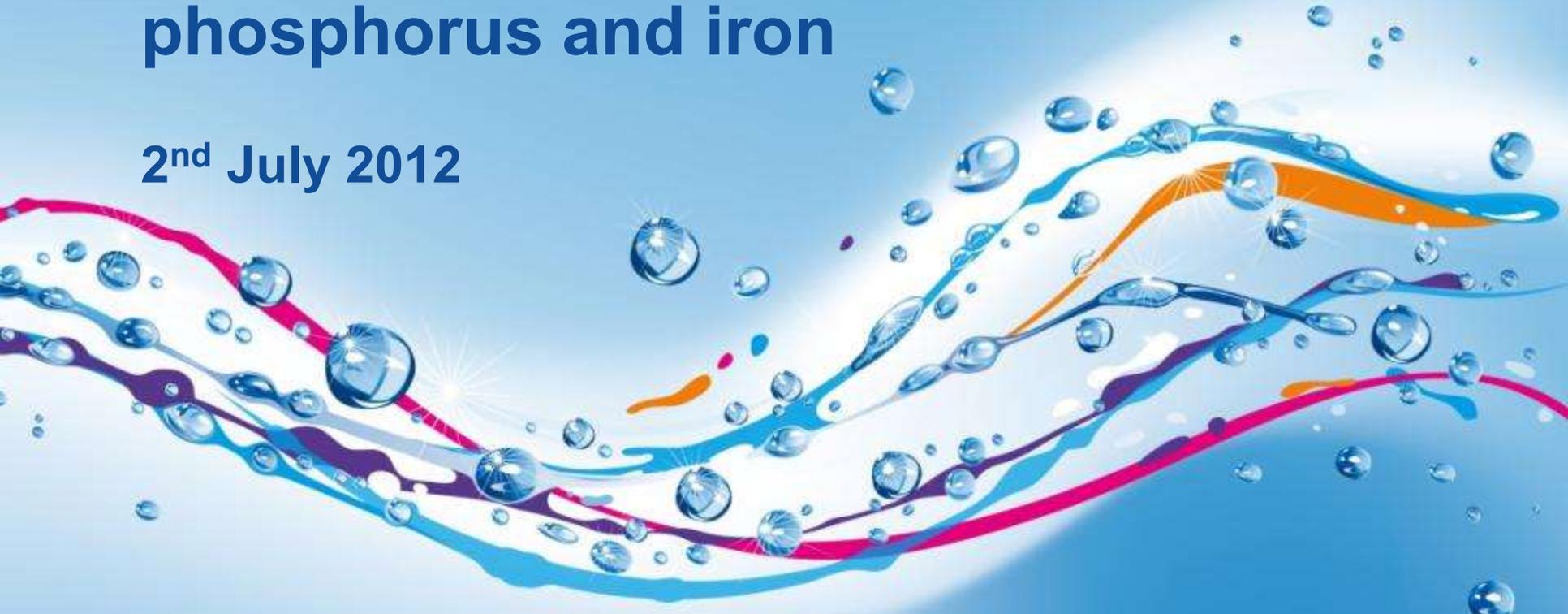


Preparing for the 2nd cycle of the WFD - the twin challenge of phosphorus and iron

2nd July 2012



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Background (1)

- Phosphorus removal from sewage now commonplace in the UK
 - by 2015 >650 WwTW will have P removal technology – serving ca. 24 million people
- To date most P consents have been driven by the UWWTD
 - for works b/w 10,000 pop and 100,000 pop = 2 mg/l total P
 - for works > 100,000 pop = 1 mg/l total P
- Under the WFD water bodies have to achieve at least 'good status' by 2015..2021...2027....
 - Phosphorus has been identified as the biggest single factor for non-compliance
 - UKTAG estimated that 65% of rivers in England will fail good status
 - Hence, there is a strong likelihood that for the 2nd cycle of the WFD we will see significantly tighter P consents

Background (2)

- The vast majority of UK WwTW employ chemical precipitation (with iron, occasionally aluminium) to remove P
- In AMP 5:
 - 360,000 to 572,000 tonnes Fe products used per annum,
 - ca. 12,000 tonnes/yr Fe discharged to aquatic environment.
 - Costing ca. £50 million per annum on Fe-based chemicals alone
- Fe has been classed as a specific pollutant by the UK, dosing Fe automatically leads to an iron consent
 - Proposals to tighten the EQS for Fe from 1 mg/l dissolved to 0.73 mg/l total have recently been published by UKTAG
- Tighter P standards will require more Fe to be dosed, so the water industry faces the twin challenge of needing to meet tighter P standards while at the same time needing to meet tighter iron standards

UK TAG is a partnership of the UK environment and conservation agencies which was set up by the UK Administrations. It is currently chaired by the Environment Agency and supported by a UKTAG Coordinator employed by Sniffer. It was created to provide coordinated advice on the the science and technical aspects of the European Union's Water Framework Directive (2000/60/EC). This Directive sets an enormous challenge in meeting the objectives of the improvement and the protection of the water environment and is the major driver for the sustainable management of water in the UK. The water environment includes all rivers, canals, lakes, estuaries, wetlands and coastal waters as well as water under the ground.

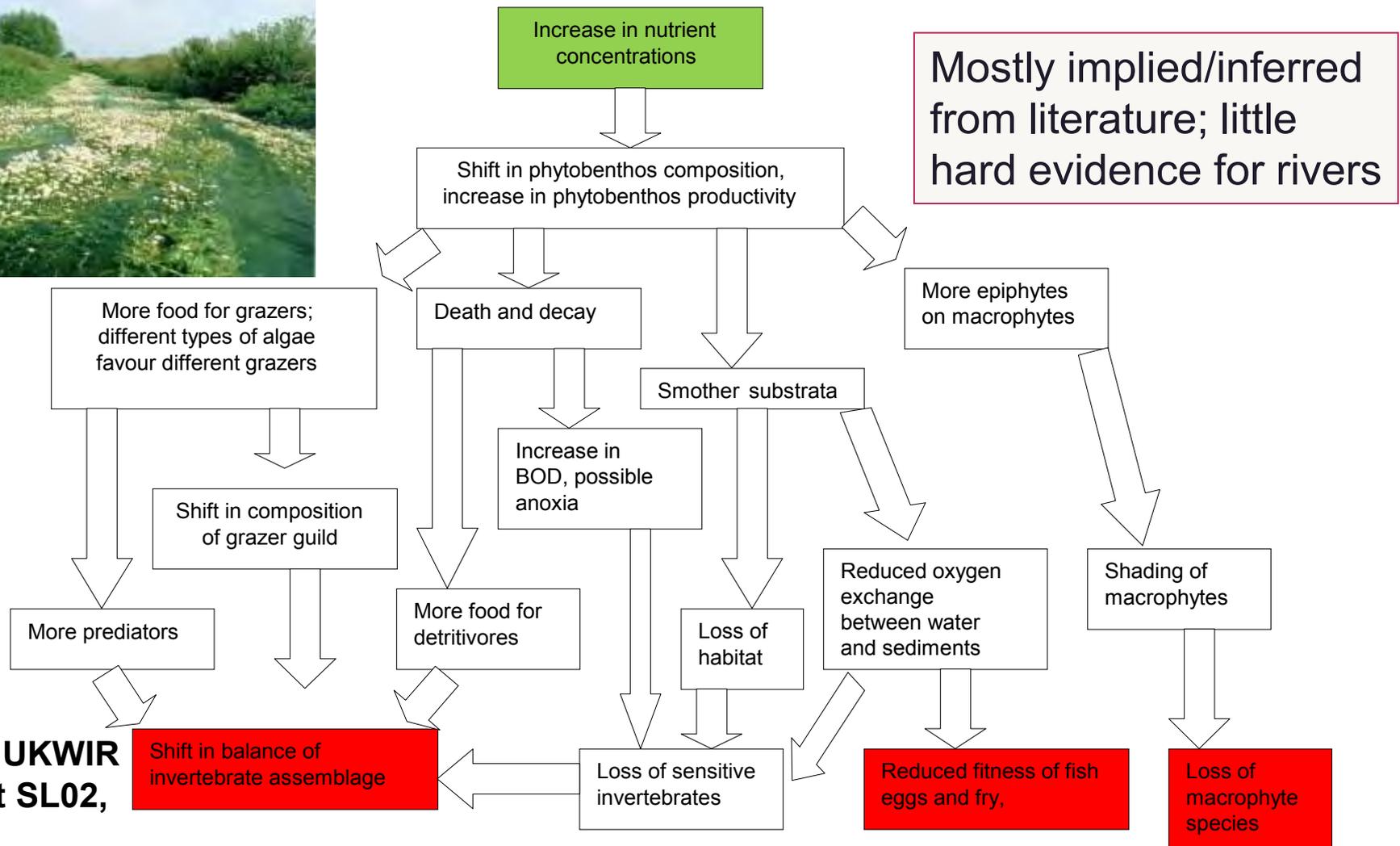
Only the UK and two other Member States have iron Consents, none of the other 24 or the USA has seen fit to classify Fe as a specific pollutant . Fe^{2+} (ferrous) iron is toxic but Fe^{3+} (ferric) is not; Fe^{2+} converts rapidly to Fe^{3+} in aerated water. If the oxygen level were so low in a water body that Fe^{2+} persisted, it would be dead because of lack of oxygen irrespective of the Fe content. Oxides and hydrous oxides of iron are abundant in soil; they are a cause of red and brown colours in soil. Maybe classifying Fe as a specific pollutant should be reappraised.

0.73 mgFe/l total is equivalent to 0.2 mgFe/l dissolved

Phosphate chemistry is very complex (slide 7) and the different forms are in dynamic equilibrium. As slide 7 shows, the various fractions are defined empirically by the laboratory procedure by which they are measured: there are then attempts to relate fractions to environmental effects – not easy when in reality ‘Bioavailable’ equilibrates with ‘Potentially (indirectly) bioavailable’, which equilibrates with ‘Not bioavailable’.

Only a small fraction of total-P in WwTW effluent is ‘soluble reactive P’ (SRP).

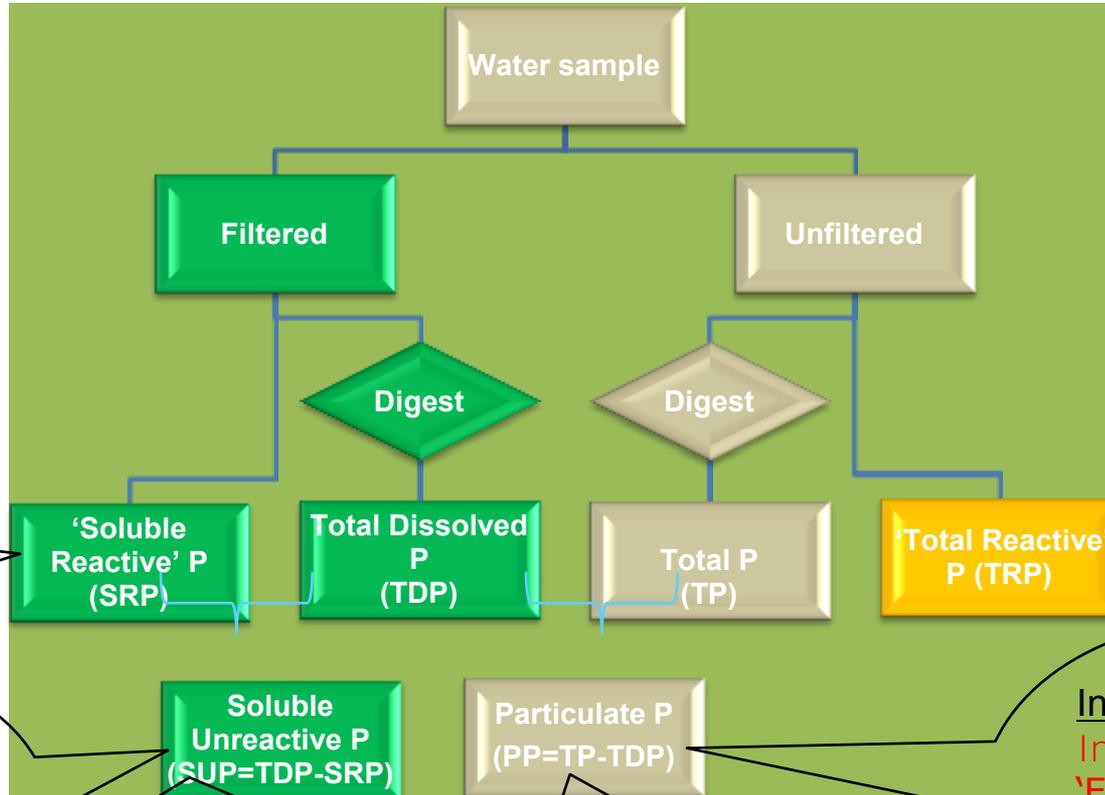
Phosphorus –Impact on Aquatic Ecology



Phosphorus chemistry in surface waters is complicated!

Which fraction should be used as the basis of WwTW consents?

From UKWIR report
WW20, 2012



Bioavailable

Potentially (indirectly) bioavailable

Not bioavailable

$H_2PO_4^-$
 HPO_4^{2-}
 PO_4^{3-}

Inorganic
Poly-P
Meta-P
Pyro-P
Colloid-P

Organic
Nucleic acid-P
Sugar-P
Phospholipid-P
Inositol-P
Humic-P

Organic (POP)
Planktonic, plant & animal biomass ?
- dead or alive

Inorganic (PIP)
Interstitial-P
'Extractable'-P
Fe-bound-P
Al-, Ca-bound-P ?
1°, 2° minerals ?

Regulating P

- In Cycle 1 of the WFD, the UK derived target for good status was designed to ensure diatom biodiversity over and above protecting for eutrophication:
 - P is only one factor that can impact diatom numbers and/or diversity, and therefore focusing only on P reduction may not bring about ecological benefit.
 - In Cycle 2 good status will be referenced on “MADpacs”, which will include macrophytes as well as diatoms, however:
- A revised phosphorus EQS is due to be published later this year (excluded from the recent UKTAG consultation document). There is a strong likelihood of a significant tightening from current EQS of 0.12 mg/l SRP.

Queries on Permitting P

- Is a change to soluble reactive phosphate permits (from total P), scientifically justifiable?
 - would match 'in river' WFD standards
 - may prevent further tightening of phosphorus consents, but
 - conflicting views on what fractions are bioavailable and therefore potentially damaging.
- Could seasonal P permitting for WwTW discharges lead to an improvement in ecological quality in a cost-neutral way?
 - Is eutrophication only an issue in spring/summer?
 - Can particulate P deposit in the winter, be stored in the sediment and be re-solubilised in the summer?

Iron –Impact on Aquatic Ecology?

- Iron has been included as a WFD Annex VIII Specific Pollutant, however the UK is out of step with most of the rest of Europe (and North America) in considering iron as a specific pollutant
- A revised draft EQS has recently been published by UKTAG, with the limit significantly tightened from 1 mg/l dissolved to 0.73 mg/l total (this equates to roughly 0.2 mg/l dissolved).
 - The consultation document states: *“In the case of iron, conventional laboratory data on toxicity are difficult to interpret. The UKTAG has applied an approach based on the use of field data to identify the concentration of total iron at which invertebrates appear consistent with good (or better) status. This threshold has been used as the basis for the proposed standard.”*

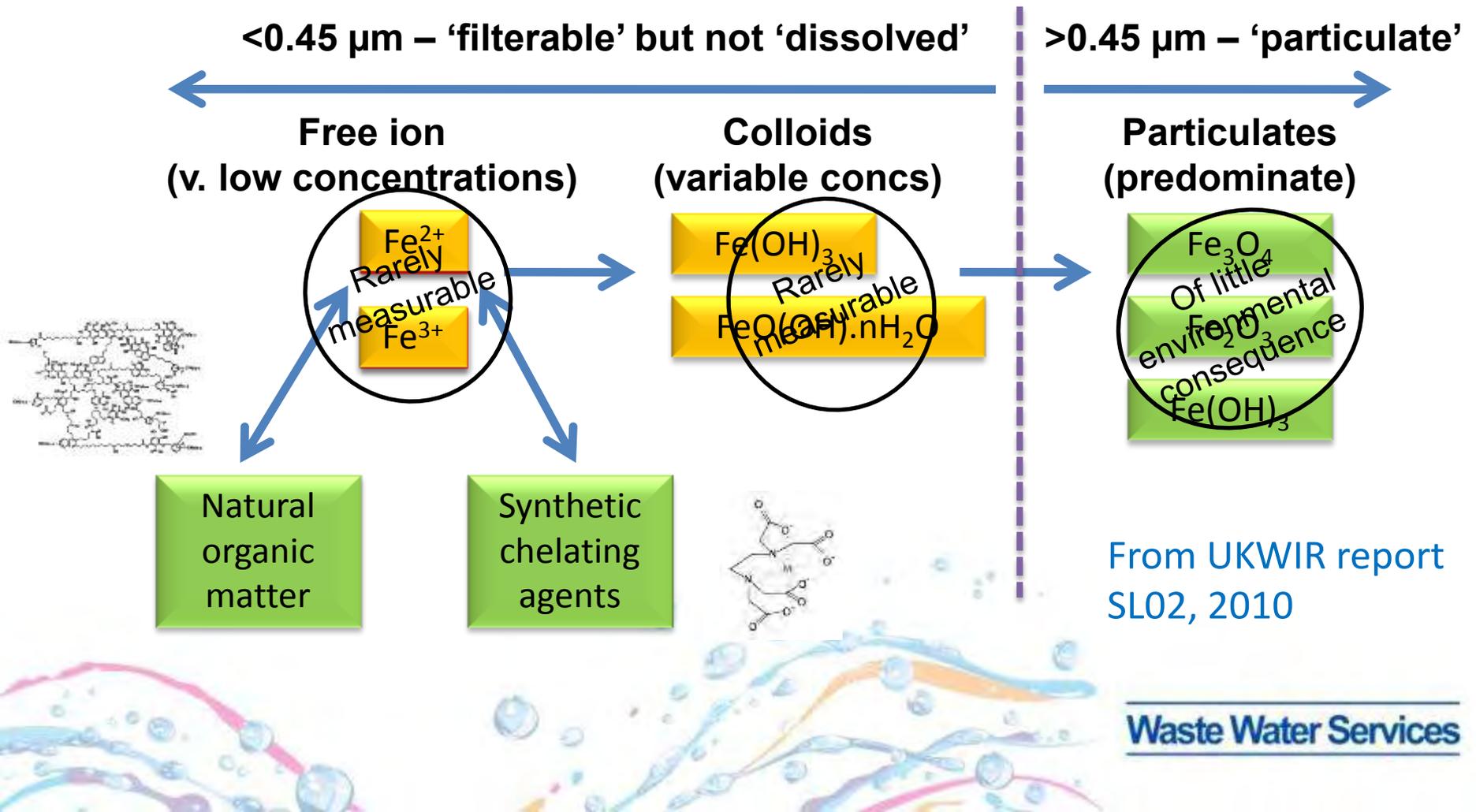
Iron – Impact on Aquatic Ecology?

- There is no evidence of significant increases in total or dissolved iron concentrations downstream of wastewater treatment works because of dosing iron for P reduction (UKWIR WW20, 2010).
- There are no significant ecological changes downstream of WwTW dosing iron (UKWIR WW20, 2010).



Iron chemistry in solution

- Iron is unstable in oxygenated water and will oxidise, and eventually precipitate.



Permitting Iron

- The current dissolved iron EQS (1 mg/l) can readily be met downstream of works dosing iron, however, the proposed tightening is likely to lead to compliance becoming increasingly difficult to achieve without additional effluent treatment.
- The lack of evidence of downstream impacts, combined with low concentrations of residual total and dissolved iron in effluents calls into question the need for a general policy to permit WwTW discharges.

STW Initiatives

Bio-P Strategy

- To put in EBPR as the preferred option; where it is technically and economically viable:
 - Now have over 10 EBPR plants,
 - By end of AMP5 ~ 4 million PE served by EBPR plants in STW region
- R&D has successfully delivered:
 - low cost EBPR plant upgrades for smaller ASP's , and
 - ways of enhancing EBPR for weaker (domestic) wastewaters
- As well as making iron compliance easier, EBPR also makes P recovery viable



RAS Fermenter at
Loughborough STW

Bio-P is less effective at high flow and at low temperature. Dynamic Consenting might be more cost-effective than static consenting.

Fe-P is invariably needed as a back-up to bio-P. The wastewater in most catchments is too weak to feed bio-P and supplemental acetic acid has to be purchased to feed bio-P. Reducing internal recycle (by dewatering liquors, which can be 25% of the P load) would improve the suitability for bio-P. STW is experimenting with fermenting the RAS to produce VFA to feed bio-P. Processes like WASSTRIP that render WAS anaerobic by mixing with primary sludge before digestion release sorbed bio-P which can be diverted using a thickener to struvite recovery.

Slide 18 the River Mease has an EQS of 0.06 mgP/l; it also has a lot of septic tanks, which can contribute more P to rivers than had been thought previously.

Most of the major detergent manufacturers have replaced P in laundry detergents in the UK already and there will be a ban in 2015. P is still used in dishwasher detergents, apparently it is more difficult to find a replacement that does not damage the items being washed. However Sweden banned P in laundry detergents several years ago and has had a voluntary ban on P in dishwasher detergents that had been so successful that there will be a ban will be introduced in 2012.

There is also the thorny issue of P dosing into drinking water to suppress plumbosolvency. This amounts to about 15,000 tP₂O₅/year, which contributes to that which has to be taken out at the WwTW.

Small to medium sized fertiliser manufacturers would welcome assured supplies of struvite that they could incorporate into their fertiliser products and give them security of P supply. For this use the physical form of the struvite is of little consequence because the struvite would be ground as part of the manufacturing process to ensure uniform blends and consistent products when co-granulated with the other components.

Collaborative work with the EA

- **Balancing Carbon and Ecology**
 - A joint initiative between STW and the Midlands EA to work towards achieving a better balance between carbon emissions, river water quality and cost to consumers.
 - Including practical field trials
 - Aiming to influence the debate (e.g. on proposed new WFD standards) from an informed position
 - Four 'briefing papers' have been published on P, Fe, Al and NH₃
- **Sustainable P Removal Task and Finish Group**
 - Instigated and led by the EA
 - looking at more sustainable ways of removing and recovering P



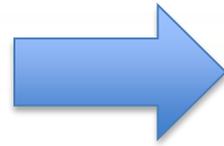
Low P Demonstration Trials (1)

- Trial Aims and Objectives
 - To install and operate, in AMP 6, five or six parallel tertiary treatment streams capable of polishing the effluent from Packington STW to meet a < 0.3 mg/l total P standard
 - To build this demonstration plant at a scale that will allow the results to be extrapolated to full scale with low risk, and to deliver a significant reduction in phosphorus load to the River Mease
 - To include in the technology assessment those technologies suited to smaller works as well as larger works, and
 - To evaluate performance in terms of whole life cost, sustainability, robustness and degree of phosphorus removal
 - To include processes amenable to P recovery

Low P Demonstration Trials (2)

- The 6 technologies that have been taken forward for more detailed design and costing are:
 - Adsorption / Ion Exchange (the design will be based on a completed Cranfield PhD project and supported by an on-going Cranfield EngD project)
 - Reed bed employing phosphorus adsorbing media (a current Wessex/Severn Trent R&D project is evaluating which media offers the most potential)
 - Iron dosed sand filters (e.g. Blue-Pro or Dynasand D2, this option will also include evaluating sand filters in series with no iron dosing)
 - Membrane filtration (micro-filtration or ultra-filtration of the effluent, possibly with iron dosing)
 - Magnetically ballasted sedimentation (e.g. Co-Mag)
 - Microalgae

P Recovery (as Struvite)



Naturally occurring

Exists in most wastewater plants
(forms easily!)

