Environment Agency

Evidence

The ecological classification of UK rivers using aquatic macrophytes

Report – SC010080/R1

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Miranda Kavanagh

Director of Evidence

Executive summary

This report describes the development and testing of a tool to classify the ecological status of rivers in the UK using macrophyte survey data. Macrophytes are water plants that are visible to the naked eye. The UK has an obligation to develop such classification tools to meet the requirements of the Water Framework Directive (WFD).

Previous use of river macrophytes for ecological assessment of rivers has ranged from the selection of sites for conservation designations, to the detection of impacts of nutrient enrichment from point sources or of the effects of water abstraction. In previous uses, there has been limited referencing to unimpacted conditions. The WFD adopts a more holistic approach to ecological assessment, based on structure and function of different biological quality elements. This is philosophically different to traditional approaches to biomonitoring in Europe, and closer to concepts of biotic integrity or ecosystem health.

The development of a classification tool for river macrophytes followed a number of steps. These included data collation, development of survey methods, construction of a river typology and identification and validation of macrophyte metrics, followed by screening of reference sites, modelling of biology expected under reference conditions, establishment of an ecological basis for class boundaries, methods for deriving a single status value from a multimetric approach, and determination of the uncertainty associated with each water body classification.

Macrophyte survey data were collated from a range of sources but most data were provided via the UK conservation agencies and the Environment Agency. Over 6,500 surveys were collated, providing comprehensive coverage of the UK river resource in terms of geographical distribution and environmental conditions. Survey data for each water body were matched to basic environmental and, where possible, pressure data, covering nutrient chemistry, catchment land cover, physical habitat modification and modelled flows. Data was obtained by two survey designs; either paired 500-m reaches or single 100-m reaches, often located upstream and downstream of a point source.

A standardised protocol for river macrophyte surveys is recommended based on surveys of 100-m reaches that covers a wider range of taxa than are used in the current Mean Trophic Rank (MTR) survey method, and which focuses recording on the active channel. Over 500 surveys to this design have been carried out within the UK environment agencies since 2005 for WFD monitoring purposes. Most European countries use a river macrophyte survey method that closely follows this design.

The UK river resource was stratified into 16 types on the basis of two environmental variables (alkalinity and slope) that have a strong influence on river macrophyte community composition and productivity. A subdivision of high-alkalinity rivers was required to separate those on hard, naturally infertile geologies in the north and west from those on easily weathered geologies in the south and east of Britain.

Metrics reflecting the composition (River Macrophyte Nutrient Index, RMNI), richness (numbers of hydrophyte taxa and hydrophyte functional groups) and abundance (cover of green filamentous algae) of vegetation were developed to reflect different aspects of the WFD normative definitions. Relationships between individual metrics and various pressures were assessed to validate their performance. Metric-based approaches are more resilient to the effects of undersampling and variable detection rates between observers than methods based on taxonomic identity. A multimetric approach offers several advantages: (i) sensitivity to a range of pressures that may have contrasting or independent effects on aquatic vegetation, (ii) compensation when compositional metrics can be derived from samples that are impoverished in terms of cover or

richness, and (iii) complementary sensitivity to key pressures such as eutrophication which appear to exhibit hierarchical effects on vegetation according to river type and degree of enrichment. Individual metrics appear to be sensitive to different pressures. However, they are best used collectively as an indicator of general degradation, rather than individually in any diagnostic sense, since different pressures may have a similar biological signature and will therefore be difficult to identify reliably, while sites will often be subject to multiple stressors.

Ecological status is the ultimate currency of the WFD. It is a measure of the degree of deviation between test sites and minimally impacted reference sites. In this project reference sites were established initially through type-specific screening using linked data on water chemistry, land cover and hydromorphology. In lowland base-rich rivers putative reference sites were only incorporated into models after they were found to lie within the range of predictions. Opportunities for reconstructing reference conditions using nineteenth century botanical records are more restricted for rivers than for lakes. Screening on biological criteria, such as minimum richness and cover, proportion of tolerant taxa and proportion of acidophiles, was used to refine site selections based on pressure data, or as a substitute when pressure data did not exist.

To predict the flora expected under reference conditions, the metric values of the population of reference sites were predicted, rather than the taxonomic composition of the flora itself. Temporally invariant and unimpacted properties of rivers, such as altitude, slope, altitude at source and distance from source, and aspects of catchment geology such as alkalinity, were used as predictors. Trials using the metric RMNI indicated that site-specific predictions using generalised linear models were superior to type-specific predictions in terms of minimising the variation between observed and expected metric values within the reference site population. Where possible, site-specific models were developed for all other metrics. In some cases, a default to a type-specific classification may be required if adequate data cannot be obtained to run site-specific models.

Observed metric values in test sites were expressed relative to values expected in reference sites in the form of an Ecological Quality Ratio (EQR). For compositional metrics, a conceptual framework was used to align class boundaries with the WFD normative definitions. This relied on the allocation of species to functional response groups describing the level of sensitivity of a species to eutrophication. The middle of moderate status was envisaged as an equilibrium point in the relative cover of tolerant and sensitive species. The good/moderate boundary was defined as the cross-over point minus the prediction error, on the basis that undesirable impacts associated with dominance of tolerant taxa are unlikely at this point. Statistical approaches based on the frequency distribution of EQR values in reference sites were used to set class boundaries for other metrics.

Different options, including averaging and worst case, were considered for combining individual metric EQRs to provide an overall EQR for the water body, on which its ecological status would then be based. An exploration of pressure-metric relationships at a type-specific level indicated that a rule based-approach would be the best option for combining metrics, to reflect contrasts in the value of different metrics in different river types or at different intensities of pressure. The rule-based approach attributes greater weight to richness metrics at higher levels of fertility. A range of case studies are provided to illustrate the large-scale geographical distribution of water bodies by type, to assess longitudinal changes in EQR in major UK rivers, and to assess the ecological status of riverine Special Areas of Conservation (SACs) designated for their aquatic vegetation.

Analysis of uncertainty in the overall river EQR, associated with sampling, temporal and spatial sources of variation, indicates that three macrophyte surveys per water body on a given sampling date are sufficient to control the effects of spatial variation on confidence of class.

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1 Introduction

1.1 The WFD and the need for classification of rivers based on macrophytes

1.1.1 Objectives

Implementation of the EU Water Framework Directive (2000/60/EC) by the UK environment agencies and other authorities in Europe requires a change in the approach to monitoring and reporting on the state of our surface waters. Under the terms of the Directive, member states are required to differentiate their surface water bodies into “types”, the types being defined by a range of physical and chemical factors (Water Framework Directive Annex II). There is then a requirement to define type-specific “reference conditions”, and to assess the ecological status of water bodies, classifying them by measuring their deviation from the reference condition. This assessment requires knowledge of a range of hydromorphological, physico-chemical and biological elements, as prescribed in Annex V of the Directive. The biological elements required for classification of ecological status in rivers and lakes include the composition and abundance of the aquatic flora, which includes macrophytes (larger plants of freshwater which are easily seen with the naked eye, including all aquatic vascular plants, bryophytes, stoneworts (Characeae) and macro-algal growths (CEN, proposed)) and phytobenthos (all phototrophic algae and cyanobacteria that live on or attached to substrata or other organisms, rather than suspended in the water column (CEN, proposed)).

This report describes the development and testing of a tool to classify the ecological status of flowing water bodies in the UK through the use of macrophyte survey data.

1.1.2 Macrophytes as a Biological Quality Element

Macrophytes represent one part of the biological quality element defined by the WFD as ‘macrophytes and phytobenthos’. The separation of these terms is vague, and macrophytes have often included macroalgae, which some definitions place within the phytobenthos. Currently, diatoms are treated as a proxy for phytobenthos, leaving a relatively clear distinction with macrophytes, although other European countries have interpreted this distinction in different ways. Tools for classifying surface waters based on macrophytes and diatoms have been developed independently, largely reflecting the traditional separation of ecological research on macrophytes and diatoms. However, in the future it will be necessary to resolve differences in classifications based on these sub-elements and to devise a method for combining them. For the present purposes, the term ‘macrophyte’ follows the Comité Européen de Normalisation (CEN) (2003) definition of ‘*larger plants of freshwater which are easily seen with the naked eye…including all aquatic vascular plants, bryophytes, stoneworts and macroalgal growths*.’

1.2 Importance of macrophytes in river functioning

At the simplest geomorphological level, rivers are water-powered conveyor belts that transport sediment of different sizes at varying rates and efficiencies. Within this image macrophytes have a relatively passive role. In general, macrophytes appear to have a more integral role in the ecology of lakes than of rivers. For example, they influence substrate and water chemistry, for instance by oxygen release in the rhizosphere or by nutrient sequestration, stabilise substrate, affect biogeochemical cycles, contribute to productivity, provide substrate for epiphytic algae and their grazers, and act as a food source for fish and water fowl (Jeppesen *et al*.,1997). However, many of the roles ascribed to macrophytes in lakes remain pertinent in rivers (Sand-Jensen *et al*., 1989), especially outside the low order streams of the uplands where allochthonous production is likely to dominate. Due to short retention times, phytoplankton standing crop is normally low in rivers. Therefore macrophytes cannot be assigned the same ecological importance they merit in shallow lakes, where stable plant-dominated ecosystems are dependent upon macrophytes to buffer predatory interactions between fish and zooplankton. On the other hand, by increasing channel roughness, especially in the middle to lower reaches of rivers, macrophytes exert a significant influence on efficiency and spatial variation in transport of both water and sediment (Dawson, 1988), as well as regulating fluxes of key nutrients (Kleeberg *et al*., 2007). In these respects they constitute a key element of hydromorphology in their own right. As refugia they are probably more important in rivers in promoting general ecological resilience to high flow events than buffering interactions between trophic levels.

Given the graphic consequences of macrophyte loss, there can be no disputing the pivotal importance of macrophytes in the functioning of lakes. Given the limited cover and dynamic nature of macrophytes in many rivers, their role would seem less critical. However, as a major source of primary production in rivers they interact directly with higher trophic levels in a variety of ways; they directly or indirectly determine the level of physical habitat support for macroinvertebrates and fish, and regulate basic geomorphological processes. Consequently, holistic assessment of the ecological condition of flowing waters would be incomplete without the inclusion of macrophytes.

1.3 Use of macrophytes for river assessment

1.3.1 Previous use

There is a long standing interest in the ecology of river macrophytes; County Floras from many of the more populated areas of Britain testify to the efforts of nineteenth century botanists to record the flora of their local rivers, with the scarcer pondweed species and their hybrids often receiving particular attention. However, at this time rivers tended to receive substantially less attention from botanists than more readily accessible and less physically challenging environments, such as canals and lakes. Consequently, prior to the 1970s, with a few exceptions such as the work by Butcher (1927, 1933), there is a dearth of systematic accounts of the full vegetation of particular sections of river. The 1970s saw a sharp rise in interest in the ecology of riverine vegetation, in the UK (Haslam, 1978; Holmes, 1983; Holmes & Newbold, 1984),and Europe (see Kohler, 1978; Wiegleb, 1981). Since 1990, with greater availability of direct gradient ordination techniques for analysing species-environment relationships, studies using national or international datasets have substantiated the view, hitherto based mainly on experimentation and site-specific observation, that factors such as alkalinity, slope, nutrient supply, substrate and flow regime are among the major causes of variation in river macrophyte communities (see Holmes *et al*., 1999; Baattrup-Pedersen *et al*., 2003; Demars & Harper, 2005).

To date, the main uses in the UK of information on river macrophytes have been (i) for conservation assessment and inventory purposes and (ii) for environmental assessment. The approaches used largely have their origins in surveys of freshwater Nature Conservation Review sites and subsequently Sites of Special Scientific Interest (SSSIs), initiated in the mid 1970s. Following the survey of over 1,000 paired 500-m sites on rivers across Britain, a TWINSPAN-based classification was performed to provide a botanical typology of rivers based on their emergent and aquatic vegetation. This generated 56 subtypes (Holmes, 1983) which were aggregated into 10 main types. This coarser typology was used, in conjunction with other criteria such as species richness, numbers of nationally and locally rare species, and diversity of *Potamogeton* species, to prioritise sites for legal protection and to justify their selection (Nature Conservancy Council, 1989). In 1999 this classification was revised following the addition of an extra pool of surveys carried out during the 1990s, which were intended to provide wider geographical coverage, assess community stability, and incorporate the influence of multiple observers (Holmes *et al*., 1999). This yielded 38 subtypes and 10 coarser types that largely followed the original classification. More latterly, this revised classification of river types has been used to guide the selection of rivers to fulfil the UK's obligations under the 1992 Habitats Directive to select, designate and protect Special Areas of Conservation (SAC), in this case Habitat H3260: Watercourses characterised by *Ranunculion fluitantis* and Callitricho-Batrachion vegetation (Hatton Ellis *et al*., 2003). In general, the primary interest in all these analyses has been in differences in river macrophyte composition over large spatial gradients, rather than in the assessment of changes within water bodies over time, and results have been used in a largely descriptive or prescriptive manner, including an assessment of the environmental characteristics of different types and subtypes. However, assessment of temporal change is now an integral component of monitoring the status of designated sites and the JNCC (Joint Nature Conservation Committee) method is now used for reporting the results of monitoring on a six-year cycle.

To date, the most notable development in the use of river macrophytes as biological indicators is the Mean Trophic Rank (MTR) developed by Holmes (1995). This system arose from an initial ranking of trophic preferences of aquatic plants pioneered by Newbold & Palmer (1979). At the same time, Haslam (1978) developed similar concepts on trophic preference of river plants. The approach used by Newbold & Palmer (1979) was developed into an expert ranking system for river plants by Holmes & Newbold (1984) and subsequently refined into the MTR system by Holmes (1995). Through the MTR river plants contribute to the requirements of the Urban Waste Water Treatment Directive (UWWTD), the system requiring a comparison of MTR values for the vegetation upstream and downstream of sewage treatment work (STW) discharges. An expert ranking system based on macrophyte response to flow is also used in water resource management in rivers (through input to Catchment Abstraction Management Strategies, CAMS). In contrast to rivers, lake macrophytes have never been used formally in the UK in statutory environmental assessment.

Compared to lakes, biological assessment of rivers in Europe is fairly advanced. Reference-based systems of assessment using macroinvertebrates have existed in many countries for several decades (Wright *et al*., 1998), while the use of diatoms in a similar fashion is gaining momentum. However, palaeoecological approaches, based on the record of diatoms, chironomids or crustaceans in lake sediment (see Bennion & Battarbee, 2007) have enabled successful reconstruction of environmental change in many lake catchments, but have rarely been used in floodplain aquatic habitats (such as Hilt *et al*., 2008). Such techniques could contribute to the derivation of reference conditions for heavily impacted lowland rivers. Schuetz *et al*. (2008) provide a recent example of the use of historical data on macrophytes (such as herbarium specimens) to derive reference conditions for the Upper Danube River, although in the UK such approaches appear to have greater potential for lakes than rivers. Most conservation-based uses of botanical data for ecological assessment of rivers, whether in the UK or elsewhere, deal with contemporary values and use expert opinion to define the quality of sites directly from this data (or via indices derived from the data), sometimes using the best and worst of what is available as a benchmark. While naturalness formed one the criteria proposed for conservation assessment by the Nature Conservation Review (Ratcliffe, 1977), there has usually been no attempt to exclude sites subject to anthropogenic pressures, provided that other criteria, such as diversity, rarity, typicalness and representativeness, can be satisfied. The focus has been to designate the best examples of what is available; deviation in the observed ecology from that expected under unimpacted reference conditions has been largely ignored. In terms of use of macrophyte survey data for environmental assessment, the limitations of upstream-downstream comparisons and their failure to control for variation in a range of independent factors, have gradually emerged. The need for an environmentally referenced model for river macrophytes, that would function in a similar way to RIVPACS by predicting the flora to be expected under a range of different unimpacted conditions, and which could then be compared to the observed flora, was recognised some time ago. Thus, PLANTPACS, a scoping study designed to investigate the feasibility of such a system (Maberley *et al*., 1999), was a precursor to this work.

1.3.2 Design of assessment systems: pressure diagnosis versus structure and function

Diagnosis of pressures is sometimes regarded as the acid test of biological assessment. Different metrics or indices have been developed for various groups of organisms to provide sensitivity to particular pressures. However, the use of community level biological data collected in the field is a world apart from single species toxicity tests conducted under highly controlled conditions. Natural environments fluctuate over the short term, or exhibit longer term climatic trends; they support genetically diverse populations of individual species that interact with one another; they differ in their connectivity with source populations or the barriers they present to dispersal; they are affected by multiple pressures with potentially additive or synergistic effects when these overlap; they have attributes that enhance the resistance or resilience of natural populations to disturbance or which directly mitigate the effects of some pressures. Therefore, expectations of the diagnostic potential of community level data need to be tempered with a large dose of realism.

In fact it is the protection of ecological structure and function, rather than diagnosis of pressures, which lies at the heart of the WFD. Success in achieving this is measured in terms of 'ecological status' which is '*an expression of the quality of the structure and functioning of aquatic ecosystems*' (WFD, Article 2.20). Ecological status is the ultimate currency of the WFD. Consequently the required model for classification is holistic assessment rather than diagnosis, which demands a philosophically different approach. In this respect the assessment goals of the WFD could be considered closer to concepts of ecosystem health or biotic integrity. Having defined ecological status in terms of structure and function in Article 2.21, the WFD makes no further mention of these terms. Consequently, it is assumed that consideration of the type and range of quality elements referred to, and correct interpretation of the associated normative definitions of ecological status, will equate to the assessment of ecological status. In the case of macrophytes, these definitions state that at high status the taxonomic composition must correspond totally or nearly totally to undisturbed conditions and show no detectable changes in average abundance. At good status, slight changes in composition or abundance are permitted provided these are not associated with 'undesirable disturbances'. At moderate status, the community is significantly more distorted than at good status, while moderate changes in abundance will be evident. Major and severe alterations equate with poor and bad status respectively.

