An evidence base for setting organic pollution targets to protect river habitat

This note brings together relevant information from various sources to help define the appropriate environmental targets to control the adverse effects of organic pollution on the characteristic flora and fauna of UK rivers. In particular it draws on work undertaken by the Centre for Ecology and Hydrology on behalf of Natural England, published as NECR023. It also uses analyses undertaken by the UK environment agencies under the Water Framework Directive. Natural England's primary aim in publishing this is to inform those involved with the review of UK Common Standards targets for rivers with special designations for wildlife. We will also use it to contribute to the debate on the control of organic pollution under the Water Framework Directive and the UK Biodiversity Action Plan.

This work is specifically designed to characterise the effects of organic pollution on the integrity of river habitats. It does not imply that organic pollution is the only significant man-made problem for riverine wildlife. A range of stresses have to be tackled to secure the ecological integrity of river habitats (Mainstone and Clarke 2008) and this note should be seen as a contribution to this wider work.

Organic pollution impacts in rivers

Modern water quality management started with the study of the response of river organisms to organic pollution. In the first half of the last century organic pollution was a major determinant of river health. Probably the most significant environmental achievement in water quality management in the UK to date is the reduction in organic pollution levels that were due to the improvements made to sewage effluents in the latter half of last century.

As organic pollution levels have progressively declined, attention has refocused on a range of other human impacts that have become more apparent over time, such as eutrophication and siltation. However, although chronic, gross organic pollution has reduced greatly, the impact of organic pollution is still being felt. Chronic mild organic pollution is a widespread effect and remaining pollution sources are more difficult and costly to control.

Targets to control organic pollution are therefore still as necessary as they ever were, but there is a need for greater subtlety in their definition and application.

Organic pollution can be defined as the human-induced entry of highly degradable organic material into environmental waters. It has a range of well-documented and inter-linked ecological effects (for example, Hynes 1970, Welch and others 2004), comprising:

- toxicity from increased ammonia (ionised and un-ionised) levels;

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- biological stress from reduced dissolved oxygen levels, due to high Biochemical Oxygen Demand (BOD) arising from microbial degradation and chemical oxidation processes;
- physical smothering and deoxygenation of the stream bed, leading to loss of usable habitat by a range of fish species (those that use the streambed for spawning), rooted aquatic plants and benthic invertebrates;
- enrichment with carbon, nitrogen and phosphorus, generating enhanced microbial communities ('sewage fungus'), and enhanced abundance of opportunistic, organic pollution-tolerant fauna, such as oligochaete worms and chironomid midges; and
- reduced abundance and loss of organic pollution-sensitive fauna, such as stone-flies, caddis-flies, mayflies and salmonids.

Organic pollution has close links with eutrophication. By tackling organic pollution through the removal of organic carbon from sewage effluents (through secondary treatment) an organic pollution problem can be transformed into a eutrophication problem. The loss of sewage-derived carbon results in a decline in microbial activity, allowing the plant community (rooted higher plants and algae) to take advantage of the high residual levels of nitrogen and phosphorus by using carbon dioxide as their carbon source (via photosynthesis).

Organic matter can be derived from a variety of sources including soil eroded from the catchment. The effects of siltation are complex and will be dealt with in a subsequent evidence paper, but can mimic those of organic pollution, particularly when the sediment source is peat. Figures 1 and 2 below show the effects resulting from enhanced delivery of organic particulates from degraded upland peat (Brown and others 2009). The observed increases in chironomids and oligochaetes and reductions in stonefly species closely resemble the effects of organic pollution.

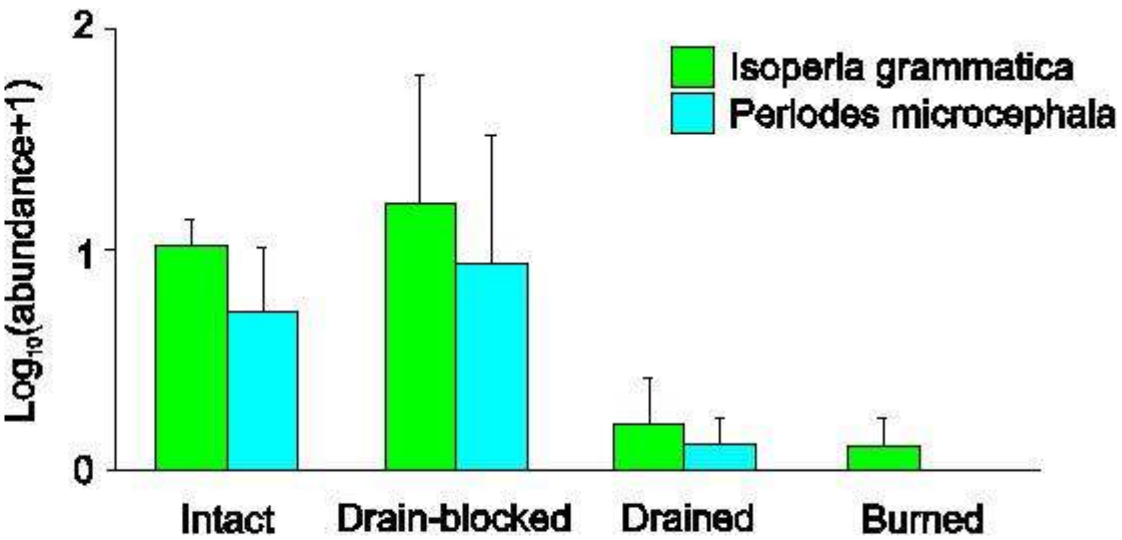

Figure 1 Abundance of two species of stonefly in upland streams draining catchments under different management regimes, September 2007. Error bars indicate 1 standard deviation. From Brown and others (2009).