Increases O & M costs

Impacts plant reliability



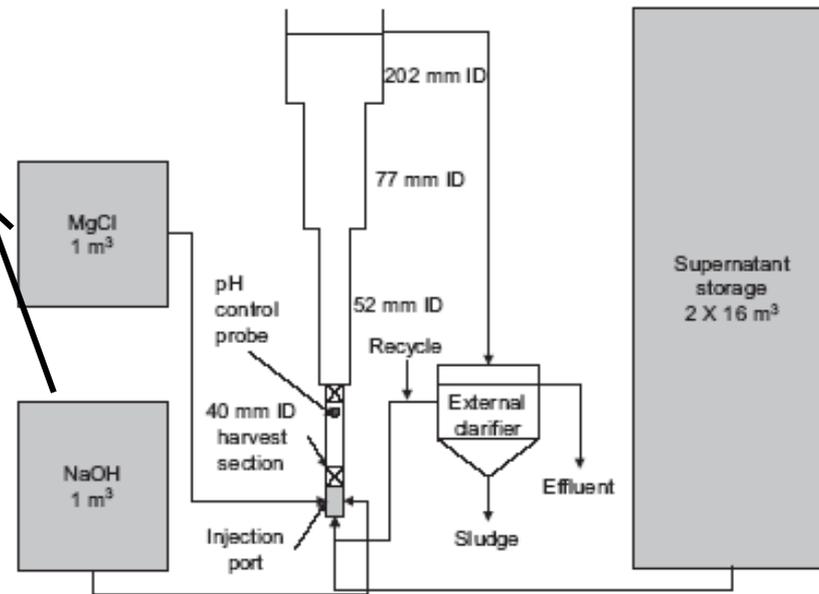
But, also a valuable
slow release fertiliser



The Ostara Pilot Plant at Derby STW



Fig. 1. Pilot-scale struvite crystallizer reactor process design.
MAP crystallizer



From Britton *et al*

J. Environ. Eng. Sci. 4 265-277 (2005)



W... ices

Full Scale Ostara Plant at Durham AWWTP,US



PHOTOGRAPH COURTESY OF OSTARA NUTRIENT RECOVERY TECHNOLOGIES INC

Conclusions

1. UKTAG have recently published proposals to tighten the Fe EQS, and it seems very likely that this will be followed by the P EQS being significantly lowered.
2. This presents a twin technological and economic challenge to the UK Water Industry as the majority of UK WwTW dose Fe to remove P.
3. It is still uncertain which fractions of P, and at what concentration, can lead to ecological disruption.
4. The evidence that Fe (in the form it is present in the aquatic environment) requires tight permit limits is far from conclusive.
5. STW is working to develop and evaluate technologies to remove and recover P that will lead to more sustainable solutions in the future.

Briefing Notes

The following briefing notes have been prepared for STW by Atkins. They summarise the situations for phosphorus, ammonium, iron and aluminium. They are intended to be informative, authoritative and unbiased so as to move the debate forward. In addition to publicly available published information, they make available information from some of the work undertaken by UKWIR which for practical purposes has been unavailable to non-members because of UKWIR's policy on pricing its publications.

Severn Trent Water Ltd
Briefing note on water quality criteria
for phosphorus

ATKINS

Executive summary

Under the Water Framework Directive (WFD), phosphorus has been selected as a supporting element to define diatom and macroalgae ecological status. As such, it has been classified as a physico-chemical standard and an Environmental Quality Standard (EQS) derived to ensure that below that value, the ecology is not adversely affected. The key driver under the WFD is diatom ecology with the focus not strictly on biomass or abundance but having the correct distribution of species present in rivers. However, almost 2,000 water bodies currently fail to achieve good status based on the EQS derived. There are also many incidences where a waterbody passes the phosphorus EQS but does not meet good status for diatoms and *vice versa*.

Almost all of the phosphorus discharged to the environment being evenly divided (on a UK basis) between agricultural runoff and wastewater treatment works (WwTW) discharges. In the absence of humans, phosphorus concentrations would be very low in the UK aquatic environment and this would be reflected in diatom populations being biased towards phosphorus sensitive species capable of ‘scavenging’ phosphorus from a variety of sources. Elevated phosphorus concentrations leads to a predominance of phosphorus tolerant diatoms which eventually out compete the phosphorus sensitive species and can eventually lead to an observable shift in ecology relating to macro algae and fish populations. There are, however, many other factors which may influence diatom communities, not just phosphorus; so the relationship between phosphorus and diatom composition is often vague and uncertainty is high. It is evident from the available data that assigning only a single variable to controlling diatom status is insufficient. The key difficulty in implementing an ecosystem-based approach to standard setting (as raised in the ammonia briefing document) is that the diatom metric (the Trophic Diatom Index – TDI) used to drive possible programmes of measures is insufficiently sensitive to show subtle changes in ecological status and is influenced by more factors than simply phosphorus, this is not a criticism of the method, simply a fact when dealing with a complex situation as observed in the river environment. It is important to note that in many water bodies the ecological quality for fish is good where diatom quality is determined to be less than good.

To meet the numeric phosphorus EQS would require major investment in both the agricultural and wastewater sectors without the comfort of being able to categorically demonstrate (or cost) the ecological benefits whether in diatom community biodiversity or in benefits to other trophic levels such as macro algae, invertebrates or fish.

Ultimately a decision at a national, Governmental scale needs to be made regarding the degree of protection required within our rivers, prioritisation of actions and what is affordable and what may be considered disproportionately expensive under WFD criteria. At this time evidence suggests that achieving good status for diatoms to the letter of the WFD is too expensive compared with the benefits likely to be achieved, particularly given that the benefits cannot be reliably measured. This suggests that alternative objectives should be considered (at least in the short to medium term) associated with ‘undesirable disturbances’ and ecosystem services focussed on ensuring diatom composition and abundance does not negatively impact on higher trophic levels. This approach would lead to target phosphorus levels likely to lie in the 0.1 to 0.2 mg/l SRP range and as a consequence, the focus for the next river basin plans and AMP6 could be to target water bodies where water quality in this range is achievable using a range of measures including source control, end-of-pipe treatment and control of agricultural diffuse inputs.

General conclusions are:

1. The Environment Agency is reviewing the existing EQS for phosphorus and will be including impacts on macrophytes as well as diatoms. Proposals presented at a recent workshop suggest that the existing EQS of 0.12 mg/l SRP should be made substantially more stringent. A proposal for consultation is expected in early 2012. The consultation period will be only 6 weeks.
2. The Environment Agency recognizes there is a large amount of uncertainty in the relationship between phosphorus and its impacts on diatoms, but seem to still want to pursue setting standards for phosphorus alone.
3. The main driver for reducing phosphorus levels in waterbodies is diatom biodiversity and composition, not simply abundance (as measured by chlorophyll analysis or biomass). Meeting the UK derived WFD target for good status drives a requirement for very low phosphorus concentrations in rivers.
4. The UK classification of diatoms for good status appears to be more stringent than equivalents elsewhere in Europe (France), based on available data for high alkalinity rivers.
5. In reality, phosphorus is one factor that can impact diatom numbers and/or diversity; however, there are many other factor which also impact diatoms, including nitrogen, shading, grazing, hydrology, hydromorphology, river management and other pollutants. As a consequence phosphorus may be between 0 and 100% of the contributing impact on diatoms.
6. The sources of phosphorus to UK rivers are dominated by agricultural runoff and WwTW effluent discharges. Over a half of the phosphorus entering WwTW is derived from anthropogenic and potentially avoidable sources. In the UK over 700 WwTW now have phosphorus permits to discharge in the range of 1 to 2 mg/l total phosphorus
7. Works dosing with iron for phosphorus reduction (the majority) to meet a 1 mg/l total phosphorus permit are likely to be discharging soluble reactive phosphorus (SRP) at around the EQS for lowland waters (0.12 mg/l SRP) based on limited available data. This suggests that reducing total phosphorus levels below 1 mg/l with iron dosing will result in effluents containing SRP levels at less than the required river quality for good status under the WFD (assuming the target is 0.12 mg/l SRP).
8. There is a lack of consistency of approach for determining SRP in water and therefore in assessing compliance with levels of interest. Orthophosphate is usually the form that is measured. This can include a fraction of particulate forms phosphorus digested from colloidal or particulate material, so monitoring tends to overstate the amount of SRP. Data indicate that the SRP contribution is typically 90 to 95% of the orthophosphate concentration but exhibits a large variation depending on site and time.
9. More obvious impacts on higher trophic levels, such as macro algae, invertebrates and fish, are not likely to occur until concentrations of phosphorus exceed a higher level than the current EQS (likely to be at least 0.2 mg/l SRP and above).
10. Where appropriate, seasonal permitting for WwTW discharges would be likely to improve ecological quality where phosphorus levels can be shown to be the factor stopping diatom good status being achieved. This may be achieved in a cost-neutral way of possibly even a saving to the water industry depending on circumstances. This approach needs to be considered on a site by site basis in light of other legislation and issues such as storage of phosphorus in sediment.

Recommendations for future planning include:

In general the Water Industry may wish to lobby for a tiered approach to meeting good diatom status, for example:

- 1) For any river where phosphorus levels are greater than 0.2 mg/l SRP consideration should be given to the most cost-effective options for reducing levels to less than 0.2 mg/l SRP.
- 2) Where SRP levels lie in the range 0.05 to 0.2 mg/l, the cause for diatoms not meeting good status needs to be identified. If phosphorus is confirmed as a major influencing factor, then the most cost-effective options for reducing levels to the EQS should be explored.
- 3) Where SRP levels are less than the EQS (e.g. 0.12 mg/l) – no further action.

Options for consideration under AMP6/RBC2:

- The Water Industry needs to lobby the Environment Agency to take a more site-specific view to dealing with non-compliance with expected diatom biodiversity/biomass. Key contributors to any non-compliance (not necessarily phosphorus) need to be identified and Programmes of Measures agreed to tackle highest priority causes. Until benefits can be justified to stakeholders (farmers and water industry) an interim alternative objective of reducing river levels to below 0.2 mg/l SRP might be considered.
- A general focus on reducing phosphorus levels in rivers to below 0.2 mg/l SRP other than at Natura 2000 sites where a higher priority may be given to reducing phosphorus levels to the EQS where justified.
- A switch to a more accurately specified and executed determination of phosphorus in rivers (e.g. as SRP).
- A general focus on where best to implement phosphorus reduction within catchments. This might possibly result in more widespread application of 1 mg/l total phosphorus permits but potentially with less emphasis on achieving less than 1 mg/l total phosphorus concentrations in effluents (which appears to be the focus of several AMP5 investigations). Iron dosed effluent with significantly less than 1 mg/l total phosphorus is likely to contain SRP at less than the lowland, high alkalinity EQS, which may not be easy to justify unless part of an integrated catchment approach to meeting the EQS (i.e. is part of a cost-neutral dosing regime).
- Pilot seasonal permitting conditions for identified catchments with: low sediment retention, high WwTW load and moderate ecology, supported by long term ecological monitoring downstream. Upstream phosphorus must be less than EQS and WwTW shown to be a major contributor to an EQS exceedance. The target should be 1 mg/l TP or less which should ensure that SRP concentrations are at the EQS or less. Ecological monitoring should include diatoms and macrophytes with supporting chemical monitoring of dissolved oxygen (diurnal cycles) to assess possible impacts on fish, phosphorus (total, SRP, TRP) and nitrogen species.
- Defra should consult on and strongly drive source control of phosphorus in food additives (where non essential or alternatives exist) and all detergents. This would likely to be a drawn out process and would probably not be achievable in the short term but should be started in AMP6/RBC2.
- Accelerated lead replacement schemes should be implemented for water company owned communication pipes and householder supply pipes, leading to marked reduction of drinking water phosphorus dosing in areas where it is cost-effective to do so. The cost to customer to so

this is potentially very high and so the need to quantify the environmental benefits with high certainty is essential.

Options for further investigation:

- Parameterisation of all factors impacting on diatom biodiversity/biomass is required. To use phosphorus as the only measure to restore good ecological status is inappropriate as it may not be the main cause of non-compliance; thus leading to expensive investment with uncertain outcomes.
- The bioavailability of phosphorus in WwTW effluents (dosed (Fe and Al), BNR and undosed) needs to be explored further.
- Improved modelling of ecological responses to phosphorus concentrations (probably based on INCA-P model) as part of an integrated catchment management approach.
- Optimised phosphorus application to land, based on improved assessments of bioavailability (e.g. DGT measurements¹).
- Development of an integrated approach to combining chemistry and ecological response (not just P and diatoms, but also N, hydrology, hydromorphology, riparian shading, suspended solids, organic pollution etc).

¹ http://www.msfp.org.au/docs/research_41.pdf

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1. Background

Phosphorus is the largest single most reason for English waterbodies not reaching good status under the Water Framework Directive (WFD) (Figure 1). ‘Good’ status under the WFD requires water quality to exhibit characteristics close to reference, pristine condition. Diatom data (numbers and type – P tolerant vs P sensitive) have been used to generate a phosphorus standard by which river quality may be judged to be in high, good, moderate or poor condition (Table 1). A summary of phosphorus data for England and Wales (Figure 2) shows the extent of non compliance based on an average of 2007 to 2009 data. The map shows that away from upland areas, in areas of intensive agriculture and urban environments concentrations are more often than not above the quality standard. However, it should also be noted that the EQS for good status in upland areas is significantly lower than in lowland, high alkalinity areas. In the UK there are no significant ‘natural’ mineralogical sources of phosphorus. In the absence of humans, phosphorus would be present in water at very low concentrations as part of the natural nutrient cycling process from the growth and death of algae, invertebrates and fish, with inputs from terrestrial organic matter.

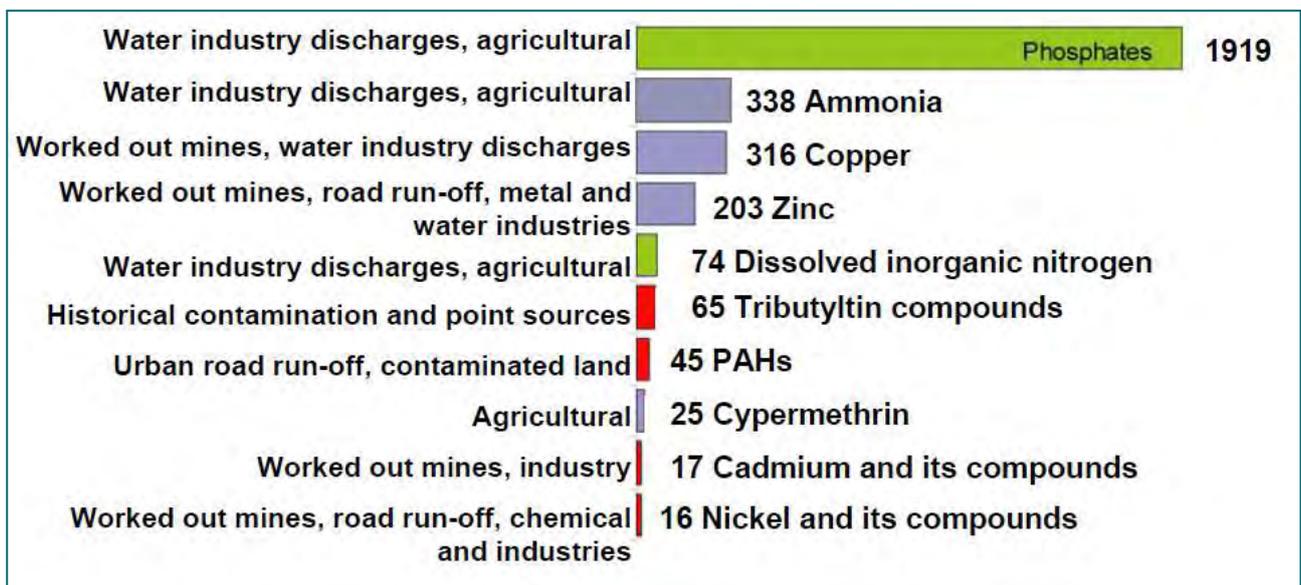


Figure 1 Number of waterbodies failing chemical standards(EA data)

Table 1 Standards for phosphorus in UK rivers under the WFD

Soluble Reactive Phosphorus (mg/l) (annual mean)				
Water type	High	Good*	Moderate	Poor
Under 80m altitude & less than 50 mg/l alkalinity	0.030	0.050	0.15	0.50
Over 80m altitude & less than 50 mg/l alkalinity	0.020	0.040	0.15	0.50
Any altitude & more than 50 mg/l alkalinity	0.050	0.12	0.25	1.0

* The target for all designated waterbodies to achieve under the WFD.

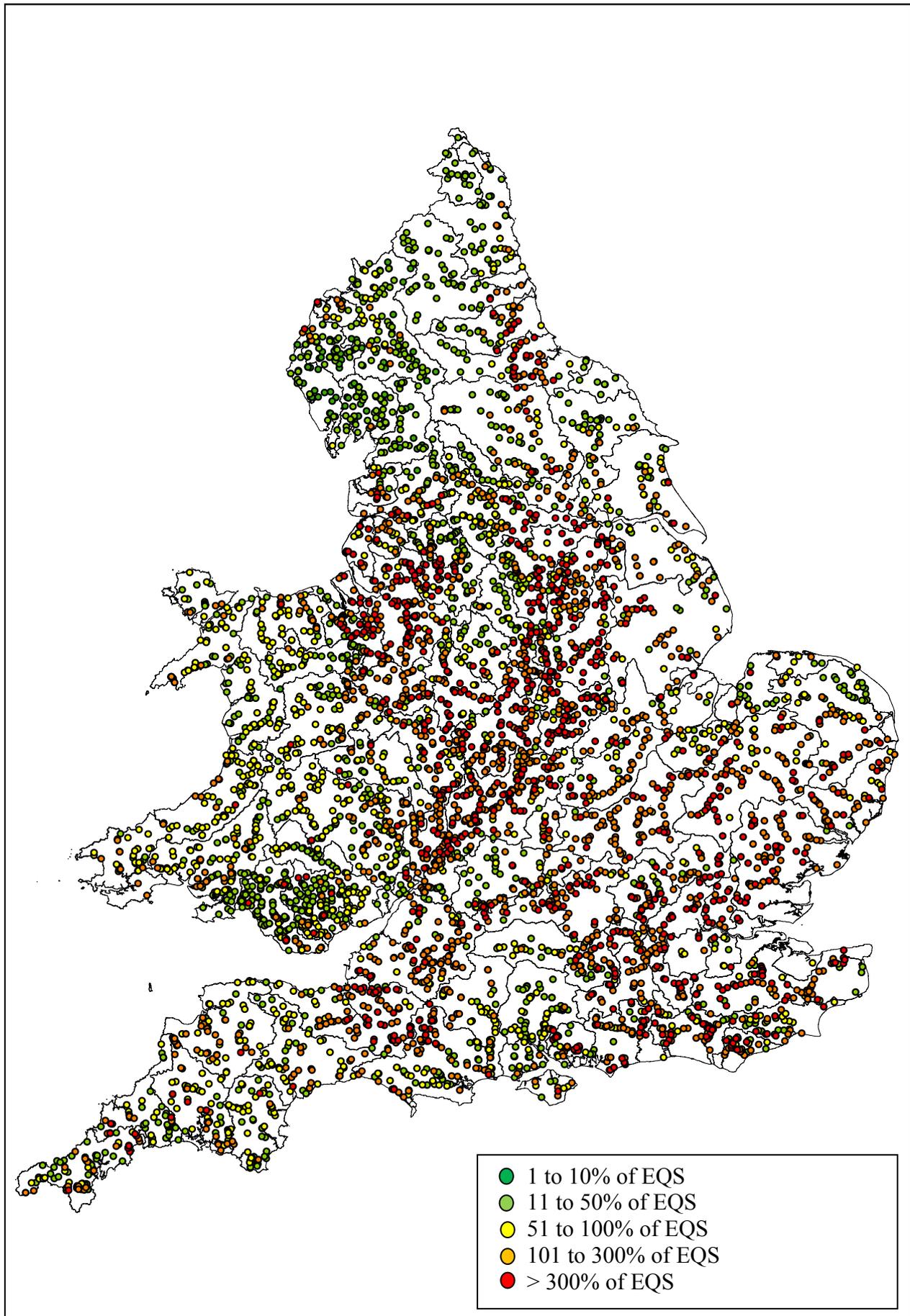


Figure 2 Mean orthophosphate concentration (2007 to 2009) as a percentage of the EQS

2. Sources of phosphorus

The impact of humans on phosphorus loads to the aquatic environment is substantial. We require phosphorus as part of a healthy diet to maintain bones, teeth and cellular processes with it being available naturally in cereals, dairy products and meat. In the absence of other uses of phosphorus in our modern world, a high human population alone will lead to several mg/l of phosphorus entering WwTW. However, on top of background dietary intake, we ingest an excess of phosphorus as a result of its presence in food additives such as soft drinks and processed meats which is then excreted in urine and faeces, therefore augmenting human derived sources of phosphorus to WwTW and thence to the environment. Other sources of phosphorus to wastewater treatment works (WwTW) include automatic dishwashing detergents, personal care products, laundry detergents (to be effectively banned in 2015) food waste, and water company dosing phosphorus for plumbosolvency control which alone contributes an average of 9% (UKWIR, 2010)² of the load (Figure 3). The other main contributor to phosphorus loads in rivers is agricultural runoff mostly derived from the addition of phosphorus fertilisers to optimise crop yields. Agricultural diffuse inputs of phosphorus comprise an estimated 39% of the river load and are of a near equivalent magnitude to that from wastewater treatment works (WwTW).

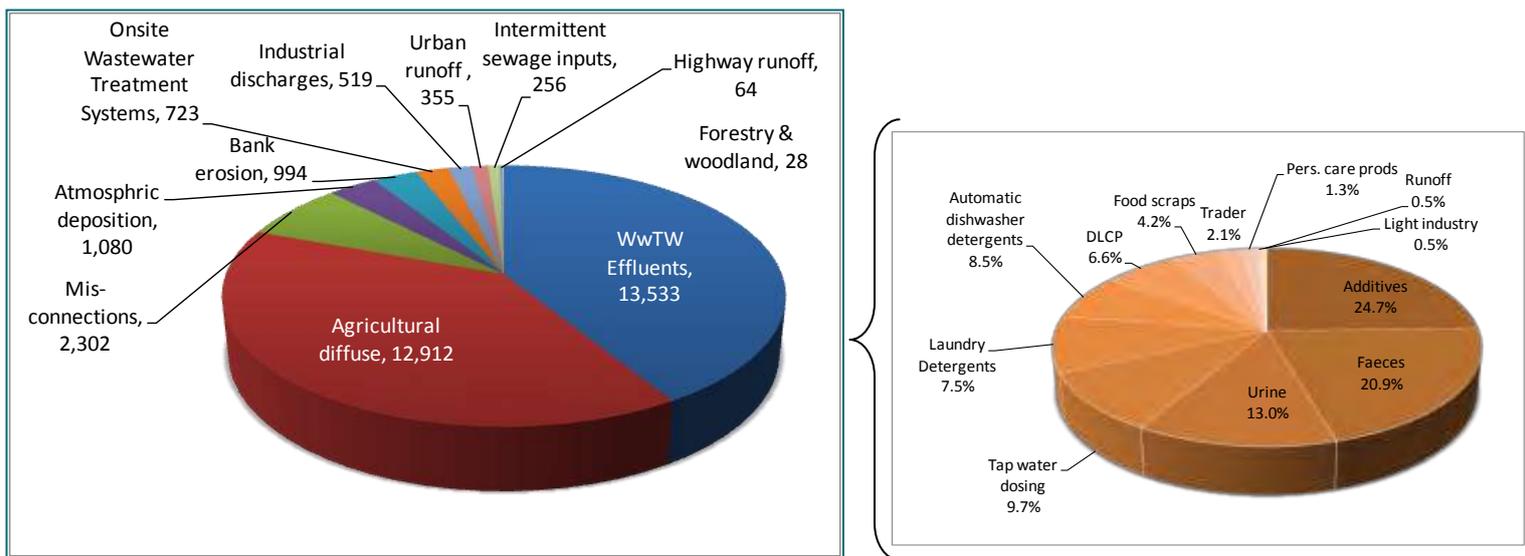


Figure 3 Sources of phosphorus to UK rivers (tonnes/year) and contribution to WwTW influent¹

² UKWIR (2010) Phosphorus Lifecycle Management (10/SL/02/9), ISBN: 1 84057 570 0, UK Water Industry Research, Queen Annes Gate, London. <http://www.ukwir.org/ukwirlibrary/93490>.

3. Phosphorus regulation and compliance

To meet the key European legislation regarding phosphorus which includes the Urban Wastewater Treatment Directive (UWWTD)³, Habitats Directive⁴ and now the WFD, a programme of phosphorus reduction measures have been carried out since the late 1990's. These have included end-of-pipe treatment at WwTW and measures to reduce diffuse agricultural inputs. There are now around 700 WwTW, serving over 23 million customers with some form of phosphorus reduction, dominated by iron dosing, but with some aluminium dosing and some utilising biological nutrient reduction (BNR), which in combination has led to around a 20% decrease in phosphorus discharged to rivers from WwTW since 2000.