Ecological status must clearly be considered in the light of the full suite of pressures to which a biological quality element is likely to be sensitive. For example, Dodkins *et al*. (2005) recently developed a multimetric model based on river plant species optima and niche breadth along gradients of silt content, pH, nitrate, dissolved oxygen and conductivity for diagnosing impacts on streams in Northern Ireland. However, the ability to accurately diagnose individual pressures must be considered a secondary aim compared to the overall assessment of ecological status. Indeed, with hindsight, the desire to develop diagnostic tools could be considered a 'red herring' in the initial stages of this and other WFD projects designed to build classification systems. The case for diagnostic biological tools can be traced to the perceived need to inform River Basin Management Plans on the most suitable Programmes of Measures to redress ecological degradation. However, under Annex II of the WFD, Member States are required to collect information on a wide range of anthropogenic pressures, and to undertake risk assessments of their surface water bodies based on this data. Environmental standards have been established for physico-chemical quality elements that should support the most sensitive biological quality elements at high or good ecological status (UKTAG, 2006). Cross-referencing environmental data to such standards ought to highlight the risk of failure to achieve environmental objectives. Consequently, while pressure diagnosis may prove a useful supporting component of biological classification tools, there should be adequate primary evidence from environmental data to reveal the causes when biological quality elements indicate that a site is below good ecological status.

The usefulness of macrophytes for biological assessment of rivers has been questioned on the grounds that many species have wide ecological amplitude and thus low indicator potential (Paal *et al*., 2007) or respond to multiple, often overlapping pressures (see Demars and Thiebaut, 2008), though in practice neither feature is unique to macrophytes. These properties would seem incompatible with a strongly diagnostic model of assessment. However, this problem can be overcome by adopting a more holistic approach to assessing ecological status. Indeed, strong diagnostic potential comes at the price of reduced sensitivity to multivariate pressures and, from this perspective macrophytes are well suited to assessment of general degradation. To make this type of assessment, it is necessary to go beyond the traditional weighted-average compositional metrics used in invertebrate and diatom-based monitoring in Europe and to introduce a broader spectrum of metrics that reflect the range of structural and functional attributes of macrophytes. Recent examples with aquatic macrophytes in standing waters suggest that compositional metrics used in isolation can be misleading on ecological quality (Croft and Chow-Fraser, 2007), or have little value alongside metrics reflecting the richness or gross structure of the vegetation (Hatzenbeler *et al*., 2004).

A model for this holistic approach is the assessment of biotic integrity of stream fisheries in the US (see Hughes *et al*., 1998, McCormick *et al*., 2001). This approach has been adapted for the development of multimetric systems for assessing the habitat quality or biological integrity of wetlands or lakes using aquatic and emergent plants in parts of North America (see Miller *et al*., 2006; Mack, 2007; Rothrock *et al*. 2008) and New Zealand (Clayton & Edwards, 2006). In Europe, the development of plant-based multimetric systems to assess aquatic habitats is still in its infancy. In the UK, the Predictive System of Multimetrics (PSYM) for ponds (Biggs *et al*., 1998) and in France, an assessment of aquatic habitats of the Rhine floodplain (Tremoliere *et al*., 2007) are among the few examples. However, despite having its origins in streams, global applications of this approach using macrophytes have strongly favoured standing waters. An assessment of the integrity of rivers in Portugal (Fereira *et al*., 2005) is among the few published studies to successfully apply the Index of Biotic Integrity (IBI) template to macrophytes in rivers.

1.4 Project objectives and report structure

This report outlines the development of a tool for classifying the ecological status of lakes and rivers in the UK using macrophytes. The project has become known by the shortened name LEAFPACS to reflect its function as a prediction and classification system using plants.

The development of classification tools follows a number of logical steps from the initial collation of data through the development of metrics for assessments and culminating in an overall EQR and class for the site. The report is arranged to reflect this sequence of steps and the aims of the project as summarised in Table 1.1. Although this sequence has been followed for both lakes and rivers, the detail and available data differs. Recognising the fact that reports on lakes and rivers are likely to be consulted by different individuals, and, to avoid producing a single, unwieldy report, lakes and rivers are covered as separate volumes.

Table 1.1 Steps in the construction of a classification tool in relation to the structure of this report

Step	Objective	Functions	This report
1	Collate, quality control and cross-match archived data	• Maximise value of large pre-existing datasets • Review data collection methods • Raw material for establishing metric-pressure relationships • Allows assessment of temporal change • Contributes to uncertainty assessment	2.2 - 2.4
2	Define and test protocol for data collection	• Provides raw data for metric calculation • Testing contributes to uncertainty assessment • Highlights opportunities for quality control	2.1
3	Define typology	• Stratify resource according to key drivers to reduce natural variability in reference conditions	3.0
4	Metric development	• Identify pressure-sensitive metrics covering attributes of the quality element covered by the normative definitions	4.0
5	Establish reference condition philosophy and identify reference sites	• Interpret normative definitions • Apply screening criteria informed by biology-pressure relationships • Identify population of sites showing minimum distortion	5.0

Step	Objective	Functions	This report
6	Predict site-specific metric values at reference condition	• Estimate metric value for any given site under reference condition using the combination of unimpacted variables that minimizes the prediction error	6.0
7	Compare observed and predicted metric values	• Calculate EQI for all metrics	6.0
8	Derive class boundaries	• Stratify EQI gradient to assign metric values to classes • Requires statistical approach based on frequency distribution of reference EQI or protocol consistent with biological interpretation of reference condition	6.0
9	Normalise and combine metrics	• Provides site EQR • Achieve overall face value classification of a site based on a suite of metrics	7.0
10	Present type-specific biology at different status	• Provides transparent biological link to classification results • Effective for communicating results to practitioners and wider public • Guiding image for restoration	
11	Determine variability in site EQR due to different sources	• Calculate Confidence of Classification (CoC) • Enable recommendation of sampling protocol (frequency, timing, spatial replicates etc) that will optimize ratio of sampling resource to CoC	8.0

2 Methods

2.1 Biological data acquisition

2.1.1 Approaches to river macrophyte surveys

Generally, the most widely used survey approach, developed originally to survey rivers using the Mean Trophic Rank (MTR) technique to test compliance under the UWWTD, is a wading survey of a standard 100-m reach, estimating cover of different taxa by eye. Material is usually identified to species level where possible, with the exception of macroalgae which are typically identified to family or genus. Voucher specimens of rare or critical taxa are usually retained for confirmation by experts. This approach has been subject to *ad hoc* trials in its development and during subsequent training of surveyors. Staniszewski *et al*. (2006) reports on inter-surveyor variability in an exercise conducted as part of the EU STAR programme. The choice of a 100-m reach reflects pragmatism over survey effort and the need to find relatively homogenous reaches for the purpose of comparing reaches upstream and downstream of point source discharges. Although rarely justified, this is the most common sample length employed in surveys of river macrophytes in Europe and most countries appear to have simply followed the example set by MTR. A second approach, based on fixed lengths of 500 m, has been widely used in the UK for conservation inventory purposes, and this unit has been recommended by the UK conservation agencies in the Common Standards Monitoring Guidance for Rivers (JNCC, 2005). A similar length is used in some European countries on very large rivers, such as the Danube. CEN (2003) guidance on river macrophyte surveys encourages a more formal investigation of relationships between species richness and river length as a means of establishing the best survey length, but this appears to have been largely ignored. Surveys conducted to the above designs are typically repeated every two to three years. Pentecost *et al*. (2009) provides a more detailed review of the state of the art.

No river macrophyte survey data was collected specifically for the development of this classification tool, although data collected in the UK since 2005 complies with the survey recommendations in this report. A related project in progress is examining sources of variability in river macrophyte surveys, to establish the most reliable and cost-effective methods for river macrophyte surveys (Davy & Garrow, 2009).

2.1.2 Problems in surveying river macrophytes

The common perception of surveying river macrophytes, based on a survey of a medium-sized, wadeable, degraded, lowland river under low flow conditions, is that such surveys are comparatively easy. Unfortunately, this scenario is the exception rather than the rule and this perception is ill-founded. Problems or sources of variability can arise when surveying rivers or in quality controlling the data collected. These need to be taken into account when interpreting data or making recommendations for new data collection. Survey variability is a concern of all ecologists working across a wide range of disciplines and environments. It is as relevant to surveys of birds in tropical rain forests or fish on coral reefs as it is to macrophytes in rivers and the basic issues of detection bias and accuracy of identification are generic. However, river-specific issues may amplify the influence of identification and detection bias on the repeatability of river macrophyte surveys.

- UK rivers span a marked gradient of oceanicity from small, generally oligotrophic, northern-atlantic streams to larger, generally fertile, lowland southern continental rivers. Relative to the total length of drainage network the number of river types is large and the total pool of species is correspondingly high.
- The number of species that occur at any one site on most rivers is relatively small (around 10-20 in a100-m reach). Even in rivers of the same type and where physical habitat is superficially similar, two sites may share surprisingly few species. This high spatial turnover reflects, amongst other things, the dynamic nature of the habitat, dispersal limitation, establishment lotteries and the quality and degree of connectance with tributary species pools or floodplain habitats. The overall result is that the pool of species that may occur at any given site is relatively large.
- The flora of rivers in the UK is dominated by several vascular plant groups (*Callitriche* and *Ranunculus*) that are morphologically variable and/or which show form reduction of critical features. Reliable identification may therefore be difficult or impossible.
- Macrophytes that occur in rivers span a wide range of groups from macroalgae to bryophytes and vascular plants. A high level of identification expertise in more than two of these groups is exceptional among surveyors.
- A recording unit of fixed size may not deal adequately with different groups of plants that may vary in size by several orders of magnitude.
- While it is possible to construct a short list of river plants, and to use only the membership of this list for classification purposes, in practice a much larger number of non-aquatic species can and do occur in or beside rivers. Some experience of these species is therefore required to distinguish them from species that are the focus of recording and classification.
- The aims of river plant surveys and subsequent approaches to treatment of data are varied and a one-size–fits-all approach may not be valid.
- Turbulence, high velocity, turbidity, a slippery or uneven bed, deep water or precipitous banks can all create problems in accessing sites safely or viewing and recording macrophytes using a common approach.
- In many rivers the available habitat for a large number of species is restricted due to factors such as substrate instability, extensive bedrock, high water velocity, tree shading. Whether surveyors locate suitable microhabitats for particular species may be a matter of chance rather than experience.
- Since water levels fluctuate with changes in flow, it is difficult to delimit the resource to be surveyed. A highly restrictive approach based on the base flow channel (for example, channel inundated for over 95 per cent of the time) may ignore much of the predominantly aquatic resource and associated species. A more general approach encompassing parts of the channel submerged for most of the year (over 50 per cent of the time) may be considered too wide-ranging and difficult to define.

- Rivers are generally sparsely vegetated and support few species at any one site. Therefore, losing or gaining one species has a large effect on the total. The highly patchy nature of the vegetation also hinders a more closely regimented sampling protocol. Except in a few well-vegetated, fairly uniform lowland rivers, standardised sampling approaches of the type employed in lakes or canals would be difficult to apply to rivers, and would carry a high chance of failing to detect any plants, even in reaches where vegetation was obviously present.
- A universal survey approach that can perform equally well in a medium-sized wadeable gravel bed river, a torrential headwater stream and a large deep lowland river, is difficult to define.

2.1.3 A standardised method for river macrophyte survey

Biological survey methods need to ensure that the data collected is fit for purpose and is collected in a manner that is as repeatable and efficient as possible, given the constraints of the environment and organisms concerned. There are numerous reasons for undertaking river macrophyte surveys. The need to classify rivers according to their ecological status, based on the extent of deviation from reference conditions, is the basic purpose for undertaking WFD surveys and is therefore rather different from the stimulus for most previous macrophyte surveys. Other reasons for carrying out macrophyte surveys include conservation assessment, monitoring of population sizes of individual species, mapping of spatial distribution of macrophyte patches, increasing autecological knowledge of rare species. Given that the goals of such surveys, and the data needed to fulfil these aims, vary widely, a single approach is unlikely to be optimal for data collection.

The basic survey method proposed for the collection of river macrophyte data for WFD purposes involves simple refinements to the existing MTR survey protocol. Macrophytes are surveyed in 100-m reaches using the same nine-point cover scale for expressing abundance. The principal differences are in the greater number of taxa to be considered, extent of the recording zone, disposition of survey reaches and emphasis on quality control. Further details are given below. The justification for this approach is elaborated on in subsequent chapters, particularly in relation to the uncertainty analysis in Chapter 8. However, the main reasons for the use of this method can be summarised as follows:

- The classification system itself is contingent upon archived data collected to one of two standard designs; consequently, it is difficult to justify a fundamentally different approach.
- The longer the reach surveyed, the greater the variability between surveyors in terms of the species recorded and the cover scores awarded to taxa common to both surveys.
- The longer the survey reach, the greater the proportion of taxa present that a surveyor will fail to record.
- The longer a survey reach, the fewer the reaches that can be surveyed per water body and therefore the more difficult it will be to accommodate large scale spatial variation.
- Longer survey reaches (500-m, one-km and so on) will tend to encounter microhabitats of limited extent that typically would be too localised to be encountered in shorter reaches. This increases the chance that locally rare

taxa will also be found in longer reaches. If such taxa are useful pressure indicators, this may reduce the ability to discriminate between reaches within a water body subject to differences in type or intensity of impacts. Shorter reaches (100-m) should be superior for this purpose.

- This method yields data suitable to compare surveys on the basis of different metrics. While comparisons based on primary species data are likely to be flawed due to incomplete detection rates, metric-based comparisons are more conservative and less strongly affected by missing species.
- Based on the uncertainty analysis carried out in this project, the variability between surveys of the same reach, when measured as an Ecological Quality Ratio, is sufficiently small for a water body to be classified with 95 per cent confidence based on three surveys, when the mean EQR is located in the middle of a class.

This is henceforth referred to as the 'standard method', not MTR, to recognise that surveys using this method have not always been conducted for UWWTD purposes and that the MTR as a system will not itself be used for WFD purposes.

UK conservation and environment agencies are currently investigating variation between river macrophyte surveys conducted to different designs, and the influence of different observers. The results of this project are discussed in Davy & Garrow (2009), and will be considered in a review of river macrophyte survey practices.

Macrophyte recording list

Fixed recording lists need to be used with caution but are necessary for recording a group of taxa that is already delimited (aquatic macrophytes), rather than everything growing in a particular place. Moreover, when the intention is to compare metrics derived from primary biological data, reference to a fixed recording list ensures a consistent basis for comparison of metric values between sites or over time. When there is a fixed list, surveyors are required to record all taxa encountered on this list at the highest possible taxonomic level, and thus lack of a record implies the species was absent or too rare to be detected. When there is an open list, the decision on what to record is left to the judgement of the surveyor. In this case differences between surveyors might reflect genuine inter-observer differences, differences in interpretation of the definition of an aquatic macrophyte, differences in interpretation of the limits of the recording zone or differences in the thoroughness of recording non-aquatic species growing as accidentals within the channel. The alternative is an explicit instruction to record everything growing within the designated recording zone. Standardisation is applied at a secondary stage when metrics are derived from primary survey data.

The main disadvantage of fixed recording lists is that they are exclusive by definition and surveyors will routinely encounter taxa not present on this list when working within the correctly interpreted recording zone at almost any site surveyed. While these non-aquatic taxa will almost always be much rarer than the listed species, there is an in-built requirement for surveyors, firstly, to be aware that not everything they encounter can or will be listed and, secondly, to discriminate between taxa on the list and those that are excluded, even if they are unable to identify all of the latter.

The decision on whether to use fixed or open lists depends partly on the purposes of the survey. If the survey is designed to evaluate the whole river corridor and the distinction between channel, banks and riparian zone is arbitrary, an open list may be appropriate. If there is a prescription to record aquatic plants associated with aquatic habitats, it is better to delimit in advance the taxa considered to fall within these definitions. The derivation of directly comparable metric values for use in classification demands that data be drawn from a fixed list of taxa.

The list of all taxa identified for inclusion was based on over 6,500 river macrophyte surveys and consultation with other national experts (Table 4.1) The list covers those taxa (vascular plants, bryophytes and readily recognisable macroalgae) that:

i. Occur obligately in river channels under base flow or below the water depth at median flow and are therefore wholly or partially submerged for more than half the year, or occupy habitats submerged for this length of time.

ii. Occur within the above zone but also sporadically higher up the inundation zone or not in strictly riverine habitats but are always of greatest abundance in rivers or the lower part of the inundation zone.

iii. Are obligate aquatic species occurring regularly in rivers, though this may not be their primary aquatic habitat.

iv. Can occur in a range of aquatic and wetland habitats but will form perennial submerged or emergent populations in rivers under base flow conditions.

The greater number of taxa (from 132 on the MTR list to 273) reflects:

i. An increase in the extent of recording zone to reflect hydromorphological pressures on water bodies rather than nutrient enrichment alone. This includes recognition of the interaction between emergent and aquatic species within the shallow water marginal zone and that some amphibious taxa can grow in a more or less permanently submerged form.

ii. Recognition that all species have some value as biological indicators and that an assemblage of species, each with wide ecological tolerance, may have as much or more information content as an assemblage represented by a single narrowly distributed taxon. Thus, all taxa score and most of the non-scoring taxa associated with the MTR list are integrated into the current list.

iii. Inclusion of some taxa, such as hybrid *Potamogeton* species, which were included in the MTR approach by implication, but not individually named.

iv. The resource to be surveyed now extends beyond the types of rivers normally covered for MTR purposes and therefore requires inclusion of some species considered too rare for inclusion in the MTR system.

v. Ecological evidence to suggest that some species previously recorded as members of an aggregate differ sufficiently to ideally be recorded to species level if diagnostic features are presented.

vi. The facility to record at various aggregate levels is important because there is an upper level of taxonomic resolution to which specimens of some taxa will be assignable. It is essential in these cases that surveyors can record such specimens at the right level, rather than ignoring them or being forced to record at a level that does not equate to the confidence of the identification.

vii. The need to maximise the biological information content of the data, and to avoid degrading records made originally at the species level down to the genera level or lower, simply for convenience of electronic data storage.

Hence, increasing the length of the recording list largely helps record at a specific level when the condition of a specimen and the expertise of the recorder permits, while retaining the option to record at a lower level if diagnostic features are lacking or a confident identification cannot be made.

