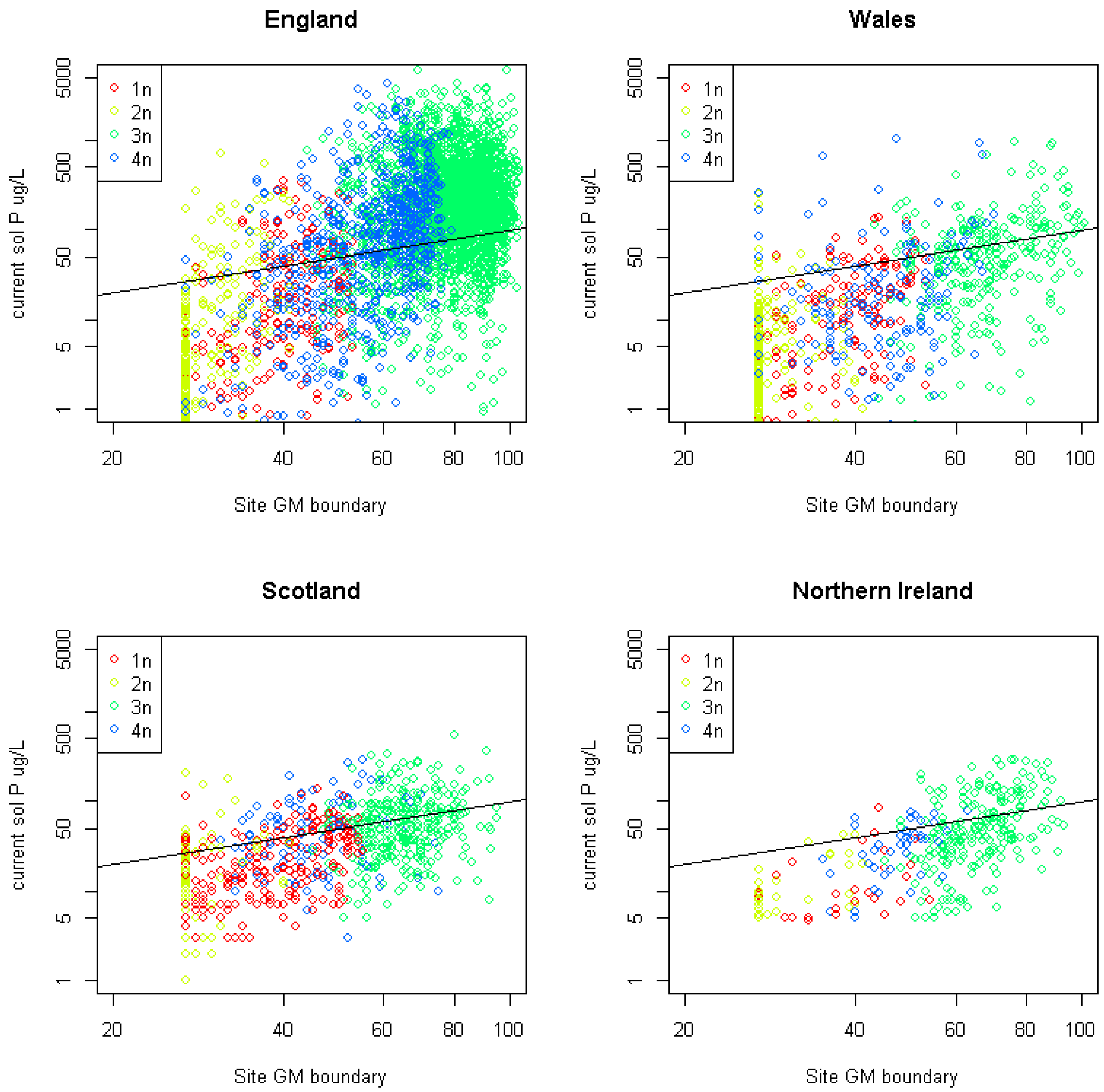

Fig A4. Relationship between proposed site specific good/moderate boundary values and observed reactive phosphorus, split by administration and river type. Diagonal line marks the position of the good/moderate boundary, sites below the line will be at good or better status and sites above the line will be moderate or worse.