Concentrations of total phosphorus in WwTW effluents not subject to phosphorus reduction are typically between 4 to 5 mg/l. Current legislation requires, (depending on WwTW size or catchment sensitivity), phosphorus reduction to 2 or 1 mg/l, with 1 mg/l total phosphorus considered the lowest reliable concentration which can be achieved using current tertiary treatment processes. Wide scale phosphorus reduction schemes were not proposed in AMP5 to meet the EQS because of one critical issue; the phosphorus standard is a physico-chemical parameter supporting ecological status, primarily diatom biodiversity and abundance, rather than it being classified as a Priority Substance or Specific Pollutant. Originally this led to phosphorus being considered as an "indirect" standard, with measures only implemented at sites 'failing' the standard where ecological status was compromised, consequently measures should be put into place to meet the EQS only as long as the ecological quality is shown to benefit.

The complex biogeochemistry of the natural environment rarely provides a black and white answer as to how to regulate phosphorus discharges and illustrates the difficulty when trying to correlate a single chemical parameter to an ecological element influenced by numerous physical, chemical and biological processes. The majority of reported relationships between diatom indices and SRP are plotted on a log-log graph which generally makes relationships look much better than they actually are. Figure 4 is a typical example, a relationship is drawn but closer examination shows that for a given TDI of say 70, SRP concentrations vary from 0.01 to 1.0 mg/l, two orders of magnitude! The reality is that phosphorus is a contributor to non-compliance for diatom good status, not necessarily the main cause. It may be accepted that above a certain threshold, phosphorus will always have an adverse impact, however, below typically 0.2 mg/l SRP there may be other factors impacting on diatom status including:

- Nitrogen
- Other pollutants
- Hydrology
- Hydromorphology
- River management
- Shading
- Other biological factors (e.g. grazing)

³ http://ec.europa.eu/environment/water/water-urbanwaste/index_en.html

⁴ http://ec.europa.eu/environment/nature/legislation/habitatsdirective/index_en.htm

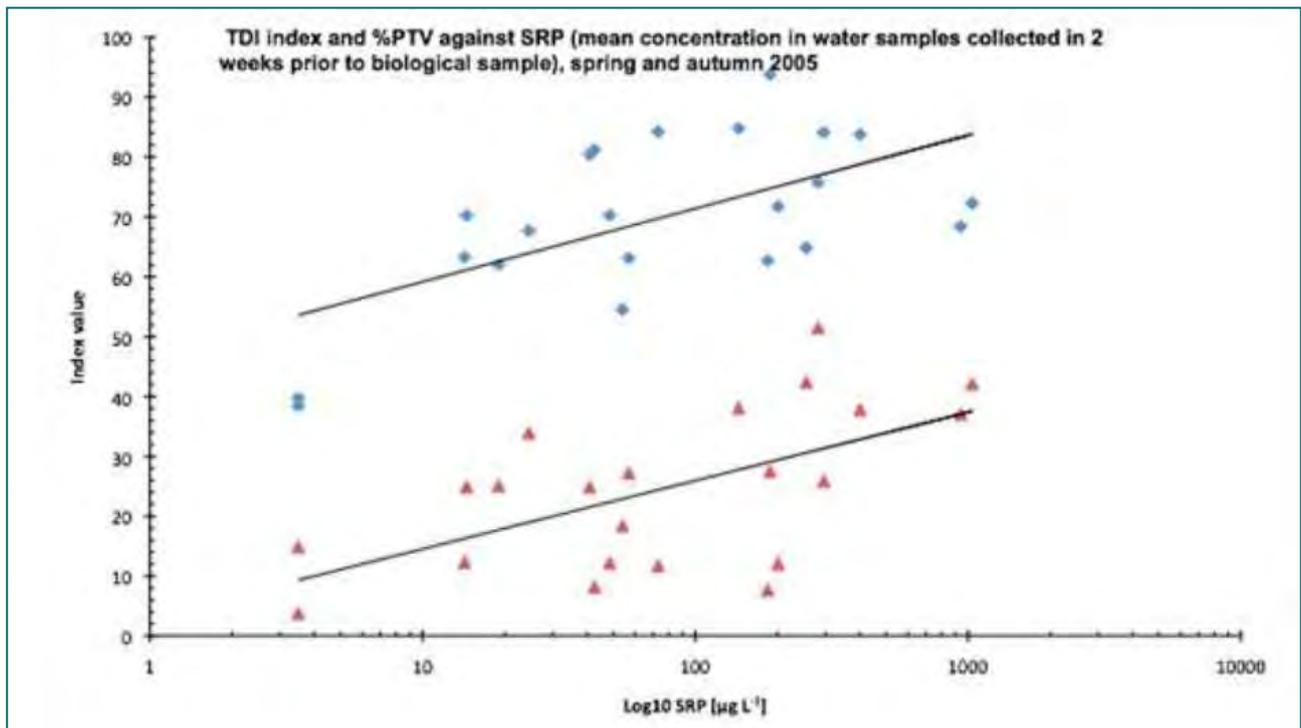


Figure 4 Relationships between Trophic Diatom Index (TDI – in blue) and associated percentage Pollution Tolerant Valves (%PTV – in red) against SRP (mean concentration in water samples collected 2 weeks prior to biological sample), spring and autumn 2005 in English lowland study streams (Natural England, 2010)⁵

Quite correctly, the first round of river basin planning and AMP5 investment has therefore concentrated on investigations to better understand the links between the ecology and chemistry and the impacts of WwTW effluents on downstream ecosystems, including a review of the EQS due out for consultation in early 2012, which will also include a measure of macrophyte quality. A recent presentation by the EA (UKTAG, Aston, Birmingham, 15th September) however, further highlighted the uncertainty in the relationship between phosphorus and diatoms/macrophytes, accepting a site-specific approach to non-compliance could be the best way of achieving WFD objectives. Early indications, however, also suggest the EQS may be lowered for some lowlands waters below the current 0.12 mg/l SRP. The issues of relying solely on phosphorus as a measure to achieve good diatom status still, however, remains. As the Environment Agency admit to the large uncertainty surrounding the phosphorus-diatom relationship, then any benefits of extensive P reduction at WwTW are likely to deliver benefits with equal uncertainty unless all of the parameters impacting diatom status are examined.

The key question now is to consider what are the options for the second cycle of river basin planning and AMP 6 investment, in order to maximise the benefits to aquatic ecology whilst achieving WFD compliance? This briefing documents explores three key areas of debate (i) the speciation of phosphorus in WwTW effluents and how compliance should be assessed; (ii) the options for seasonal permitting and (iii) measurement of ecological benefits of phosphorus reduction at WwTW. The recommendations from the briefing note aim to provide recommendations as to what can be achieved in AMP 6, what may be achievable in subsequent AMP cycles and where additional information is required.

⁵ <http://naturalengland.etraderstores.com/NaturalEnglandShop/NERR034>

4. Phosphorus speciation and its use in permitting

The regulation of phosphorus under different EU Directives is confusing and in some cases conflicting. The UWWTD requires compliance against a total phosphorus (TP) permit for WwTW, the Habitats Directive, total reactive phosphorus (TRP) in receiving waters and the WFD soluble reactive phosphorus (SRP) in receiving waters. ALL WwTW permits are expressed as TP which largely reflects the fact that the UWWTD standards were the first to be introduced and also offers a precautionary approach to phosphorus discharged into the environment which may at a later date become bioavailable.

In reality the forms of phosphorus in WwTW effluents cover a spectrum of bioavailability and vary considerably depending on whether iron dosing occurs. Figure 4 provides a comparison in phosphorus form and composition between those WwTWs that are iron dosing and those that are not. Figure 5 shows that upon iron dosing a large proportion of the SRP reacts with the iron and precipitates into the sludge. Therefore not only is the overall TP concentration decreased in the effluent, but the majority is particulate with lower bioavailability. The approximate 27% of soluble phosphorus is divided between that which is reactive (detectable using the molybdate analytical method) and that which is not reactive (assumed to be poly phosphates or organic species). In WwTW not dosing iron, SRP generated via the biological treatment processes employed at the works predominates in the effluent. Recent UKWIR data suggests that in both cases the particulate phosphorus amounts to around 0.26 mg/l (UKWIR, 2011)⁶. This shows the double benefits of iron dosing, overall concentrations are reduced, but so is the proportion of SRP in the effluent discharged. This does, however, suggest that any works investing in treatment to meet a permit of 1 mg/l TP is actually delivering effluent quality which will have only a minor proportion of phosphorus classified as ‘soluble reactive’.

The situation is further complicated by the fact that the measurement of phosphorus does not define its form accurately. The Environment Agency measure ‘orthophosphate’ which is essentially TRP as it is the molybdate SRP method applied to unfiltered samples. Although correct for compliance assessment with Habitats Directive requirements it is likely to over estimate SRP in rivers where there is appreciable solids because the acid used in the reagents will strip some phosphorus off the unfiltered solids presenting the sample. Again this provides a precautionary assessment as it can be argued that the acid removes the phosphorus on the solids which may become available in the river. However, the digestion is actually quite aggressive, using highly acidic 0.4M sulphuric acid and so may over state the phosphorus which may become available within a river water environment. Although this approach is pragmatic and is in line with the Agency’s precautionary principle, if not accurately quantified it could drive large amounts of potentially unnecessary investment.

Orthophosphate (TRP) comprises around 80% of the total phosphorus in effluent at WwTW dosing iron, but available Agency data for SRP, again agrees with the UKWIR data suggesting SRP only comprises a median contribution of approximately 14% of the total phosphorus concentration (Figure 6). This suggests that a significant proportion of the orthophosphate reported in WwTW effluents is being leached from the solids during the analysis rather than being actually SRP, which is quite different to river samples where SRP typically comprises 90% or more of the orthophosphate measured. The actual bioavailability of the phosphorus classified as orthophosphate, but not SRP is unknown. This data for WwTW suggests that a works dosing to 1 mg/l total phosphorus is discharging SRP at very close to, if not at the typical SRP EQS for good status level

⁶ UKWIR (2011) Phosphorus contributions from WwTW discharges to watercourses and their long term impacts relative to other sources. UK Water Industry Research, Queen Annes Gate, London.

of 0.12 mg/l. This suggests that seeking to optimise iron dosing to reduce total concentrations in effluent to below 1 mg/l TP could lead to the water company discharging SRP at less than the EQS. It therefore questionable whether focus should be on reducing BAT for phosphorus or considering wider scale dosing to 1 mg/l total phosphorus.

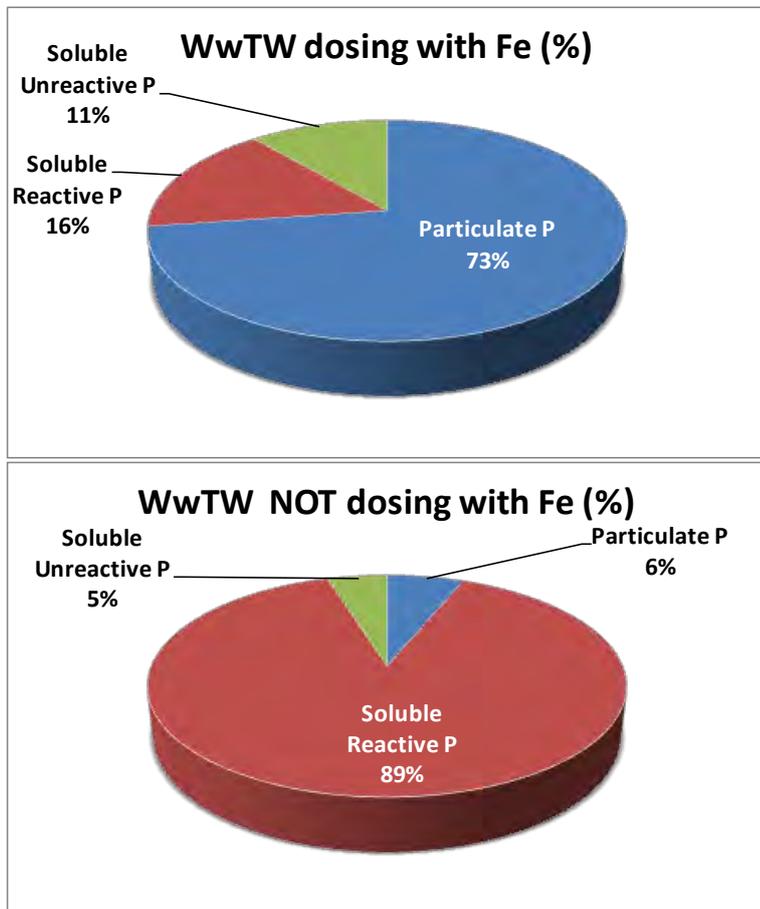


Figure 5 Typical forms of phosphorus in WwTW effluents (UKWIR, 2011)²

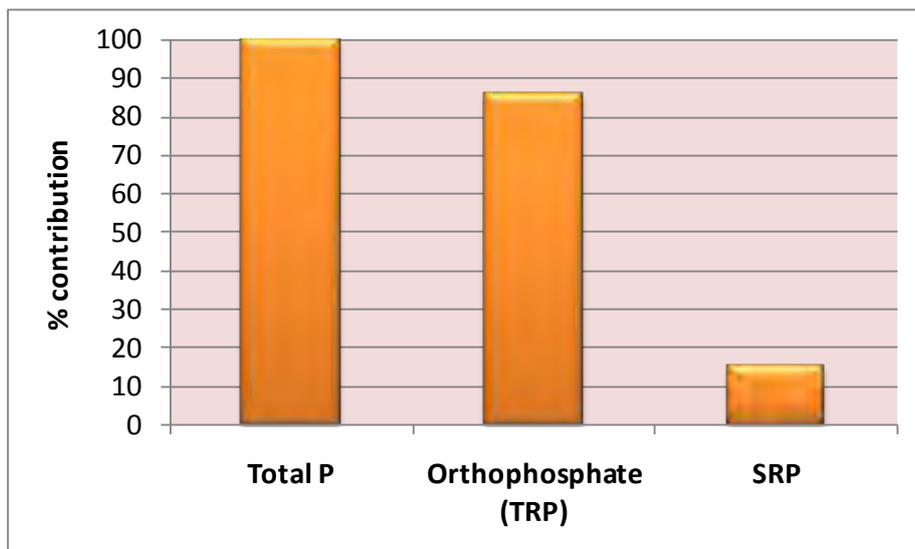


Figure 6 Typical forms of phosphorus in WwTW effluents as median contributions (EA WIMS data)⁷

⁷ UKWIR (2011) Data provided for UKWIR WW02 source apportionment project.

An analysis of Agency data suggests the majority of river monitoring data comprises orthophosphate data and so if used to assess compliance will potentially overstate SRP because the acid reagents will leach some phosphorus from solids or colloidal material, particularly in areas where suspended solids levels are high. Data suggest that the median percentage of SRP as orthophosphate is between 90 and 95% (Figure 7) but the range is large, with a significant proportion of samples, particularly in the midlands region where the SRP percentage is in the range 50 to 80%. This will translate into less failures of the EQS within the river should genuine SRP be determined rather than orthophosphate, although not by a substantial degree. This data is quite different to that for WwTW effluent dosed with iron where SRP only contributes a small percentage of the measured orthophosphate.

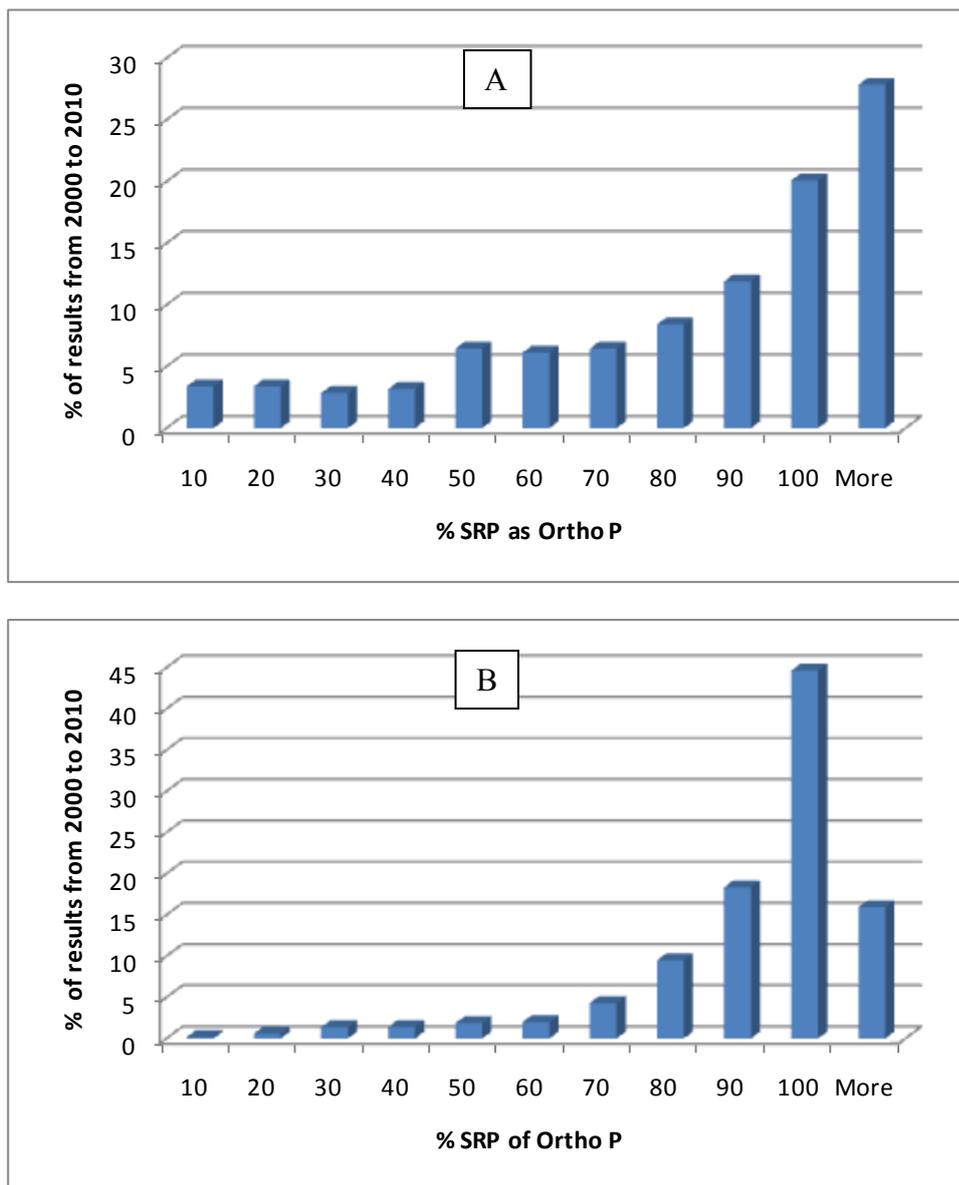


Figure 7 Histogram of phosphorus results for river samples taken from 2000 to 2010 showing the percentage of the calculated SRP proportion of measured Orthophosphate for Midlands (A) and Anglian (B) EA region

There is little ‘true’ SRP monitoring data available (in effluents or receiving waters) which should be remedied in the longer term. Interestingly a large proportion of WwTW compliance data is also reported as orthophosphate (TRP) rather than TP, which when assessing compliance with the UWWTD would underestimate TP in the effluent by may be 15% (Figure 5). What is clear is that there is a lack of consistency in the current monitoring and compliance assessment regime which leads to a lack of confidence in the decision-making process when considering future investment.

The following recommendations may be proposed which would provide more a more accurate assessment of phosphorus speciation and impacts:

- Accept that for the sake of custom, experience, simplicity and broad comparability with previous data that the molybdate method will continue to be used, but that variants of the procedure involving different concentrations of acid and different “digestion” times should be avoided.
- Consider a defined filtration step that excludes the very variable “non-settleable” fraction and focuses on a specific and widely accepted cut off for particulate phosphate.
- Determine the effect of the establishment of such an improved and more clearly defined methodology on the perceived concentrations of phosphate in different waterbodies and different types of waters.
- Assessment of the impact of any differences on any interpretation of compliance and ecological risk in the different waters investigated.

5. Ecological benefits of phosphorus reduction

As already described, there is little data available to quantify the benefits of reducing phosphorus levels in rivers and considering phosphorus alone as the only measure to address diatom status is a flawed principle. A previous comprehensive review of the impacts of phosphorus reduction on downstream ecology¹ provided the following conclusions as to why improvements have not been observed after investment in P reduction schemes at WwTW:

- **Upstream SRP concentrations exceed the EQS:** For the studies on the Ribble, for example, all of the sites upstream of the WwTW exceeded the 0.12 mg/l SRP EQS, which means that any remedial action taken at WwTW did not lead to EQS compliance downstream owing to significant additional inputs to the waterbodies. As a result of the EQS exceedance upstream, changes in ecology downstream of the WwTW may have been masked by background levels still exceeding the EQS and therefore by definition (assuming the EQS accurately reflects diatom pressures) should be having an adverse impact on the in-stream ecology. As can be seen from Figure 2, there are many upland areas well away from urban centres where monitoring data shows phosphorus concentrations already a significant proportion of the EQS.
- **Data relevance:** Diatom measurements using the Trophic Diatom Index (TDI) are limited with most ecological data available for invertebrates which are not necessarily the most relevant ecological metric.
- **Data unavailability:** The Mean Trophic Rank (MTR) and TDI data are collected at best, twice a year, however, both macrophyte and diatom communities have a high level of natural variation, which can easily obscure changes in water quality due to WwTW phosphorus reduction.
- **Sediment partitioning:** Although WwTW discharges may decrease, some form of re-equilibration from sediment-bound phosphorus can occur to stabilise downstream water column concentrations.
- **Timescales:** Improvements in water quality are often rapid and measurable, but improvements in biological status tend to be slow or non-existent, with full recovery potentially taking a decade or more. Long term monitoring is required to demonstrate the subtle changes in biodiversity the ecological indicators are seeking.
- **Other factors:** Eutrophication in flowing water is not as clear as in lakes where algal blooms are generally very obvious. In rivers, the EQS for phosphorus is based on subtle community changes. In reality, there are many factors which control the growth of algae in rivers including:
 - Other nutrient sources (e.g. nitrate, silicate)
 - Hydrology and hydromorphology
 - Other chemicals present (metals, pesticides, persistent organic pollutants)
 - Suspended sediment concentrations and/or shading
 - Climate change

Consequently, it is not unsurprising that benefits of P reduction at WwTW have had little bearing on diatom status downstream. There is therefore a temptation to draw the conclusion that phosphorus reduction at WwTW is pointless without additional measures to reduce diffuse sources of phosphorus from agriculture. Where, phosphorus levels in the river are not the limiting factor, then this conclusion holds, but if phosphorus can be shown to be the limiting factor then benefits may be realised. For example, a comprehensive review by Natural England has demonstrated a positive correlation, some of which are non linear, between improving ecological status (for diatoms and macrophytes) with reducing phosphorus concentrations, particularly where phosphorus concentrations lie in the range 0.1 to 0.2 mg/l. Above a concentration of around 0.2 mg/l phosphorus then the significance of phosphorus as a limiting factor becomes less evident and relationships then to break down.⁸ This suggests that for many UK rivers which exhibit phosphorus concentrations of less than 0.2 mg/l that there is scope for ecological improvement with reducing phosphorus loads if phosphorus can be shown to be the limiting factor in achieving good status for diatoms.

Where justified, the question to be answered is how best to reduce phosphorus loads to rivers in the most cost-effective way to maximise benefits to the environment. An option which should not be ignored is source control which if implemented (banning of P in detergents, food additives and cessation of P dosing for plumbosolvency) then loads to WwTW could be reduced by around 40% (Figure 3). Some, if not all of these options may be expensive and in some cases unpopular, but need to be considered in light of the benefits gained and costs for tertiary treatment. The wide scale exceedance of the standard also suggests that a prioritisation is required and a dose of pragmatism applied. In other words effort should be placed on improving water quality in areas where the EQS may be achieved via WwTW treatment alone as well as areas which would most benefit from nutrient reduction (high quality, rural rivers, rather than heavily impacted urban ones). One other option may be seasonal permitting which is considered below.

One important point to make, which also fits in line with conclusions regarding iron and ammonia, is that there is evidence that the UK classification is more stringent and/or wide ranging than neighbouring EU Member States such as France, Belgium and Netherlands (at least for high alkalinity lowland waters). The UK regulators are currently reviewing the EQS, with particular reference to macrophyte impacts and high alkalinity waters. By 2012 a revised standard is likely to be available, but it is thought that it is unlikely to be significantly different to the existing value. The derivation of this standard will require careful scrutiny and serious consideration of why classification in somewhere such as France appears more relaxed than the UK (at least for chalk-based rivers).