The recording list is intended to provide a highly parsimonious view of the number of river plants that can be considered aquatic. Szoszkiewicz *et al*. (2006) also proposed an expansion of the range of species covered by the MTR system, citing some of the reasons listed above to support their case. Revisions to this list that involve removing species would potentially require the removal of relatively large numbers of other taxa to maintain a consistent definition. Similarly, small numbers of species cannot be readily added to this list without distorting the criteria for inclusion or exclusion. Thus, adding one or two species would imply raising the elevation of the inundation zone in which species are considered for inclusion and this would demand the addition of much larger numbers of taxa to maintain a consistent definition. However, compromises over the desirable level of identification may be feasible, if after several years of standardised data collection, sufficient evidence exists that some identifications, to for example genus level, would not materially affect classification results.

The list was devised to cover all species that a surveyor might reasonably expect to find in the channel or lower part of the inundation zone of British rivers (minus a few highly localised rarities) at a moderate level of abundance. Many other non-aquatic species can and do occur, atypically and at very low abundance in the recording zone and surveyors should be aware that this list, by necessity, cannot cover every species they might possibly encounter.

The fixed recording list excludes a number of categories of species:

- Species that occur, often frequently, by rivers in the upper part of the inundation zone but whose populations are never subjected to frequent or prolonged submergence. This includes vascular plant species (such as *Petasites hybridus*, *Symphytum officinale*, *Filipendula ulmaria*) that are widely distributed alongside rivers and often abundant but, critically, also occur frequently in non-riverine terrestrial situations (meadows, woods, verges and so on). A number of riverine bryophytes are strongly associated with the upper part of the inundation zone and may be largely dependent on a flow regime that ensures humidity, delivery of propagules, renewal of gaps and so on, but these species cannot be considered truly aquatic.

- Species that occur in habitats characterised by near-permanent soil saturation but which frequently occur independently of rivers (such as mires, heaths, flush zones) and are more indicative of the riparian zone (*Carex nigra*, *Carex panicea*, *Hydrocotyle vulgaris*, *Molinia caerulea*).

- Exceptionally rare species which, although aquatic and occurring in rivers, are extremely unlikely to be encountered, either because they are rare at a national level and rare where they occur (some *Potamogeton* hybrids, for example), or because they are more characteristic of lakes and are therefore only likely to occur sparsely in rivers in the immediate vicinity of lake outflows (such as *Nuphar pumila*, *Pilularia globulifera*).

- Species that occur more or less only in montane headwaters where there is considerable overlap between the flora of streams and flush zones or other humid microhabitats such as rock ledges.

- Species that occur more or less at or below the tidal limit of rivers.

The fixed recording list does not preclude recording of other species if desired, but this information will not be used in site classification for WFD purposes. Conversely, the absence of a cover value for any taxa in the standard list implies that the taxa was absent from a site at the time of survey (or was so rare as to have been undetectable). Hence there are no ‘optional taxa’; if a listed taxa is encountered it must be recorded.

Extent of recording zone

Surveyors should record where they are most likely to record aquatic plants – in permanently submerged parts of the channel or within the saturated zone at the margins or the lower part of the inundation zone. Aquatic species are considered those which occupy channel habitats that are inundated for more than half the year. This implies recording up to a bank height equivalent to the water depth associated with the median flow (the Q50 or flow that is exceeded 50 per cent of the time). Surveyors must be aware that the difference in elevation between water depth at summer base flow (Q90-Q95 – flow conditions under which most macrophyte surveys will take place) and at median flow in British rivers will be relatively *small* (median flows are typically two to seven times the summer base flow, increasing from stable lowland rivers to flashy upland streams). It is difficult to be precise about the limits of recording due to differences between sites in flow regime and channel geometry. However, an absolute elevation of 20-50 cm above summer base flow will generally enclose that part of the inundation zone where water level fluctuations lead to frequent and/or prolonged inundation sufficient to maintain aquatic taxa. In high-gradient, geologically constrained channels a level above this may be appropriate in areas of high turbulence and splashing. Familiarity with a site will help greatly in delimiting the recording zone. Surveyors should also check online gauging station information which will often provide data on ambient flow conditions relative to the water depth at base flow.

With practice, surveyors will be able to recognise the limits of the inundation zone within which recording should take place. The following should be borne in mind:

- Desiccation and water logging tolerances of different groups of plants can be used as a guide to the position of the Q50. Most aquatic bryophytes are intolerant of prolonged desiccation (Englund *et al*., 1997). Consequently mid-channel or marginal boulders and bedrock that are well-vegetated with pleurocarpous mosses, such as *Fontinalis*, *Hygrophypnum*, will almost certainly be fully submerged at Q50 flows. Conversely, most riparian trees, including alders and willows are likely to be situated near or above the Q50 and no woody species will have overhanging foliage submerged by Q50 flows. A large number of herbaceous vascular plants characteristic of fens and mires (such as *Filipendula ulmaria*, *Valeriana officinalis*, *Epilobium hirsutum*) will occur alongside rivers in habitats that are perennially water logged. Although many of these species will tolerate prolonged inundation by winter floods, they are not well adapted to cope with regular inundation during the growing season and thus are unlikely to occur widely below the Q50.

- The lower edges of the inundation zone are likely to be sparsely vegetated and to include frequent bare gravels, wet mud or damp sand splays.

- Trash lines and other evidence of high flows such as material trapped in trees should not be used as a guide to the stage associated with the Q50. Any such material still visible in the summer is likely to reflect the maximum height of water reached during the preceding winter and will represent flow events of a much lower recurrence (such as less than Q5).

- In the majority of cases it will not be possible to carry out macrophyte surveys when a river is running at Q50 since the depth and velocity of water will prevent safe wading. Elevated turbidity is also likely to impede viewing of the stream bed under such flows.
- The fixed species list covers taxa characteristic of the recording zone and those that surveyors are most likely to encounter there in abundance. If surveyors begin to routinely encounter unlisted species, they may have strayed above the limits of the recording zone and this will certainly be the case if unlisted species occur at cover values of three or above. When recording the less strictly aquatic taxa, only that part of the population of a species growing within the recording zone should be considered, not populations (or that part of the population) growing higher up the bank.

Distribution and frequency of recording

Although the focus in river macrophyte surveys is on the individual site, the WFD requires the classification of whole water bodies. In terms of rivers these range from a few km to more than 50 km in length. This raises the question: How many different surveys are required to adequately represent a water body, and how often should these be carried out?

Spatial variation within a water body and temporal variation at a site is considered in greater detail in Chapter 8, but is summarised here for the sake of completeness. Where multiple surveys of the same water body have been carried out on a given date, the variation between surveys is such that three standard surveys of a water body should be sufficient to enable any water body to be classified with 95 per cent confidence, when the average EQR across these surveys is located in the middle of a class. In terms of temporal variation, evidence from repeat surveys of a site indicates that one survey repeated every three years should be sufficient to accommodate temporal variation of this scale.

In terms of positioning surveys, an analysis of longitudinal change in status along river water bodies suggests there is no advantage or disadvantage to locating sites towards the bottom of a river, except in rivers, mainly in northern England, which have sparsely populated upper catchments, but flow through areas of dense population and industry in their lower courses. In general, the subdivision of rivers into equal thirds, followed by the random placement of a site in each third of the water body, is likely to suffice.

Specific approach to data collection in the field

In vegetation surveys, there is a basic requirement to record everything present (although the taxonomic resolution of recording often varies between groups). However, **all** surveys are merely a sample of the true population of taxa present, with the tendency to under-sample increasing with size of the sampling unit, spatial aggregation of the vegetation, difficulty of observation, and severity of the environment from a human perspective. Consequently, surveys of 100-m units of river channel in which vegetation is intrinsically patchily distributed, depth and turbulence constrain observation and surveyors are liable to be cold and wet, are probably more prone to under-recording than a survey of a 2 x 2 m quadrat in a mesotrophic grassland where vegetation is more uniformly distributed, there are few constraints on observation and the environment for recording is comparatively comfortable. Issues of under-sampling in rivers apply equally to novices and to experienced botanists. The latter may under-sample less if they can discriminate more readily between critical taxa in the field and know, on the basis of experience, where to look for certain plants. However, all experienced botanists will individually under-sample a river relative to the true population as detected by a group of surveyors with the same level of experience carrying out simultaneous surveys. An individual will also under-sample a site in a single survey compared to further surveys of the site by the same individual in the same growing season due to factors unrelated to the phenology of different species. Unfortunately, there have been few attempts to quantify the extent of under-sampling and it seems likely that under-sampling will vary between rivers depending on the patchiness of vegetation, difficulty of observation and characteristic dominant taxa.

This raises the question of whether it is possible to design a systematic and more structured sampling strategy for river vegetation that increases detection and reduces variability between observers. For example, the survey design for site condition monitoring of macrophytes in lakes is based partly on surveys of replicate 100-m shoreline transects with data recorded within one square metre quadrats at five points along each transect, and at a series of water depths at each point (Gunn *et al*., 2004). The common standards monitoring approach for canals (Eaton & Willby, 2003) requires surveys to be carried out as 150-m units with vegetation recorded at 10 equidistant points per site by visual assessment of emergent species coupled with sampling of submerged vegetation using a grapnel. In the Danish approach to river macrophyte surveys (Baattrup-Pedersen & Riis, 1999) data is collected in one-m quadrats that are laid consecutively along a transect across the channel, with sufficient transects being sampled to generate 100 recording points per site. However, in these examples, vegetation is relatively uniformly distributed and the physical differences between sites, that might dictate modifications to the sampling protocol, are fairly small.

Current recommendations for river macrophyte surveys in the MTR method (Holmes *et al*., 1999b), that will apply to most UK rivers, are that surveyors should subdivide each reach into 10-m segments and zigzag across the channel a minimum of four times in each segment, noting all taxa present or collecting samples. Some surveyors appear to separately record cover values for taxa in each segment before aggregating this information at a reach scale. This advice attempts to create a more systematic sampling approach but is not prescribed and the number of times to cross the channel is open to interpretation in wider rivers. Alternative provisions are made for deep narrow channels where surveys should be carried out by walking both banks, or for large slow-flowing channels, where the prescriptions for wading surveys are replaced by the use of a boat, while retaining the same strategy of criss-crossing the channel. The current recommendations are symptomatic of the difficulty in being highly prescriptive over sampling strategies for rivers in which the physical characteristics and size of the channel vary strongly between sites and the distribution of vegetation ranges from relatively uniform to highly patchy. In a 10-m wide lowland river, the recommended strategy of zigzagging across the channel has a fairly high probability of encountering most taxa present, yet in an upland river of the same dimensions where vegetation is patchily distributed the same sampling strategy would potentially under-record many species. Simply walking along each side of the channel about one metre in from the water's edge would probably record more species in an upland river than the zigzagging trajectory (although a meaningful assessment of cover might then be difficult to achieve). There are no equivalent recommendations in the JNCC survey method, the method simply specifying that surveys will usually be carried out by wading and that surveyors should discriminate between the occurrence of a species in the channel or on the banks, using a scale of one to three to reflect cover.

The overall aim of WFD river macrophyte surveys for monitoring purposes should be to collect a standardised *sample* of the vegetation at a site based on a comparable effort, such that the vegetation of a site is characterised adequately to assess its ecological status. Unfortunately, there have been no formal tests of the effectiveness of different sampling protocols in different river types, which makes it impossible to advise on improvements to the current recommendations. These recommendations provide a more systematic basis to the survey of river macrophytes and are important because they attempt to standardise the survey effort invested at a site, even though the sampling efficiency may differ between different types of sites.

With greater survey effort or a more efficient search strategy tailored to a particular type of river, it will always be possible to record more taxa regardless of expertise. For example, standardised sampling strategies are weighted in favour of widely dispersed taxa and are less likely to detect locally distributed taxa, even if their overall cover within a reach is similar (although it is a basic biological law that widely distributed species are normally more abundant where they occur than rare species). Greater survey effort and exhaustive searching of a site will undoubtedly detect more species, but if these levels of effort are required to find such taxa it is debatable whether they can be regarded as being characteristic of a site for the purposes of assessing ecological status. There are entirely justifiable reasons for adopting type-specific and more efficient sampling strategies when rivers are surveyed for other purposes, such as inventory, conservation assessment or to locate rare species. Hence, the degree to which the recommendations have been followed by different surveyors is a potentially significant source of inter-observer variability in the results of surveys, especially those of some river types in which vegetation cover is more sparse and patchily distributed. Until formal comparisons have been made, it is recommended that the stated sampling strategy be retained. A survey undertaken for several different purposes cannot serve all these purposes equally well. Surveyors who carry out surveys under contract are likely to undertake surveys for a range of different organisations and to suit different purposes and may therefore be in the habit of carrying out surveys to a personalised design. Surveyors should be adequately acquainted with the purposes of surveys and the type of sampling strategy that is to be followed.

An additional recommendation relates to search time at a site. Even adhering to the recommendations for sampling strategy, it is impossible to prescribe how long a river macrophyte survey should take, since this will vary with the width of the channel, complexity and extent of the vegetation, and logistical factors such as depth, water velocity and uniformity of substrate size, which will govern how quickly a surveyor can move safely around a site. A survey of a 100-m reach might therefore take anything from about one to three hours. However, in the interests of pragmatism and efficiency, it is recommended that surveyors note when they record each new taxa to see the rate of accumulation of new records, or consider if a survey is potentially complete when a period of 10-15 minutes have elapsed during which no new taxa have been recorded.

2.1.4 Training and accreditation for river macrophyte surveys

Apart from confirmation of voucher specimens by recognised national experts, checks on the validity of data are largely restricted to comparison of records for rarer species against known geographical distributions and consideration of the experience of the recorder. Thus, accreditation is an important step in training surveyors in *both* survey techniques and identification. Novices should be encouraged to ‘shadow’ experienced surveyors and should never lead survey teams. All surveyors are expected to have attended at least one residential field course on macrophyte identification plus a survey training workshop, and that those leading a survey team will have at least two full seasons of previous experience as a survey assistant.

The ultimate aim in river macrophyte surveys is to ensure that macrophytes are recorded accurately and in the least selective manner (low bias between recorders in terms of what they record). The best way of achieving this is via a training program designed to ensure that:

- Plants observed are identified correctly and to the highest possible taxonomic level given the characters available.
- Surveyors can distinguish between similar species on vegetative criteria where this is possible.
- Surveyors can distinguish between plants on a recording list and potentially confusable non-aquatic species that commonly occur adjacent to some rivers.
- Surveyors recognise important microhabitats in the field.
- Identification expertise is spread across a range of groups covering all species that surveyors have a high likelihood of encountering in the area where they work, rather than having a high level expertise in one group (such as vascular plants) that extends to, for example, rare hybrids, which there is an extremely low chance of encountering.

Training should **not** aim to ensure that surveyors can identify every species on a recording sheet, or to ensure that they record every species that can be found in a section of river, since both are unachievable and therefore futile aims. All surveys of all biota in all habitats are a sample of the true population. Other things being equal, surveyors are less likely to record species that are truly rare at a site, and different surveyors are more likely to record different rare rather than different common taxa. Rarity is not, however, the only cause of detection bias; surveyors may overlook abundant yet inconspicuous taxa, sometimes including species that are literally 'staring them in the face', or species which they have become over-familiar with in recent surveys. While each surveyor might find only 50-70 per cent of the true number of species at a site, the overlap between different surveyors in terms of what is actually recorded will be higher, and is likely to be increased in sites of shorter length. To be of most use, identification courses should be tailored for beginners, with advanced courses offered for those with at least one full season of field experience. A more worthwhile goal of training would be to build a comparable level of proficiency in identification across a range of plant groups (and thus habitats) with the aim of reducing *differences* in detection bias between observers.

Working in groups or pairs is an effective way to reduce errors in recording or identification, as well as improve safety. Nevertheless, unforced errors such as in field recording and transcribing data are inevitable, and all biological databases will certainly contain a small number of 'unrecoverable' errors.

2.1.5 Quality control of macrophyte surveys

Macrophyte surveys (of lakes or rivers) are rather different to surveys involving most other quality elements. The data is derived by field observation and recording with the eye of the surveyor usually serving as the only sampling device. Thus, it must be assumed that what is recorded is a correct reflection of what was actually seen. Similarly, when a species is not recorded (is absent by inference) it must be assumed that the correct techniques were employed to find that species (and that the surveyor would be capable of recognising it) and consequently that it was indeed absent or too rare to be detected within the area sampled. The overall sample is never retained. In the case of diatoms and invertebrates it is possible to revisit or audit preserved samples, but such opportunities do not apply to macrophyte surveys.

One option is to exercise quality control over surveys by resurveying a proportion of the sites covered using an independent surveyor. However, this then raises the issue of the acceptable level of similarity between surveys. Given that several experts surveying a river at the same time will produce different results (find different species, identify the same taxa differently, award different cover scores, or delimit the recording zone differently) the margins of error between different surveyors carrying out a survey of the same river on different dates must be set realistically. Even the same surveyor recording the same river on a different date would undoubtedly produce a different set of results in terms of species detected and cover scores allocated that could not be readily attributed to seasonality or conditions at the time of survey. Once the additional influence of season and present and antecedent flow conditions are incorporated, it becomes clear how difficult it is to identify the fraction of inter-survey variability that can be associated purely with *inter-surveyor* variability. Following the adage that it is impossible to step in the same river twice, it could be argued that it is impossible to undertake the same river macrophyte survey twice. Surveys for quality control purposes should be conducted 'blind' so that surveyors are unaware that they are repeating a survey and do not have access to the initial survey data; similarly, those undertaking the initial survey should not be told that this survey will be repeated on a later date. Comparisons for the express purposes of comparison with an earlier survey are in themselves biased if species found in the first survey are explicitly searched for, since this may involve a higher level of survey effort than was originally invested and will thus increase the chance of finding species not originally recorded.

Quality control of surveys should be based primarily on confirming the identity of species included in the initial survey, and ascertaining whether any species were omitted that would materially alter the character of a reach. The difficulty of controlling for time of year and viewing conditions is likely to render most comparisons of cover values meaningless.

2.1.6 Safety issues in relation to macrophyte surveys

There are risks associated with all fieldwork activities and these risks increase when work is carried out near water. It is therefore important that these risks are identified and managed by adherence to protocols.

River macrophyte surveys involve extensive time in the water for surveyors and are more comparable to fisheries surveys than to routine collection of samples for identification of benthic invertebrates or diatoms. It is strongly recommended that all surveys are carried out by two personnel. Apart from the safety advantages, double manning probably reduces survey time and is also likely to reduce survey variability between pairs of observers.