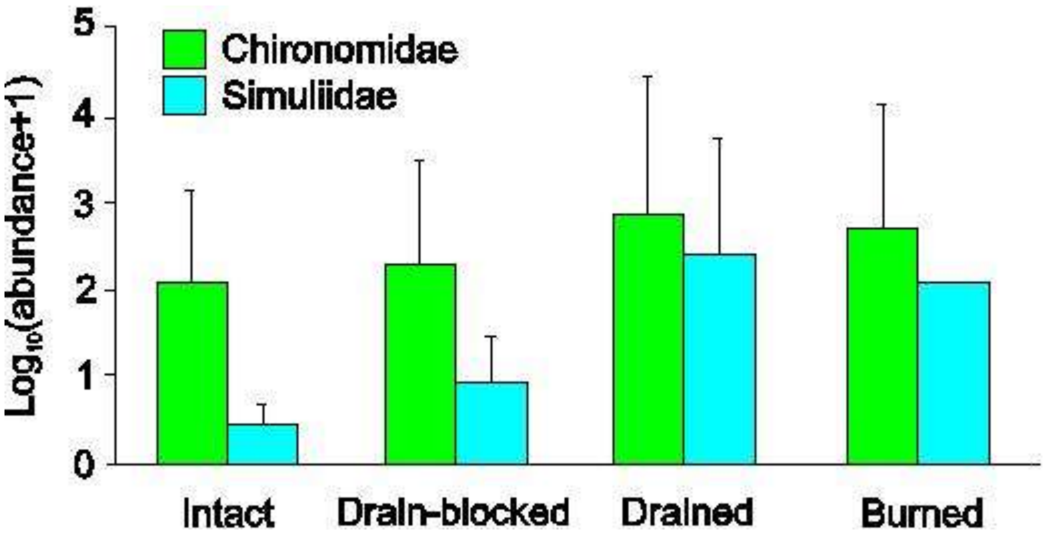

Figure 2 Abundance of two groups of true fly in upland streams draining catchments under different management regimes, September 2007. Error bars indicate 1 standard deviation. From Brown and others (2009).

Climate change has the potential to alter the biological impact of any given level of organic pollution load to rivers. Increased water temperatures will increase microbial decomposition rates, giving rise to greater levels of BOD and lower levels of dissolved oxygen.

Climate-induced changes to the flow regime, predicted to involve lower summer flows and higher winter flows (Hulme and others 2002), will reduce dilution rates and increase retention times in summer, increasing the pollution stress experienced by biological communities.

Recent research suggests that addressing mild organic pollution may help considerably in minimising the impact of climate change. Durance and Ormerod (2009) found that climate-induced increases in water temperatures over the period 1989 to 2007 (2-3 deg C in winter and 1 to 1.5 deg C in summer) were insufficient to detect a predicted negative change in the macroinvertebrate community over the positive changes apparently brought about by a reduction in the level of mild organic pollution over the same period.

Key quantitative evidence for thresholds

The published literature

There is a wealth of literature describing the quantitative responses of river biota to organic pollution, either in its entirety or in terms of selected components of the overall stress (particularly increased ammonia levels and reduced dissolved oxygen levels).

Key published evidence has been collated by Jones and others (2009), comprising laboratory experiments, field mesocosm studies and analysis of field survey data.

Much of the literature relates to the response of the benthic macroinvertebrate community, which is amenable to routine monitoring and experimental study and exhibits considerable sensitivity to organic pollution. Fish communities are also highly sensitive, particularly salmonids, but there is less literature on this.

Studies vary considerably in design, from those involving conventional chronic toxicity tests on individual species, to the evaluation of tolerance to transient peaks in ammonia or troughs in dissolved oxygen.

The biological end-points studied vary from sub-lethal effects (including behavioural responses such as drift in stream channels), to mortalities, and up to population-level effects.

No attempt is made here to re-present published material collated in Jones and others (2009), which should be seen as an integral part of this evidence base. However, some comments are made below about the difficulties in comparing data derived from different research designs.

First, the response of biota in conventional chronic exposure tests in the laboratory is highly modified compared to that in real systems, where:

- organisms can withstand higher ammonia concentrations and lower dissolved oxygen levels for short periods;
- behavioural responses in real systems can increase the ability of organisms to withstand organic pollution stress; and
- the combined effect of different components of organic pollution stress in real systems (increased ammonia levels, reduced oxygen levels, increased carbon enrichment) alters the observed apparent relationship between the biota and the individual stressors.

In addition to the realities of biological response, the resolution of measurement of organic pollution stressors varies widely between monitoring designs. In particular, analyses involving routinely collected macroinvertebrate and water quality data from the field generally use data on ammonia, dissolved oxygen and BOD from monthly discrete sampling within daylight hours. Available comparisons between such discrete sampling regimes and continuous monitoring programmes show high variability in water quality that is not captured by routine monitoring but which forms a key part of the exposure regime to which resident biota are responding.

In addition, routine monitoring of ammonia is only undertaken in relation to total ammonia (ionised and un-ionised combined).