In summary, the uncertainty that still exists in the relationship between phosphorus concentrations and ecological impacts on diatoms may be considered as insufficiently developed to support major investment in wastewater investment in the next AMP/ river basin cycle. The Water Industry may wish to lobby for a tiered approach to meeting good diatom status for example:

- 1) Any river where phosphorus levels are greater than 0.2 mg/l SRP then the most cost-effective options for reducing levels to less than 0.2 mg/l SRP should be explored.
- 2) Where SRP levels lie in the range 0.05 to 0.2 mg/l then the cause for diatoms not meeting good status needs to be identified. If phosphorus is confirmed as a major influencing factor, then the most cost-effective options for reducing levels to the EQS should be explored.
- 3) Where SRP levels are less than the EQS – no further action.

⁸ Natural England Research Report NERR034. An evidence base for setting nutrient targets to protect river habitat, 11 November, 2010

6. Seasonal permitting

Algal growth is driven by water temperature, nutrient availability and sunlight largely. As a consequence it is the spring and summer months when greatest activity occurs in the water column. During the winter months, growth is significantly diminished and river flows are higher and nutrients are flushed more readily from the system. There is, therefore, a potentially rational case to seek to limit phosphorus discharges during the summer months, possibly accepting that the current Best Available Technique - 'BAT' limit of 1 mg-P/l must be reduced, but allowing reduced or cessation of dosing in the winter. Previous UKWIR projects have suggests site-specific benefits may be achievable (Figure 8) with significant reductions in summer SRP concentrations downstream as the total phosphorus concentration in the effluent is reduced. The figure below shows the predicted river SRP concentration based on the total concentration of phosphorus discharged from the two main WwTW on the river Wensum in East Anglia. The figure shows that significant downstream reductions in SRP may be achieved by applying a permit to the effluent of 2 mg/l TP or less. The data also shows that owing to higher flows in winter the concentration of phosphorus in the WwTW effluents have much less impact on the downstream concentrations, suggesting that even above 2 mg/l TP the EQS (0.12 mg/l SRP) would be comfortably achieved.

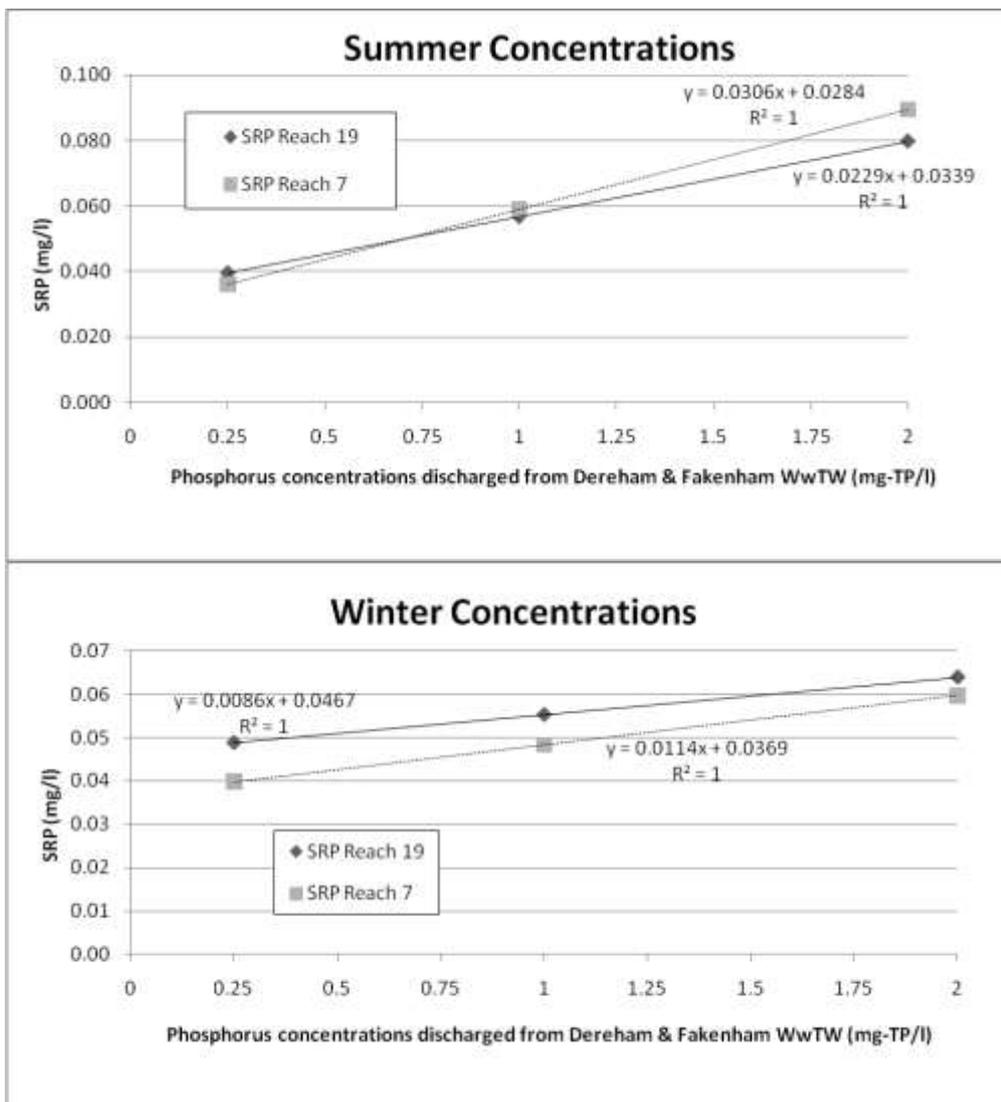


Figure 8 Summer and Winter predicted downstream SRP concentrations as a result of different permitting conditions at two WwTW¹.

However, a recent UKWIR report on ‘Better Regulation’⁹ assessed options for seasonal consenting made an important point regarding meeting an annual average phosphorus EQS:

“If the EQS for phosphorus is to be met as an average over the whole year, any increases in concentration in winter must be compensated for by reduced levels in the summer. The effect of switching iron dosing on in summer and off in winter was examined. This approach does not at present seem to be a viable proposition. Whilst treatment is only effective during the summer dosed period, in winter (undosed) the increase in concentration is, for most rivers, not compensated for by increased river flow. Hence the overall annual mean value is higher than that specified in the EQS.”

This suggests that the way the EA assess compliance would have to be reviewed and the benefits of lower summer phosphorus concentrations balanced against levels in winter which could exceed the EQS, but which may have no significant environmental impact. Aspects such as storage of “winter phosphorus” in sediments with subsequent release in summer would have to be considered. Current evidence suggests that a site-specific assessment needs to be implemented with pilot-studies employed to assess the potential benefits of seasonal permitting to both the receiving water and the water company in terms of practicality and costs.

⁹ Better Regulation: Integrated Catchment Regulation UKWIR 10/WW/23/6 - ISBN: 1-84057-566-2, UK Water Industry Research, Queen Anne's Gate, London.

7. Recommendations for future permitting

Based on the substantial amount of research carried out in the last 5 years it is possible to provide a series of recommendations as to how to move issues regarding phosphorus permitting forward:

Options for consideration under AMP6/RBC2:

- The Water Industry needs to lobby the Environment Agency to take a more site-specific view to dealing with non-compliance with expected diatom biodiversity/biomass. Key contributors to any non-compliance (not necessarily phosphorus) need to be identified and Programmes of Measures agreed to tackle highest priority causes. Until benefits can be justified to stakeholders (farmers and water industry) an interim alternative objective of reducing river levels to below 0.2 mg/l SRP might be considered.
- A general focus on reducing phosphorus levels in rivers to below 0.2 mg/l SRP away from Natura 2000 sites (where greater emphasis on meeting the EQS may be given, provided it can be demonstrated phosphorus is the limiting factor).
- A switch to measurement of SRP in rivers and (for the purpose of apportionment) WwTW final effluents for consistency. A 1 mg/l total phosphorus permit will still have to be retained at larger WwTW to meet UWWTD requirements. At smaller sites, however, permitting based on SRP should be considered.
- A general focus on where best to implement phosphorus reduction within catchments, possibly resulting in more widespread application of a 1 mg/l total phosphorus permits but potentially with less emphasis on achieving less than 1 mg/l total phosphorus concentrations in effluents as this is likely to lead to effluent quality to being below the EQS which may not be easy to justify unless an integrated catchment approach shows by 'over achieving' at one site provides more benefit in terms of downstream water quality than dosing at a number of smaller sites.
- Pilot seasonal permitting conditions for identified catchments with: low sediment retention, high WwTW load and moderate ecology, supported by long term ecological monitoring downstream. Upstream phosphorus must be less than EQS and WwTW shown to be a major contributor to an EQS exceedance. The target should be 1 mg/l TP or less which should ensure that SRP concentrations are at the EQS or less. Ecological monitoring should include diatoms and macrophytes with supporting chemical monitoring of dissolved oxygen (diurnal cycles) to assess possible impacts on fish, phosphorus (total, SRP, TRP) and nitrogen species.
- Defra should consult on and strongly drive source control of phosphorus in food additives (where non essential or alternatives exist) and automatic dishwashing detergents. This would likely to be a drawn out process and would probably not be achievable in the short term but should be started in AMP6/RBC2.
- Accelerated lead replacement schemes should be implemented for Water Company owned communication pipes and householder supply pipes, leading to cessation of phosphorus dosing in areas where it is cost-effective to do so. The cost to customer to do this is potentially very high and so the need to quantify the environmental benefits with high certainty is essential.

Options for further investigation:

- Parameterisation of all factors impacting on diatom biodiversity/biomass is required. To use phosphorus as the only measure to restore good ecological status is inappropriate as it may not be the main cause of non-compliance; thus leading to expensive investment with uncertain outcomes.
- The bioavailability of phosphorus in WwTW effluents (dosed (Fe and Al), BNR and undosed) needs to be explored further.
- Improved modelling of ecological responses to phosphorus concentrations (probably based on INCA-P model) as part of an integrated catchment management approach.
- Optimised phosphorus application to land, based on improved assessments of bioavailability (e.g. DGT measurements¹⁰) - this is a WFD measure, rather than permitting.
- Development of an integrated approach to combining chemistry and ecological response (not just P and diatoms, but also N, hydrology, hydromorphology, riparian shading, suspended solids, organic pollution etc).

Future AMP cycles:

- Maximum source control where cost effective.
- Seasonal permits based on SRP with significantly lower summer values, combined with limited or potentially no dosing in winter.
- Significant reductions from agricultural diffuse sources via land and 'smart' fertiliser and land management.
- Optimised iron dosing to minimise SRP rather than total phosphorus concentrations.

¹⁰ http://www.msfp.org.au/docs/research_41.pdf

Severn Trent Water Ltd
Briefing note on water quality criteria
for ammonia

ATKINS

Executive summary

1. Current regulation of ammonia in surface waters is based largely on meeting the requirements of the Freshwater Fish Directive (FFD). This directive sets limits for two forms of ammonia - total ammonia and un-ionised ammonia, the relative proportions of which are determined by water pH and temperature. Provision is made for derogations against the total ammonia standard where it can be shown that ammonia levels are not detrimental to fish development. Wherever two limits apply one will be more important than the other. For most surface waters in England and Wales the total ammonia limit value is the more stringent of the two – unless derogations apply for total ammonia, in which case the un-ionised ammonia criterion becomes more important.
2. As part of the implementation of the Water Framework Directive (WFD), Member States have to define water quality criteria for ammonia that are consistent with the achievement of good ecological status in surface waters. UK regulators have undertaken an assessment of ammonia by both toxicological and field-based methods. The resulting proposals by UKTAG involve an ecologically-based ammonia limit value of 0.6 mg N/l as total ammonia. This represents, in most cases, little change in regulation compared with the published FFD limit for total ammonia. However, where derogations are already in place the new proposal is likely to be substantially more stringent. The basis of the proposed new limit is also questionable because it fails explicitly to account for the relationship between ammonia toxicity, its speciation (form) and water quality.
3. Any increase in the stringency of regulation should be based on evident need and should be supported by an assessment of its consequences on the resulting requirement for investment. It is not certain either that need has been demonstrated or that implications have been fully considered. Estimated levels of non-compliance at the water body level, comparing the proposed with current limits, are predicted to increase – though not markedly. However, if, for example, the less than good sites represent waters with relatively high ammonia levels it is possible that the improvement (and investment) required in order to achieve compliance with the new standard will be substantially greater than that needed to meet the existing one.
4. Comparison with regulation in other developed countries shows that the current FFD guidelines are stringent with respect to limits set by the US EPA, Environment Canada and regulators in Australia and New Zealand. All water quality criteria for ammonia are based on the appreciation that the un-ionised form is the more toxic, but that some form of protection with respect to ionised ammonia is needed at pH values less than 8. The current limit for un-ionised ammonia is accepted widely. For total ammonia there is less agreement about an appropriate level, but both the FFD and the proposed UKTAG value appear stringent in relation to limits applied elsewhere.

However, recent evidence relating to the toxicity of ammonia to sensitive species such as freshwater mussels is leading several overseas regulators to give consideration to setting flexible limits that can be more or less demanding - as necessary, at a local level. This approach of adapting the quality criterion to meet local water management objectives reflects a move by many regulators away from the imposition of uniform standards that do not take into consideration the potential benefits that might be realised if such standards are met. By contrast, current UKTAG proposals involve the establishment of a new essentially fixed criterion to be established for WFD compliance purposes.

5. In short, examination of the large amount of regulatory research on ammonia (in Europe and particularly work done independently in the US) indicates that the Freshwater Fish Directive (with derogation for total ammonia) seems to strike the right balance between protection of the environment and over-regulation. It is recommended that future ammonia limits should:
 - a. Be based on current FFD experience or on the new national assessment, combined with existing provisions for derogation;
 - b. Retain the element of control over both total and un-ionised forms of ammonia;
 - c. Take account of local needs in terms of the present conditions and potential of the surface water under consideration (note this focus on local conditions does not necessarily mean lower water quality, but it is intended to focus resources where they are most required and likely to be most effective).
6. The recommended approach for the Water Industry is therefore to continue to develop a management strategy for ammonia that is based on the above principles. In particular, consideration should be given to an appropriate choice of limits for total ammonia in waters that either are evidently of good status with current concentrations of ammonia or which have little prospect of early improvement in status for reasons other than ammonia.

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1. Background

1.1. Introduction

The implementation of the Water Framework Directive (WFD) involves reconsideration of the regulation of ammonia concentrations in surface waters. New criteria for level of ammonia acceptable for the protection of aquatic life have been proposed by the UK technical advisory group for WFD implementation (UKTAG). These new criteria are more stringent than those applied under the Freshwater Fish Directive (FFD). Water Companies are concerned that the proposed limits for ammonia are both overprotective and do not take into account the increased cost to the water consumer that will arise as a consequence of the proposed changes. These proposals have serious implications for wastewater treatment works consents, leading to the potential need for very costly investment in further treatment in order to meet the latest incremental tightening of regulation.

As an example, figures have been provided to show that for one large wastewater treatment works (WwTW) in the Midlands the cost of upgrading to meet an ammonia consent of 3 mg N/l (from a previous value of 5 mg N/l) was £57 million in capital costs and £1 million per year in ongoing operational costs.

Severn Trent Water has asked Atkins to provide this review of the regulation of ammonia in surface waters as a background to further discussion of the way forward.

2. Forms of Ammonia in Water

At the pH of most biological systems, ammonia exists predominantly in the ionic form (NH_4^+). The fraction of total ammoniacal nitrogen that is present in the un-ionised form (NH_3) depends on three principal factors: the pH value of the water, its temperature and the amount of dissolved salts in the water (this last quantity is usually expressed in terms of the electrical conductivity of the water).

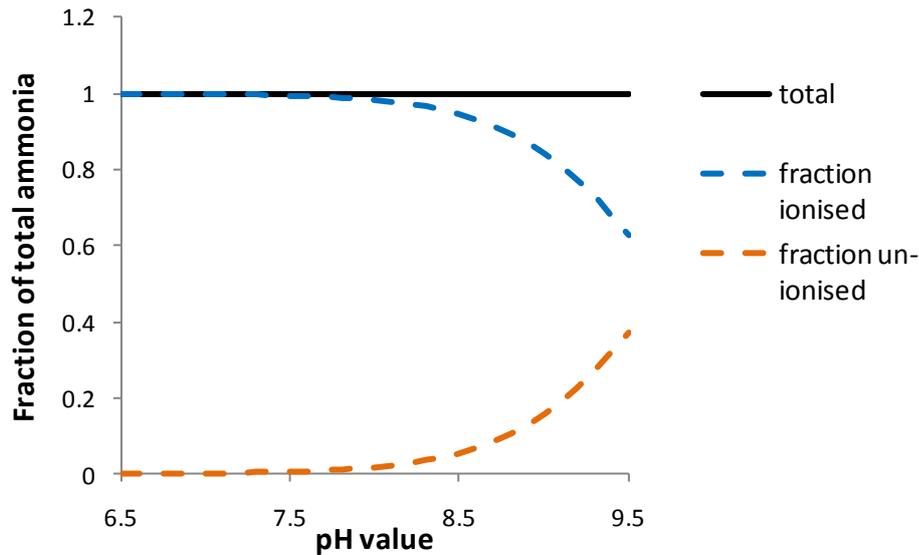


Figure 2-1 Illustration of ammonia speciation versus pH (temperature 10 °C)

The relative importance of these three factors is illustrated below in terms of the change in the fraction of un-ionised ammonia produced by a plausible change in each factor, the others being assumed to be constant.

Table 2-1 Effects of pH value, temperature and ionic strength (electrical conductivity) on proportion of ammonia present in un-ionised form

Factor	Change	Comment	Percentage change in fraction of un-ionised ammonia (%) from baseline conditions*
pH value	From pH 7.5 to 8.0	The difference between a typical sewage effluent and a well buffered surface water	+212
Temperature	From 10 °C to 15 °C	Approximate range of mean daily temperature for 2006 in 437 sites in salmon rivers across England and Wales ¹	+ 47
Electrical conductivity	From 0 to 600 µS/cm	The difference between pure water and a well buffered lowland river	-10

*Assuming baseline conditions for factors not changed to be: pH 7.5, Temperature 10 °C, Electrical conductivity 0 µS/cm. The fraction of total ammonia present as the un-ionised form under these condition is 0.0058.

From this it can be seen that, within these plausible ranges of variation, the pH value is the predominant influence on the fraction of un-ionised ammonia, followed by temperature and then conductivity.

¹ Changing water temperatures: a surface water archive for England and Wales. H.G. Orr, S. des Clers, G.L. Simpson, M. Hughes, R.W. Battarbee, L. Cooper, M.J. Dunbar, R. Evans, J. Hannaford, D.M. Hannah, C. Laize, K.S. Richards, G. Watts and R.L. Wilby. BHS Third International Symposium, Managing Consequences of a Changing Global Environment, Newcastle 2010

3. Aquatic toxicity of ammonia

The speciation relationships discussed in the previous section are important to ammonia toxicity because un-ionised ammonia (NH_3) is considerably more toxic than ammonium ion (NH_4^+). The importance of un-ionised ammonia was recognised when it was observed that increased pH caused total ammonia to appear to be much more toxic (Chipman², 1934; Wuhrmann and Woker³, 1948).

It is, however, also accepted that the ionised form does exert some toxicity. This toxicity is evident in the residual toxic effects of ammonia that are observable at low pH value where the concentration of un-ionised ammonia is vanishingly small. Observations that ammonia toxicity is relatively constant when expressed in terms of un-ionized ammonia come mainly from toxicity tests conducted at pH values of greater than around 7.5, where the main toxic effect appears to be exerted by un-ionised ammonia. At lower pH, toxicity varies considerably when expressed in terms of unionized ammonia and under some conditions is relatively constant in terms of ammonium ion (Erickson⁴, 1985). Studies have also established that mechanisms exist for the transport of ammonium ion across gill epithelia (Wood⁵, 1993), so this ion might contribute to ammonia exchange at gills and affect the build up of ammonia in tissues if its external concentration is sufficiently high. This is not to say that ammonium ion is as toxic as unionized ammonia, but rather that, despite its lower toxicity, it might still be important because it is generally present in much greater concentrations than un-ionised ammonia. This is illustrated by an idealised illustration (based on EPA 1999⁶) of ammonia toxicity.

² Chipman, W.A., Jr. 1934. The Role of pH in Determining the Toxicity of Ammonium Compounds. Ph.D. Thesis. University of Missouri, Columbia, MO. 153 pp.

³ Wuhrmann, K., and H. Woker. 1948. Beitrage zur Toxikologie der Fische. II. Experimentelle Untersuchungen uber die Ammoniak- und Blausaurevergiftung. Schweiz. Z. Hydrol. 11:210-244.

⁴ Erickson, R.J. 1985. An evaluation of mathematical models for the effects of pH and temperature on ammonia toxicity to aquatic organisms. Water Res. 19:1047-1058.

⁵ Wood, C.M. 1993. Ammonia and urea metabolism and excretion. In: The physiology of fishes. Evans, D.H. (Ed.). CRC Press, Ann Arbor, MI. pp 379-424.

⁶ United States Environmental Protection Agency. 1999. 1999 Update of ambient water quality criteria for ammonia. EPA-822-R-99-014. National Technical Information Service, Springfield, VA.

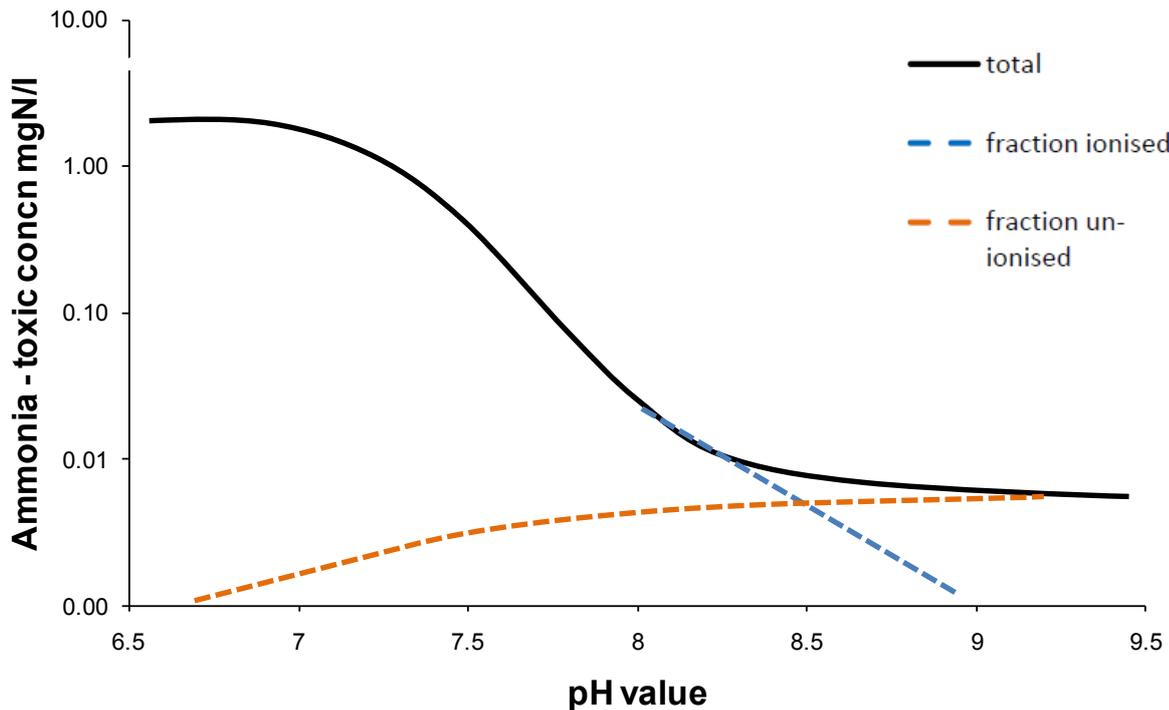


Figure 3-1 Idealised illustration of ammonia toxicity

The two forms thus appear to differ in toxicity to fish species by approximately two orders of magnitude. This difference has led to the differentiation, in term of the recommended quality standards, between the two forms of ammonia.