Life jackets should be worn as a basic minimum when working in freshwater. Unless streams are small and shallow it is recommended that surveyors wear membrane dry suits rather than waders or chest waders. Dry suits conserve body heat, prevent wet clothing, are naturally buoyant, and reduce the risk of being submerged or swept away if surveyors inadvertently venture into deeper fast flowing water. Heavy soled boots and chest or thigh pockets are useful features on dry suits.

2.2 Environmental data

River macrophyte surveys have been carried out for different purposes including conservation inventory and site designation, surveillance, compliance and basic ecological research. While the focus has been on the collection of biological information, supporting environmental data is essential to help screen reference sites, develop river typology, and enable site-specific prediction of expected metric values.

There is an explicit awareness of the influence of physical habitat on the distribution and cover of river macrophytes. Consequently, most surveys have attempted to collect data on variables, such as channel width, depth, substrate and water clarity. The link between biological and environmental data in rivers extends well beyond what is the state of the art for lakes, yet there have been few attempts to evaluate this physical habitat data and its collection has largely been treated as a secondary exercise to demonstrate the comparability (or otherwise) of upstream and downstream sites surveyed for UWWTD purposes.

The recommendations for collecting physical environmental data, originally employed in JNCC surveys, have largely been transferred to MTR surveys, thereby retaining a standard approach. Both methods recommend the collection of data on channel width, depth, substrate and habitat type (pool, riffle, run, slack, rapid), while shading, water clarity, and bed stability are specified additionally for MTR surveys (although the latter two variables do not seem to be consistently recorded). The approach used is categorical, the percentage of the reach belonging to each category of a variable being recorded. Thus for width, for example, the percentages of a reach in each of the following width categories are estimated; under one, one to five, five to ten, 10-20 and over 20 metres. The collection of data on most of these variables has been routine in other surveys, and, while the method or recording categories are often different, it has usually proved possible to harmonise the data into units that are directly comparable (for example, by converting substrate composition assessed as visual classes, or by direct measurement of the particle size composition of the armoured layer, into a weighted mean phi grade for the reach).

Where datasets have been collected to interpret large-scale distribution patterns, or to account for the relative importance of different variables in explaining macrophyte distributions, it has been normal to link the data collected to basic geographical variables. Thus for all JNCC data, and some research datasets, survey locations have been matched to information on key variables such as site altitude, slope, distance from source and dominant catchment geology. The same objectives have not applied to MTR data collected for routine monitoring purposes and consequently this data has no explicit link to geographical-scale environmental data.

By contrast, for compliance and surveillance monitoring, data is frequently collected from sites monitored for other purposes. Consequently a range of other supporting local biological and environmental data is readily available that is well matched in space and time and can be used to interpret the results of biological surveys. The review of the MTR method (Dawson *et al*., 1999) represents a major attempt to explore the link between river macrophyte surveys and data obtained from routine water chemistry monitoring. This study was extended to cover JNCC surveys but the exercise was restricted to sites in England and Wales and was compromised further by the small number of sites with a satisfactory spatial and temporal match. Consequently, it proved difficult to derive an underlying chemical basis for the distribution of river botanical types or to explore possible causes for variation in community composition along individual rivers that might be related to reductions in water quality. Research projects have usually collected environmental data or sites have been deliberately located to exploit environmental monitoring networks.

Current approaches to the field collection of supporting environmental data should be maintained to allow subsequent interpretation of macrophyte survey data when necessary. The variables water clarity and bed stability may be too temporally variable or subjective to be of value. Width may also be of comparatively little use since it is highly correlated with distance downstream and might be derived more accurately by measurements from aerial photography or maps. However, given the need to consolidate understanding of the relationships between pressures, hydromorphology, hydraulic conditions and ecology, collection of other physical habitat data should be maintained for the time being. Physical habitat characteristics measured in this way cannot be used as predictors of the biota expected under reference conditions, since these parameters are directly vulnerable to modification by processes, such as bed aggradation, channel incision or sedimentation, that may reflect anthropogenic pressures on the reach or elsewhere in the catchment.

2.3 Data sources and database compilation

Data used in this project was derived from several sources and surveys carried out for a range of purposes (Table 3.1). Only a small percentage of the data in the database (10 per cent) was collected post-2000, although national agencies subsequently carried out river macrophyte surveys according to the recommended design, with approximately 1,000 river macrophyte surveys undertaken in England Wales, Scotland and Northern Ireland since 2005. Consequently, the classification approach developed in this project was applied to surveys carried out more recently by or for the agencies, although the tool itself was developed independently of this data.

Macrophyte survey data used in tool development was generally more contemporary in origin than was the case for lakes (median survey year of 1996 for rivers compared to 1989 for lakes, with 17 per cent of surveys prior to 1985 for rivers, compared to 20 per cent for lakes). The bulk of the river macrophyte survey data derived from two sources:

i. Surveys carried out, according to the MTR design, by Environment Agency staff during the period 1993-2002. This body of data mainly comprises surveys of a small number of sites per river (for example, upstream and downstream of a discharge), with surveys repeated annually.

ii. Paired 500-m surveys carried out between 1976-1997, initially under the auspices of the Nature Conservancy Council, and subsequently by the national conservation agencies for the JNCC. This body of data mainly comprises one-off, end-to-end surveys of a river, with sites located three to seven km apart, that, in some cases, have been resurveyed on a single occasion 10-20 years later, generally by a different surveyor.

Although this second group of surveys were conducted to a different design their inclusion was deemed necessary, since, generally, they relate to higher quality rivers, have mostly been carried out by experienced surveyors and at the outset of the project, provided the vast bulk of data on river macrophytes in Scotland. To provide a more widespread geographical coverage, and to ensure inclusion of less impacted rivers, data collected to the MTR design was complemented by several smaller bodies of contemporary data collected to the same basic design (100-m survey lengths with cover assessed on a nine-point scale).

Contemporary surveys of some lowland river types cannot provide examples of reference conditions, given the scale and pervasiveness of human impacts on rivers in densely populated catchments. In the case of lakes, archived historical data extending back to the mid-1800s was used to help fill gaps in the reference site network. However, a survey of historical data for rivers indicated that most records were difficult to localise (beyond the name of the river), were unlikely to provide a full picture of the vascular plant flora due to selective recording, and systematically ignored lower plants which potentially represent a major component of the reference condition vegetation. Consequently, no formal attempt was made to include historical data in the development of a classification tool for rivers.

Data sources are tabulated below. These relate to the supplier of the data and not necessarily the owner or body which funded the collection of the data. All data had permission to be used in this project, this use being licensed where necessary.

Table 2.1 Datasets made available to this project and used in classification tool development

Data provider	Method	No. surveys	Time period	Notes
Centre for Ecology and Hydrology[1]	100m	600	1998-2002	Linked to Countryside Survey, mostly in England & Wales
JNCC	Paired 500m	2,226	1976-1997	For conservation inventory and typing purposes. Distributed throughout GB and NI, typically of higher quality
Environmental Protection Agency	100m	100	2003	Reference sites in Ireland selected for use in classification tool development
Industrial Research and Technology Unit	100m	281	1998	Range of sites in NI; for assessment and research
Environment Agency	100m	3,264	1993-2002	Operational and surveillance monitoring sites in E&W, typically of poorer quality
N Willby	100m	109	1996-1997	Sites throughout GB mainly of high quality, surveyed for research purposes

The current database amounts to 6,600 macrophyte surveys of almost 1,000 named rivers providing an extremely wide and comprehensive coverage of flowing waters in the UK (Figure 2.1). For all these surveys, the standard of data collected is adequate to calculate the full suite of metrics required for classification.

Given the coverage provided by these surveys, no further attempts were made to trawl the literature for additional sources of survey data. The most conspicuous dataset that was not used in this project was that collected by Sylvia Haslam, which was based on visual recording of the dominant macrophytes from bridges. Since the data was not available electronically and used a substantially different survey approach to the rest of the data in this project, it was not considered further. However, findings based on this data (Haslam, 1978, 1982, 1987) contributed to the interpretation of analyses here.

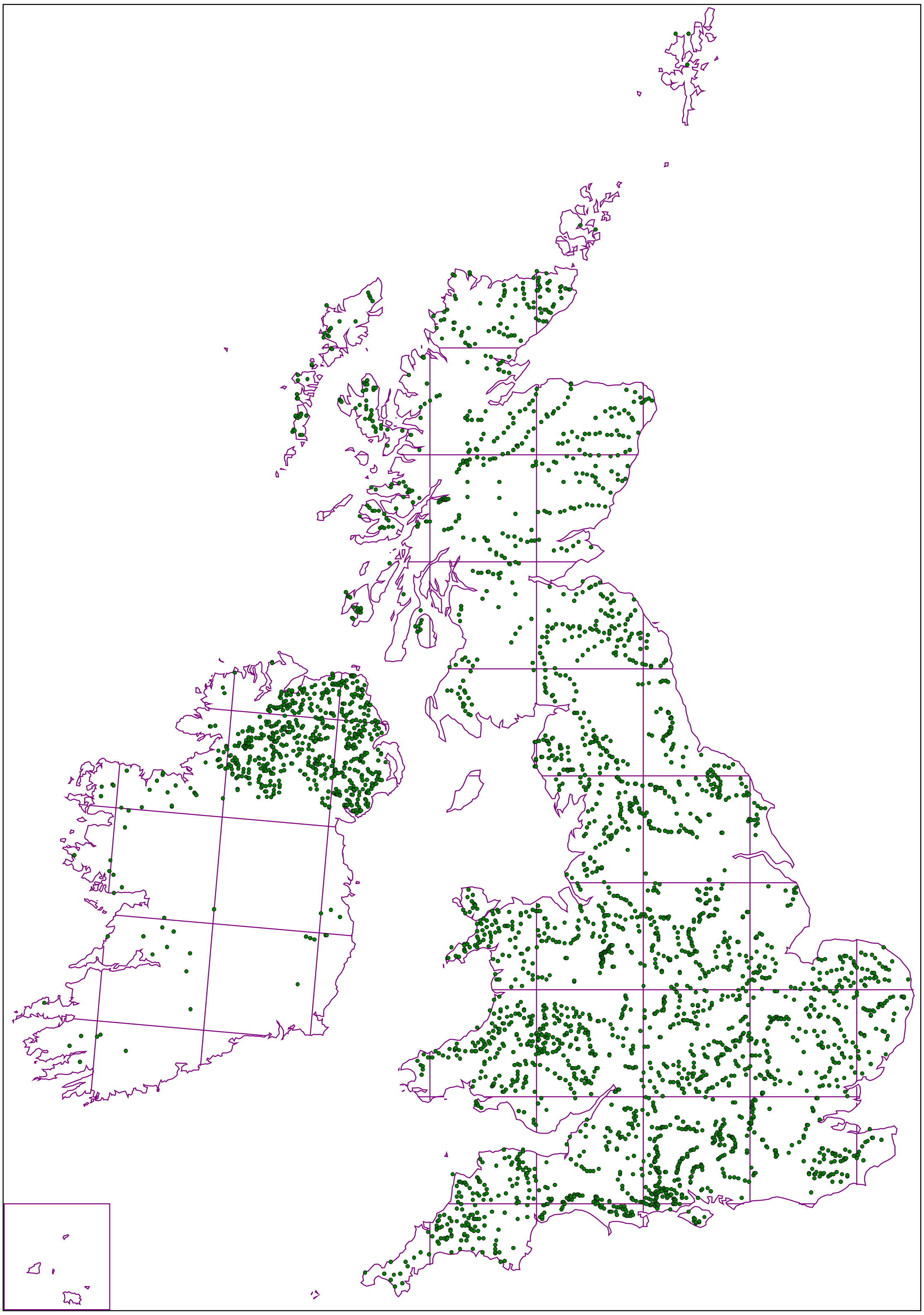


Figure 2.1 Availability of river macrophyte surveys within Britain and Ireland used in classification tool development.

2.4 Treatment of data

2.4.1 Biological data

The biological data provided was used ‘as is’, the only modifications being to misspellings and to deal with synonyms. Suspected errors in identification or transcription were corrected where it was possible to locate the primary source of the data.

The data and analyses reported here cover both aquatic and emergent vegetation, since the surveys available have conventionally given similar attention to all elements of the channel flora. In rivers, many helophyte species are not confined to marginal habitats and overlap in their distribution with more strictly aquatic species, or, in some cases have populations that are both aquatic and emergent (such as *Berula erecta*, *Myosotis scorpioides*). Moreover, populations that are exclusively emergent or amphibious under low flow will often be submerged under median flows. This applies particularly to many bryophyte species. Emergent vegetation in rivers exerts significant control on the movement of sediment and organic matter, protects banks and creates marginal habitats (Brookes, 1988). The competitive interactions between emergent and more strongly aquatic taxa in rivers, and the sheltering effects of emergent plants, have been documented by Haslam (1978). In this project, it was considered important to include data on emergent species to permit more effective assessment of a range of pressures, including those hydromorphological impacts likely be focused on the channel margins.

The list of vascular plants, macroalgae and bryophytes recorded on British rivers is large. For example, the JNCC Rivers Conservation database contains records for over 980 taxa. In practice, almost any plant species could occur in or beside a river as an accidental and surveys which incorporate the ‘banks’ include a large pool of terrestrial species that are not specific to rivers (Lansdown, 2008). An initial assessment was made of data on channel and bank species in JNCC surveys. We assessed the proportion of cover and records of all species within the channel, as a proportion of both channel and banks, to identify those species most strongly associated with the channel. However, the data in JNCC surveys for ‘banks’ included numerous species physiologically unable to tolerate inundation for 50-85 per cent of the time (the definition of banks in the JNCC survey method). Consequently, because the interpretation of banks is, at worst, likely to have included essentially terrestrial habitats and, at best, likely to be have been inconsistent, we restricted analysis to the channel component of the vegetation in all JNCC surveys.

Several metrics described here require the use of continuous scale numeric data, expressed in the form of percentage ‘cover’. Surveys of rivers usually follow a common approach in estimating by eye the extent of the bed that is vegetated within a length of channel. Values are recorded as percentages or, more usually, assigned to a cover class which is associated with a range of percentage cover. The range and scaling of cover values differs between MTR and JNCC surveys and harmonisation is therefore required prior to analysis. JNCC surveys were also stored in the form of a pair of 500-m surveys centred on a single point. Since all the available environmental data was relative to a central point, rather than each component 500-m, these surveys were pooled to give data for a one-km reach.

Table 2.2 Conversion of cover scores from different river macrophyte survey methods to standard scheme

JNCC cover score	MTR cover score	% equivalent
1	1	0.05
	2	0.5
2	3	1.7
	4	3.8
	5	7.5
3	6	17.5
	7	37.5
	8	62.5
	9	87.5

The harmonisation of cover scores follows Table 2.2. Thus, each cover value within each constituent JNCC survey was rescaled to its percentage equivalent and the two surveys were then averaged. The resulting average was assigned back to a cover score on the one to nine scale used in MTR. A re-amalgamation of MTR scores to fit the range of cover scores used by JNCC indicated that a JNCC cover score of two matched up most closely with an MTR cover score of three, while a JNCC cover score of three matched up most closely to an MTR cover score of six. There is, inevitably, substantial uncertainty over the true MTR equivalent score when a JNCC cover value of three is applied, since the range of possible percentage cover values greater than five per cent (the definition of a JNCC cover score of three) is so large. However, directly assessed cover values of eight and nine have a very low frequency in the MTR database, suggesting that the conversion of a JNCC cover score of three to an MTR equivalent of six is a reasonable approximation.

Where data was obtained from MTR surveys the data used included both scoring and non-scoring species, provided these had been recorded and entered on the database.

River macrophyte surveys are undertaken almost exclusively during the period May-September. Unfortunately, in some instances, the precise date of sample collection was not recorded or not available electronically. Consequently, the exact survey timing was not included as a variable in subsequent analyses.

The original survey data were supplied with six-figure grid references. A subset of well defined sites was chosen by selecting only those sites that could be “snapped” (automatically relocated) to sit on the 1:50,000 scale Ordnance Survey river network without ambiguity. Software was written to clean up the river names by removing words such as river, brook, burn and other ambiguous characters to leave just the river name itself. These “cleaned” sites were then run through the Centre for Ecology and Hydrology Intelligent River Network (CEH-IRN) Geographical Information System (GIS) where the river names were compared with the river name of the river stretch that the sampling point had linked to. CEH-IRN allocated a score to each site depending on the quality of the match. A site with a score of one, where the sample point river name matched the river network name, was accepted as good quality. All other values were deemed a poor match and were manually connected to the blue line network after comparing site descriptions with Landranger maps. Sites which could not be snapped unambiguously to a river were excluded from further analysis. About 200 widely dispersed sites had to be excluded because they had no national grid reference (NGR), or an incorrect NGR. All datasets were given a unique numeric ID because some sites in different regions had been given the same ID by Environment Agency staff.

2.4.2 Environmental data

All surveys which had adequate or correct locational information were snapped to the CEH digital rivers network and, from this, cross-matched to the nearest measurement points for water chemistry, flow gauging, invertebrate sampling and the location of River Habitat Survey (RHS) sites. Basic contextual information for each site on easting, northing, slope, site altitude, source altitude, distance from source (distance to the furthest upstream point on the stream network) and catchment area was derived via the digital rivers network, or if necessary, manually. For rivers situated in Great Britain the solid and drift geology composition of the upstream catchment of each site was derived from the appropriate British Geological Survey (BGS) geology layers. Solid geology was summarised in terms of the relative proportion of calcareous and siliceous rock in the catchment, based on the BGS classification used in the initial WFD water body typing exercises and, in terms of an alternative classification, based on a finer subdivision of base status, referred to as the Acid Sensitivity Class. Drift geology was summarised as percentage, sand, clay, peat or rock (no drift). The process was carried out independently for rivers in Northern Ireland with results expressed according to the same scheme. The composition of land cover in the catchment at each site in GB was extracted from Land Cover Survey 1990 maps.

