Comments on “UK TAG (Oct 2012) A revised approach to setting Water Framework Directive phosphorus standards”

http://www.wfduk.org/stakeholders/stakeholder-review-phosphorus-and-biological-standards

Consultation period end 28th February 2013

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The UK TAG team are suggesting new site specific estimation of soluble reactive P standards from altitude and stream water alkalinity to assess the nutrient pressure on river ecosystems. They then suggest a new nutrient index reducing the mismatch between the phosphorus and revised biological WFD classifications[1].

The present document open to consultation is an extract from a wider analysis commissioned by the UKTAG into the management of nutrients in freshwater (Willby et al 2012). One would have expected the full report (Willby et al 2012) to be published before opening the consultation process on the extract. This said, several points of contention were noted based on our expertise in nutrient cycling and aquatic ecology in river basins. The comments below have strong implications for the “UK TAG (2012) Proposed recommendations on biological standards” simultaneously open for consultation.

(1) (Near) natural nutrient concentrations ?

Key publications have shown that background dissolved inorganic nutrient (N and P) concentrations in (near) natural ecosystems are generally extremely low (Meybeck 1982; Edwards et al 2000; Perakis & Hedin 2002; Demars & Harper 2002; Demars & Edwards 2007a).

Previous P standard concentrations were indeed too high compared to studies that monitored stream water quality above major and minor point sources (e.g. in the UK, Edwards et al 2000, Demars & Harper 2002, 2005a; Demars & Edwards 2007a). This left (near) natural sites unprotected (cf Slavik et al 2004; Benstead et al 2007; Davis et al 2010; Gudmunsdottir et al 2011) and biased the assessment of impacted sites towards a far too lenient state (Moss 2008). **The absence of N standards remains very problematic however.**

[1] UK TAG for WFD (2012). Proposed recommendations on biological standards. Consultation

The new suggested standards are more in line with the peer-reviewed literature (Mainstone 2010) on nutrient concentration studies (recalling that the different analytical methods used makes the comparison very difficult, especially under low concentrations such as below 10 µg P L^{-1} of reactive P). This is to be welcomed if we are to be able to protect nature adequately or/and better assess the value of rivers.

This said, the reactive P standards under low altitude and high alkalinity are still too high, particularly at the high/good and good/moderate boundaries (see Moss 1988; Demars & Harper 2002, 2005a).

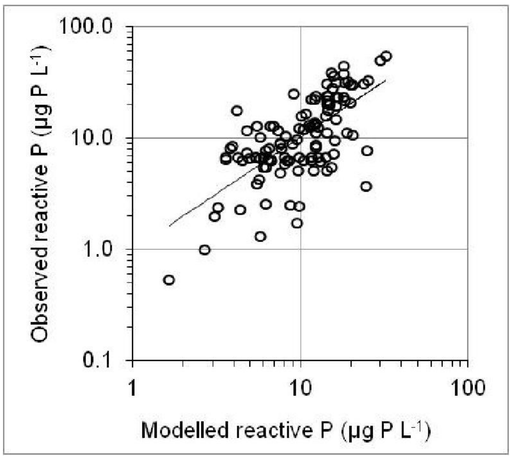
(2) Objective and logical modelling?

It is not clear how the 116 reference sites (114 sites in the statistical analyses) were chosen for the development of the P standards: e.g Pardo et al (2012) worryingly interpreted catchment areas with up to 50% intensive agriculture and excessive nutrient concentrations as (near) natural conditions. Such compromise (see also Kelly et al 2008) puts nature at risk, although slightly more stringent conditions were also used in previous studies (following van de Bund 2009). If reference conditions cannot be found within an area, other approaches need to be considered (e.g. Dodds & Oakes 2004; Baattrup-Pedersen et al 2008, 2013). This is especially true in areas where even groundwater is contaminated (Holman et al 2010). The geographical distribution of the reference sites should be plotted on a map (as in Kelly et al 2008).

The expected reactive P under reference conditions is now calculated as a function of alkalinity and altitude (Eq 1 in the report). In (near) natural systems, nutrient concentrations tend to be very low (<1-10 µg P L^{-1} of soluble reactive P), and this can be independent of alkalinity (e.g. Robach et al 1996; Edwards et al 2000; Perakis & Hedin 2002; Demars & Edwards 2007a). For example, sandstone (with siliceous cement) may weather as fast as calcareous rocks and release a similar flux of P for a very different alkalinity. How can we be sure that the model is not biased by the choice of reference sites? Lowland, alkaline rich stream are likely to be more impacted in the selection of reference sites than upland, base poor streams. A table with proportion of land cover and other chemical attributes for the reference sites would help give more context.