Since the un-ionised form is much more toxic than the ionised form, and the relative proportions of the two can vary considerably depending on circumstance, this leads to ammonia data that is relatively poor at indicating ammonia toxicity.

Perhaps the most important conclusion to draw from the published literature is that the targets derived from the evidence base need to be relevant to real environmental conditions and take into consideration the way in which organic pollution stress is operationally evaluated.

The evidence presented in subsequent sections focuses on analyses of the response of the macroinvertebrate community in real environmental conditions, using large datasets from routine monitoring. Of the available evidence these are arguably the most amenable to the generation of environmental targets. However, the limitations of these data need to be borne in mind and are considered later in this paper.

New studies involving real-world analyses of biological responses to the organic pollution pressure gradient have recently been published. These are too recent to have been included in the review undertaken by Jones and others. (2009), but are summarised below:

- In a study of long-term trends in macroinvertebrate communities in streams across southern England, Durance and Ormerod (2009) found that biological changes in chalk streams over the period 1989 – 2007 were most strongly related to reductions in the level of organic pollution/enrichment, even though organic pollution levels were only 'mild' in the conventional sense. Increased abundances of taxa sensitive to organic pollution (including the caddis-fly families *Limnephilidae* and *Lepidostomatidae*) were correlated with reductions in mean BOD from around 1.6 mgl^{-1} to 1.3 mgl^{-1}.
- In an analysis of environmental change in the Wye catchment, Herefordshire, Clews and Ormerod (2009) found that observed improvements in the macroinvertebrate community over the period 1989-2000 were best explained by reductions in the level of organic pollution, from around 1.5mgl^{-1} mean BOD to around 0.7 mgl^{-1}. This level of organic

pollution was observed as widespread across the Wye catchment.

- In a recent study of nearly 600 Danish streams over an 11-year period, Friberg and others (2010) observed strong relationships between the abundance of certain pollution-sensitive invertebrate taxa and mild levels of organic enrichment, which were not explicable through inter-correlation with key habitat variables. The stonefly genus *Leuctra* showed the highest sensitivity, with occurrence declining sharply at BOD levels above 1.6 mgl^{-1}. Other genera with similar relationships included the caddis-fly genera *Sericostomatidae* and *Glossosomatidae*, and the stonefly genus *Isoperla*, whilst pollution-tolerant taxa such as the midge genus *Chironomus* showed a positive exponential relationship with BOD.

Such correlative analyses do not provide proof of cause and effect, but the lack of clear relationships with other possible environmental influences does strongly suggest a mechanistic link.

Water Framework Directive analyses

An analysis of routinely collected macroinvertebrate and chemical data was undertaken by the environmental agencies to inform the derivation of targets for organic pollution under the Water Framework Directive (WFD) (Guthrie and others 2006). This work was specifically designed to identify standards for High and Good Ecological status (HES/GES) as defined by WFD, but can be used to inform conservation objectives as well. This analysis provides a basis for setting biological targets that relate to organic pollution stress, as well as chemical targets to limit organic pollution stress itself.

Characterising the biological response

Routinely collected macroinvertebrate data in the UK are based on family-level taxonomic resolution, summarised into weighted scores based on the judged organic pollution sensitivity of each family.

Routinely used indices are:

- Biological Monitoring Working Party (BMWP) score.
- Average Score Per Taxon (ASPT).
- Total Number of (BMWP-scoring)Taxa.

These are all corrected for the biological community expected at reference (putative minimally impacted) conditions using the RIVPACS reference database and prediction system. This correction results in observed-to-expected ratios called the EQI.

The data analysis for the WFD focused on ASPT as the most sensitive indicator of the organic pollution stress gradient. Scores are based on the presence rather than the abundance of each taxa, so evaluation of changes in community composition are restricted to taxon loss and gain.

As part of the WFD evaluation, the environmental agencies undertook a series of analyses to look at the ecological consequences of changes in ASPT associated with the organic pollution stress gradient. This considered:

- the ratio of sensitive to insensitive taxa;
- loss of taxa; and
- loss of key taxonomic groups.

The work was based on a lumped analysis of all routinely monitored sites across the UK (some 6000 sites in all), with variations in ASPT due to natural habitat conditions between sites dealt with through the use of EQIs.

The environmental agencies acknowledge that at the time of these analyses the RIVPACS reference database contained sites which were not of reference quality. At that time, the database was judged to span a range of quality conditions from the top of High Ecological Status to the bottom of Good Ecological Status (Guthrie and others 2006).

For this reason, RIVPACS predictions of the community of a site under reference conditions were significantly under-estimated, yielding an over-optimistic ratio of observed-to-expected values.

This was probably most apparent for chalk rivers, where the under-estimation of reference ASPT can be considerable, although no objective analysis of the extent of the problem in different river types was possible with information to hand. As a consequence of this problem, the environmental agencies set the boundary between High and Good Status at a ratio of 1.0 (the mean value of EQIs in the RIVPACS database).

Since the time of this analysis, the RIVPACS database has undergone a further screening exercise to eliminate sites that were at less than high quality at the time the reference dataset was assembled. The new River Invertebrate Classification System (RICTS) is built on this screened RIVPACS reference database (Davy-Bowker and others 2008). It is not possible at this time to evaluate whether this screening has dealt with the problem of over-estimating EQI ratios in its entirety.