To select environmental data for statistical analysis, each plant survey site had to be linked with an appropriate chemistry, RIVPACS and RHS site. This was achieved using the CEH Intelligent River Network GIS, which is capable of linking sites in one dataset with those of another using the connectivity of the river network. The software loaded the plant site dataset and one of the other datasets and then proceeded to search the network to identify any of the chemistry, RHS or RIVPACS sites which were on the same river network, within 10 km (although occasionally larger distances were set if no nearer data could be found), and had the same river source as the plant site, that is, they were not on a tributary. A table was then produced giving, for each plant site, a list of all other sites within the set distance, recording: a) the distance from the plant site to the other site, b) the Strahler stream order of the macrophyte survey site and c) the Strahler stream order at the other site.

For RHS and RIVPACS (1995 GQA survey sites) sites, the selection with the shortest distance to the plant site, either upstream or downstream, was chosen, with the additional proviso that the Strahler stream order did not differ between sites by more than one, that is, no major tributary had entered the system between the two sites.

The MTR survey data, which were generally collected upstream and downstream of a Sewage Treatment Works (STW), had an additional quality control procedure applied. Using the same approach, links were made between each MTR site and STW recorded in the Environment Agency's database of STW. A table of linked sites similar to the other datasets was derived and if a STW was identified between an MTR site and the nearest chemistry site, an alternative chemistry site, the nearest on the same side of the STW as the plant site, was chosen instead.

All chemistry data contained information on determinand, number of measures, maximum, minimum, mean and standard deviation of values, and first and last date of samples included within the average. Where detection limits were stated, values included in the calculation of the mean were set arbitrarily at half the detection limit if the concentration was found to fall below this. A quality index was provided for each determinand at each chemical sampling point indicating if there was sufficient data (over 12 samples) available:

i. in the three years prior to the plant survey;
ii. in the five years prior to plant survey;
iii. in the five years after the plant survey;
iv. between 10 years before and 10 years after the survey;
v. from the whole period for which samples were available.

In subsequent modelling (for example, to derive site-specific expected metric values), the input predictive environmental data for a water body was considered: (a) to be fixed and (b) to be representative of the site surveyed in its unimpacted state. Thus, in relation to (a), the value used for temporally variable data, such as alkalinity, is the long term average for the site, not the spot sample or annual mean from the year of biological sampling. Therefore, the expected metric value for a site should be considered temporally invariant. Consequently, where multiple surveys of a site exist, fluctuations in the EQR must reflect variations in the observed biology and not the predictive variables. If better long-term data becomes available for a site, the expected value will need to be revised and EQRs will change accordingly. Any revisions to long-term mean values must be applied retrospectively to earlier assessments prior to making judgements of change in ecological status. Generally, environmental data available for rivers is better resolved temporally than the equivalent data for lakes, with most means based on the average of at least 12 values.

In relation to (b), it is important, firstly , to resolve spatial mismatches between the location of biological and chemical sampling points (an issue that largely does not apply to lakes), how these are located in relation to one another, and whether any significant discharges enter a river between the location of biological and chemical sampling stations. It is also important to use values for predictive environmental variables that are directly representative of the site surveyed. Thus, where there is substantial fine-scale variation in factors such as slope, as might occur naturally in some upland rivers, use of average valley slope would be inappropriate in a short low-gradient reach dominated by rooted vascular plants. Conversely, where slope has been obviously impacted (for example by impoundment or loss of sinuosity), the natural valley slope should form the predictor, otherwise any impact associated with reach-scale slope modifications will be concealed. The same is true of the use of alkalinity as a predictor. Measurements based on samples collected downstream of point sources, especially on naturally lower alkalinity systems, should be used with extreme caution unless they are supported by samples taken at other points within the water body.

In addition to the aforementioned environmental data, predicted flows generated by CEH Low Flows 2000 models were available for the downstream end of water bodies in England and Wales, and corresponding to the position of river macrophyte survey locations in Scotland. These were used primarily to identify sites where anthropogenic factors influenced flow (by more than ten per cent deviation from naturalised flow).

Where available, this environmental data was used to select reference sites, validate pressure-sensitive metrics and develop supporting environmental standards. Data on RIVPACS metrics was used to support the interpretation of pressure-response relationships and to examine relationships between macrophyte and invertebrate metrics. Core variables, such as slope and altitude, were used in modelling the values of different metrics to be expected under reference conditions.

3 A river typology for macrophytes

3.1 Background

Following the allocation of surface water bodies to basic categories (in this case lakes and rivers), the WFD stipulates that surface water bodies should be separated according to type. This is an essential step in the establishment of type-specific reference conditions. Even when the final objective is a site-specific predictive system, (as in the present case) the typing process provides a framework for sifting out a pool of unimpacted sites. In this instance the typology should be seen as a means to an end rather than the end in itself. Two approaches are given in the WFD to assist the typing process:

- System A is a fixed typology with a set of variables (such as altitude, catchment area, and geology) each of which is stratified into prescribed classes (such as catchment area classes of 10-100, 100-1,000, 1,000-10,000 and more than 10,000 square km).
- System B provides a set of obligatory (such as altitude and geology) and optional (such as slope, width, depth, substrate) factors that can be used to construct a typology.

In the case of rivers the GB typology is based on altitude (mean altitude in catchment), geology (assigned to calcareous, siliceous or organic) and catchment area (using the subdivisions indicated above). For rivers, this typology serves a dual purpose since it is also the basis for subdividing river networks into discrete water bodies that are then assessed individually. Consequently a water body can only belong to a single river type. The GB typology and constituent water bodies are derived directly from a GIS layer which generates 18 river types distributed across 10,000 water bodies covering almost 65,000 km of water course. In practice, five of these 18 types are poorly populated and could be aggregated with an adjacent type. However, typologies are largely for reporting purposes and it is highly unlikely that a single typology, such as that described, would be optimal for all biological quality elements. For example, slope is not considered within this typology (other than indirectly through covariation with the typing variables) yet slope is widely regarded as a key determinant of macrophyte composition in rivers (Holmes *et al*., 1998; Baattrup-Pedersen *et al*., 2003; Demars & Harper, 2005). Consequently, it is necessary to confirm whether the typology for rivers is the best basis for stratifying the variation in macrophyte assemblages.

3.2 Approach

Holmes *et al* (1999b) report the results of a TWINSPAN classification of river macrophyte compositional data based on JNCC surveys. Rather than run a new analysis, we used their analysis as a reasonable reflection of the range of variation in the structure of river plant communities in the UK. We used the 16 cluster analysis based on five divisions which generated a large minimum group size (n=22) and, at this level, was insensitive to the idiosyncracies in data collected by different surveyors.

These types were then related to an environmental dataset composed of temporally invariant, ‘unimpactable’ variables, including geology, slope, site altitude, source altitude, distance from source, and various geological characteristics. One-Way Analysis of Variance (ANOVA) was used to compare the ability of different variables (after transformation), and to identify threshold values of different variables, to discriminate between botanical clusters. Where variables could not be normalised a non-parametric Kruskal Wallis test was used for the same purpose. Canonical Correspondence Analysis (CCA) of the global sites x species composition dataset constrained by the sites x environmental dataset was used to verify the hierarchy of environmental determinants as indicated by ANOVA. However, CCA alone is unsuitable for establishing a typology.

The survey database covered sites impacted to differing degrees and this step was not, therefore, designed to achieve a classification of unimpacted sites, or to use subsequent procedures, such as Multiple Discriminant Analysis, to predict reference assemblages (as in the approach adopted for RIVPACS).

3.3 Results

3.3.1 Overall typology

Slope was the primary variable in explaining variation in river macrophyte composition. This was followed by the altitude of the source and the distance of a site from its source. Alkalinity came next, a rather diminished ranking compared to its dominant position in the lake typology, although this may partly reflect the poorer availability of information on river alkalinity compared to the other geographical variables. When viewed as box plots (Figures 3.1 to 3.3), with botanical types ranked along the base, there is a largely predictable gradient from base-rich, lowland, sluggish-gradient rivers through to base-poor, upland, steeper-gradient streams, reflecting the major environmental SE-NW axis in Britain.

Slope exerts a pre-eminent influence on riverine vegetation, greatly exceeding the strength of effect of depth on lake plants. There is a clear link between plants and slope, since slope determines whether deposition of fine sediment is likely, and governs hydraulic forces on plants. Slope does not form a component of the UK typology (being replaced by average catchment altitude) but its use is appropriate in a macrophyte typology given the strength of its influence. Consequently the typology used in this project is as defined in Table 3.1. The botanical clusters that were used to inform the setting of thresholds for environmental variables are summarised in Table 3.2. Differences in typology between quality elements, or between one quality element and a general reporting typology, are of little consequence. The quality-element specific typology is merely a method to minimise the unexplained variation in biology in unimpacted sites, thus increasing the ability to detect pressures within impacted sites of the same type. Any site, regardless of quality element or supporting typology, can be mapped onto a reporting typology.

3.3.2 High-alkalinity rivers

The lake typology required a subdivision of high-alkalinity lakes to differentiate between naturally fertile base-rich lakes on soft calcareous geologies, such as chalk, and naturally infertile base-rich lakes on slow weathering geology, such as limestone, hard sandstone and various metamorphic rock types. The same argument has been followed here, in order to separate fertile streams in SE England from those of a similar alkalinity but draining areas of hard calcareous geology, such as the south Pennines. These have been separated into southern-continental and northern-Atlantic subtypes respectively. Rivers in the second group will generally rise at higher altitude (over 200 m) but this variable is not considered within the alkalinity x slope typology. The subdivision of rivers into geographical subtypes has only proved necessary for high-alkalinity rivers (50-200 mg/l $CaCO_3$) since values of alkalinity higher than this are confined to chalks and clays in the south and east of England.

Table 3.1 Summary of river typology used in this project and the various notations used to represent the types

<table>
<tr><td colspan="2" rowspan="2"></td><td colspan="4">Slope or Gradient (m/km)</td></tr>
<tr><td>Gentle or Very Low
(<1)</td><td>Low
(1-3)</td><td>Moderate
(3-8)</td><td>Steep or High
(>8)</td></tr>
<tr><td rowspan="5">Alkalinity (mg/l $CaCO_3$)</td><td>Low
(<10)</td><td colspan="2">Low alkalinity, low slope/gradient
LA_LS
LA_LGr</td><td>Low alkalinity, moderate slope/gradient
LA_MS
LA_MGr</td><td>Low alkalinity, high/steep slope/gradient
LA_HS
LAS
LA_HGr</td></tr>
<tr><td>Moderate
(10-50)</td><td colspan="2">Moderate alkalinity, low slope/gradient
MA_LS
MA_LGr</td><td>Moderate alkalinity, moderate slope/gradient
MA_MS
MA_MGr</td><td>Moderate alkalinity, high/steep slope/gradient
MA_HS
MAS
MA_HGr</td></tr>
<tr><td rowspan="2">High
(50-200)</td><td>High alkalinity, very low slope/gradient, Northern Atlantic
HAN_VLS
HA_VLGr_N</td><td>High alkalinity, low slope/gradient, Northern Atlantic
HAN_LS
HA_LGr_N</td><td>High alkalinity, moderate slope/gradient, Northern Atlantic
HAN_MS
HA_MGr_N</td><td rowspan="2">High alkalinity, high/steep slope/gradient
HA_HS
HAS
HA_HGr</td></tr>
<tr><td>High alkalinity, very low slope/gradient, Southern Continental
HAS_VLS
HAC_VLS
HA_VLGr_S</td><td>High alkalinity, low slope/gradient, Southern Continental
HAS_LS
HAC_LS
HA_LGr_S</td><td>High alkalinity, moderate slope/gradient, Southern Continental
HAS_MS
HAC_MS
HA_MGr_S</td></tr>
<tr><td>Very high
(>200)</td><td>Very high alkalinity, very low slope/gradient
VHA_VLS
VHA_VLGr</td><td>Very high alkalinity, low slope/gradient
VHA_LS
VHA_LGr</td><td colspan="2">Very high alkalinity, moderate slope/gradient
VHA_MS
VHA_MGr</td></tr>
</table>

Alkalinity category	Low	Moderate	High (Southern Continental) High (Northern Atlantic)	Very High
Notation used where slope is not considered	LA	MA	HA_C HA_N	VHA

Table 3.2 Botanical classification of GB river plant communities, showing 38 subtypes and 16 coarser types used to develop an environmental typology (from Holmes *et al*., 1999)

Code	Description
Ia	Large, lowland rivers with high base flow
Ib	Fast-flowing, coarse-bedded lowland rivers of low gradient
Ic	Lowland, very low-gradient rivers with fine substrates
IIa	Small 'classic' clay rivers
IIb	Clay rivers with diverse substrates and flow patterns
IIc	Clay-dominated rivers with impoverished flora
IIIa	Classic chalk rivers
IIIb	Chalk/oolite streams and high base-flow rivers
IVa	Base-rich/neutral, impoverished rivers, normally close to source
IVb	Base-poor, impoverished ditch communities
IVc	Upland rivers with impoverished floras
Va	Mesotrophic upland hard limestone/sandstone rivers
Vb	Small, lowland, base-rich sand rivers or winterbournes
Vc	Small, lowland, impoverished, sand/clay rivers
Vd	Western, stable rivers on sandstone and shales
Ve	Lowland, large rivers in south-west England and Wales
VIa	Lowland, large mesotrophic rivers on limestone or sandstone
VIb	Large, lowland reaches of meso-eutrophic rivers with upland sources
VIc	Middle reaches of upland rivers traversing richer strata
VId	Small, low-gradient meso-eutrophic rivers
VIe	Small, basic, upland rivers
VIIa	Small, shallow, high-altitude hard limestone and sandstone rivers
VIIb	Mesotrophic rivers with strong calcareous influence
VIIc	Lowland, mesotrophic rivers with acidic feeders
VIId	Mesotrophic, upland plateau rivers
VIIIa	Steep-gradient, low-altitude, sand/shale rivers
VIIIb	Moderate-gradient, shale/sandstone rivers below uplands
VIIIc	Base-rich, meso-oligotrophic, upland rivers
VIIId	Large, low-gradient, lowland reaches of upland rivers
VIIIe	Small, oligo-mesotrophic reaches of highland rivers
IXa	Lowland, low-gradient, oligotrophic rivers dominated by higher plants
IXb	Hard rock, 'lowland' rivers with vascular plants dominant
IXc	Base-poor rivers with mixed communities
Xa	Highland rivers with atypically shallow gradients
Xb	Low-altitude bedrock rivers
Xc	High-altitude, steep-gradient rivers, rarely on base-poor rocks
Xd	Oligotrophic rivers of the west coast of Scotland
Xe	Small, shallow, oligotrophic rivers

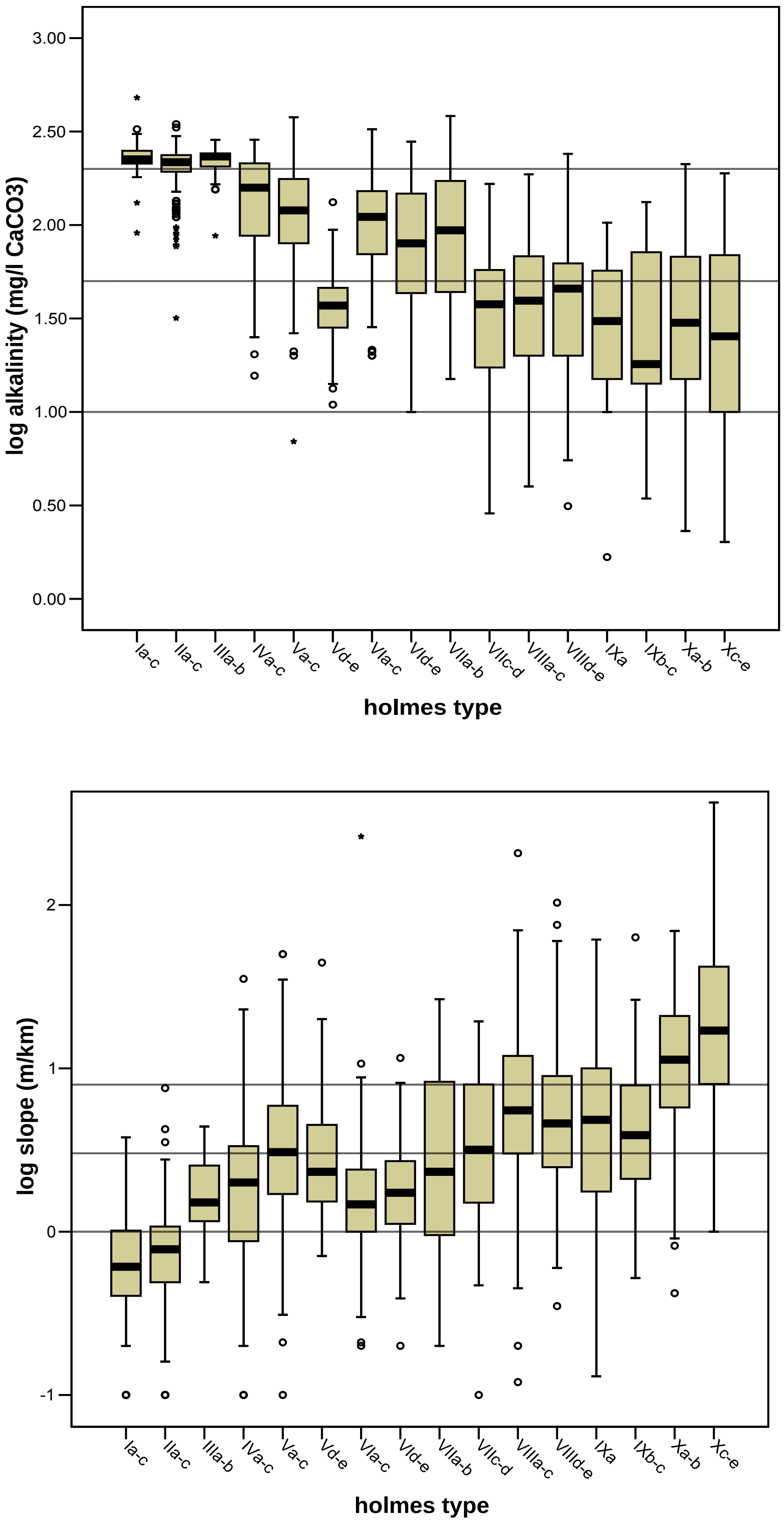


Figure 3.1 Distribution of Holmes (1999) sixteen pre-existing river botanical types in relation to dominant typing variables: alkalinity (upper) and slope (lower).