It is not proposed to carry out a detailed review of the toxicology of ammonia here – this has been done many times elsewhere (^{7 8 9 10 11}) and the results used to frame water quality standards. Hence, these standards can be seen as summaries of toxicology data in that they are designed to reflect the prevailing view of the threat posed by aquatic life by ammonia. So we will consider below the existing standards and any proposals for new standards.

⁷ Seager J, Wolff E W and Cooper V A, 1988 Proposed Environmental Quality Standards for List II substances in water: ammonia. TR 260. Prepared for the Department of the Environment (DoE). Medmenham, Buckinghamshire: WRc.

⁸ US Environmental Protection Agency (US EPA), 1999 Update of ambient water quality for ammonia. Technical Version EPA-823-F99024. Washington, DC: US EPA.

⁹ Alabaster J S and Lloyd R, 1982 Water quality criteria for European freshwater fish. London: Butterworth Scientific.

¹⁰ Szumski D S, Barton D A, Putnam H D and Polta R S, 1982 Evaluation of EPA unionised ammonia toxicity criteria. Journal of the Water Pollution Control Federation, 54, 281–291.

¹¹ Nixon S C, Gunby A, Ashley S J, Lewis S and Naismith I, 1995 Development and testing of General Quality Assessment schemes: dissolved oxygen and ammonia in estuaries. NRA Project Record 469/15/HO. Bristol: National Rivers Authority.

4. Ammonia Limit Values in Surface Waters

4.1. Existing EU criteria for supporting freshwater fish

In European Commission designated waters, the water quality limits for the protection of freshwater fish¹² are 0.021 and 0.78 mg N/l, for unionised ammonia and total ammonia, respectively. These are mandatory values expressed as 95th percentiles. Lower “guide” values have been set at 0.005 and 0.04 mg N/l, respectively.

The Freshwater Fish Directive (FFD) also contains a provision for Member States to set a higher quality criterion for total ammonia than the 0.78 mg N/l specified, under the following conditions:

- Where the water is of low temperature or where nitrification is reduced, or
- Where the competent authority can prove that there are no harmful consequences for the balanced development of the fish population.

This clause in the Directive offering the possibility of derogations from the stated criteria is a potentially important issue to which we will return later.

4.2. Proposed UK WFD limits

As part of the implementation of the WFD, Member States have to define criteria for ammonia that are consistent with the achievement of good ecological status in surface waters. UK regulators have undertaken an assessment of ammonia by toxicological and field-based methods.

a) Toxicologically based proposals

This assessment was proposed by UKTAG, based on the EU recommended toxicologically-based approach¹³ to the establishment of EQSs.

¹² Directive 2006/44/EC On the quality of waters needing protection or improvement in order to support fish life – the Freshwater Fish Directive (FFD)

¹³

[http://ecb.jrc.ec.europa.eu/DOCUMENTS/Practical_Course_on_Environmental_Quality_Standards/01_TGD_%20EQS%20v.%208.0%20for%20formatting%20\(12%20May%202011\).pdf](http://ecb.jrc.ec.europa.eu/DOCUMENTS/Practical_Course_on_Environmental_Quality_Standards/01_TGD_%20EQS%20v.%208.0%20for%20formatting%20(12%20May%202011).pdf) accessed 02062011

Table 4-1 UKTAG proposed standards based on toxicology

Receiving medium/exposure scenario	Proposed PNEC* (mg N/l as un-ionised ammonia)
Freshwater/long-term	0.0011
Freshwater/short-term	0.0068
Saltwater/long-term	0.00066
Saltwater/short-term	0.0057

*The PNEC (predicted no effect concentration) is not a standard, but it is often used as the basis for a water quality standard. The usual process is to set a standard equal to the PNEC divided by an “assessment factor”; a safety factor, which is graded inversely with respect to the reliability of the data.

Assessment factors vary from approximately 1000 for very weak data to as low as 2 for highly reliable data for a full range of possible target organisms using a species sensitivity distribution. The point to note here is that the EQS is almost always numerically lower than the PNEC. Thus the very low PNEC values listed above pose great difficulties in terms of potential compliance and of compliance monitoring. The assessment factor for both the above PNEC values for freshwater was 10 times and for saltwater 100 times (long-term) and 10 times (short term).

b) Ecologically based standards

It is understood that UKTAG recognised this difficulty with the toxicology based proposals and concluded that the standard EU process for EQS setting (having been developed for man-made organic chemicals) was not necessarily suitable for naturally occurring substance such as ammonia. Hence an alternative solution was sought. UKTAG has therefore proposed (at a later date and presumably with correspondingly greater endorsement) a draft water quality standard based on ecological considerations.

Table 4-2 UKTAG proposed standards based on ecology (mg N/l total ammonia)

Type of water body	Standard to be consistent with high ecological status	Standard to be consistent with good ecological status	Standard to be consistent with moderate ecological status	Standard to be consistent with poor ecological status
Upland low alkalinity	0.2	0.3	0.75	1.1
Lowland high alkalinity	0.3	0.6	1.1	2.1

Standards are to be interpreted as 90 percentile values

Upland low alkalinity waters are those at an elevation of greater than 80m with alkalinity less 200 mg CaCO₃/l or those at any elevation having an alkalinity of less than 50 mg CaCO₃/l

Lowland waters high alkalinity waters are those below 80m having an alkalinity of greater than 50 mg CaCO₃/l or those at any elevation with alkalinity of greater than 200 mg CaCO₃/l

The key distinction for assigning criteria for compliance with the WFD objective is to meet the standards for good ecological status highlighted in the above table.

4.2.1. Different approaches to standard setting

Toxicological

Water quality criteria based on toxicological tests have the great strength that they are carried out under laboratory conditions whereby all relevant factors can be controlled, leading to reasonably clear and precise measures of toxicity to the test organism or organisms.

The main weakness of the approach is the mirror of its strength – realism is to some extent sacrificed in order to achieve the control exercised in laboratory tests. Furthermore, it is not possible to test a given organism across a range of environmental conditions that go beyond its natural tolerance. For example, a fish might not be able to undergo a test at water pH values in the range 5 – 8.5 because extremes of pH have the potential to cause adverse effects independently of any toxicant. Finally, the validity of laboratory tests might be questioned on the grounds a) that laboratory organisms might not represent their counterparts in the environment – for example they might not have the same inbred tolerance to pollution gained by acclimation or b) the form of the pollutant might not be the same in the laboratory test as in the wider environment.

Despite these drawbacks, water quality standard setting on the basis of toxicological tests are the norm. They have been the recommended approach in the UK and elsewhere for the last 50 years. However, more recently as quality criteria have been proposed for an ever wider range of substances, some clear shortcomings of the purely toxicological approach have been recognised. Amongst these is the issue of assessment (safety) factors for naturally occurring substances such as metals and ammonia. In these cases the existence of an ambient concentration has posed problems. When assessment factors (see above) are applied to account for shortage of information about all possible target organisms, the resulting quality criterion can be well below the background or ambient level. This leads the approach into disrepute – how can the criterion be set at a value below the levels at which aquatic life evidently thrives? Consequently alternative, more “realistic” approaches have been sought.

Ecological – quality standards based on field data

The ecological approach can be summarised as one that relies on empirical evidence gathered in the form of field data. It asks “what concentration of ammonia is consistent with observed good ecological status?” Its strength is that it is a practical demonstration that the expected range of organisms flourish to the expected extent even in the presence of X mg of ammonia per litre. This is a very persuasive argument and might appear difficult to challenge; except with respect to the details. These are many, including:

- How to define what is a “good” ecological condition (as distinct from “moderate” or “high”);
- How to ensure that definitions of “good” are comparable/transferrable amongst different environments or different regions (eg of Europe);
- How to cope with uncertainty inherent in the classification of status and in the monitoring of the substance of interest;
- How to deal with confounding factors such as the presence of other pollutants, and factors relating to hydrology and hydromorphology.

A great deal of effort has gone into resolving these questions. With respect to ammonia, many key issues are discussed in the UKTAG briefing note on the methodology used to derive the proposed

standards for ammonia (and dissolved oxygen) – see Guthrie *et al*¹⁴. Within the context of setting a water quality standard in this way there is much merit in the approach adopted. However, two main criticisms of the ecological approach remain:

1. The ecological approach shows the ammonia concentrations that are observed to be consistent with good ecological status; it does not clearly define the limiting concentration of ammonia that is associated with the first signs of ecological damage. For this reason it might tend to be overly protective. Neither does the approach address temporal effects in the same way as standards based on short- and long-term toxicological tests;
2. This approach has moved away from expressing the standard in terms of the form of ammonia that is acknowledged to be by far the most critical one in terms of toxicity. Hence by controlling total ammonia, rather than the un-ionised form, it loses the focus on what matters most. As a result the link with temperature and pH value is lost. For the great majority of water types this lead to somewhat more stringent water quality criteria than those prevailing under the FFD, though for some unusually warm waters of pH value greater than 8.0 it is less protective (see below).

4.3. Consequences – how do the current FFD and proposed new quality criteria compare?

For the vast majority of water bodies - which are classifiable as lowland and of high alkalinity (and which no derogations apply) the comparison is between a total ammoniacal nitrogen of 0.6 mg N/l as a 90%ile and the Freshwater Fish Directive (FFD) un-ionised ammonia standard of 0.021 mg (N)/l or total ammonia limit of 0.78 mg N/l as a 95%ile. The question is how do these compare? Firstly, we might consider the face value comparison – ignoring the difference between the 90%ile of the former and the 95%ile of the latter. Under this assumption the answer depends on the pH and temperature of the water concerned. If, for the sake of simplicity, we ignore the effect of ionic strength as insignificant and we assume no derogations from the total ammonia limit, Table 4.3 illustrates the position.

The figures in the table represent the concentration of total ammonia as mg N/l that corresponds to just meeting either the unionised FFD standard of 0.021 mg N/l or the 0.78 mg N/l total values – whichever is the more stringent. The colour coding shows the points at which the FFD limiting value exceeds the proposed new standard water of 0.6 N/l, (corresponding to the good / moderate boundary for lowland waters). The new standard is slightly more stringent than the current FFD values for all waters of pH value less than 8 and temperature lower than 18 °C. The estimated change in the incidence of failure to comply with a status of “good or better” in England is reported by UKTAG to be from 14.6% to 17.3%. This does not seem like a substantial change, though it is not possible to fully evaluate the consequences on data presented. If, for example, the less than good sites represent waters with relatively high ammonia levels it is possible that the improvement (and investment) required in order to meet the new standard will be substantially greater than that needed to achieve compliance with the old one.

¹⁴ Guthrie R, Duncan W and Owen R (Jan 2006). The development of oxygenation condition and ammonia regulatory values in UK rivers. Paper prepared for UKTAG Rivers Task Team. <http://www.wfduk.org/LibraryPublicDocs/DevtofOxyConandAmmonRegValinUKRivers>

With respect to upland waters the proposed value of 0.3 mg N/l represents a marked tightening of quality criteria.

Table 4-3 Current (FFD) standards for total ammonia at different pH and temperature

pH	7.5	7.6	7.7	7.8	7.9	8	8.1	8.2
Temperature deg C								
0	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
1	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
2	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
3	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
4	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
5	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
6	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
7	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
8	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
9	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.78
10	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.73
11	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.68
12	0.78	0.78	0.78	0.78	0.78	0.78	0.78	0.63
13	0.78	0.78	0.78	0.78	0.78	0.78	0.73	0.59
14	0.78	0.78	0.78	0.78	0.78	0.78	0.68	0.54
15	0.78	0.78	0.78	0.78	0.78	0.78	0.63	0.51
16	0.78	0.78	0.78	0.78	0.78	0.73	0.59	0.47
17	0.78	0.78	0.78	0.78	0.78	0.68	0.55	0.44
18	0.78	0.78	0.78	0.78	0.78	0.63	0.51	0.41
19	0.78	0.78	0.78	0.78	0.74	0.59	0.47	0.38
20	0.78	0.78	0.78	0.78	0.69	0.55	0.44	0.35
21	0.78	0.78	0.78	0.78	0.64	0.51	0.41	0.33
22	0.78	0.78	0.78	0.75	0.60	0.48	0.38	0.31
23	0.78	0.78	0.78	0.69	0.56	0.45	0.36	0.29
24	0.78	0.78	0.78	0.65	0.52	0.42	0.34	0.27
25	0.78	0.78	0.76	0.61	0.49	0.39	0.31	0.25
26	0.78	0.78	0.71	0.57	0.45	0.36	0.29	0.24
27	0.78	0.78	0.66	0.53	0.42	0.34	0.28	0.22
28	0.78	0.77	0.62	0.49	0.40	0.32	0.26	0.21
29	0.78	0.72	0.58	0.46	0.37	0.30	0.24	0.20
30	0.78	0.67	0.54	0.43	0.35	0.28	0.23	0.19

Table shows prevailing ammonia standard in mg N/l. Note: for all waters of pH lower than 7.5 the 0.78 mg N/l FFD limit for total ammonia would apply – **provided no derogations apply**

Dark blue – FFD limit for total ammonia applies – this is less stringent than the proposed value of 0.6 mgN/l

Light blue - FFD limit for un-ionised ammonia applies – this is very slightly less stringent than proposed value of 0.6 mg N/l

Green - FFD limit for un-ionised ammonia applies – this is more stringent than proposed value of 0.6 mg N/l

Table 4.4 shows the same comparison but with an assumed derogation of the ammonia limit to a value of 1.3 mg N/l – the value applied in current Environment Agency policy. This revised limit represents a reasonable option that might be argued as justifiable after consideration of the standards set in other countries, as discussed in Section 5 below.

Table 4-4 Current (FFD) standards for total ammonia at different pH and temperature (with derogation of total ammonia to 2 mg N/l)

pH	7	7.1	7.2	7.3	7.4	7.5	7.6	7.7	7.8	7.9	8	8.1	8.2
Temperature deg C													
0	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30
1	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30
2	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30
3	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.27
4	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.17
5	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.08
6	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.25	1.00
7	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.16	0.92
8	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.07	0.86
9	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.24	0.99	0.79
10	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.15	0.92	0.73
11	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.07	0.85	0.68
12	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.24	0.99	0.79	0.63
13	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.15	0.92	0.73	0.59
14	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.06	0.85	0.68	0.54
15	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.24	0.99	0.79	0.63	0.51
16	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.15	0.92	0.73	0.59	0.47
17	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.07	0.85	0.68	0.55	0.44
18	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.24	0.99	0.79	0.63	0.51	0.41
19	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.16	0.92	0.74	0.59	0.47	0.38
20	1.30	1.30	1.30	1.30	1.30	1.30	1.30	1.08	0.86	0.69	0.55	0.44	0.35
21	1.30	1.30	1.30	1.30	1.30	1.30	1.26	1.00	0.80	0.64	0.51	0.41	0.33
22	1.30	1.30	1.30	1.30	1.30	1.30	1.17	0.93	0.75	0.60	0.48	0.38	0.31
23	1.30	1.30	1.30	1.30	1.30	1.30	1.09	0.87	0.69	0.56	0.45	0.36	0.29
24	1.30	1.30	1.30	1.30	1.30	1.27	1.02	0.81	0.65	0.52	0.42	0.34	0.27
25	1.30	1.30	1.30	1.30	1.30	1.19	0.95	0.76	0.61	0.49	0.39	0.31	0.25
26	1.30	1.30	1.30	1.30	1.30	1.11	0.88	0.71	0.57	0.45	0.36	0.29	0.24
27	1.30	1.30	1.30	1.30	1.30	1.03	0.83	0.66	0.53	0.42	0.34	0.28	0.22
28	1.30	1.30	1.30	1.30	1.21	0.97	0.77	0.62	0.49	0.40	0.32	0.26	0.21
29	1.30	1.30	1.30	1.30	1.13	0.90	0.72	0.58	0.46	0.37	0.30	0.24	0.20
30	1.30	1.30	1.30	1.30	1.06	0.84	0.67	0.54	0.43	0.35	0.28	0.23	0.19

Dark blue – Relaxed FFD limit of 1.3 mg N/l for total ammonia applies – this is less stringent than the proposed value of 0.6 mg N/l

Light blue - FFD limit for un-ionised ammonia applies – this is less stringent than proposed value of 0.6 mg N/l

Green - FFD limit for un-ionised ammonia applies – this is more stringent than proposed value of 0.6 mg N/l.

Given the prevalence of light and dark blue values in the above tables it appears that the new criterion of 0.6 mg N/l is generally more stringent. However, we still need to consider the fact that the new standard is a 90%ile and the old FFD limit is set as a 95%ile. This complicates matters because the issue of stringency is now shown to depend on the frequency distribution of ammonia concentrations. In short – is the 0.6 as a 90%ile more stringent than a 0.78 as a 95%ile?

Figure 4.1 illustrates the relationship between the frequency distributions of data that comply with each criterion – for three different frequency distributions of ammonia, ie for distributions of different skewness. A logNormal model is chosen because this is the accepted form of data distribution that is used universally for consenting purposes. The green curve shows a data distribution that just complies with the new 0.6 mg N/l 90%ile value; the blues curve shows a distribution of an identical basic shape that just complies with a 0.78 mg N/l 95%ile. By looking at the positions of these curves on the “x” axis it is possible to compare the levels of ammonia that represent marginal compliance.

The uppermost diagram shows a relatively symmetrical distribution. In this case the tolerable ammonia level for the FFD regime, is higher up the concentration “x” axis is than that of the green new regime. This means that the new regime is more stringent in that it tolerates lower ammonia levels. In other words the tolerable ammonia load under the new regime is less. The middle diagram corresponds to a data distribution that is more typical of riverine ammonia levels in that it shows a moderate degree of skewness. In this case, the two regimes are to all intents and purposes identical – they are just defined in a different way. Finally, the bottom diagram shows the case for an extremely skewed distribution. Here the curves are still very similar but mathematically it can be shown that the new regime is very slightly less stringent than the old.

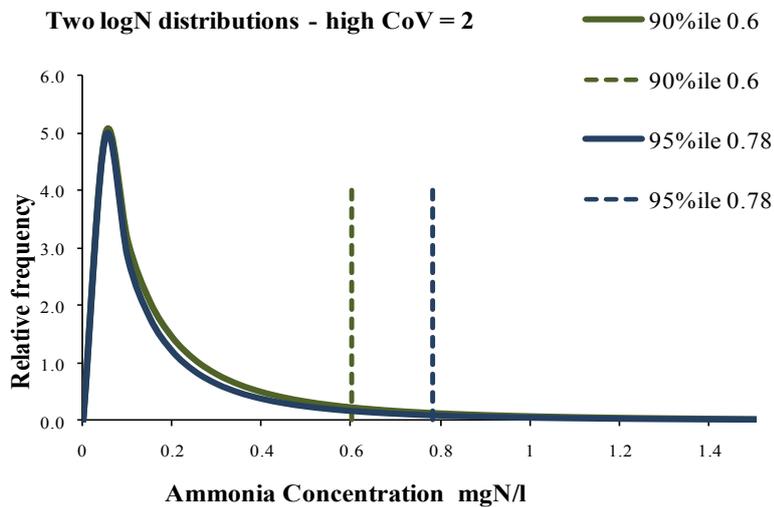
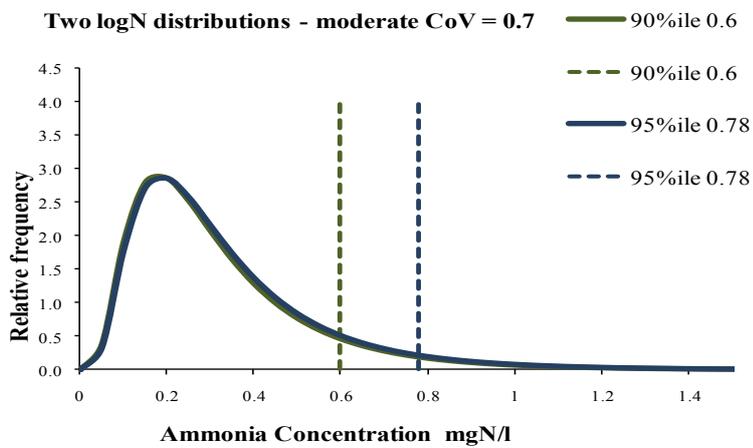
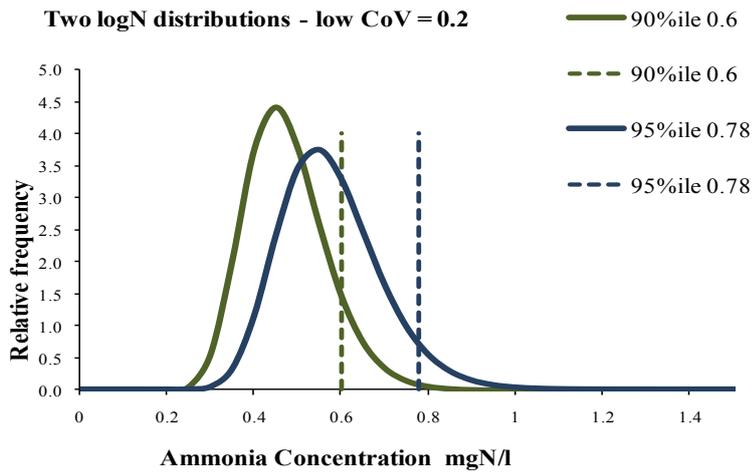


Figure 4-1 Illustration of the way different percentiles compare for FFD and UKTAG regimes

“CoV” is coefficient of variation – the standard deviation of the logNormal data divided by the mean

Conclusions - Current and new UK limit values

In summary, we can see that:

1. Taking the FFD limits as stated, for all waters of temperature below 18 C and of pH less than 8 (ie most waters) the FFD limiting water quality is defined by the value set for total ammonia. Justifiable derogations to higher limits than 0.78 mg N/l for total ammonia represent regulation that strikes a proper balance between protecting aquatic life and not requiring unnecessary improvements in water quality. For other waters, of higher temperature or pH value, the current limit for un-ionised ammonia applies.
2. Where it can be shown that a derogation for total ammonia is justifiable (in the example above a value for total ammonia of 1.3 mg N/l was used), the limit for un-ionised ammonia becomes more important, applying to a greater extent to waters in the more typical range of pH and temperature values.
3. The proposed UK TAG limit of 0.6 mg N/l is effectively the same as the corresponding FFD value; the apparently lower numerical standard, being offset by its definition as a 90%ile in comparison with the 95%ile FFD value. This field based standard has the potential to be overprotective owing to its derivation as a “no effect” standard and because it does not recognise the disparity in toxicity between the two forms of ammonia. Compared with a derogated value of 1.3 mg N/l the new limit represents a considerable tightening of regulation.

4.4. Standards in other countries

4.4.1. US EPA

The US EPA has recently updated the numeric freshwater acute and chronic aquatic life criteria¹⁵ for ammonia to ensure they are protective of freshwater mussels (classified in the family *Unionidae*)¹⁶. This was in response to findings that ammonia is particularly toxic to *unionids* and the fact that many such species are now considered to be endangered and requiring protection. The strategy was adopted of having a two tier system according to whether or not freshwater mussels are present. A fallback position for the protection of fish fry (in the absence of mussels) is also provided.

15 EPA-822-D-09-001 Environmental Protection Agency 4304T December 2009. Update Aquatic Life Ambient Water Quality Criteria For Ammonia – Freshwater.

http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/pollutants/ammonia/upload/2009_12_23_criteria_ammonia_2009update.pdf accessed 2/6/2011

¹⁶ In Great Britain four species of unionids occur; *Unio tumidus* and *Unio pictorum* in the rivers of Southern England. The pearl mussel, *Unio margaritifera*, occurs in the mountain streams of Scotland and Ireland. *Anodonta cygnea* - the swan mussel is found in large ponds.

Table 4-5 Comparison of EU FFD criteria for ammonia with 2009 values published by the US Environmental Protection Agency

	Temperature °C: (refers to pH 8.0)	0	25
EU FFD criteria (protection of freshwater fish) for total ammonia expressed as mg N/l	Total ammonia mg N/l	0.78	0.39
US EPA criteria (protection of freshwater fish)	Total ammonia mg N/l	2.32	1.84
US EPA criteria (protection of freshwater fish and <i>unionid</i> mussels)	Total ammonia mg N/l	0.82	0.26

This illustration applies to water of pH 8.0, other criteria apply at other pH value but the basis for change in response to different value if pH and temperature is similar see Figure 4.1.

These standards are expressed as mg N/l but using pH and temperature criteria to take account of the proportions of un-ionised ammonia (effectively the “mussels absent” criteria) are less stringent than the FFD values in table above. The US criteria for when mussels are present are comparable.