Large uncertainties remain in the background (expected) reactive P model (Eq 1 in the report), see Fig 1 below. And the model was not tested on independent sites, so it is not possible to evaluate its predictive strength. There were also very few sites above 20 µg P L^{-1} (only 6 sites), yet the suggested standards under low altitude, high alkalinity are mostly above 20 µg P L^{-1}. This suggests that very few sites were available to define the reference conditions under low altitude, high alkalinity and contrasts with lower background reactive P concentrations in previous publications (Moss 1988; Demars & Harper 2002, 2005a).

(a)



(b)

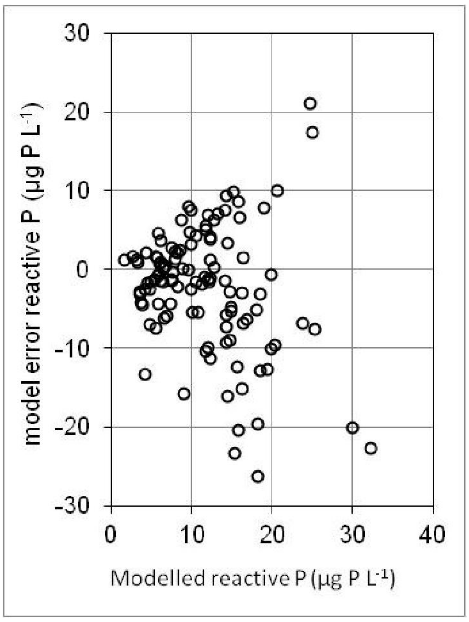


Fig 1. Large uncertainties remain in the reactive phosphorus model results (Equation 1) used to calculate the reference conditions for reactive P standard. **(a)** Fig A1 of the report is re-plotted with original P values (rather than LOG_{10} transformed values); **(b)** the residuals of the model calculated from the back-transformed values of model-observed at the 114 reference sites.

The UKTAG report recognises the difficulty of linking diatom/macrophyte indices to reactive P (p 8-9). The report suggests a new approach to minimise the chances of misclassification. It rather means minimising the discrepancies between reactive P standards and biological EQR. The boundary reactive P is calculated (Eq. 2 in the report) as a function of both the biological EQR and the expected P (under reference conditions). Then the reactive P EQR (equation in annex 1) is calculated using the boundary reactive P (Eq. 2 in the report) and expected reactive P. **This means that for a given altitude and alkalinity the reactive P EQR depends solely on the biological EQR. Hence it is not surprising that reactive P EQR correlates** (r=0.58, Table A2) **with** log transformed **biological EQR** (Fig. A2 in the report) **because the two are not independent.**

This approach is therefore flawed and misguiding, particularly because very few reliable correlations have been found between independent diatom/macrophyte composition/indices and reactive P (see below).

(3) No reason to leave out dissolved inorganic N

Dissolved inorganic nitrogen (ammonium and nitrate) impact is equally important to phosphorus in aquatic ecosystems (Elser et al 2007; Gudmundsdottir et al 2011). For now N is only used as a weak (because of the very high N concentration chosen) criterion to define reference conditions (Pardo et al 2012). Why is N continuously ignored by the UK TAG team? This is despite a clear rational for including it (e.g. Moss 2008) and mounting evidence of potential effects on aquatic plant growth rate (Robe & Griffiths, 1994; Boedeltje et al, 2005) and macrophyte occurrence/biomass and diversity in rivers and receiving water bodies (Sosiak 2002, Carr et al 2003; James et al., 2005; Ruhl & Rybicki 2010; Lambert & Davy, 2011). Even the Mean Trophic Rank (MTR) and its current version renamed River Macrophyte Nutrient Index (RMNI) were both previously found to correlate as strongly with nitrate (or dissolved inorganic N) than reactive P in Great Britain (Demars & Harper 1998, Dawson et al 1999, Demars & Edwards 2009, Willby et al 2009; but see below). The revised Trophic Diatom Scores were also validated against a SRP and nitrate nutrient pressure (Kelly et al 2008, but see below).

(4) Evidence based?