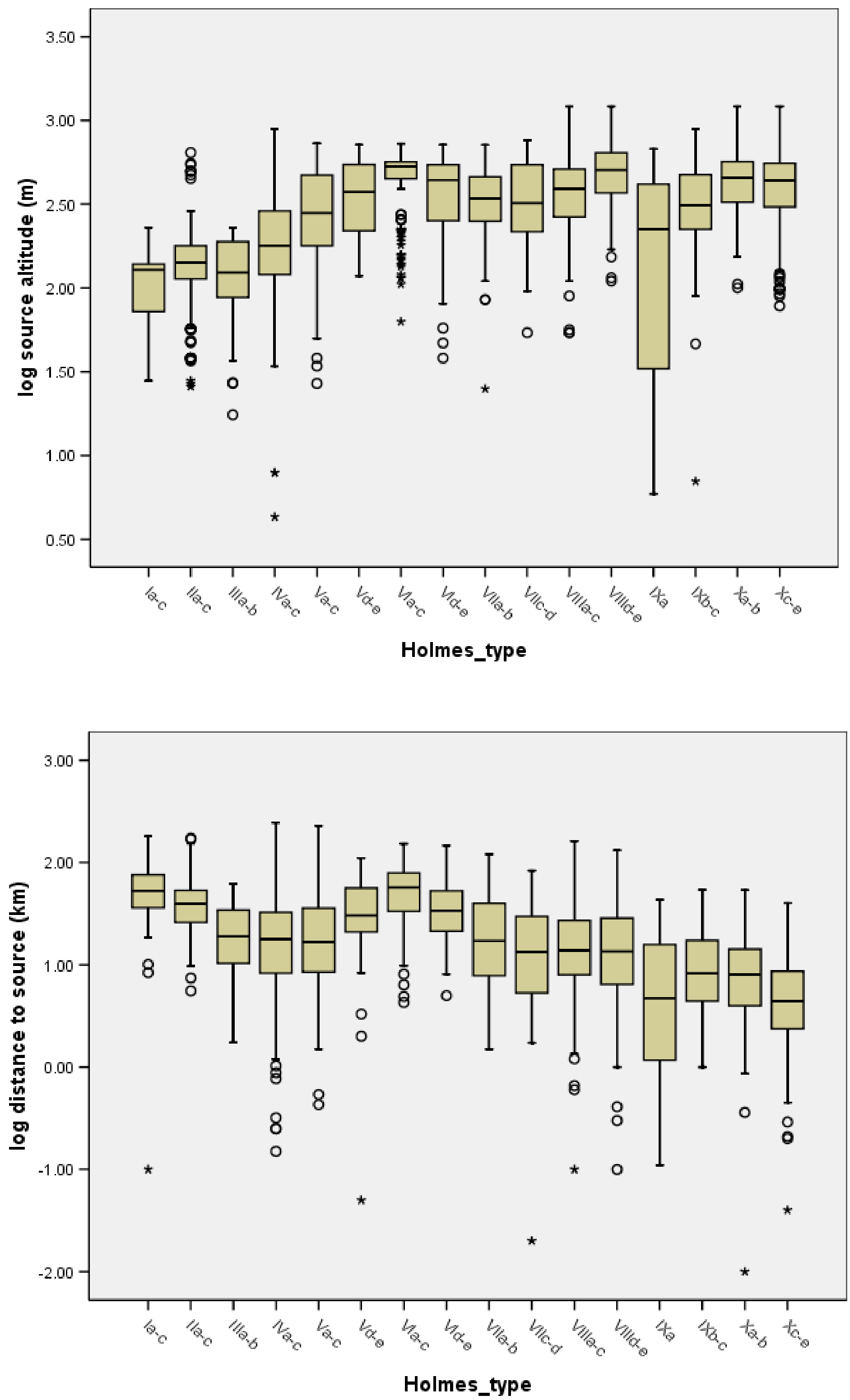


Figure 3.2 Distribution of Holmes (1999) sixteen pre-existing river botanical types in relation to source characteristics: Upper panel, altitude of source; lower panel, distance of site from river source.

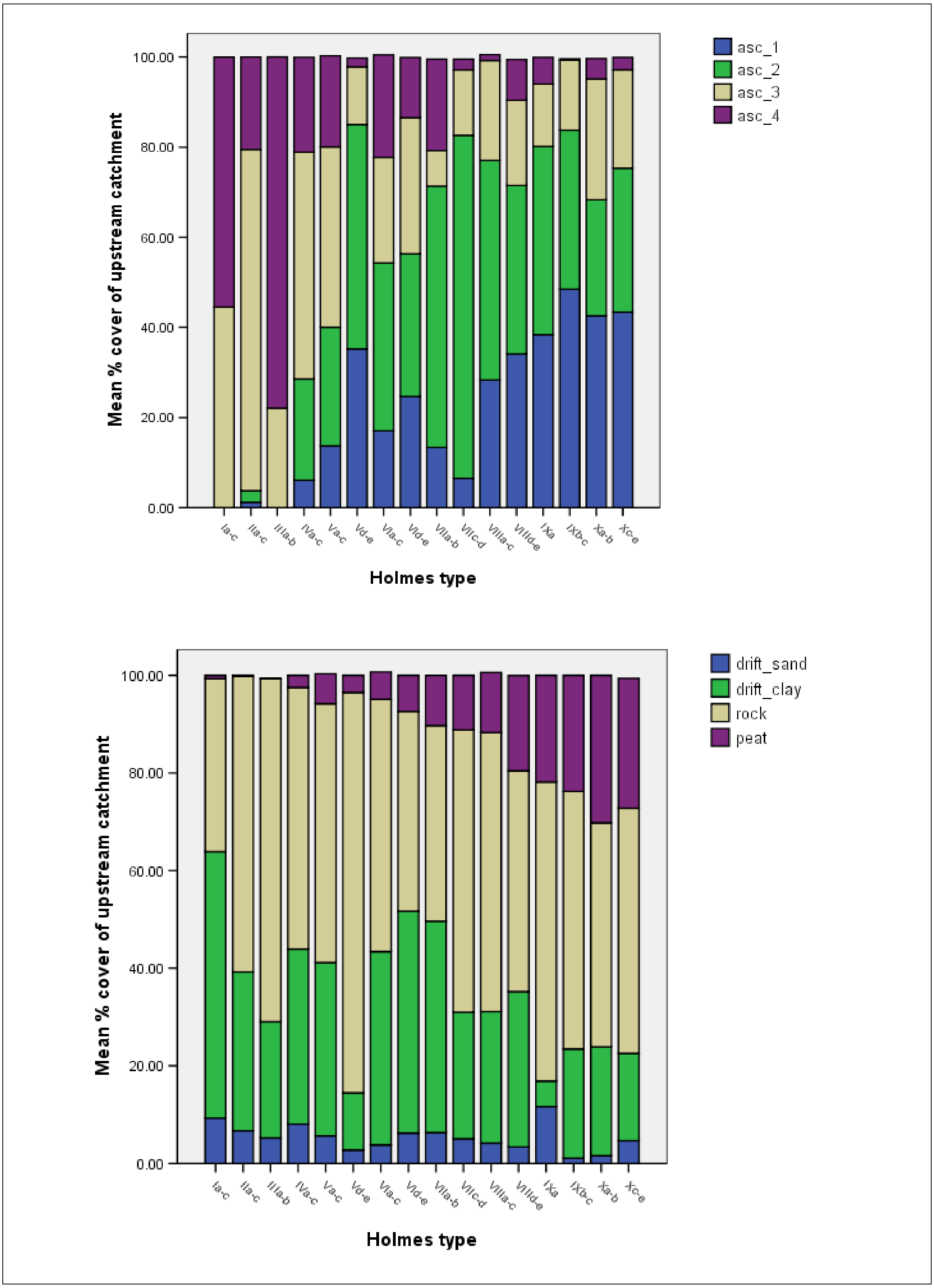


Figure 3.3 Geological characteristics of stream catchments stratified by Holmes (1999) 16 group classification of river botanical types. Upper panel, solid geology, expressed as distribution of acid sensitivity class (ASC) values. Lower panel: simplified drift geology classification.

3.4 Assessment of typology

3.4.1 Relationship between botanical and environmental types

A cross-tabulation of botanical types versus the proposed environmental river typology (Table 3.3) indicates that the alignment between biology and environment at this level is far from clear. For example, each river type contains on average 10 of the 16 botanical clusters, while each botanical cluster occurs, on average, in 10 of the 16 river types. There are, however, clear patterns. Thus, low-slope, low-alkalinity rivers are dominated by botanical type IX, steep-slope, low-alkalinity rivers are dominated by type X and low-slope, moderate alkalinity rivers are dominated by botanical types Vd-e to VIa-c. At the other extreme, very high-alkalinity, very low-gradient rivers are dominated by types I and II, while type III dominates in very high-alkalinity, low-slope rivers. Moreover, there are only a few river types (for example high-alkalinity; low slope; northern) in which several botanical clusters are common (more than 20 per cent of the number of surveys in that type), while each botanical type is assigned principally to a few key environmental types. The core river typology cannot by definition cover a range of other variables that influence botanical composition; natural variation in botanical type within a given river type is therefore to be expected. Since the observed botanical river type may also be the product of a variety of impacts, it is likely that a range of botanical types will overlap within a given river type for non-natural reasons.

3.4.2 Relative importance of other environmental factors

The results of a Canonical Correspondence Analysis (CCA) in which site biology is constrained by data on a wide range of environmental variables confirms the overriding importance of slope in structuring river macrophyte communities (Figure 3.4). Thus, the first axis of this analysis is dominated by a gradient from steep, upland rivers on the right, through to base-rich lowland rivers on the left. The strength of this gradient is such that this axis explains more variation in species composition than all subsequent axes combined. The significance of a range of variables vulnerable to anthropogenic influence (such as nutrient levels and substrate characteristics) is also illustrated.

Figure 3.5 offers a refined perspective by mapping all surveys in ordination space (Detrended Correspondence Analysis, axes 1 and 2) coded by their river type. The two dominant themes to emerge are that:

i. There is a strong gradient of increasing alkalinity from left to right. This is consistent with extensive empirical evidence that alkalinity is a key determinant of river macrophyte composition.

ii. Turnover between surveys decreases with increasing alkalinity, as reflected by the diminishing range of scores on axis 2. This is the inverse of the pattern exhibited by lakes where the highest alkalinity lakes show the greatest turnover between sites, reflecting the influence of differences in hydraulic disturbance between small sheltered and large open sites, while low-alkalinity lakes support a narrow range of species. By contrast in rivers, the turnover between low-alkalinity rivers on low gradients where rooted vascular plants dominate, and steep gradients where bryophytes dominate, is much larger than the turnover across the relatively constrained range of slope that occurs at the highest alkalinities.

The forward selection analysis (Table 3.4) highlights a pool of variables and their derivatives (slope, source altitude, distance from source and alkalinity) that together account for 60 per cent of the explainable variation in river macrophyte composition. In general therefore, it seems that a typology built on alkalinity, slope and geographical distribution will adequately describe the variation in macrophyte composition in UK rivers, even if these are (sometimes) correlates of, rather than the specific variables driving, differences in plant composition.

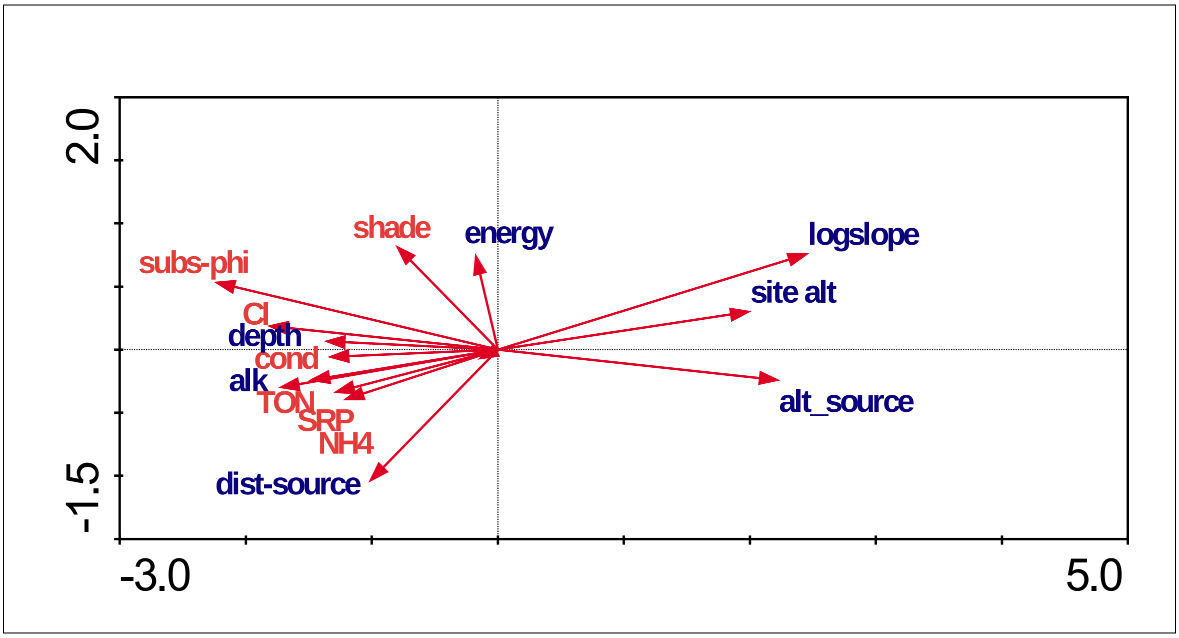


Figure 3.4 Environmental variable plot from a Canonical Correspondence Analysis of river macrophyte surveys data. All variables shown explained a significant amount of variation in species data (p = <0.001). Intrinsic variables (considered for typology) in blue; impacted variables in red.

Table 3.3 Cross-tabulation of membership of river botanical clusters in relation to proposed environmental typology

Alkalinity[1]	**Low**	**Low**	**Low**	**Mod**	**Mod**	**Mod**	**High**	**High**	**High**	**High**	**High**	**High**	**High**	**Vhigh**	**Vhigh**	**Vhigh**
Slope[2]	**Low**	**Mod**	**High**	**Low**	**Mod**	**High**	**Vlow**	**Low**	**Mod**	**High**	**Vlow**	**Low**	**Mod**	**Vlow**	**Low**	**Mod**
NA/SC[3]	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**SC**	**SC**	**SC**	**SC**	**SC**	**SC**
Ia-c							1.9				1.9	0.9		70.8	22.6	1.9
IIa-c				1.0			4.0	3.0			11.1	5.0	0.5	55.8	18.6	1.0
IIIa-b								1.1			2.1	9.6	2.1	12.8	60.6	11.7
IVa-c	1.3	0.4		11.5	2.2	2.2	9.7	11.9	10.6	0.9	7.1	7.5	6.2	6.2	13.7	8.4
Va-c	0.6	0.6	0.6	6.4	7.3	3.4	6.1	20.7	18.7	9.2	3.4	7.5	7.0	1.4	3.9	3.4
Vd-e	3.6			55.4	25.0	3.6			7.1	3.6		1.8				
VIa-c				24.4	2.4		18.9	32.3	11.0	1.8	3.7	0.6	0.6	1.8	2.4	
VId-e				37.7	8.7		13.0	18.8	11.6	2.9	1.4	2.9		2.9		
VIIa-b	2.0	2.0		8.2	4.1	12.2	16.3	26.5	6.1	8.2	2.0	2.0	2.0	4.1	2.0	2.0
VIIc-d	14.3	4.1	6.1	24.5	16.3	14.3		6.1	4.1	6.1	4.1					
VIIIa-c	3.7	6.4	11.0	19.3	19.3	13.8	1.4	4.1	8.7	9.6			2.8			
VIIId-e	7.1	12.3	8.5	8.5	13.3	13.7	1.4	6.6	6.6	19.0		0.5	2.4			
IXa	18.2	22.7	13.6	9.1	9.1	9.1		4.5		13.6						
IXb-c	43.6	23.1	17.9		10.3	2.6				2.6						
Xa-b	8.7	20.5	41.7	0.8	2.4	16.5			5.5	3.1			0.8			
Xc-e	3.2	13.7	51.6	0.5	2.7	14.6		0.5	1.4	9.1		0.9	1.4		0.5	
N-types	**11**	**10**	**8**	**13**	**13**	**11**	**9**	**12**	**11**	**13**	**9**	**11**	**10**	**8**	**8**	**6**

[1]Alkalinity as mg/l $CaCO_3$ where thresholds for low, moderate, high and very high are as given in Table 3.1.
[2]Slope as m/km where thresholds for very low, low, moderate and high as in Table 3.1.
[3]NA/SC distinguishes between northern-atlantic and southern continental rivers on base-rich rock types of differing fertility. Hard sedimentary or metamorphic rocks in north, soft and easily weathered calcareous substrates in south and east.