Ratio of sensitive to insensitive taxa

Taxa were divided into those deemed sensitive or insensitive to organic pollution, and the relative proportion of both groups (in terms of contribution to EQI score) was then plotted against ASPT EQI (Figure 3).

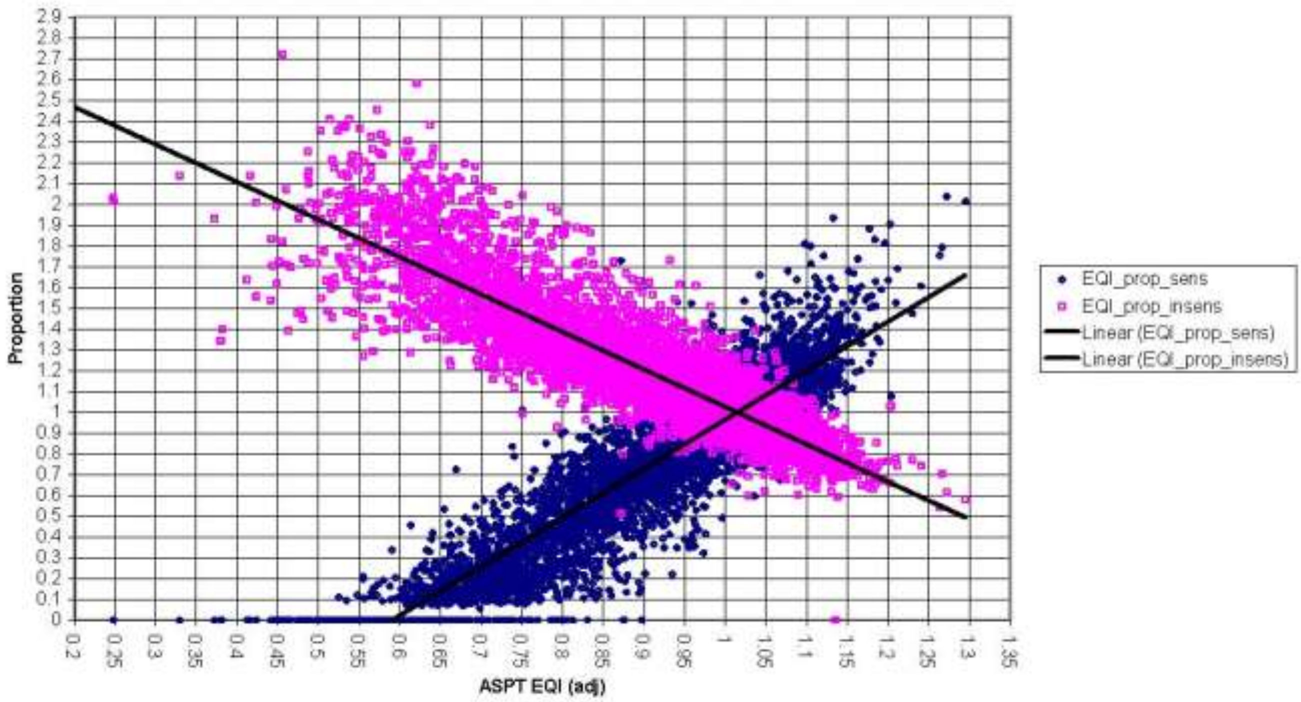

Figure 3 Analysis of changes in the ratio of sensitive to insensitive invertebrate taxa along the ASPT EQI scale (Guthrie and others 2006)

There was some nominal adjustment made in these ratios to account for the fact that RIVPACS predictions at that time were based on a database of sites that included impacted sites - it is not possible to judge whether these adjustments fully addressed the problem. Taken at face value, the graph indicates a steady decline in the relative proportion of sensitive taxa from ASPT EQI values of around 1.2 or 1.3, reaching a cross-over zone centred on an EQI value of around 1.05.

This area of the graph encompasses the bulk of the RIVPACS reference database, and the position of the boundary between High and Good Ecological Status recommended by Guthrie and others is at the lower end of these values.

Loss of taxa

Figure 4 shows a plot of the number of sensitive taxa predicted by RIVPACS to be missing from each of the 6000 sites, against ASPT EQI. Table 1 shows the average number of missing sensitive taxa at a range of EQI values. These show that at an EQI of 1.0 there is already an average of 1.6 taxa missing, whilst at an EQI of 0.9 there is an average of 4.6 taxa missing.