4.4.2. Environment Canada

The guideline for long-term control of ammonia in effluents is essentially the same as that of the FFD:

“An objective for ammonia releases to surface water is to achieve and maintain a concentration of ammonia in the effluent that will ensure the protection of freshwater life. Wastewater systems should not release ammonia in quantities or concentrations resulting in a concentration of unionized ammonia greater than 0.019 mg N/L in the aquatic environment.”¹⁷

4.4.3. Australian and New Zealand (ANZECC guidelines)¹⁸

The ANZECC water quality guidelines were published in 2000 and are currently under review. The approach adopted by regulators in Australia and New Zealand is to formulate guideline concentrations of different toxicants that are intended to be used with a tiered approach to water quality management. This begins by defining management aims and taking account of local issues by following a decision tree to assess risk at different levels.

The rationale and technical background to the derivation of the pH dependent standards is identical to that applied in Europe and the US, except that toxicity to antipodean species has been considered and no temperature dependence is explicitly taken into account. The ANZECC guideline values are shown on the graph below in relation to other national quality criteria. The guidance document notes that:

¹⁷ <http://www.gazette.gc.ca/archives/p1/2004/2004-12-04/html/notice-avis-eng.html#i2> accessed 06/06/2011

¹⁸ Australian and New Zealand guidelines for fresh and marine water quality 2000 ISBN 09578245 0 5 (set) <http://www.mfe.govt.nz/publications/water/anzecc-water-quality-guide-02/anzecc-water-quality-guide-02-pdfs.html> accessed 06/06/2011

“The 95% figure provided) is considered sufficiently protective of most slightly-moderately disturbed systems. However, this figure may not be sufficiently protective of the freshwater clam *Sphaerium novaezelandiae* and related species. If these are significant at a site, site-specific studies or a higher protection level may be warranted..... The freshwater trigger value could be divided by a factor of 2 to protect this species¹⁹”.

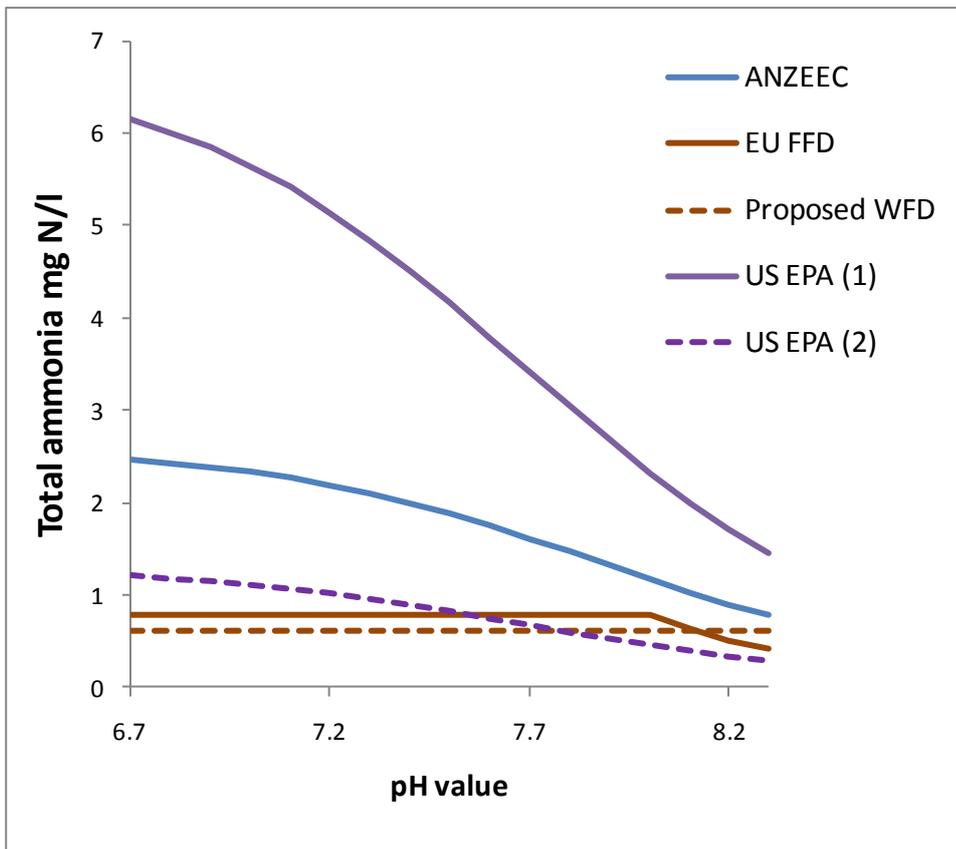


Figure 4-2 Comparison of ammonia quality criteria

Notes:

ANZEEC values under review (2011) – option to reduce limits to protect sensitive species such freshwater clams

US EPA limits shown (1) – to protect fish fry and (2) - to protect freshwater mussels (revised 2009)

Where temperature is accounted for the diagram refers to 15 °C

The proposed WFD criterion shown is the one suggested for lowland waters

¹⁹ Hickey CW, Martin ML. Arch Environ Contam Toxicol. 1999 Jan; 36(1):38-46. Chronic toxicity of ammonia to the freshwater bivalve *Sphaerium novaezelandiae*.

5. Implications for the Water Industry in the UK

Recent monitoring of effluent quality as part of the UKWIR Chemical Investigations Programme (CIP) has provided the following data. For the 34 Severn Trent WwTWs that are involved in the relevant part of the CIP the profile of average ammonia concentrations is as shown in Figure 5.1.

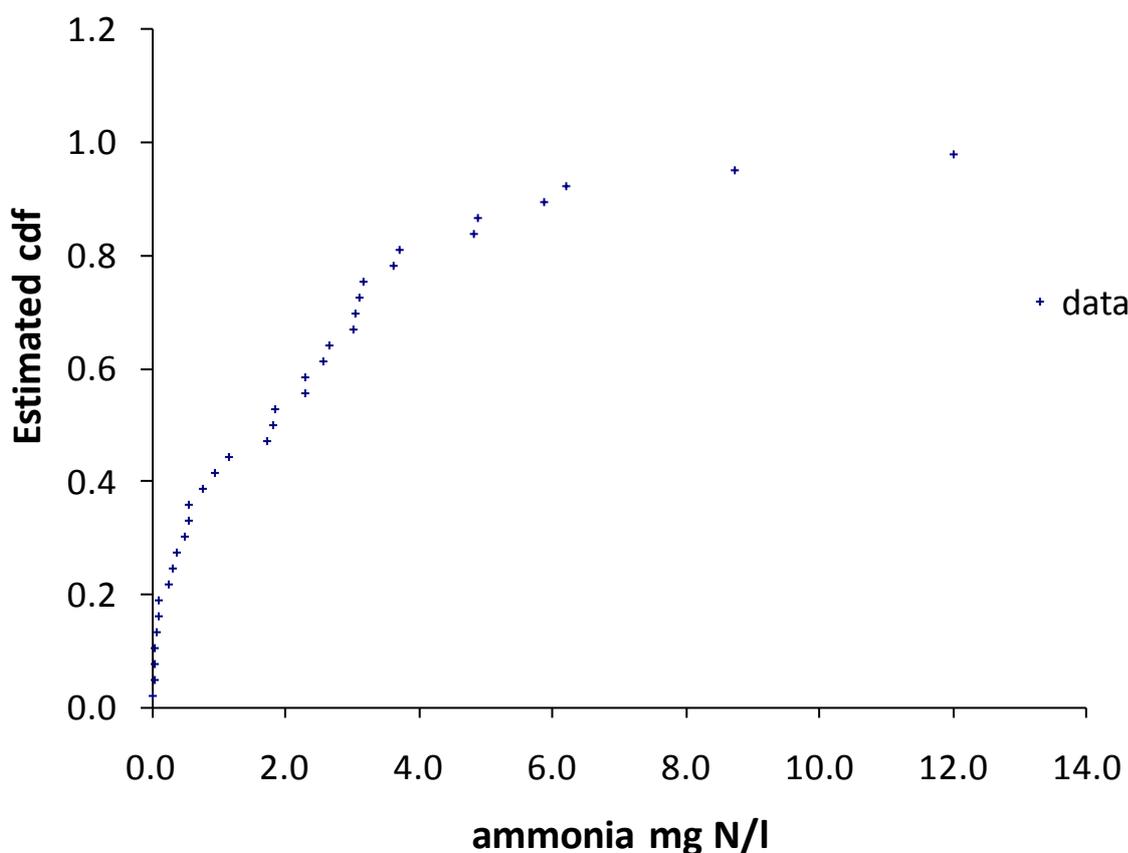


Figure 5-1 Profile of average ammonia concentrations in WwTW effluents (Severn Trent)

The cumulative distribution function (cdf) represents the proportion of average effluent ammonia concentrations that are less than the corresponding X axis value. Hence half the values are less than 2 mg N/l and 90% are less than approximately 6 mg N/l.

As an illustration of how these effluents compare with the effluent quality required to meet a 90%ile downstream concentration of 0.6mg N/l it is possible to use the Environment Agency river quality planning tool to model the average effluent quality required to meet the quality limit value - for different assumed levels of effluent dilution. The results of this exercise are shown in the table below.

Table 5-1 Average effluent concentrations needed to achieve 0.6 mg/l as 90%ile

Effluent dilution factor = flow of river / flow of effluent	Average effluent concentration to just meet an in-river 90%ile value of 0.6 mg N/l
3x	1.7
4x	2.1
5x	2.5
10x	4.5

Assumptions

Coefficients of variation for flow (river and effluent) = 0.4

Coefficient of variation for effluent quality = 0.7

Correlation between river and effluent flow = 0.6

If we assume that effluents receive only a modest discharge dilution of three fold (for which the above table shows a required average effluent concentration of 1.7 mg/l), we can refer to Figure 5.1 to see what proportion of works would be at risk of not achieving the required average concentration. This involves looking up on the diagram for an X axis value of 1.7, to read from the curve a Y axis value of approximately 0.5 – ie nearly half of works have a mean ammonia concentration of greater than 1.7) Of course this analysis accounting for dilution would need to be done on a case by case basis but this exercise suggests that reduction of ammonia concentrations would be required at a significant number of works in order to meet the limit value, without derogations. Even without considering the costs of doing this, it is clear that this would represent a considerable commitment of resources.

6. Water quality criteria for ammonia - conclusions

- 1) All credible water quality criteria for ammonia are based on the appreciation that the un-ionised form is the more toxic, but that some form of protection with respect to ionised ammonia is needed at pH values less than 8. This understanding is incorporated into the current criteria given in the Freshwater Fish Directive which specifies both a value of un-ionised ammonia and for total ammonia. The former value is more stringent (0.021 mg N/l) in comparison with the latter (0.78 mg N/l), reflecting the difference in toxicity. The relative importance of the current Freshwater Fish Directive total and un-ionised limits for ammonia depends crucially on the possibility of derogations to the total ammonia limit. Without derogations, the total limit applies in most cases. Where derogations for total ammonia can be put in place, the importance of the un-ionised form in determining the overall limits increases.
- 2) The current guidelines given in the Freshwater Fish Directive are stringent in comparison with standards set by the US EPA, Environment Canada and regulators in Australia and New Zealand. The current limit for un-ionised ammonia is accepted widely. For total ammonia there is less agreement about an appropriate level, but both the FFD and the proposed UKTAG value appear stringent in relation to limits applied elsewhere. However, recent evidence relating to the toxicity of ammonia to sensitive species such as freshwater mussels is leading to these regulators giving consideration to setting more demanding standards, at least at a local level.
- 3) This approach of adapting the quality criterion to meet local water management objectives reflects a move by these regulators away from the imposition of uniform standards, without regard to the potential benefits that might be realised if such standards are met. By contrast, current UKTAG proposals involve the establishment of a new essentially fixed criterion, (fixed, other than with respect to “typology” which is itself unalterable) to be established for Water Framework Directive compliance purposes.
- 4) For waters of “lowland” typology (that constitute the great majority of receiving waters in England and Wales) the proposed criterion of 0.6 mg N/l is broadly comparable (allowing for the difference in percentile level set) with the existing limits set in the Freshwater Fish Directive. However any loss of the ability to derogate the limit for total ammonia should be considered carefully because it could lead to a step change in stringency. For the small proportion of waters in the “upland” class, a proposed criterion of 0.3 mg N/l is a marked reduction in limit value (given that these waters are of predominantly low pH and would previously have attracted an FFD limit of 0.78 mg N/l limit for total ammonia).
- 5) The change in regulation should be examined to provide reassurance that the proposed change will not have an effect on costs that will disproportionate to its apparently small effect on compliance. If, for example, the less than good sites represent waters with relatively high ammonia levels it might be that the improvement (and investment) required in order to meet the new more stringent standard will be substantially greater than that needed to achieve compliance currently. Furthermore, the adoption of a fixed, pH independent limit is at odds with regulatory practice across the globe; it may be somewhat overprotective at low pH and underprotective at high pH values. For waters that are by

British standards of high temperature and pH, the Freshwater Fish Directive calls for considerably better control of ammonia levels than the proposed new criterion.

- 6) In summary, it is recommended that the regulatory regime for ammonia should:
- Retain the current provision for derogations against the current FFD (or new UKTAG) limit for total ammonia; and,
 - Retain the differentiation in limit values between total and un-ionised ammonia.

Severn Trent Water Ltd
Briefing note on water quality criteria
for iron

ATKINS

Executive summary

The implementation of the Water Framework Directive (WFD) means that the existing Environmental Quality Standard (EQS) is under review and may be revised. Any revision of the EQS downwards (as has been the case for most other substances reviewed under the WFD) could lead to tightening of permits to discharge to surface waters, which in turn could have a significant impact on the Water Industry. There are few alternatives to iron that are not either more costly or potentially more toxic, so any restrictions on its application have the potential to raise compliance and sustainability issues. UKWIR recently commissioned a thorough review of the impacts of iron dosing in the UK and possible implications of a change in EQS. This lobbying document summarises the findings from that report.

Conclusions and recommendations:

1. The UK seems to be the only country in Europe considering iron as a specific pollutant under the WFD.
2. From the available data, collected and collated from across England, as well as that modelled using SIMCAT, there is no evidence of significant increases in total or dissolved iron concentrations downstream of wastewater treatment works because of dosing iron for phosphorus reduction.
3. The water quality data is supported by the fact that there are no significant ecological changes downstream of WwTW dosing iron; in many cases the number of sensitive ecological species present actually increases. Furthermore, estimated particulate concentrations in WwTW effluents dosing iron display a similar content (ca. 6%) of iron to that in background stream sediments, which suggests that there should be no significant accumulation of particulate iron in sediments downstream of WwTW.
4. The outcome of the modelling exercise suggested that the current dissolved iron EQS can readily be met downstream of works dosing iron for phosphorus reduction. However, reduction of the current EQS below a threshold of approximately 0.5 mg/l dissolved iron or rigid implementation of the 'no deterioration' policy, would lead to compliance becoming increasingly difficult to achieve without additional effluent treatment. This in turn would lead to commensurate rises in costs to customers and increases in carbon emissions. Assuming the retrofitting of sand filters to works currently without them, figures for a single company (Anglian Water, *pers comms*) suggests that the Capex would be £38.3 million with an Opex of £2.9 million, leading to an increase in customers' bills of £6 per household over 5 years.
5. Sand filters appear to be able to provide only modest improvements in effluent iron concentrations. Owing to the presence of colloidal and complexed iron that is not retained by the sand filters, a consistent effluent quality of significantly less than 1 mg/l (total iron) would be unlikely to be achievable.
6. The lack of evidence of downstream impacts, combined with low concentrations of residual total and dissolved iron in effluents calls into question the need for a general policy to permit WwTW discharges. It is therefore recommended that the Agency reconsider the use of widespread iron permitting.
7. Where permits are applied, a change from the maximum discharge limit currently imposed to the use of a percentile and a look up table is recommended, either by use of existing permits as 95 percentile values, or use of the Agency's RQP model to generate appropriate percentile values. The setting of a percentile limit based on current WwTW performance is not recommended as it does not take account of available in-river dilution and does not allow for within WwTW process changes to, for example, increase iron dosing to meet increasingly stringent phosphorus permits.

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1. Background

1.1. Introduction

Iron salts are currently used at approximately 700 WwTW serving over 20 million customers as a coagulant within wastewater treatment to precipitate phosphorus into sludge in order to meet European Directives for the control of phosphorus discharged to the aquatic environment, with the majority of works dosing to achieve phosphorus permits of either 1 or 2 mg/l. Where iron is added during the treatment process a permit to discharge is generally applied by the Environment Agency to ensure that the receiving water Environmental Quality Standard (EQS) is not exceeded, thus ensuring there is no negative impacts on downstream ecology. The current EQS set under the Dangerous Substances Directive (DSD) is 1 mg/l as dissolved iron.

The Water Framework Directive (WFD) is a relatively new European Directive which set very stringent targets for water quality and ecology. As part of its implementation the DSD is being repealed and new EQS are being derived for a large range of potentially toxic substances. Iron is not a particularly toxic metal owing to its instability in water, however, excessively high concentrations can cause toxicity and so has meant an EQS was set under the DSD. Iron may now be included as a WFD Annex VIII Specific Pollutant and a revised EQS set by the Environment Agency, is likely to be published for consultation in 2011. Given that the WFD EQS setting methodology invariably leads to tighter EQS than those set under the DSD, there is risk to the Water Industry of a downward pressure on new and existing iron permits in the future. Furthermore, in many cases phosphorus permits are also being tightened to meet WFD objectives and are likely to become more extensive in the future, which puts pressure on water companies to use more iron, recent estimates (UKWIR, 2010) also suggest that to achieve the phosphorus EQS, many more wastewater treatment works (WwTW) may require some form of phosphorus removal (potentially at least an additional 1,000 WwTW) in an attempt to meet in-river phosphorus EQS.

Any decrease in iron permit values would put significant pressure on the Industry to install extensive tertiary treatment (sand filters) to reduce effluent concentrations or seek alternatives to its use. Concerns regarding cost and increasing carbon emissions have led to the Water Industry via UK Water Industry Research (UKWIR) to review the current permitting policy, the fate and behaviour of iron in receiving waters; impacts on downstream ecology and water quality; costs of additional treatment and to offer options for possible future permitting regimes (UKWIR, 2011)¹. Decisions regarding the regulation of iron in the UK have yet to be made; this lobbying document is designed to assist in discussions between the Water Industry and the Regulators by summarising the key outcomes of the industry's research.

¹ UKWIR (2011) A Review of the Setting of Iron Limits for Wastewater Treatment Works Effluents (11/WW/20/4). <http://www.ukwir.org/ukwirlibrary/93693>

2. Forms of iron in Water

It is important to note that iron is one of the Earth's most abundant elements and is present in the crust at around 5% by mass. In effluents and surface waters iron is chemically unstable in solution and should be present as solid iron hydroxides. Concentrations of the toxic forms of iron, namely the dissolved free metal ion, are therefore very low and in most cases, not measurable. Only in acidic waters in areas rich in iron minerals would appreciable concentrations of dissolved iron be detected, prior to precipitation downstream as an orangey, 'ochrous' deposit. Low concentrations of filterable iron may be detected in the aqueous phase owing to inadequacies in the determination of dissolved iron. An operationally defined particle size cut off of 0.45 μm is used for filtering samples and classifying a metal as 'particulate' or 'dissolved'. Colloidal iron (oxidised iron hydroxides in an intermediate phase between dissolved and precipitation as a solid) passes through the filter and is therefore included in the 'dissolved' fraction; 'filterable iron' would be a more accurate description. Because colloidal species are relatively unreactive, their toxicity is much less than the free metal ion. In some cases its slow oxidation or binding to substances which maintain its solubility in rivers and effluents, such as dissolved organic matter or in the case of effluents, synthetic chelating agents used in detergents, can also mean iron is detectable in the water column.

3. Aquatic toxicity of iron

Owing to its lack of solubility, iron is not an especially toxic metal and has always been viewed in this light by the regulator. It has not been classified as a Priority Substance or Priority Hazardous Substance by the EU, who have left individual Member States to decide whether to include it as a Specific Pollutant or not. Discussion with key Environment Agency staff (*pers comms* P Whitehouse, 9th July, 2011) suggests the UK to be potentially alone in considering iron as a specific pollutant across the EU. The existing DSD EQS is 1 mg/l dissolved metal and exceedances would only be expected in iron-mineral rich areas. Furthermore, its instability in water complicates toxicity testing, as there are often uncertainties over the form of iron to which an organism is being exposed, as well as its concentration. Consequently, the derivation of EQS and permit conditions are complex and generally imprecise. Recent standard setting methodologies have in some cases (see Ammonia Briefing Note) set EQS based on firstly identifying waterbodies of 'good' ecological status then setting a chemical standard based on observed concentrations. However, because iron is unstable in water and will tend towards the solid phase, water concentrations will be low in most types of water (pristine, polluted or otherwise) of any ecological quality and significantly less than any toxicity threshold, which would lead to a very low EQS being set based on observed concentrations.

4. Iron permitting

Current Permits: Current WwTW iron permits are set as total iron and are typically 4 mg/l (Figure 1A), which owing to the complex chemistry of iron rarely reflects available dilution (Figure 1B). This de facto emission limit value has been generated via negotiation and knowledge of effluent quality achievable, fate in receiving water rather than strict adherence to modelling outputs and Environment Agency policy. For compliance purposes, under the DSD all metal permits are in theory set as maximum values, which now is accepted as a statistically flawed approach, with percentile values being preferred for permitting under the WFD.

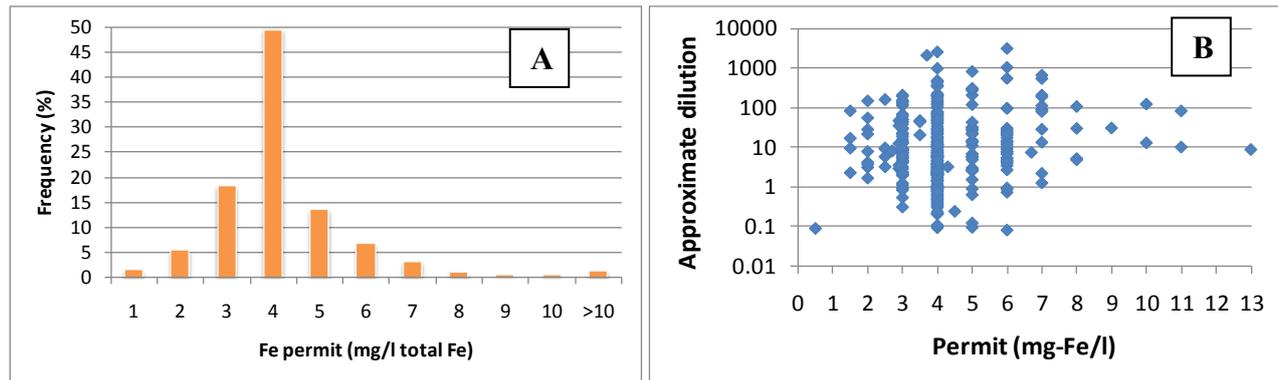


Figure 1 (A) Percentage of permits applied to iron for WwTW based on available data (453 WwTW). (B) Estimated dilution (River: WwTW flow) for WwTW with an iron permit based on available data

Where dosing iron, WwTW effluents exhibit concentrations significantly less than the permitted discharge limit. This is not because of the use of extensive tertiary treatment, to meet a permit expressed as a maximum allowable concentration (MAC), but because the physico-chemical characteristics of iron will result in its precipitation and accumulation in the sludge rather than discharge in the effluent (Figure 2). Dissolved concentrations of iron in dosed effluents are typically 25% of the total values. Exceedances of the permitted MAC in WwTW effluents are ephemeral and associated with the discharge of high concentrations of suspended solids. The instability of iron in solution would make setting permits on 'dissolved' or 'filterable' iron difficult owing to likely changes in form between sampling and receipt by the laboratory, thereby necessitating filtration on-site. /this will need to be considered on the basis of how any new EQS is expressed.