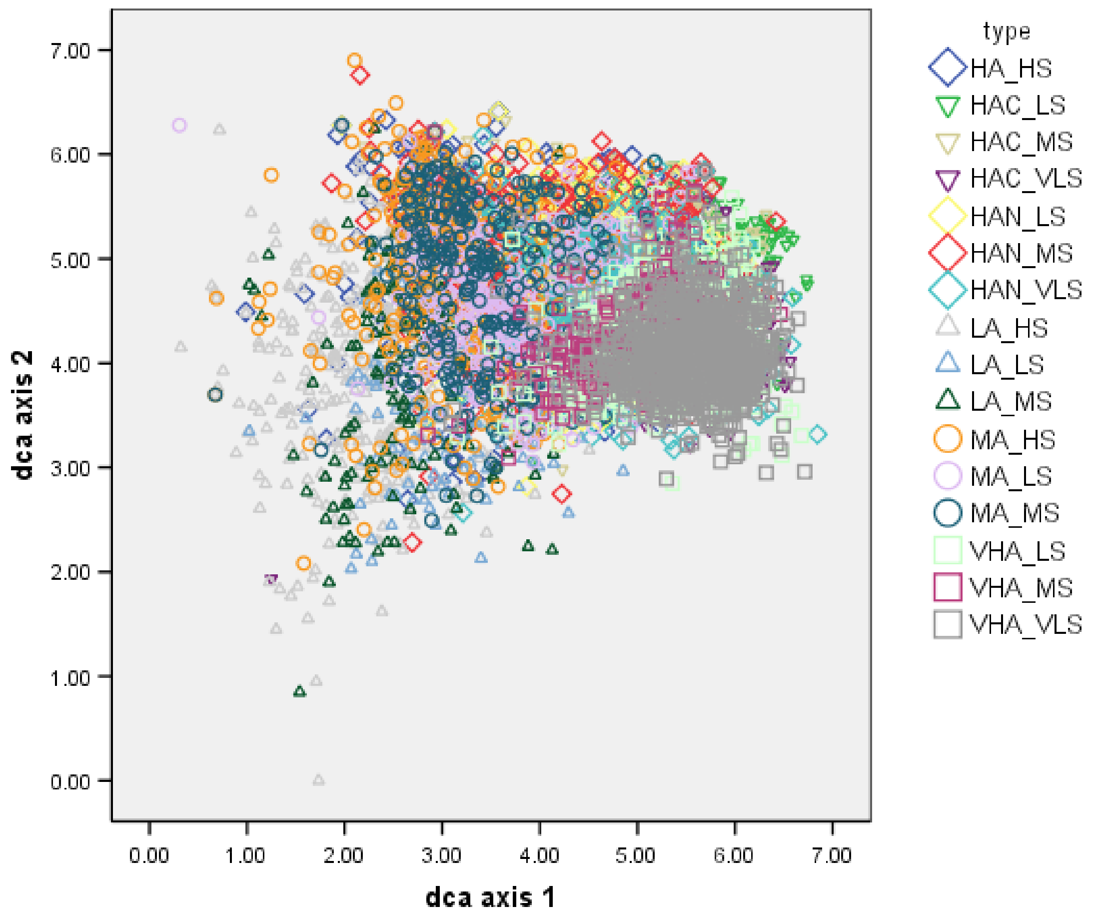


Figure 3.5 Results of Detrended Correspondence Analysis (DCA) of river macrophyte survey data stratified by river type. Note the trend of increasing productivity on axis 1 from left (high slope, low-alkalinity) to right (low slope, very high-alkalinity). Also note the tendency for greater turnover between surveys on axis 2 at lower productivity.

Table 3.4 Results of forward selection analysis within Canonical Correspondence Analysis, indicating conditional effects of environmental variables as explanatory factors of variation in river plant species composition

Variable	LambdaA	P	F	intrinsic
slope (Log_{10})	0.29	0.002	204.15	*
source altitude	0.15	0.002	105.22	*
substrate phi grade	0.07	0.002	50.55	
% urban land cover	0.05	0.002	39.77	
distance from source	0.06	0.002	40.56	*
weighted energy	0.04	0.002	31.28	*
alkalinity	0.03	0.002	24.35	*
alkalinity (Log_{10})	0.03	0.002	24.54	*
temperature	0.04	0.002	23.14	
distance from source (Log_{10})	0.02	0.002	17.48	*
site altitude	0.02	0.002	16.5	*
site altitude (Log_{10})	0.03	0.002	19.65	*
mean width	0.02	0.002	13.75	
pH	0.01	0.002	12.41	
SRP (Log_{10})	0.02	0.002	14.27	
TON (Log_{10})	0.02	0.002	15.16	
% intensive land cover	0.02	0.002	11.54	
slope	0.01	0.002	11.66	*
shading	0.02	0.002	10.61	
conductivity	0.01	0.002	9.36	
catchment area (Log_{10})	0.01	0.002	8.48	*
mean depth	0.01	0.002	7.86	
$N\text{-}NH_4$ (Log_{10})	0.01	0.002	5.25	
chloride	0	0.002	5.22	
Suspended solids	0.01	0.004	4.6	

4 Developing metrics for ecological assessment of rivers using macrophytes

4.1 Metric-based approaches to the use of river macrophytes for ecological assessment

Macrophytes have been used extensively in Europe since the 1970s to assess the quality of rivers, although most work in this field has been carried out in the UK, France and Germany. A number of approaches to the assessment of riverine ecosystems using macrophytes can be distinguished (Thiebaut *et al*., 2002):

i. Classical phytosociological approaches, as used mainly in France and Germany.
ii. Measurements of cover or biomass.
iii. Typing of communities based on their composition.
iv. Assessment based on comparison of vegetation samples according to combinations of drainage order and geology.
v. Application of expert-based or empirically derived water quality indices to community level data.
vi. Description of vegetation in terms of biological traits or functional groups.
vii. Use of diversity indices.

The suitability of these approaches depends on the methods used to collect the data, the limitations of this data and the purposes for which this data is to be used. Thus, standardised survey approaches based on fixed lengths of river will embrace a range of different microhabitats, and are clearly not aligned to the concept of phytosociological units. Given a need to relate differences in vegetation composition between sites, or between a site and its reference condition, to different pressures, primary biological data may also be resistant to further interpretation unless converted to another format, such as index or metric values.

Approaches that are reliant directly or indirectly on finding or not finding individual species are open to the criticism that macrophyte surveys are subject to significant detection bias and that this bias varies between observers. Consequently, it is difficult to be confident that an individual taxon is, or was, absent from a site. Therefore, proving that a newly recorded species was previously genuinely absent, or that a species previously recorded but no longer found is now truly extinct, may be impossible (Lansdown, 2007). Phytosociological and typological approaches are especially vulnerable in this respect because they use raw lists of species to generate associations or types and require data of the same format to classify new observations. For example, the current common standards monitoring assessment approach for riverine SACs depends firstly on correctly assigning a reach to its botanical type, and secondly assessing whether the composition of the reach is typical of the type based on a constancy table (JNCC, 2005). Both these steps are reliant on finding specific taxa and identifying them correctly. In their current WFD classification systems for river macrophytes, several European countries have based their approach on the proportion of characteristic type-specific taxa that can be found in a reach (see Schaumburg *et al.*, 2004). Dispersal limitation creates an additional complication in the case of water plants, since, in contrast to mostly mobile macroinvertebrates, most macrophyte taxa have a low probability of occurrence at any given site, yet their absence cannot be reliably interpreted as being due to lack of suitable habitat.

Approaches such as the measurement of biomass, cover or richness, recording of biological traits or derivation of a weighted indicator score for a site could all be classed as metric-based approaches. These may require the collection of primary biological data but the metrics themselves are normally an abstraction of this data. There are a number of advantages associated with the use of biological metrics for ecological assessment. These include

i. Less sensitivity to inter-surveyor variability and detection bias because site metric scores appear relatively conservative, showing comparatively little change with the addition of new species or removal of existing species, once the number of taxa exceeds a certain level (Ewald, 2003). Moreover, because several taxa can have similar specific scores they are treated as ecologically equivalent, while strongly taxonomic approaches will treat them as discrete entities. Two sites may thus have few species in common, while still having similar scores for a given biological metric. Some metrics will bypass completely issues of detection bias (such as biomass or cover), while others, such as richness, will be more sensitive.
ii. Ease of modelling numerical values to estimate metric values under reference conditions when no contemporary reference sites are available.
iii. Amenable to conversion to an Ecological Quality Ratio (EQR) which is required to express classification results.
iv. Ease of integrating different aspects of community structure and function, expressed as individual EQRs, into an overall measure of class.

Perhaps the biggest disadvantages of current metric-based approaches are firstly their lack of immediacy with biological data, secondly, the rather weak evidence base from which to validate the use of some metrics and to provide a basis for their interpretation, and, thirdly, the focus on single pressures (typically chemical water quality) that is incompatible with the need for a more holistic assessment. The following analyses are designed to identify a pool of candidate metrics for use in assessment of ecological status in rivers based on macrophytes and to provide empirical support for their use.

4.2 Developing a metric to detect nutrient-based pressures

4.2.1 Background

There is a relatively long tradition in biomonitoring of freshwater environments to attaching numerical ranks to taxon names in order to derive some form of site index that relates the biota to environmental variation, whether natural or pressure-related, and measured or merely inferred. In the case of river plants, this approach can be traced back to the early 1970s and the ideas of Carbiener in France and Kohler in Germany. Such ranks may be derived from empirical data, for example in terms of the species optima on a given environmental gradient, as determined by weighted or reciprocal averaging, or they may be based on expert judgement. Various refinements, such as weighting ranks by cover of species, and/or by the indicator values (tolerance) of individual species, are also possible. In this project, a method was devised for empirical adjustment of expert ranks which might be considered a hybrid between the strict empirical and expert approaches. The site scores derived from this ranking were subsequently validated against measured pressure data.

Diekmann (2003) reviews the practical and philosophical strengths and weaknesses of empirical or expert ranking systems.. Some of these are discussed below.

Expert ranking systems reflect a diffuse evidence base incorporating, on the one hand, literature reports or anecdotal observations or compositional changes over time at individual sites, measured environmental data or inferences made from landscape or land cover, and on the other hand, understanding of general relationships between biological traits and environment. In this sense, such systems could be argued to be closer to the concepts of ecological structure and function advocated by the WFD, but equally it is difficult to extract precisely what such systems are measuring. Consequently, such indices can be related (or not) to measured environmental data which might be expected to reflect the pressure to which the index is designed to respond, but the index itself is effectively unfalsifiable. Expert ranking systems arguably reflect the basic principles of biological monitoring most faithfully, since they attempt to represent aspects of the environment that are not readily measurable. The more heavily contingent a biological metric is upon a measured environmental variable, the more redundant the biological metric will become.

By contrast, empirical systems are directly contingent on the environmental data supplied. What is actually being measured is thus more transparent. However, it is misleading to believe one can establish precisely what a metric is measuring, given the degree of intercorrelation amongst environmental variables. Thus, empirically-derived metrics do not have unique meaning (although there is usually a common cause of intercorrelation among co-linear groups of variables). Deriving phosphorus optima for a range of macrophyte species based on measurements of nutrient chemistry at the sites where these species occur is straightforward, but the optima may still more closely reflect variation in alkalinity or some other covariable. It is also unclear whether a single directly measured environmental variable (such as annual mean water column soluble reactive phosphorus (SRP)) can adequately capture a broad pressure, such as eutrophication, which influences macrophytes via multiple routes. An index which synthesised information on loading rates, land cover, sediment nutrient concentrations, organic nutrient fractions and nutrient concentrations derived over different averaging windows might prove more effective.

It also seems likely, given widely observed lags in response of macrophytes in lakes and rivers to changes in nutrient concentrations, that building empirical metrics from paired contemporary nutrient and biological data carries the risk of misrepresenting the nutrient optima of a species unless concentrations are stable. Empirical metrics are also data hungry and for this reason tend to be based on data collected in discrete geographical areas and can only provide reliable information on relatively widespread taxa. The validity of extrapolation to other regions is therefore questionable, while scores for sites in which rare taxa account for a large proportion of the total assemblage will have a large error attached. Validation of metric-nutrient relationships is often based on sacrificing a small proportion of sites from the metric development stage and then using these to test the model. However, many paired biology x chemistry datasets are concentrated in a small geographical area. Thus, if the objective is to roll out a metric for use in a wider geographical area, using a subset of the initial dataset is liable to over-validate the model. A further concern relates to the use of chemistry from routine water sampling in the derivation of metric values, since routine sampling represents a highly biased sample of the resource and is likely to greatly undersample higher quality sites (Irvine, 2002). Aggregating chemistry data from multiple sources and regions also raises concerns since the comparability, and sometimes reliability of such data can be questionable, yet this becomes ‘buried’ within the resulting species scores. An obvious problem with environmental and biological data derived from independent sources is that the measured chemistry rarely corresponds exactly in space and time to the macrophyte survey data. Consequently, it is artificial to represent individual species by chemical optima derived from a different time and location. In the case of macrophytes, on the one hand, it may be less problematic to associate macrophyte composition with older environmental data, due to the likely lag phase in response of macrophytes to environmental change compared to other biological quality elements. On the other hand, macrophytes are highly sensitive to antecedent flow conditions so links to older environmental data may introduce additional noise. The consequence of spatial mismatches will probably depend on river type and associated physical heterogeneity.

Compositional metrics could be applied to any environmental variable where there is a desire to reflect that variable though the biology. However, in the case of macrophytes compositional metrics are usually applied in the context of nutrient enrichment. Other pressures that could be considered usefully via this approach include hydromorphological alteration, for example in the form of flow regime change, channel or bank modification, or alterations to substrate characteristics.

4.2.2 Approach

In developing compositional metrics to assess river ecological status, the initial focus has been on the assessment of nutrient impacts. A CCA confirms that major nutrients still have a major influence on the composition of river macrophytes, once the influence of intrinsic factors such as slope, source altitude and alkalinity have been accounted for. Thus, when considered independently of other factors nutrients explain about five per cent of the variation in macrophyte composition in UK rivers (Table 3.4). Evidently the influence of nutrients in river is weaker than their influence on macrophytes in lakes, and a focus on nutrients in rivers to the exclusion of hydromorphological variables, such as substrate, could not therefore be justified. However, nutrients represent an important residual effect on riverine vegetation once the effect of key intrinsic factors such as slope and geology are accounted for and their concentrations are highly prone to anthropogenic influence (see Section 7.8.1 for further discussion). Consequently, there remains a strong empirical basis for assessing nutrient-related pressures using macrophytes.

In the UK, the Mean Trophic Rank (MTR) devised by Holmes (1996) has been the approach used hitherto to infer information on the nutrient status of rivers based on their macrophyte communities. The MTR is an expert system based exclusively on catchment characteristics, such as land use and geology, and observations of floristic differences upstream and downstream of major nutrient point sources. Many European countries, including Sweden, the Netherlands, Poland, Austria and the Czech Republic, operate ranking systems derived and structured in similar ways, in some cases simply adopting the MTR itself, with minor revisions for national use. The rank used to refer to a species essentially represents the point on an arbitrary scale at which that species should occur at greatest relative abundance. However, for some species the range of tolerance may be wide but this may not be apparent from the rank that is applied. The basis for deriving species optima is not a toxicological dose-response type relationship. Only a handful of species are likely to demonstrate a toxic response to nutrient concentrations over the range of concentrations regularly found in UK rivers. Indeed, grown in monoculture, most species would be expected to perform better under higher nutrient concentrations; it is only interspecific interactions and differences in nutrient affinity and growth rates that reinforce characteristic distributions of species in relation to nutrient supply, and even these differences are likely to be greatly diminished if flow variability is sufficient to buffer the effects of competitive interactions.

One of the difficulties of expert systems is that they reflect only the experience of one or more individuals. While this experience may be extensive, the mental ability to assimilate huge volumes of species compositional data and subsequently readjust species scores is limited. There have been attempts to improve expert systems, by ground-truthing them against measured environmental data (see Ertsen *et al*., 1998; Warnelink *et al*., 2005), or by adjusting scores statistically to reflect the underlying patterns of co-occurrence of different species in large datasets. Examples of the latter include the adjustment of European Ellenberg scores to more closely fit the UK flora based on an analysis of plant quadrat data collected as part of the 1978 and 1990 UK Countryside Survey (Hill *et al*., 2000) and the adjustment of invertebrate Biological Monitoring Working Party (BMWP) scores based on a 17,000 sample dataset (Walley and Hawkes, 1996).

This project used an algorithmic approach, similar to that of Hill *et al.* (2001), to form an expert index for riverine plants. Recalibration of expert indices using such approaches is discussed by Diekmann (2003). The adjusted Ellenberg N scores for UK flora and original MTR species scores harmonised to the same scale formed the basis of this index since, together, these covered a wider range of species and provided UK wide geographical coverage. The approach used is described in detail in Appendix 1. In summary, the algorithmic approach calculates a site score based on the average of the known scores of species and then performs a DCCA where the site scores are constrained by the unadjusted expert scores. In the case of species without expert scores, these are obtained based on a regression between axis 1 species scores and expert scores of known species. A second iteration is performed in which the site score based on all species is used to constrain the ordination. The species scores derived from this analysis are used as the compositional metric. Various small refinements described in Appendix 1 are used to deal with species with small numbers of records and to avoid various forms of bias. After calculation, all scores are rescaled to run from one to ten. In deference to other indices (Ellenberg, Trophic Diatom Index, Trophic Ranking Score) high scores are associated with the most nutrient-tolerant species. To clearly discriminate from the MTR system, this expert nutrient metric is referred to as the River Macrophyte Nutrient Index (RMNI). The RMNI operates on a continuous rather than ordinal scale, thus circumventing criticisms (see Diekmann, 2003) that site scores based on averaging of species ranks should strictly be based on the median rather than average rank of the taxa present.