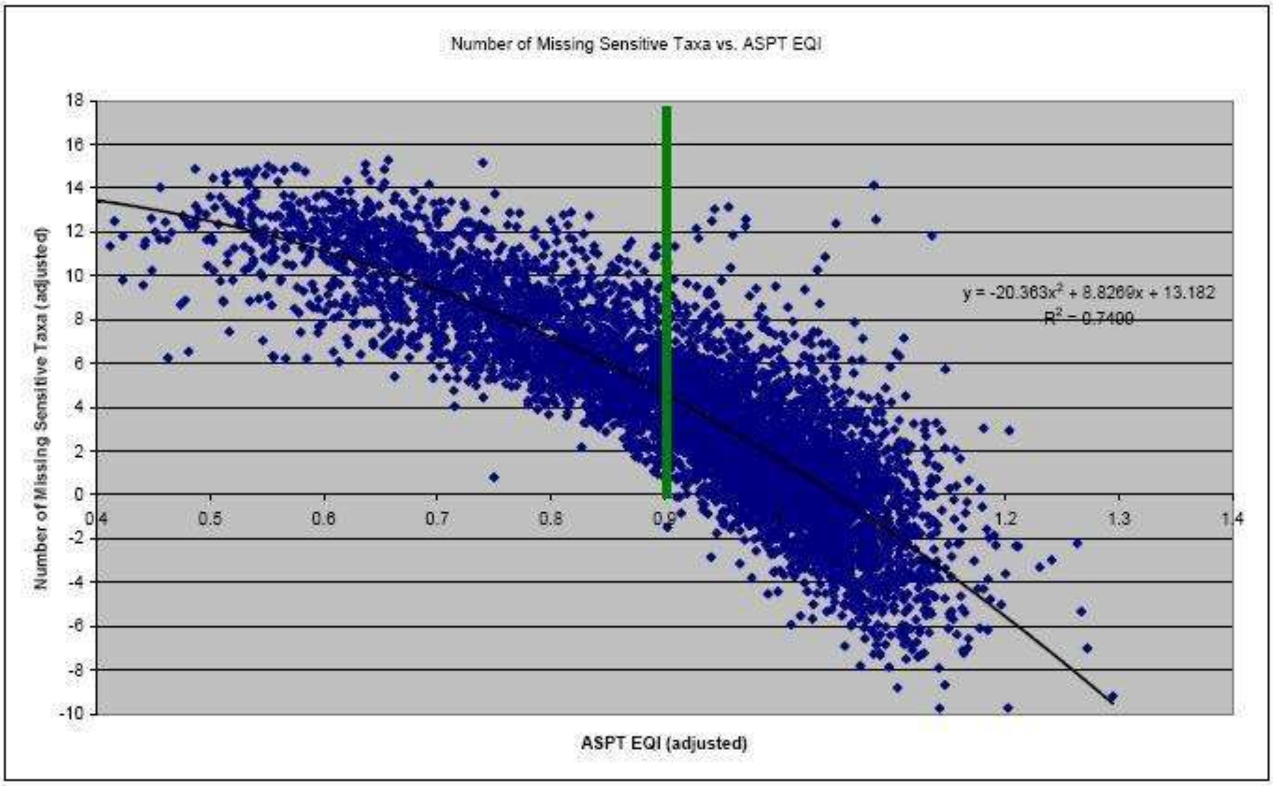

Figure 4 Changes in the number of missing sensitive taxa (families) along the ASPT EQI scale (Guthrie and others 2006)

Table 1 Numbers of missing taxa at points along the ASPT EQI scale, showing judgements of the location of WFD ecological status boundaries (Guthrie and others 2006)

Status description	ASPT EQI	Number of missing sensitive taxa
High/Good Boundary	1.0	1.6
Mid Good	0.95	3.2
	0.94	3.5
	0.93	3.8
	0.92	4.1
	0.91	4.4
Good/Moderate Boundary	0.90	4.6
	0.89	4.9

Loss of major taxonomic groups

Guthrie and others defined a major taxonomic group as an order, for example, mayflies *Ephemeroptera*, and stoneflies *Plecoptera*. Figure 5 shows the missing orders predicted by RIVPACS for each of the 6000 sites, against ASPT EQI. Whilst the graph is difficult to read, Guthrie and others report that an EQI value of 0.9 equates to an average loss of 1 missing order. An EQI of 1.0 equates to an average loss of no missing orders.