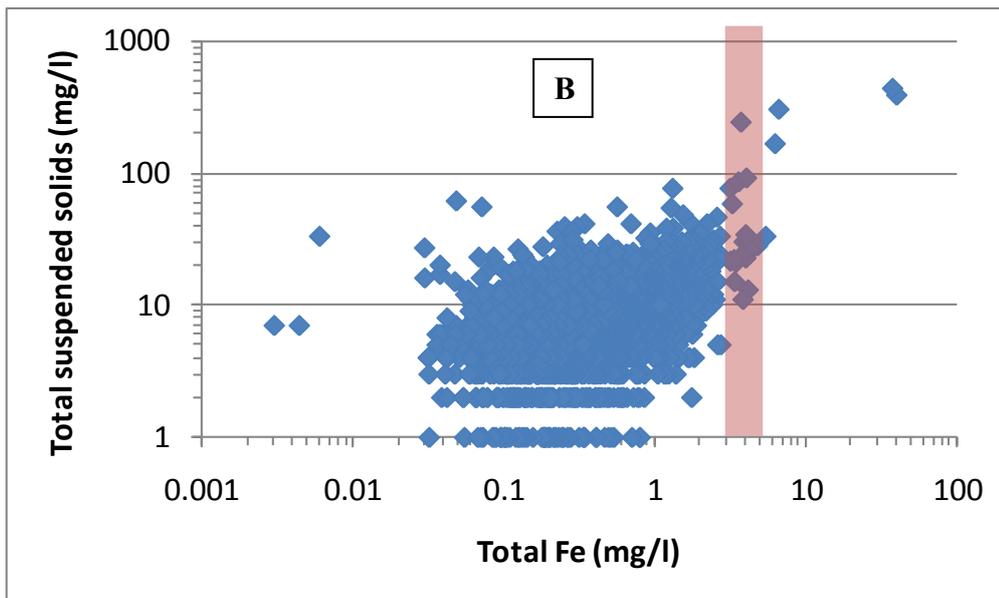
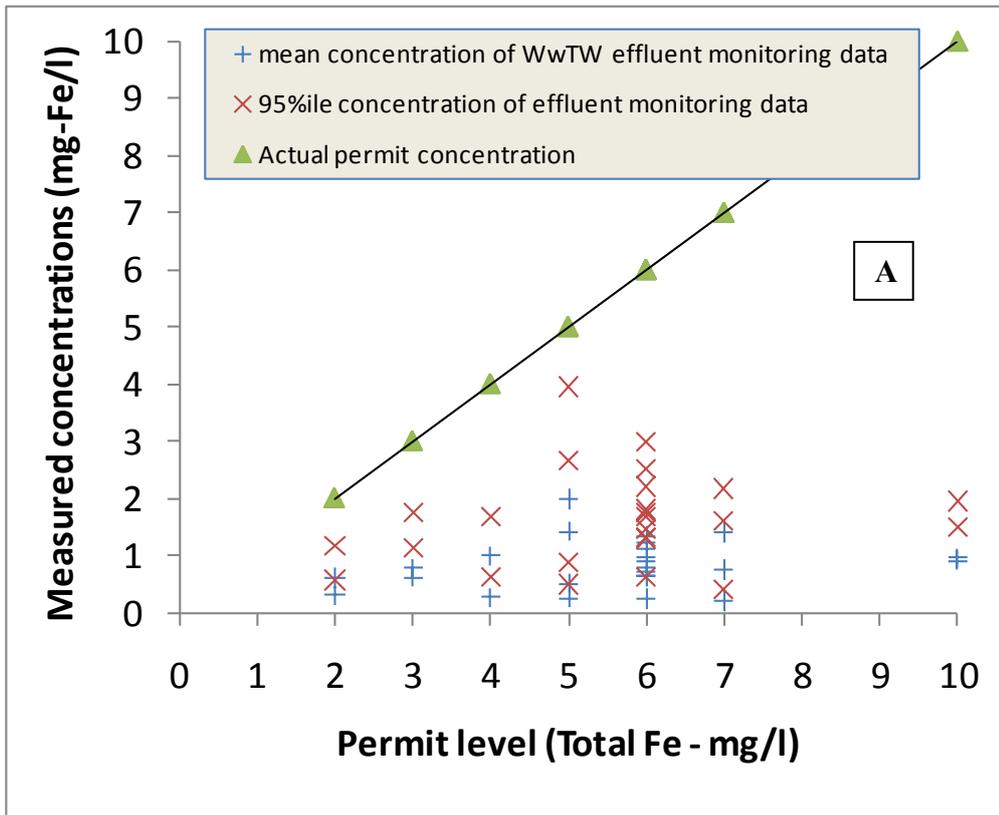


Figure 2 (A) Measured iron levels against permit conditions for one company. (B) Relationship between total iron & suspended solids. Transparent bar denotes typical WwTW permit range of 3 to 5 mg/l

It appears that a minimum average effluent concentration, achievable using conventional secondary treatment, is around 0.6 mg/l total iron (although there is obvious variation around this figure). Sand filters actually only appear to offer a marginal benefit, owing to iron complexation with natural and/or synthetic chelating agents maintaining its presence in solution probably as a colloid. The data suggests that it may therefore not be technically feasible to achieve permits significantly below 1 mg/l (especially as a MAC) owing to complexation and stabilisation in the dissolved phase.

5. Downstream impacts of iron dosing

5.1. Chemistry

Examination of river monitoring data provided by the EA reflects the lack of stability of iron in the water column, with measured concentrations of total or dissolved iron downstream of WwTW pre- and post iron dosing showed no significant change (Figure 3). A case study on the Warwickshire Avon has been used to illustrate this.⁵ Simcat water quality modelling was used to confirm this conclusion and illustrated that even dosing up to the permitted concentration still had little impact on downstream concentrations.

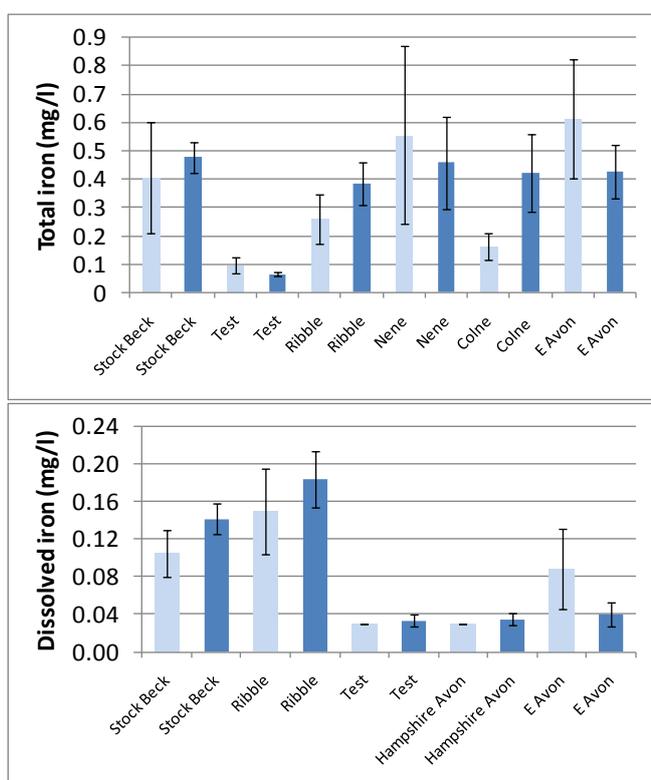


Figure 3 Dissolved and total iron concentrations in rivers immediately downstream of a WwTW pre- and post iron dosing for phosphorus reduction. Dark bars denote post iron dosing, light bars pre dosing

5.2. Ecology

Extensive ecological quality monitoring has been undertaken in the UK over the past 30 years or so. A ranked system (RIVPACS2) has been developed to compare observed and expected biodiversity of invertebrates (bottom-dwelling organisms) sampled by disturbing the river bed and capturing and counting the species present then ranking their ‘quality’ described as BMWP score (e.g. may flies and caddis flies rank highly as they are associated with clean water and chironomid worms low as they are associated with low oxygen, sediment rich poor quality water), the number of taxa present (NTAXA) and the average score per taxa (ASPT).

² [http://www.limnetica.com/Limnetica/Limne10-1/Limnetica-vol10\(1\)-pag15-31.pdf](http://www.limnetica.com/Limnetica/Limne10-1/Limnetica-vol10(1)-pag15-31.pdf)

Examination of ecological data downstream of WwTW pre- and post-dosing failed to show any evidence of systematic degradation in ecological status. Indeed, in more cases than not, improvements were observed (Table 1). Given that the ecological measurements were based on invertebrate surveys on organisms which would most likely to be impacted by ‘smothering’ caused by precipitation of iron onto gills, and included data sets likely to be used to set any revised EQS for iron, the significance of this conclusion should not be under stated.

Table 1 Summary of differences in biological scores following commencement of iron dosing at nine WwTW

WwTW	BMWP	Statistically significant? ¹	NTAXA	Statistically significant?	ASPT	Statistically significant?
Ashford	↑	✓	↑	✓	↑	✓
Coventry	↑	✓	↑	✓	↑	✓
Marlborough	↓	x	↓	✓	↑	✓
Needham Market	↓	x	↓	x	↓	✓
Redditch	↑	x	↑	x	↑	✓
Stowmarket	↓	✓	↓	✓	↓	X

Note: **up arrows** show increased score, **down arrows** the opposite, heavier shading shows statistical significance

¹ t-Test: Two-Sample Assuming Equal Variances (95% confidence interval)

² Insufficient data for statistical assessment.

Further investigation into those sites with a significant decrease in biology scores shows that corresponding upstream sites have also demonstrated a decline in scores (although not significantly) (Figure 4). It is not possible therefore to conclude that iron dosing has caused a negative impact on ecology at these sites.

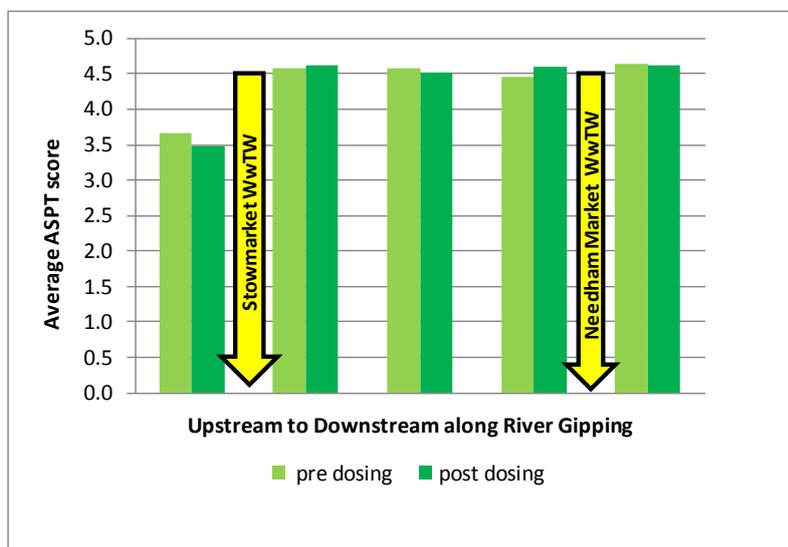


Figure 4 Cumulative impact on ASPT score from iron dosing at Stowmarket & Needham Market WwTWs

5.3. Sediment accumulation

Particulate iron concentrations in 10 WwTW effluent discharges dosing iron were calculated for 1,973 sampling occasions. Average iron content was 6% (Figure 5) which falls in the middle of that observed in UK stream sediments (Figure 6). Available data therefore suggest no impact of WwTW dosing iron on downstream sediment iron concentrations, largely owing to the abundance of iron in UK soils and sediments.

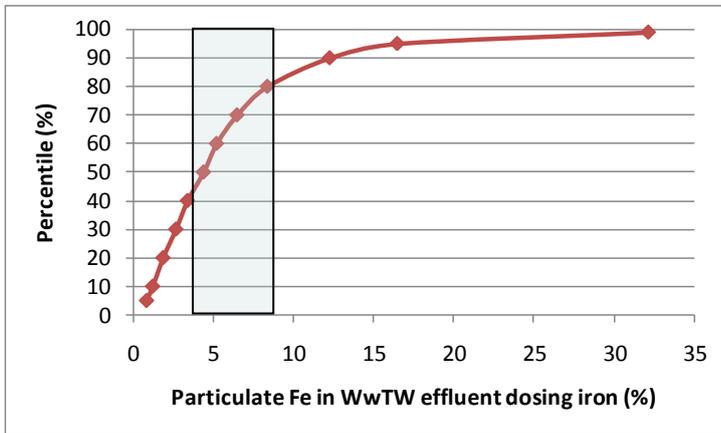


Figure 5 Cumulative distribution of particulate Fe percentages in WwTW effluent (shaded area represents typical composition range of iron in UK sediments)

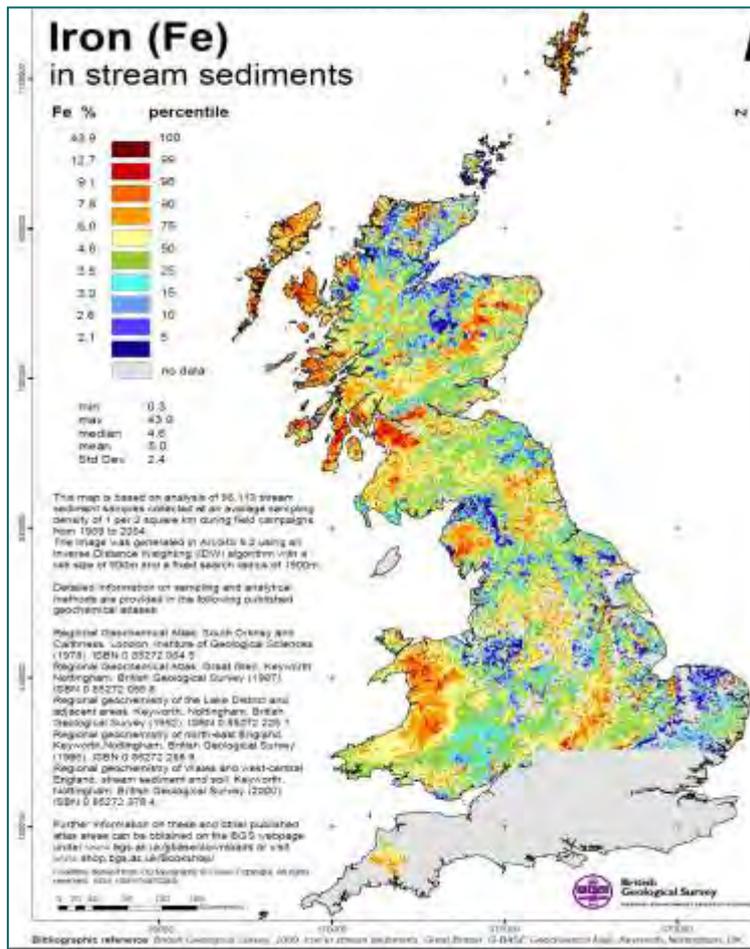


Figure 6 Cumulative distribution of particulate Fe percentages in stream sediment iron content (%), BGS (Note no data is available for the south of England)

6. Possible impacts of EQS changes

Possible Impacts of EQS Changes: The data suggests that a modest reduction in the value of the current EQS (1 mg/l dissolved iron as an annual average) may be accommodated based on existing treatment technologies (potentially down to an EQS of 0.5 mg/l dissolved iron depending on permitting policies). However, any significant reduction in the EQS or strict adherence to a no deterioration policy, would lead to the requirement for additional treatment (with commensurate costs and carbon emissions), which may not be achievable using existing tertiary technologies. Figure 6 shows two scenarios of possible additional costs should in-river iron targets change and show that depending on the assumed current effluent quality, downstream targets of less than 0.5 mg/l dissolved iron lead to an increasing risk of the necessity of tertiary treatment with commensurate rises in costs and carbon emission.

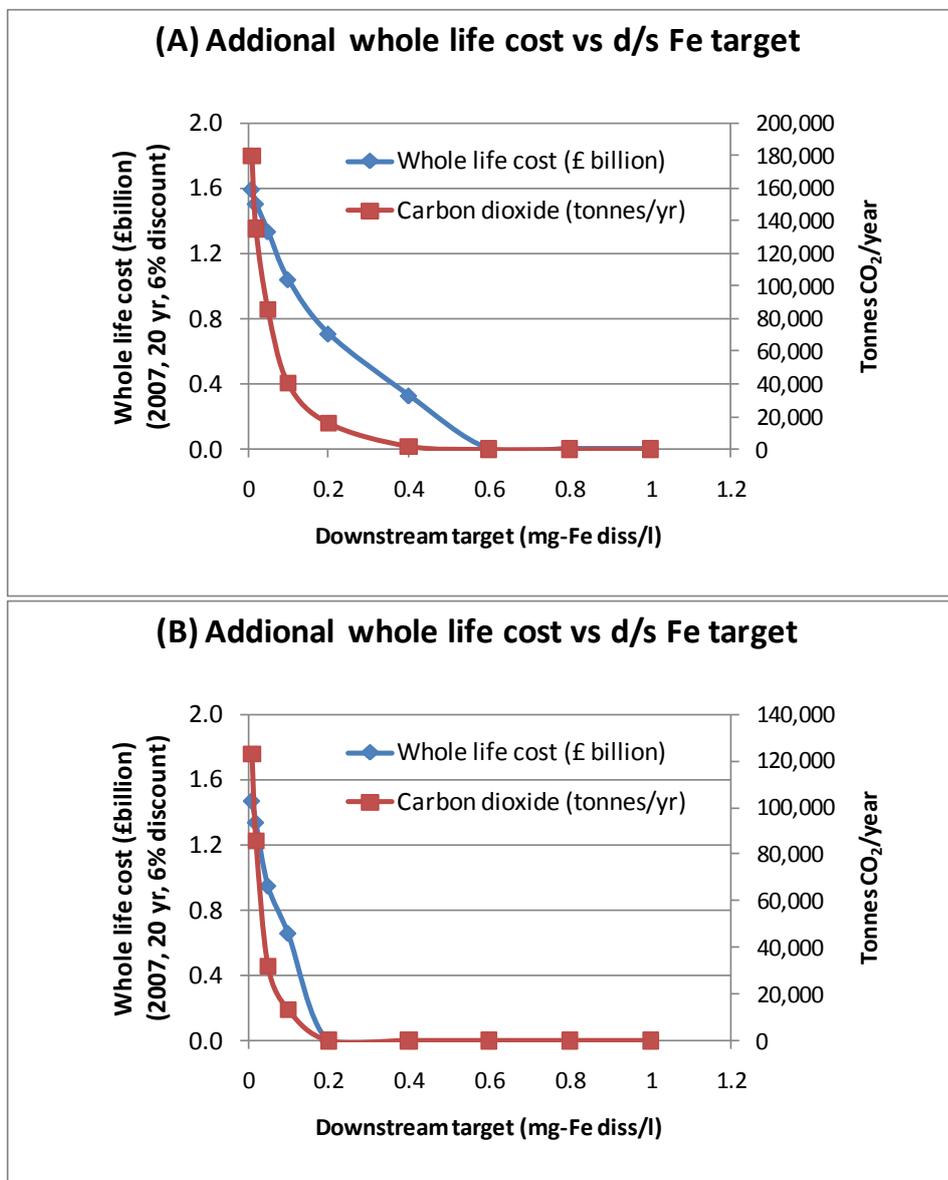


Figure 7 Estimate cost curves for two iron permitting scenarios. (A) WwTW assumed to be discharging 1 mg-Fe diss/l, - roughly equivalent to a 4 mg-Fe tot/l permit. (B) WwTW assumed to be discharging 0.2 mg-Fe diss/l – typically achieved at moment

7. Recommendations for future permitting

The outcome from the review suggests that iron dosing at WwTW for phosphorus removal has no significant impact on downstream sediment, water chemistry or ecology. The data questions the need for iron permitting at all, unless it can be justified, the fact that other EU Member States have not identified iron as a Specific Pollutant further supports this statement. If deemed necessary, the UKWIR research concludes that a 4 mg/l total iron limit would be protective of the existing EQS and should be at the vast majority of WwTW without the need for sand filters. Alternatively, application of the EA's river needs consenting policy taking account of dilution and setting a permit as a 95th percentile appears also to be effectively a cost-neutral option but would lead to significant relaxation of permits at WwTW with high dilutions with some tightening of permits at WwTW with very low dilutions. Setting a 95thile permit based on existing effluent quality would represent a significant tightening compared with current permit values and as it does not take dilution into account, does not reflect any of the impacts (or lack of them) that the WwTW has on the receiving water.

Severn Trent Water Ltd
Briefing note on water quality criteria
for aluminium

ATKINS

Executive summary

1. The widespread occurrence and unique behaviour of aluminium in the Environment makes aluminium something of a special case amongst metals. It is universally present in rivers and streams, both in the water column and in sediments. It occurs as a component in a large range of minerals. It is present in many chemical forms, the great majority of which are essentially non bioavailable and non toxic to aquatic life. However, some risk can be posed by low molecular weight dissolved forms of aluminium that, under specific conditions, can be precipitated and thereby have adverse effects on fish though a mechanism of smothering at the gill surface. The possibility that aluminium ions might exert direct toxicity in, for example, algae has been raised, but is essentially unproven at least for waters of pH greater than 6.5 (ie the vast majority of waters).
2. Risk from aluminium in surface waters depends not only the presence of the potentially reactive forms of the metals, but also on the prevailing environmental conditions in the water of interest. The circumstances under which precipitation of undesirable forms of aluminium occurs are where: a) reactive forms of the metal are present; and, b) where these forms are subject to a pH value (or, more importantly, a change in pH value) across the range 6.5 to 7.5 – the pH range at which insoluble forms of the metal tend to predominate.
3. Regulation of aluminium in waters needs to take account of these factors. Merely applying the conventional approaches used for most other metals leads to unworkable outcomes. Notably, any focus on total metal fails to address risk in a meaningful way because of the presence of aluminium in the clay particles that make up sediments. Even consideration of dissolved aluminium does not address directly the forms of the metal that are might have adverse effects. The Environment Agency has recognised this and for a number of years has been exploring alternative approaches. A number of recent and encouraging developments include:
 - a. The establishment of measurement techniques for reactive forms of aluminium in analytical laboratories has progressed significantly as a result of the inclusion of this determinand in the UKWIR chemical investigations programme. This offers the prospect that regulation might be framed and compliance assessed in environmentally relevant terms;
 - b. Environment Agency research on a range of surface waters across England and Water has not linked the specific presence of aluminium to measurable adverse effects on freshwater ecology. Having this evidence base should make it possible for the regulator to justify a more rational approach to aluminium quality standards. It is understood that the Environment Agency will shortly produce proposals for consultation;
 - c. Monitoring and research seem to indicate that concentrations of aluminium in wastewater treatment works' effluents are low in relation to the level of limit value that is being discussed.
4. This is generally a positive picture from a water industry perspective, but as always, until there are firm proposals to consider, uncertainty remains. Some factors may need to be kept in mind when forthcoming proposals are considered. Specific issues include:
 - a. It is likely that separate limit values will be require for upland, low pH / low alkalinity waters. This is because reactive aluminium is more likely to be present

naturally in such waters (and that adding more via discharges would not be desirable). This may mean that for the limited number of receiving waters more restrictive limits might apply (this caveat is likely to apply to less than 5% of the total number of river reaches in England and Wales, but a greater proportion in Scotland and Northern Ireland);

- b. The level and form of consents and hence of compliance risk will depend on the exact details of any future regulations. Specifically, three factors are likely to be important: the numeric limit chosen; and, the way in which it is specified statistically - (a percentile value is likely to be chosen rather than an average). The details of what is eventually proposed will need to be looked at carefully, ie
 - i. Details of the way in which regulation is translated into consents;
 - ii. The way in which a consent is likely to be expressed;
 - iii. How compliance will be monitored; and
 - iv. The uncertainties associated with any aspect of any proposed approach.

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1. Background

1.1. Introduction

The implementation of the Water Framework Directive (WFD) involves consideration of the regulation of a range of substances in surface waters. The WFD categorises two types of substances for which pollution control measures need to be examined. Substances of greatest concern, the “Priority Substances”, are to be regulated by the publication of quality criteria that apply across the EU. Other classes of substances, the “Specific Pollutants”, are identified for regulation at the Member State level.

UK Regulators have begun addressing specific pollutants, either, by developing new quality criteria, or by examining the need to revise existing quality standards.

Aluminium is included in the list of specific pollutants. It is of particular interest to the Water Industry because aluminium containing chemicals are used widely in purification of water for potable use and to a lesser extent in wastewater treatment. Hence any changes in the way aluminium is regulated might have an impact on the options open to water companies regarding which treatment processes they might implement. The application of aluminium in drinking water treatment is already well controlled such that operations lead to minimal discharges of aluminium to the water environment, so potable water use will not be considered further here. Use of aluminium in wastewater treatment raises concerns about how much of the metal might be discharged into rivers and streams. Hence this activity is of potentially greater concern.

2. Aluminium in the Water Environment

The three important issues relating to the form and behaviour of aluminium in water are the effects of other water quality parameters, the effects of precipitation of aluminium minerals and kinetic effects on speciation. These are discussed below.