The information **extracted** from Mainstone 2010 is rather biased towards positive findings including several unpublished reports and abstracts from meetings not easily accessible. It fails to report that the over simplistic rational adopted by the river UK TAG team is weak. That is, there are **no reliable direct links** between diatom / plant composition or indices and nutrient concentrations (Demars & Harper 2005b, Paal et al 2007, Demars & Edwards 2007b, 2009; Demars & Thiébaut 2008; Demars & Trémolières 2009, Demars et al 2012, Demars 2013).

Not only the full report (Willby et al 2012) was not yet published but other key reports were also unavailable on the web sites (UK TAG, Environment Agency) or British Library: Kelly et al (2011); Barahona (2009). In future consultations, all documents cited should be accessible. Citing abstracts from meetings and other unpublished reports as often done also makes it difficult to understand or defend their conclusions.

The effects of dissolved N and P concentrations have been detected in several studies but these effects were invariably found to be either weak or confounded with other variables (e.g. Demars & Harper 1998; Carr et al 2003; Stevenson et al 2008; Demars et al 2012). In other studies with a wide range of nutrient concentrations, no direct effect was found (e.g. Demars & Harper 2005b; Demars et al 2012). This is developed further below as comments to the simultaneous open consultation on biological standards.

Complex interrelationships between the actors (microbe, plant, invertebrate, fish) and the environment (hydrology, pCO_2) within an ecosystem need to be taken into account (e.g.

Jones & Sayer 2003). Generally, more compelling evidence of nutrient enrichment impact on stream ecology is provided by experimental work, notably in whole ecosystem experiments (e.g. Slavik et al 2004; Benstead et al 2007; Davis et al 2010; Gudmunsdottir et al 2011).

Better evidence of eutrophication may be found in river backwaters, but these are not surveyed by the regulatory agencies with their narrow definition of rivers limited to the main channel, or in downstream ecosystems (lakes, estuaries and coastal ecosystems), subject to different nutrient standards (total P and dissolved inorganic nitrogen for lakes and estuaries respectively, Moss 2008). This again is questioning the reliance on soluble reactive P only. Receiving water bodies are neither protected from either particles, colloidal P sources nor from nitrogen (House & Edwards 1998; Hilger et al 1999, Moss 2008).

The worst eutrophication problems in UK rivers (and elsewhere) are not solely the result of nutrient enrichment, but the product of multiple stresses such as past river engineering work impeding the flow, fish overstocking destroying the macroinvertebrate grazers, lack of riparian shading and water abstraction (drought). Hence risks of eutrophication should be assessed in a broader context of river health and management.

(5) Revised biological standards

The nutrient pressure (reactive P) is now suggested to correlate best with the lowest score of the diatom and macrophyte indices. It is worth noting that neither of these indices has ever been clearly and strongly linked to reactive P.

The diatom index was related to reactive P and nitrate (Kelly et al 2008). However, confounding effects were not tested, despite a large part of the variability in the diatom index being related to variables other than reactive P and nitrate. The latter are often correlated in large national or continental scale studies to alkalinity or conductivity (e.g. Stevenson 2008). Uncertainties were also not reported adequately in key papers, as discussed by Demars (2013).

The overall macrophyte index (LEAFPACS) was said to mostly reflect the ecological status of the river (ecosystem structure and functioning) in the original report, and so there was no claim that the final metric would respond to nutrient (Willby et al 2009). Yet it was clear that the authors had essentially devised a pressure response tool. Hence, the individual metrics were tested on an independent dataset, but found not to respond to the pressures (notably reactive P and sediment bio-available P; Demars et al 2012).

The macrophyte index has now been simplified: most notably the hydro-morphological index was dropped, species diversity has been downweighted, calculations were corrected and simplified. But now, the multi-metric LEAFPACS (version 2) is presented as an indicator of nutrient pressure. This is a major change of emphasis. Corrections of LEAFPACS v2 is highly correlated to v1 (Fig 2, using data from Demars & Harper 2002, 2005b), so all previous findings and conclusions regarding nutrients in Demars et al 2012 are unchanged. LEAFPACS v2 was found neither correlated to water or sediment reactive P (Fig 3, using data from Demars & Harper 2002, 2005b).