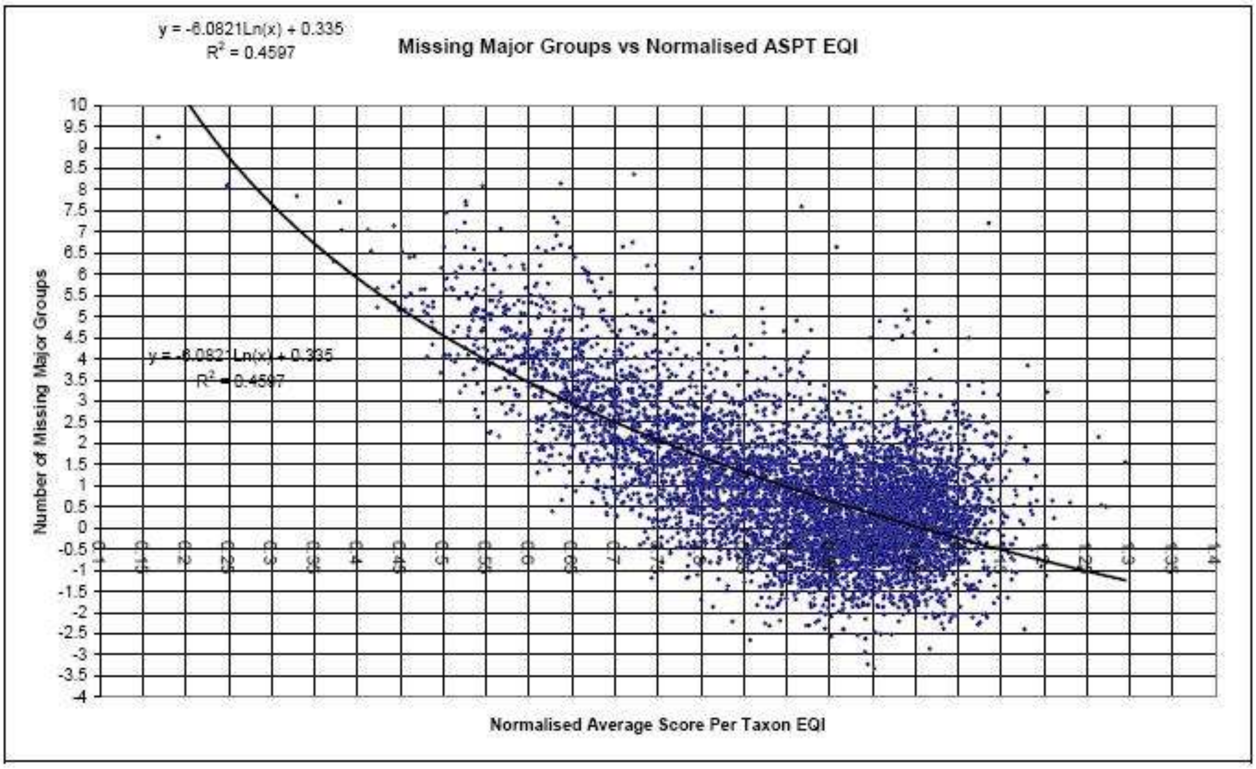

Figure 5 Changes in the number of missing Orders along the ASPT EQI scale (Guthrie and others 2006)

On the basis of this analysis, Guthrie and others proposed an ASPT EQI of 0.9 as the boundary between Good and Moderate Ecological Status.

Quantifying the biological response along the organic pollution stress gradient

Guthrie and others proposed standards (see table 2 overleaf) to protect High and Good Ecological Status in relation to DO, BOD and Total Ammonia. These were derived by grouping sites judged to be at GES and HES (according to ASPT values) within a series of river types and, for each group of sites, generating a frequency distribution of 90%ile BOD and Total Ammonia and 10%ile DO values from each site.

Table 2 Thresholds proposed by Guthrie and others (2006) to protect High and Good Ecological Status

River type	Sub-types*	Dissolved Oxygen (% sat, 90%ile)		BOD (mgl-1, 90%ile)		Total Ammonia (mgl-1, 90%ile)	
		H/G	G/M	H/G	G/M	H/G	G/M
Upland and low alkalinity	1,2,4 and 6 below	77	72	3.4	4.2	0.20	0.30
Lowland and high alkalinity	3,5 and 7 below	68	60	4.2	4.8	0.33	0.55

Figures are pollution levels suggested as being consistent with the boundaries between HES and GES (H/G) and between GES and Moderate Ecological status (G/M).
Note that these figures were rounded during the process of finalising proposals by the WFD UK Technical Advisory Group.

* Subtypes shown below

Site Altitude	Alkalinity (as mg/l $CaCO_3$)				
	<10	10 to 50	50 to 100	100 to 200	>200
Under 80 metres	Type 1	Type 2	Type 3	Type 5	Type 7
Over 80 metres			Type 4	Type 6	

A fixed percentile value of each distribution was then taken as the proposed standard – the 90%ile values of BOD and Total Ammonia and 10%ile values of DO within each frequency distribution were used. The assumption made by Guthrie and others is that if 10% of sites judged to be at GES or HES within a river type can withstand the specified level of stress from organic pollution, then the other 90% of the sites in the same river type that are judged to be at the same ecological status should be able to withstand that level of stress, even though they are not currently experiencing it.

The analysis is not flexible to consider different judgements of acceptable levels of biological quality ie the data have been packaged to provide answers in relation to the biological definitions of HES and GES used by Guthrie and others. Equally, the frequency distributions themselves are not available, and so the possibility of using different percentile values from the distributions (to consider different levels of environmental precaution) cannot be considered.

Analyses by the Centre for Ecology and Hydrology

This work (Jones and others 2009) was commissioned by Natural England to provide a clearer picture of the degradation of the biological community along the organic pollution gradient than was generated by the WFD analysis.

The data analysis involved a national macroinvertebrate dataset at family and (where available) species level, and used multivariate analysis to plot the disappearance of taxa as organic pollution pressure increases.

The analysis lumped data from all river types but accounted for natural variation in communities within the multivariate analysis. The family-level plot is shown in Figure 6.