2.1. The effect of other water quality parameters

The static, equilibrium speciation of aluminium is relatively complicated. Speciation depends on reactions with different substances in water such as hydroxide ion and naturally occurring organic complexants. If the chemical characteristics of the water changes, for example when an aluminium-containing effluent is discharged into a river, changes in the distribution of forms of aluminium would be expected. The pH value of the water is the most important factor in determining dissolved phase speciation at equilibrium.

Dissolved aluminium species may be categorised as follows:

- Free hydrated Al^{3+} ions;
- Aluminium ions complexed by inorganic ligands (e.g. hydroxide, fluoride);
- Aluminium ions complexed by organic ligands (in waters and biological systems these ligands are often of unknown identity).

In waters that contain inorganic and organic ligands, the above species tend to exist in a state of equilibrium with one another.

Figure 2.1 shows the forms of aluminium that might be expected to exist at equilibrium – excluding the possibility of precipitation of solid species. Figure 2.1 might be considered to illustrate aluminium speciation shortly after reactive aluminium is introduced into a medium hardness water of the specified pH value. The main features revealed in Figure 2.1 are:

- The free (hydrated) aluminium ion, Al^{3+} , is likely to constitute only a small proportion of the total at pH values larger than approximately 5.0.
- In the pH range between 5 and 7, speciation is controlled by competition between complexation by fluoride, hydroxy species and organic ligands.
- For pH values in the range 7 to 9, two hydroxy complexes predominate ($\text{Al}(\text{OH})_3$ and $\text{Al}(\text{OH})_4^{-}$). These two principal hydroxy forms differ from the complexes found at lower pH values in that they are of higher molecular weight, and one is uncharged and one is negatively charged. These differences are likely to have important consequences for the solubility and for the bioavailability of the metal (see below).

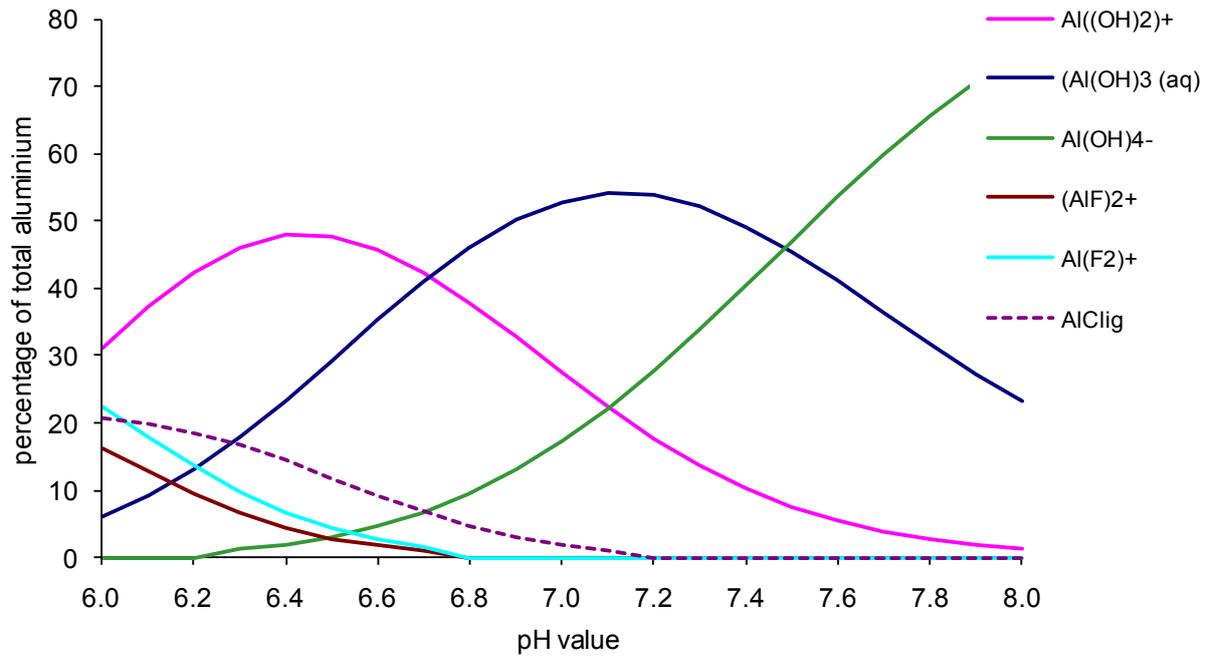


Figure 2-1 Illustration of the different forms of aluminium for a typical surface water at different pH values

Notes to Figure 2.2: Water is of approximate hardness 100 mg CaCO_3/l ; concentrations of fluoride and some organic complexing agents (Clig) are assumed to be at a concentration similar to that of aluminium (10-6M); a log K value for the association of aluminium with the organic ligands of 9 has been assumed.

2.2. Precipitation effects

In practice, aluminium speciation cannot be regarded as static (as in the discussion above) because of the marked tendency for dissolved aluminohydroxy and hydroxysilicate species to precipitate. Precipitation results in the removal of some dissolved forms from solution and consequent and continuing shifts in the equilibria of dissolved species. This effect is particularly marked at pH values in the range from 6 to 8 – the pH range of most surface waters.

Initially, over a period of minutes to hours, solid forms of aluminium hydroxide would be expected to begin to precipitate, initially as unstable colloids, then as recognisable minerals

2.3. Kinetic effects

In the case of aluminium speciation, kinetic considerations may be important on account of:

- The establishment of solution equilibria. Inorganic equilibria may be established in a few seconds or less; organic equilibria may take minutes or hours to complete. Different species behave in different ways;

- The precipitation of insoluble forms of aluminium – this may take place in the course of a hour or so (Gardner and Comber (2003)¹, Jugdaohsingh et al. (2001)² - the fate, behaviour and toxicity of solid forms will differ from soluble aluminium;
- Ageing (mineralisation) of precipitated forms - The differentiation between dissolved and particulate forms is made more complicated by the fact that portions of the precipitated fraction are continuously interacting with the dissolved forms of aluminium. It is thought that freshly precipitated forms of alumino-hydroxide and aluminosilicates can redissolve relatively easily (and hence be exchangeable with dissolved forms). After an interval of hours to days, precipitated minerals undergo a process of ‘hardening’ or maturation, after which they are less readily redissolved – and presumably less bioavailable.

In summary, if free aluminium ion is released into a surface water, several processes begin to take place. These processes are:

- hydrolysis of the free ion, according to the pH of the medium, to give inorganic hydroxy-species (this is very fast – less than seconds, these are still essentially monomeric and can dissociate to Al³⁺ easily);
- complexation by organic complexing agents of a proportion of the inorganic forms to produce organic monomeric aluminium (this probably occurs in a few minutes or less, dissociation of the complexes is possible, in response, say, to pH reduction);
- coagulation of hydroxy-species to form colloids (this takes tens of minutes to a few hours, these forms are not strictly monomeric and since they have already precipitated might pose less risk to the fish gill – but they can dissociate to give monomeric forms that might pose a risk);
- ripening of coagulated hydroxy-species to form aged precipitate (this occurs over several hours – days, these forms are more stable and less likely to release monomeric Al by reverse reactions);
- mineralisation of ripened forms – to inert clay type minerals (this occurs over days).

¹ Gardner M J and Comber S D W, 2003. Aluminium speciation in effluents and receiving waters. *Journal of Environmental Monitoring*, 5, 902–905.

² Jugdaohsingh R, Salim N, Mccrohan C R, White K N, Thompson P H and Powell J J, 2001. Rapid non-equilibrium aluminium–ligand interactions: studies on the precipitation of aluminium by laser light scattering, ultrafiltration and centrifugation. *Journal of Inorganic Biochemistry*, 87, 29–35.

3. Establishment of an EQS for Aluminium

The special properties, as outlined above, of aluminium mean that consideration needs to be given to modifications to the conventional approach to regulation, in which primary reliance is placed on ecotoxicology, for the following reasons:

- Ecotoxicological testing often has not accounted adequately for aluminium speciation;
- Risk posed by aluminium relates primarily to rarely-determined low molecular weight forms (referred to here as “reactive” aluminium, but called variously “active”, “fast reactive” or “monomeric”. All these terms are empirical based on similar methods that depend on an operational selection of the most chemically reactive fraction). Until recently, this has led to lack of credible environmental and ecotoxicological data;
- Risk also relates to water pH value and the potential for pH change (hence to pH and buffer capacity – in approximate terms, to alkalinity);
- Lack of ecotoxicology data and the uncertain quality of what is available leads – via the conventional EQS setting approach to the use of such large assessment factors that the resulting EQS are unrealistically low – usually below the background level.

a) Toxicologically based proposals

A large amount of toxicological testing has been carried out on aluminium. The greater proportion of this has been carried out in relation to “acid” waters – those of low pH value and alkalinity. This has happened because high concentrations of reactive aluminium are found in such waters, leading to issues of fish mortality being identified and subsequently followed up. The bias in favour of research into acid waters has also been exacerbated by the direction of research funding towards the once very topical issues of surface water acidification. The outcome is a validation of the maxim that if you subject anything to enough research, it becomes recognised as a “problem”. By contrast, relatively little work has been done on the more neutral and alkaline waters that make up the great majority of surface waters, at least in England and Wales. Figure 3.1 shows the distribution of mean pH value measured by the environment Agency for 2002-2004. Very few waters are of low enough pH value for much of the aluminium toxicity work to be clearly applicable.

Nevertheless, it is accepted that the main toxicological concern and the only one that has been explored/demonstrated by thorough testing, is the precipitation / smothering effect on fish gills. A second concern, direct toxic effect on growth rate inhibition of algae, is more speculative, having been supported by one study³ of which the title (the test medium is acidified synthetic soft water) only indicates its questionable relevance.

³ Parent L and Campbell P G C, 1994 Aluminium bioavailability to the green alga *Chlorella pyrenoidosa* in acidified synthetic soft water. *Environmental Toxicology and Chemistry*, 13, No. 4, 587–598.

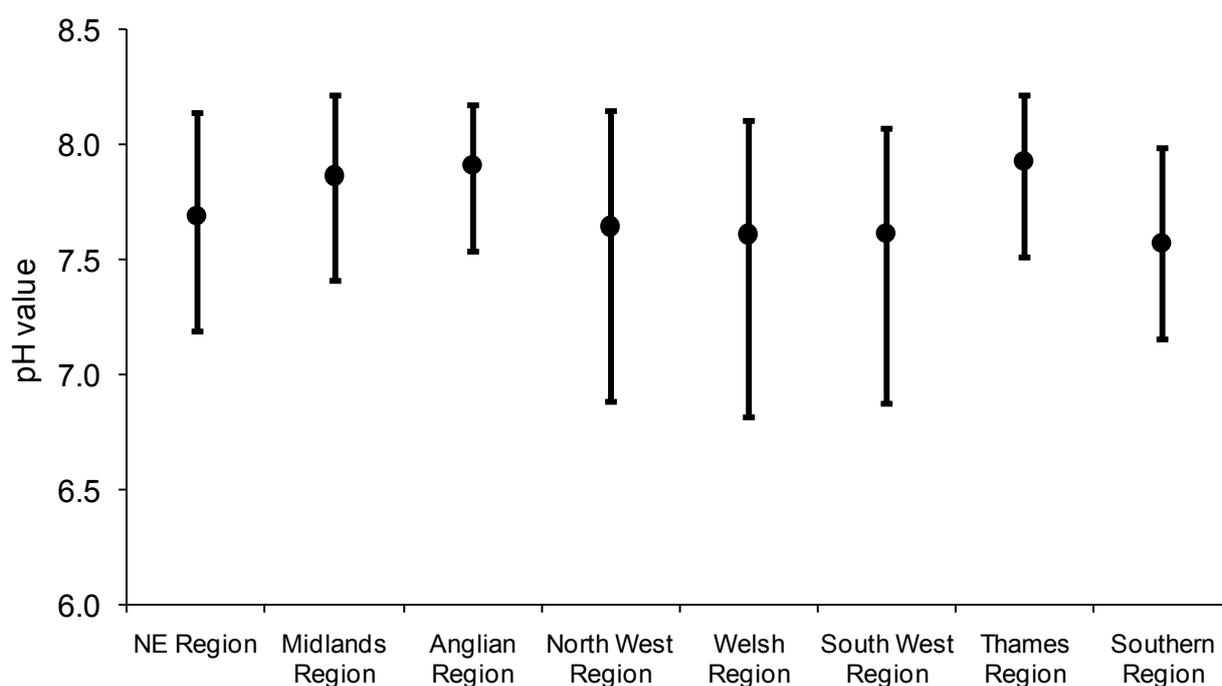


Figure 3-1 Mean, 5%ile and 95%ile pH values for monitored surface waters in England and Wales 2002-2004

Despite these reservations, an assessment has been proposed by UKTAG⁴, based on the EU recommended toxicologically-based approach⁵ to the establishment of EQSs. The proposed values are shown in Table 3.1.

Table 3-1 Toxicologically based proposes PNEC values

Receiving medium/exposure scenario	Proposed PNEC ($\mu\text{g/l}$ reactive Al)	Assessment factor used
Freshwater/long-term	0.05	150x
Freshwater/short-term	0.25	30x
Saltwater/long-term	0.005	1500x
Saltwater/short-term	0.025	300x
Secondary poisoning	0.4	

*The PNEC (predicted no effect concentration) is not a standard, but it is often used as the basis for a water quality standard. The usual process is to set a standard equal to the PNEC divided by an “assessment factor”; a safety factor, which is graded inversely with respect to the reliability of the data.

⁴ UKTAG (2008) Proposals for Environmental Quality Standards for Annex VIII Substances Final Report SR1 – 2008 (June 2008) UK Technical Advisory Group on the Water Framework Directive. Science Report: SC040038/SR1

⁵

[http://ecb.jrc.ec.europa.eu/DOCUMENTS/Practical_Course_on_Environmental_Quality_Standards/01_TGD%20EQS%20v.%208.0%20for%20formatting%20\(12%20May%202011\).pdf](http://ecb.jrc.ec.europa.eu/DOCUMENTS/Practical_Course_on_Environmental_Quality_Standards/01_TGD%20EQS%20v.%208.0%20for%20formatting%20(12%20May%202011).pdf) accessed 02062011

The resulting limit values based on the above are all considerably below background levels and are therefore recognised as impractical and inappropriate. Hence, an alternative solution was sought.

In order to address these issues the Environment Agency and other organisations (including Severn Trent Water) have supported a programme of research going back to the 1990s^{6,7,8}. This has involved defining the issues⁹, conducting investigations on the behaviour of aluminium in waters and effluents^{10,11} and, latterly, attempting to set an ecological benchmark for aluminium in surface waters^{12,13}. In the parallel briefing note on ammonia¹⁴ we raised some objections to ecologically based standard setting. The relevance and importance of these objections varies with context. In the case of aluminium the main issues are:

- The most obvious issue is confounding – in any real situation there is likely to be a number of factors that operate simultaneously, either additively or in a more complex way. It is not usually possible to isolate the impact of one factor. This is not a reason to dismiss ecological approaches, but it is a source of uncertainty that makes the approach less definitive than might be desired;
- Looking for relationships between a contaminant concentration and ecological quality can often be limited by the availability of different types of comparisons. If there were a full range of combinations of low and high aluminium and good and poor ecological status (and setting aside confounding effects from other contaminants) it might be possible to delineate the threshold aluminium concentration that is associated with the first signs of harm. This might then form the basis of a water quality limit value that is balanced with respect to restrictions on water use and environmental protection. However, in practice, this range of combinations is not available. In the case of aluminium, in most lowland waters of moderate to high pH value, it is not clear that the presence of aluminium can be associated at all with degraded ecological conditions. This means that the ecological approach might be of value in setting a lower limit of “no effect”, but may not be so reliable in defining the threshold concentration of aluminium that actually begins to have an adverse impact. In other words, in the absence of clear and widespread evidence for an effect, the proposed limit value is likely to be overprotective.

Do these objections matter? Well, in the case of aluminium, possibly not. If the proposed limit value for reactive aluminium is high in relation to concentrations that might be present and therefore of concern to dischargers, there might be little reason to object to a quality standard derived in this way – provided, perhaps, that such a standard is framed in a way that recognises its conservative characteristics. This might mean defining a limit value either towards the upper part of

⁶ Dixon, E.M. and Gardner, M.J. (1997) Importance of Aluminium Speciation to the Design and Interpretation of Toxicity Studies. Proceedings of Expert Workshop on Feasibility of a Chronic Neurotoxicity Study of Aluminium in Drinking Water. Sept. 3-4, published by Health Canada.

⁷ Dixon E, Whitehouse P, Gardner M and Mitchell R, (1998) Proposed Environmental Quality Standards for aluminium in water. Draft R&D Technical Report P43. Medmenham, Buckinghamshire: WRc.

⁸ Dixon, E. and Gardner, M (1998). Reactive aluminium in UK surface waters. *Chemical Speciation and Bioavailability*, 10 (1), 11-17.

⁹ Environment Agency (2006). Feasibility of EQSs for aluminium in surface waters. Environment Agency R&D Project Record 7326.

¹⁰ Gardner, M.J. and Comber, S.D.W. Aluminium Speciation in Effluents and Receiving Waters. *Journal of Environmental Monitoring* (2003) 5 902-905.

¹¹ Comber S D W, Gardner M J, and Churchley J, (2005) Aluminium speciation: implications of wastewater effluent dosing on river water quality. *Chemical Speciation and Bioavailability*, 17, 117–128.

¹² Environment Agency (2007) Water Framework Directive - Further Development of Approaches to the Regulation of Aluminium in Surface Waters - Risk Category survey. Environment Agency R&D Project Record 7462.

¹³ Gardner, M J, Brown B, Whitehouse, P and Birch, M ((2008) Towards the establishment of an environmental quality standard for aluminium in surface waters. *Journal of Environmental Monitoring*, 10, 877-882.

¹⁴ Gardner, M.J. (2011) Briefing note prepared for Severn Trent Water on the regulation of ammonia, Atkins report (draft).

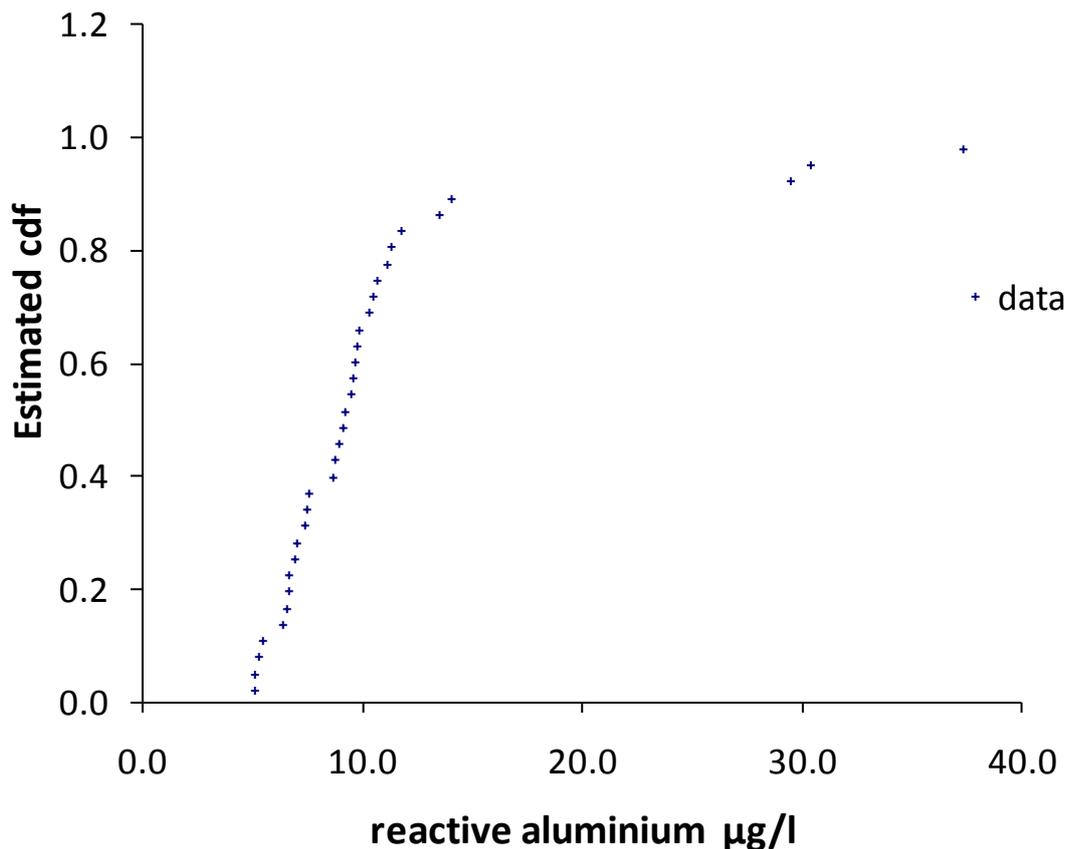
the proposed possible effects range or by setting it as a lower percentile (90 or 80%ile) instead of a more customary 95%ile.

An EQS value of 50 µg/l, as reactive aluminium, has been proposed following a recent (2010) review of the approach to setting an EQS for aluminium. This value is still under discussion and may or may not be published in the planned consultation document on the subject. A further key issue is the percentile value of any standard. For any given numerical standard, if we assume a particular statistical distribution of data, the higher the percentile chosen, the lower the tolerable level of contaminant will be. This is evident in the fact that an average standard is less stringent than a 90%ile. For the former (given a symmetrical distribution of data) it is acceptable for individual concentrations to exceed the limit for up to half of the time, whereas for the latter only a 10% “failure” rate is applicable

4. Implications for the Water Industry in the UK

Recent monitoring of effluent quality as part of the UKWIR Chemical Investigations Programme (CIP) has provided the following data. For the 34 Severn Trent WwTWs that are involved in the relevant part of the CIP the profile of average reactive aluminium concentrations is as shown in Figure 5.1. Given the uncertainty about the details of any future limits / consents for aluminium the results below are offered for information – there is little else in which to base a more sophisticated interpretation. However, it might be noted that the great majority of results are considerably lower than the mooted 50 $\mu\text{g/l}$ (but that at a very few sites there might be need for further investigation of sources of aluminium).

Figure 4-1 Profile of average reactive aluminium concentrations in WwTW effluents (Severn Trent)



The cumulative distribution function (cdf) represents the proportion of average effluent aluminium concentrations that are less than the corresponding x axis value. Hence 90% of values are less than 13 $\mu\text{g/l}$. Individual Severn Trent WwTWs do, however, exhibit high values of greater than 30 $\mu\text{g/l}$ (eg Maltby, 100 $\mu\text{g/l}$ [not shown on the above graph], Snarrows, and Newthorpe).

5. Water quality criteria for aluminium - Conclusions

1. The ubiquity of aluminium combined with its unusual chemical properties mean that the conventional approaches to regulation that are applied for most other metals are not appropriate. Regulation in terms of total metal does not make sense because aluminium is a constituent of particulate matter such as sediments and soils. Control in terms dissolved aluminium whilst more appropriate does not address the forms of the metal that actually constitute the principal environmental risk. These low molecular weight forms – loosely and operationally defined as reactive aluminium – usually constitute a small and variable fraction of the dissolved portion of the metal. Even reactive aluminium by itself does not necessarily pose a risk to aquatic life; risk to fish is greatest when the conditions in the water favour precipitation of the aluminium-hydroxide solids that have an adverse effect at the fish gill. These conditions can be defined primarily as the presence of sufficient reactive aluminium in a water that is of pH in the approximate range of 6.5 to 7.5.
2. Long standing issues relating to the monitoring of reactive aluminium – few laboratories were set up to determine it and consequently there was not a lot of experience or confidence in its measurement – have recently been addressed. Incorporation of this determinand into the UKWIR chemical investigation programme has prompted analytical laboratories to implement the methodology. This programme has generated almost 2000 measurement of WwTW effluents to a limit of detection low enough to address most concerns. Data received are consistent with previous more research based work and are comparable between laboratories. This implies that measurements on river waters should not pose too many problems and opens the way to regulation and compliance assessment in terms of the relevant metric (rather than some less reliable surrogate as had previously been proposed).
3. Recent studies of the impact of reactive aluminium on the ecological status of waters have progressed to the point where the Regulator is about to propose a limit value that protects surface waters whilst not falling into the error of previous limits that did not account properly for aluminium speciation.
4. It is therefore recommended that the Water Industry should examine the forthcoming consultation document in the light of the above.