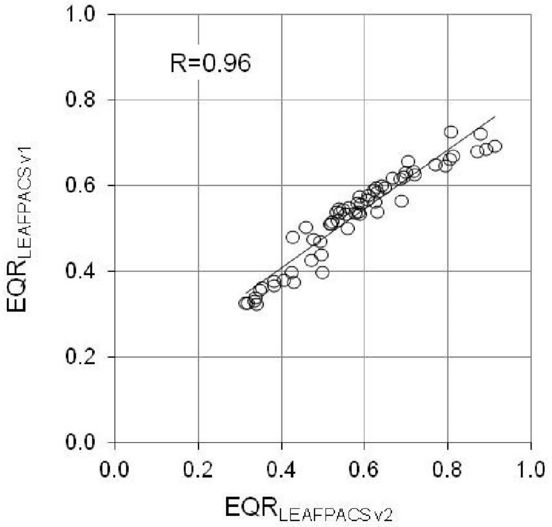


Fig 2. The first and second version of LEAFPACS are highly correlated. Data from Demars & Harper 2002, 2005b.

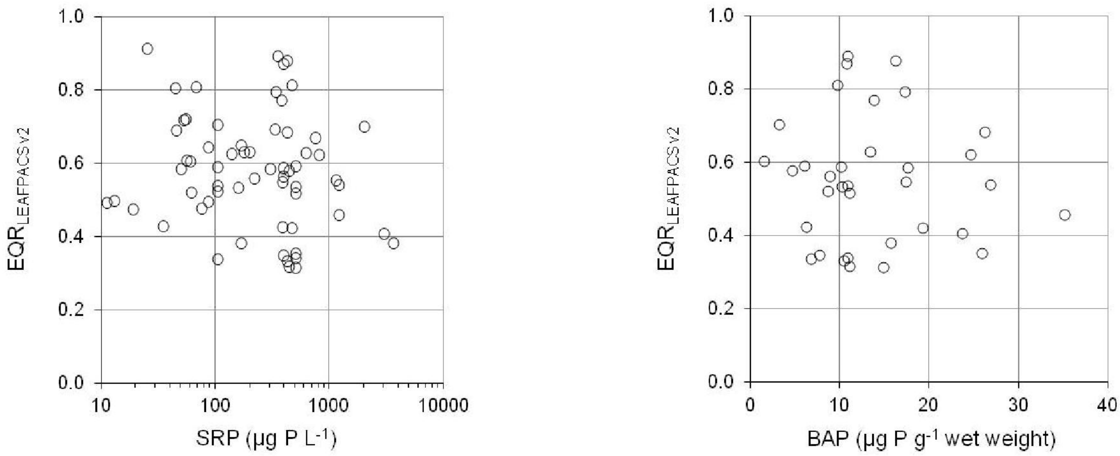


Fig 3. LEAFPACS v2 is unrelated to reactive P, both from the water (SRP) and the sediment (BAP=bioavailable sediment P). Data from Demars & Harper 2002, 2005b.

(6) What to do then?

Unless better tools are provided to the UK TAG, little is likely to change. What the UKTAG team need to realise however is that this is not by 'improving' the current tools that we will succeed. Radical changes are necessary, not just in Britain, but in Europe. Some have been more inventive with macrophytes (e.g. Baattrup-Pedersen et al., 2008, 2013), but there is a much wider range of approaches in the literature (nutrient spiralling, nutrient ratios, eco-enzymatic stoichiometry, whole stream metabolism) which could be used to provide strong evidence linking nutrient enrichment (and other pressures) to the rich diversity of organisms and functions of our rivers (e.g. Mainstone 2010). Do we really need then to both monitor nutrient and biota at several thousands of sites (e.g. Murphy & Davy-Bowker 2005, Willby et al 2009)? This does not sound cost efficient (perhaps a few hundred monitoring sites for the UK), particularly if reference conditions are still poorly appraised. Both a new monitoring strategy and a mechanistic system such as the CADDIS initiative by US EPA must be developed so that restoration (programme of measure) is considered in a more holistic and integrated way than at present (e.g. Zalewski et al 2008, Palmer & Febria 2012) and reconcile, where possible, regulatory obligations with conservation aims (Irvine 2009).

Conclusion:

The present reactive P standards are now more stringent and more in line with the scientific peer-reviewed literature, although the reactive P standards under low altitude and high alkalinity are still too high (particularly at the high/good and good/moderate boundaries).

The underlying rational and modelling linking nutrients and ecological response is flawed and should not be used. Consequently, the biological standards for river nutrient pressure, as presented in the "UK TAG (2012) Proposed recommendations on biological standards", are inadequate and should not be used.

The UK TAG team need to realise that a paradigm shift in thinking is needed to assess nutrient (and other) pressures, as some have realised (e.g. Mainstone 2010, Kelly 2011).

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