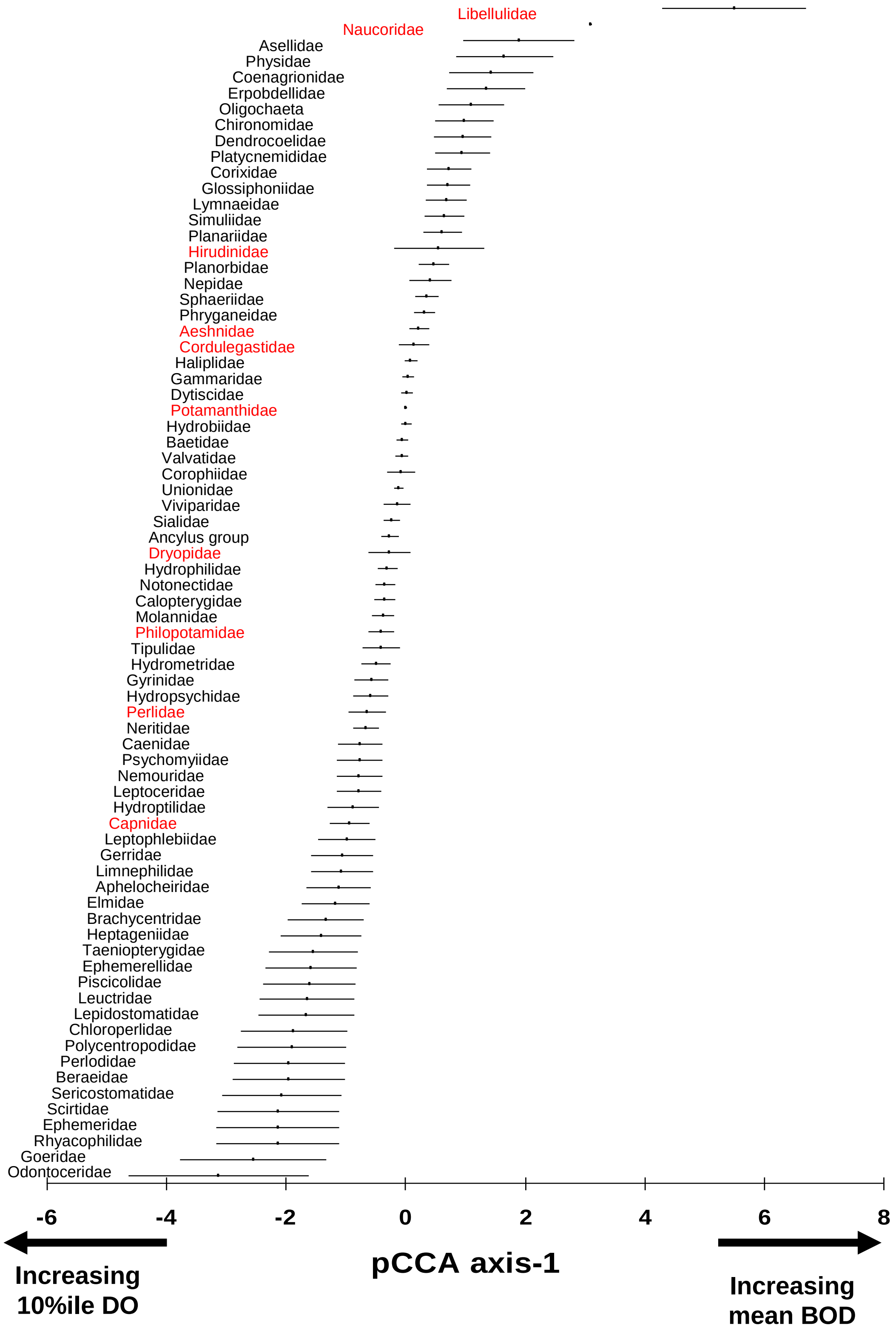

Figure 6 Occurrence (optimum and range) of BMWP families along the organic pollution gradient (first axis of Canonical Correspondence Analysis), ranked by sensitivity (Jones and others 2006). Families in red are of low frequency in the dataset and were not used to construct the ordination.

The pattern of taxon sensitivity varied quite considerably from similar work using Artificial Intelligence networks undertaken by the Centre for Intelligent Environmental Systems (CIES - see Jones and others 2009 for a comparison). It is not clear whether this is a product of different but valid analytical approaches or problems with one or other approach. A species-level plot was generated but species data were too sparse to provide a robust picture of species-level responses.

Jones and others found that, whilst extreme percentile values of ammonia (90%ile value at each site) and dissolved oxygen (10%ile value at each site) best characterised the relationship of these parameters with the invertebrate community, the mean value at each site best characterised the relationship with BOD.

This suggests that, whilst the main influence of ammonia and dissolved oxygen relates to the magnitude of transient peaks in pollution, BOD is providing an indication of both transient pollution peaks and the level of consistency of the supply of organic material (ie whether or not there is chronic carbon enrichment that opportunistic detritus feeders can exploit to dominate the invertebrate community).

Thresholds for DO, Total Ammonia and BOD were derived from the family-level regression equations for a number of the most sensitive taxa (those with adequate levels of occurrence to generate a robust relationship), using sufficient taxa to ensure that all river types were adequately covered.

Thresholds were set at a level at which 20% of sites containing the sensitive taxa were exposed to more polluted conditions. This contrasts with the 10% figure used in the WFD analyses described above, providing greater confidence that these thresholds will protect the fauna, ie that they are not set within an area of the frequency distribution subject to excessive levels of uncertainty, associated with various types of sampling error and unevaluated sources of variability.

The thresholds were then applied to river types on the basis of the predicted occurrence of these sensitive taxa within each type.

Two sets of thresholds were generated (Table 3), which were assigned to river types as in Table 4. Jones and others concluded that river types assigned Set 2 (less stringent) thresholds would not safeguard low frequency (rare) occurrence of more sensitive taxa relating to Set 1 thresholds, and recommended that for such protection Set 1 thresholds would ideally be used across all river types for rivers with special designations for wildlife.

Table 3 Sets of chemical thresholds for the most sensitive taxa of different river types generated by Jones and others (2009)

Determinand	Set 1	Set 2
10%ile DO (% saturation)	85	79
Mean BOD (mgl-1)	1.8	2.0
90%ile Total Ammonia (NH_3-N, mgl-1)	0.23	0.29

Table 4 Suggested application of CEH thresholds to river types used in Common Standards, based on frequency of occurrence of sensitive taxa in those types (Jones and others 2009)

Dominant Catchment Geology		Threshold	
	1. Headwater	2. River	3. Large river
A Hard upland geologies	I	II	II
B Other Cambrian-Devonian geologies	I	I	II
C Jurassic and Cretaceous limestones	I	II	II
D Triassic sandstones and mudstones	I	I	II
E Mesozoic clay vales and Tertiary clays	I	II	II

It should be noted that no direct numerical comparision can be made between the BOD thresholds defined for HES and GES by the WFD analysis and those proposed by the CEH analysis, as they employ different summary statistics (the 90th percentile in the case of the WFD analysis, and mean BOD in the case of the CEH analysis). The relationship between the mean and 90%ile values varies depending on the shape of a site's frequency distribution of raw values but, for the purposes of broad comparison, information provided by CEH (pers. comm. Iwan Jones) suggests that a mean BOD threshold of 1.5mgl^{-1} typically equates to a 90%ile value of around 3 mgl^{-1}, and a mean BOD threshold of 2mgl^{-1} typically equates to a 90%ile value of around 3.5 mgl^{-1}.

Key messages

Key message1

The most amenable data for deriving organic pollution targets relevant to the real world relate to the benthic macroinvertebrate community and routine chemical sampling. The inadequacies of these data should be noted, in particular:

- Species-level data are rarely collected and therefore species-resolution of impacts is not currently possible. This is a major issue in relation to evaluating effects on biodiversity.
- Standard indices used to report ecological status are traditionally based on presence/absence of taxa and do not allow changes in the relative abundance of different taxa to be evaluated (this situation is changing but affects analyses based on historical data).
- The RIVPACS tool, which predicts the macroinvertebrate community of a site under reference conditions, has historically been over-optimistic in its judgement of biological quality (particularly for chalk streams), but recent changes to the reference database and models may have resolved this issue.
- Routine, discrete chemical sampling data provide a very partial temporal picture of pollution stress.
- The highly toxic un-ionised ammonia fraction is not routinely sampled and has therefore not been part of data analyses to date (although it can be derived from routinely collected total ammonia and environmental data).
- The benthic macroinvertebrate community comprises only one component of riverine communities, albeit a relatively sensitive one.

These points need to be borne in mind when generating targets to protect characteristic biodiversity of rivers.

Key message 2

The CEH analysis has highlighted the importance of average levels of BOD as an indicator of carbon enrichment. BOD is often criticised as not being ecologically relevant, with DO assumed to provide adequate expression of the ecological consequences of BOD.

However, BOD seems to be providing an additional window in on organic pollution impacts, associated with the trophic effects of carbon enrichment, which is not provided by DO or ammonia but is important in the setting of environmental targets.

Key message 3

Recent published literature and the CEH analysis described above suggests the need for control of mild levels of organic pollution, of the order of less than 1.5 – 2 mgl^{-1} mean BOD and even less than 1 mgl^{-1} mean BOD to adequately protect the biodiversity of benthic macroinvertebrate communities.

Key message 4

Real-world datasets contain high levels of statistical and ecological variability. Positioning of thresholds using such datasets is strongly influenced by how this variability is dealt with, which relates to the level of environmental precaution applied.

There is justification in adopting different levels of environmental precaution for different policy drivers.

Key message 5

Available evidence points to climate change increasing the ecological stress caused by any given level of organic pollution, suggesting the need for more stringent control of organic pollution levels than might otherwise be the case.

Comments on the state of the evidence base

1. In comparison with the evidence base for certain other types of environmental targets in rivers, the published evidence base on organic pollution is extensive. However, there are certainly areas in which it could be improved.

2. There is a considerable body of evidence on the response of river biota to organic pollution, but there is a good deal of conflicting quantitative information brought about by differences in experimental and analytical designs, biological end-points and levels of environmental reality.

More effort could be made in rationalising data in the published literature on the subject, to explain differences in results and build quantitative process models of organic pollution that capture the evidence base in an assimilable way for water quality managers.

Such work could be linked to models of eutrophication and siltation processes to build a better and more understandable picture of interactions between these overlapping stressors.

3. More research is needed on the characterisation of biological responses at the mild end of the organic pollution spectrum, and on improving the taxonomic resolution of the biological response.

4. A more extensive monitoring dataset of species-level macroinvertebrate information across a full range of sites would provide greater scope for resolving species-level responses and better inform a biodiversity appraisal of the organic pollution stress gradient. Species from the *Plecoptera* (stone-flies), *Trichoptera* (caddis-flies) and *Ephemeroptera* (mayflies) are of clearest concern, since these groups are known to be highly sensitive to organic pollution and contain numerous rare and threatened species.

5. Better characterisation of the response of real macroinvertebrate communities to un-ionised ammonia regimes (as opposed to the less toxicologically relevant total ammonia), is needed as part of future analyses of routine macroinvertebrate and water quality data.

6. It would be valuable to have more focused studies on species of high conservation value likely to be most affected by mild organic pollution, involving a combination of strategic field surveys and experimental manipulations.

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Further information

Natural England publications are available to download from the Natural England website: **www.naturalengland.org.uk**. In particular see the research report:

- NECR023: *Review of the evidence for organic pollution thresholds to protect rivers with special designations for wildlife*
- NERR034: *An evidence base for setting nutrient targets to protect river habitat*

For further information contact the Natural England Enquiry Service on 0300 060 0863 or e-mail **enquiries@naturalengland.org.uk**.

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