Table 4.1 Species scores for the metrics RMNI and RMHI, aquatic species and functional group membership

Taxa	RM NI[1]	RM HI[2]	Aq[3]	FG[4]	Taxa	RM NI	RM HI	Aq	FG
Acorus calamus	9.49	9.82			Myosotis secunda	4.74	5.44		
Alisma lanceolatum	8.47	9.12			Myosotis sp(p).	7.00	7.06		
Alisma plantago-aquatica	7.82	8.02			Myriophyllum alterniflorum	3.44	5.20	1	7
Anthelia julacea	2.70				Myriophyllum spicatum	8.26	7.91	1	7
Apium inundatum	4.34	5.89	1	8	Myriophyllum spp indet	5.89	6.58	1	7
Apium nodiflorum	8.64	8.08	1	8	Myriophyllum verticillatum	7.53		1	7
Azolla filiculoides	9.71	8.98	1	1	Nardia compressa	1.05	2.89	1	23
Baldellia ranunculoides	4.34	3.70	1	4	Nardia scalaris	2.73	3.00	1	23
Batrachospermum sp(p)	5.46	6.10	1	19	Nardia sp.	1.40	3.04	1	23
Berula erecta	8.24	8.17	1	8	Nitella flexilis (agg.)	4.39	5.56	1	2
Bidens cernua	8.13	7.90			Nitella opaca	4.31	5.10	1	2
Bidens tripartita	8.39	8.16			Nitella sp	4.59	5.76	1	2
Blindia acuta	1.09	3.24	1	22	Nitella translucens	4.17	6.15	1	2
Blue-green algal scum/pelts	5.10	5.20	1	3	Nostoc commune	5.14	5.48	1	3
Bolboschoenus maritimus	7.65	8.19			Nostoc parmelioides	4.12	4.97	1	3
Brachythecium plumosum	2.92	3.87	1	21	Nostoc sp	4.66	5.19	1	3
Brachythecium rivulare	3.56	4.30	1	21	Nostoc verrucosum	4.71	5.11	1	3
Bryum alpinum	3.83	3.21			Nuphar lutea	8.42	9.16	1	12
Bryum dixonii	5.22	5.65			Nymphaea alba	5.69	8.42	1	12
Bryum pseudotriquetrum	2.71	3.87			Nymphoides peltata	9.37	9.80	1	10
Butomus umbellatus	8.89	8.61	1	13	Octodiceras fontanum	6.54	6.70	1	22
Calliergon cuspidatum	3.49	3.72			Oenanthe aquatica	6.06	6.91	1	8
Callitriche brutia var hamulata	4.51	5.81	1	6	Oenanthe crocata	6.22	6.48	1	8
Callitriche hermaphroditica	5.75	7.60	1	5	Oenanthe fistulosa	8.27	8.30		
Callitriche obtusangula	8.04	7.98	1	6	Oenanthe fluviatilis	8.57	8.54	1	8
Callitriche platycarpa	7.56	7.74	1	6	Orthotrichum rivulare	4.71	5.57		
Callitriche spp.	6.67	7.18	1	6	Palustriella commutata	4.61	3.75		
Callitriche stagnalis	6.47	7.17	1	6	Pellia endiviifolia	6.50	6.49		
Callitriche stagnalis/platycarpa	6.21	6.14	1	6	Pellia epiphylla	3.34	5.09		
Callitriche truncata	6.47	7.15	1	6	Pellia sp.	4.67	5.64		
Caltha palustris	4.20	5.24			Persicaria amphibia	8.20	8.33	1	10
Carex acuta	7.19	7.30			Persicaria hydropiper	6.97	7.53		
Carex acutiformis	8.21	8.10			Phalaris arundinacea	7.52	7.24		
Carex aquatilis	3.90	5.02			Philonotis caespitosa	2.74	3.08		
Carex elata	4.54	6.23			Philonotis fontana	2.66	3.09		
Carex lasiocarpa	3.41				Phragmites australis	7.70	8.94		
Carex paniculata	7.49	7.96			Platyhypnidium lusitanicum	4.35	3.82	1	21
Carex recta	5.42	6.83			Platyhypnidium riparioides	5.16	5.29	1	21
Carex riparia	9.06	8.89			Porella cordaeana	4.95	5.00	1	23
Carex rostrata	2.64	4.71			Porella pinnata	4.91	5.20	1	23
Carex vesicaria	3.68	5.39			Potamogeton alpinus	4.96	6.26	1	16
Catabrosa aquatica	8.70	7.57			Potamogeton berchtoldii	7.35	7.76	1	14
Ceratophyllum demersum	9.73	9.32	1	5	Potamogeton compressus	8.33	9.00	1	14
Chaetophora sp.			1		Potamogeton crispus	8.02	7.86	1	17
Chara globularis	3.30		1	2	Potamogeton filiformis	6.00	7.62	1	15
Chara sp.	3.85		1	2	Potamogeton friesii	8.19	9.09	1	14
Chara vulgaris	3.77	5.66	1	2	Potamogeton gramineus	4.24	5.69	1	16

Taxa	RM NI	RM HI	Aq	FG	Taxa	RM NI	RM HI	Aq	FG
Chiloscyphus pallescens	4.78	4.75			Potamogeton lucens	8.54	8.79	1	17
Chiloscyphus polyanthos	4.05	4.77	1	23	Potamogeton natans	5.69	7.54	1	16
Cinclidotus fontinaloides	5.37	5.68	1	22	Potamogeton nodosus	7.05	8.79	1	16
Cladophora aegagropila	5.66	6.23	1	19	Potamogeton obtusifolius	5.84	6.80	1	14
Cladophora glomerata	7.50	6.84	1*	19	Potamogeton pectinatus	9.59	8.58	1	15
Cladophora glomerata/Rhizoclonium hieroglyphicum	8.66	7.18	1*	19	Potamogeton perfoliatus	8.16	8.14	1	17
Collema dichotomum	4.42	5.15	1	3	Potamogeton polygonifolius	1.71	4.69	1	16
Cratoneuron filicinum	5.02	6.34			Potamogeton praelongus	7.81	8.83	1	17
Dermatocarpon spp. (aggregated)	3.51	4.85			Potamogeton pusillus	7.47	8.45	1	14
Dichodontium flavescens	2.94	3.60			Potamogeton trichoides	7.24	9.31	1	14
Dichodontium palustris	1.68	3.52			Potamogeton x bottnicus	6.41	8.00	1	15
Dichodontium pellucidum	3.07	4.02			Potamogeton x cooperi	6.07	6.87	1	17
Draparnaldia	3.04	2.64	1	19	Potamogeton x fluitans	6.51	5.51	1	16
Drepanocladus fluitans	3.73	4.45			Potamogeton x gessnacensis	3.88	5.97	1	16
Elatine hexandra	4.17	6.15	1	11	Potamogeton x lanceolatus	4.24	6.50	1	17
Eleocharis acicularis	5.35	6.79	1	4	Potamogeton x nitens	6.17	5.45	1	17
Eleocharis palustris	4.54	5.79			Potamogeton x olivaceus	5.44	6.41	1	17
Eleogiton fluitans	2.06	5.36	1	15	Potamogeton x salicifolius	6.36	7.01	1	17
Elodea canadensis	7.65	7.60	1	5	Potamogeton x sparganifolius	3.87	3.78	1	16
Elodea nuttallii	9.44	8.62	1	5	Potamogeton x suecicus	6.02	6.31	1	15
Equisetum fluviatile	3.92	6.01			Potamogeton x zizzii	4.19	3.75	1	16
Filamentous green algae	7.61	7.04	1*	19	Potentilla palustris	2.88	5.04		
Fisidens polyphyllus	3.84		1	22	Racomitrium aciculare	1.89	3.37	1	22
Fissidens crassipes	6.20	6.00	1	22	Ranunculus (sect Batrachian) sp or hybrid indet1	7.33	7.75	1	18
Fissidens curnovii	3.94	5.03	1	22	Ranunculus (sect Batrachian) sp or hybrid indet2	7.33	7.75	1	18
Fissidens osmundoides	3.06		1	22	Ranunculus (sect Batrachian) sp or hybrid indet3	7.33	7.75	1	18
Fissidens rivularis	5.95	6.35	1	22	Ranunculus aquatilis var aquatilis	5.67	6.63	1	18
Fissidens rufulus	4.70	5.26	1	22	Ranunculus aquatilis var diffusus	7.65	7.74	1	18
Fissidens serrulatus	5.27	3.39	1	22	Ranunculus circinatus	9.42	8.85	1	5
Fissidens sp. (aggregated)	5.80	5.94	1	22	Ranunculus flammula	2.56	4.39		
Fissidens viridulus	4.66	5.51	1	22	Ranunculus fluitans	7.97	7.44	1	18
Fontinalis antipyretica	5.40	5.95	1	21	Ranunculus hederaceus	5.47	5.64		
Fontinalis squamosa	3.66	5.02	1	21	Ranunculus omiophyllus	3.43	4.78	1	11
Glyceria declinata	6.66	6.25			Ranunculus peltatus var baudotii	9.06	7.79	1	18
Glyceria fluitans	5.25	5.77			Ranunculus peltatus var peltatus	6.22	6.71	1	18
Glyceria fluitans agg	5.81	6.01			Ranunculus penicillatus	8.25	7.64	1	18
Glyceria maxima	9.64	8.96			Ranunculus penicillatus ssp. penicillatus	6.29	6.35	1	18
Glyceria notata	8.28	7.49			Ranunculus penicillatus ssp. pseudofluitans	7.92	7.53	1	18
Glyceria x pedicillata	7.12	7.15			Ranunculus penicillatus subsp vertumnus	5.87	6.51	1	18
Gongrosira incrustans	7.46	5.84	1	20	Ranunculus sceleratus	9.86	8.43		
Groenlandia densa	7.96	8.12	1	5	Rhodochorton violaceum	4.14	4.35		
Heribaudiella fluviatilis	5.49	5.68	1	20	Riccardia chamaedryfolia	4.91	6.00		
Hildenbrandia rivularis	6.03	6.07	1	20	Riccardia multifida	5.25	6.74		
Hippuris vulgaris	5.94	8.22			Riccia sp.	4.86	9.00	1	1
Hottonia palustris	6.93	8.85	1	7	Rivularia	4.77		1	20
Hydrocharis morsus-ranae	8.77	9.69	1	1	Rorippa amphibia	9.20	8.51		
Hydrodictyon reticulatum	8.79	7.74	1*	19	Rorippa nasturtium-aquaticum	8.42	8.08		
Hygroamblystegium fluviatile	5.41	5.29	1	21	Rorippa palustris	7.32	7.45		

Taxa	RM NI	RM HI	Aq	FG	Taxa	RM NI	RM HI	Aq	FG
Hygroamblystegium sp.	6.55	5.98	1	21	Rumex hydrolapathum	8.65	8.11		
Hygroamblystegium tenax	5.27	5.60	1	21	Sagittaria sagittifolia	9.24	9.32	1	12
Hygrobiella laxifolia	2.76	2.06	1	23	Scapania sp. (aggregated)	2.14	3.83	1	23
Hygrohypnum duriusculum	3.33		1	21	Scapania subalpina	3.21	2.18	1	23
Hygrohypnum eugyrium	4.28	3.90	1	21	Scapania uliginosa	2.66	1.97	1	23
Hygrohypnum luridum	2.80	3.83	1	21	Scapania undulata	2.05	4.00	1	23
Hygrohypnum ochraceum	2.96	3.87	1	21	Schistidium agassizii	2.23	3.45		
Hyocomium armoricum	1.96	3.60			Schistidium rivulare	5.16	3.48		
Hypericum elodes	2.66	4.70			Schoenoplectus lacustris	8.44	8.83	1	13
Iris pseudacorus	6.92	7.57			Schoenoplectus tabernaemontani	7.43	8.02		
Isoetes lacustris	3.02	5.90	1	4	Scirpus sylvaticus	6.45	6.85		
Juncus articulatus	3.10	4.27			Scorpidium revolvens	4.29	5.01		
Juncus bulbosus	1.89	4.35	1	4	Sium latifolium	7.08	7.93		
Jungermannia atrovirens	2.28	3.62	1	23	Sparganium angustifolium	2.26	4.81	1	13
Jungermannia exsertifolia	3.87	4.44	1	23	Sparganium emersum	8.32	8.58	1	13
Jungermannia obovata	2.97	4.40	1	23	Sparganium erectum	8.34	8.26		
Jungermannia paroica	4.00		1	23	Sparganium natans	3.59		1	13
Jungermannia pumila	3.29	3.01	1	23	Sparganium sp.	4.11			
Jungermannia sp.	2.41	3.49	1	23	Sphagnum denticulatum	4.84			
Jungermannia sphaerocarpa	3.08	2.48	1	23	Sphagnum sp(p)	1.07	2.92		
Lemanea fluviatilis	4.51	5.26	1	19	Spirodela polyrhiza	8.99	8.90	1	1
Lemanea sp(p.)	4.53	5.17	1	19	Spirogyra	6.45	6.37	1*	19
Lemna gibba	10.0	9.14	1	1	Stigeoclonium tenue	6.62	5.69	1*	19
Lemna minor	8.80	8.59	1	1	Stigonema sp	4.32	6.62	1	19
Lemna minuta	9.21	8.87	1	1	Tetraspora lubrica/gelatinosa	6.72	6.07	1	3
Lemna sp.	7.60	9.80	1	1	Thamnobryum alopecurum	4.22	4.89	1	21
Lemna trisulca	8.21	8.66	1	1	Tolypothrix penicillata	2.96	3.35	1	3
Leptodictyon riparium	7.57	6.58	1	21	Triglochin palustris	4.07	5.17		
Littorella uniflora	1.96	4.84	1	4	Typha angustifolia	7.57	9.05		
Lobelia dortmanna	2.72	5.26	1	4	Typha latifolia	8.87	8.42		
Luronium natans	4.37	5.57	1	4	Ulva flexuosa	9.52	8.43	1*	19
Lythrum salicaria	7.33	8.11			Utricularia intermedia	2.74		1	9
Marsupella aquatica	3.17	1.00	1	23	Utricularia minor	3.77		1	9
Marsupella emarginata	1.06	2.85	1	23	Utricularia sp	3.23		1	9
Marsupella sp.	1.24	2.75	1	23	Utricularia vulgaris s.l.	3.72		1	9
Mentha aquatica	6.27	6.71			Vaucheria sp(p)	8.41	7.60	1*	19
Menyanthes trifoliata	3.14	5.69			Veronica anagallis-aquatica	8.45	8.25		
Mimulus guttatus	5.79	5.67			Veronica beccabunga	7.31	6.99		
Mimulus sp./hybrid	5.60	5.42			Veronica catenata	9.32	8.48		
Monostroma sp.	6.86	6.43	1	3	Veronica catenata x anagallis-aquatica	8.34	7.92		
Montia fontana	3.35	3.56			Veronica scutellata	2.35	4.60		
Myosotis laxa	4.82	5.47			Zannichellia palustris	9.01	8.43	1	15
Myosotis scorpioides	6.83	6.98			Zygnematalean alga	6.45		1*	19

[1]River Macrophyte Nutrient Index
[2]River Macrophyte Hydraulic Index
[3]Species counted as aquatic in N_ATAXA
[4]Functional group membership
* Indicates taxa included in calculation of green filamentous algal cover

4.2.3 Validation

Part of the rationale for biological indicator scores is that they reflect environmental determinants of species distribution that cannot, or cannot readily, be measured directly. Consequently it is difficult to falsify species ranks, although it is possible to partition and test their underlying environmental basis. The most obvious way to validate a compositional metric is to correlate it with primary data from the same set of sites for an environmental variable that is sensitive to the pressure that the metric is designed to detect. There is an element of trade-off here since, while the lack of any correlation might lead to the conclusion that the metric was independent of the pressure and therefore of little utility, a perfect or near perfect correlation would equally mean that the biology was reflected closely by a readily measured environmental variable and was thus effectively redundant.

Figure 4.1 describes the relationship between phosphorus (as annual mean SRP) and RMNI and nitrogen (as Total Oxidised Nitrogen, TON) and RMNI, where RMNI is the cover weighted average of the expert-based metric described above. The stratification by survey method shows that JNCC survey data tends to have a somewhat smaller scatter but that the overall form of the relationship is unaffected by survey method.

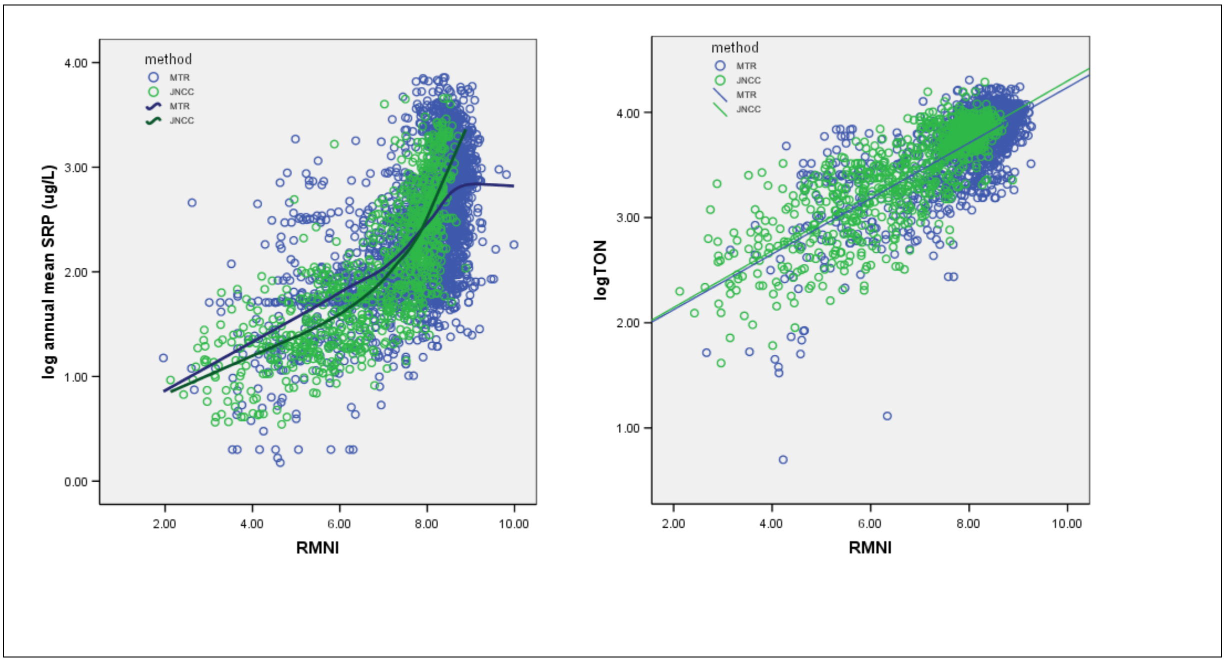


Figure 4.1 RMNI versus SRP and TON concentrations and the effect of survey method

The strength of relationship between a metric and a pressure is likely to be sensitive to the quality of biological data from which the metric value was derived. The RMNI metric is potentially sensitive to the number of contributing taxa, which may be small at any given site. When the number of taxa at a site was used to filter the surveys used to derive the RMNI versus SRP relationship it was found, by sequentially changing the filter, that the relationship was optimised when sites with five or fewer taxa were excluded. When the number of species recorded is very small (under five), a site may be grossly impacted by nutrient-related pressures but, given that exclusion of such surveys apparently strengthens the pressure-metric relationship, many impoverished sites are likely affected by other pressures, have been subject to recent natural disturbance, or are simply poorly surveyed (for example, the survey was performed by inexperienced surveyors, was incomplete or carried out under suboptimal conditions).

4.2.4 Comparison with MTR scores

At a global level there is a close correlation between site scores based on RMNI and site scores based on MTR (Figure 4.2). This suggests there may be little advantage in substituting RMNI for MTR.

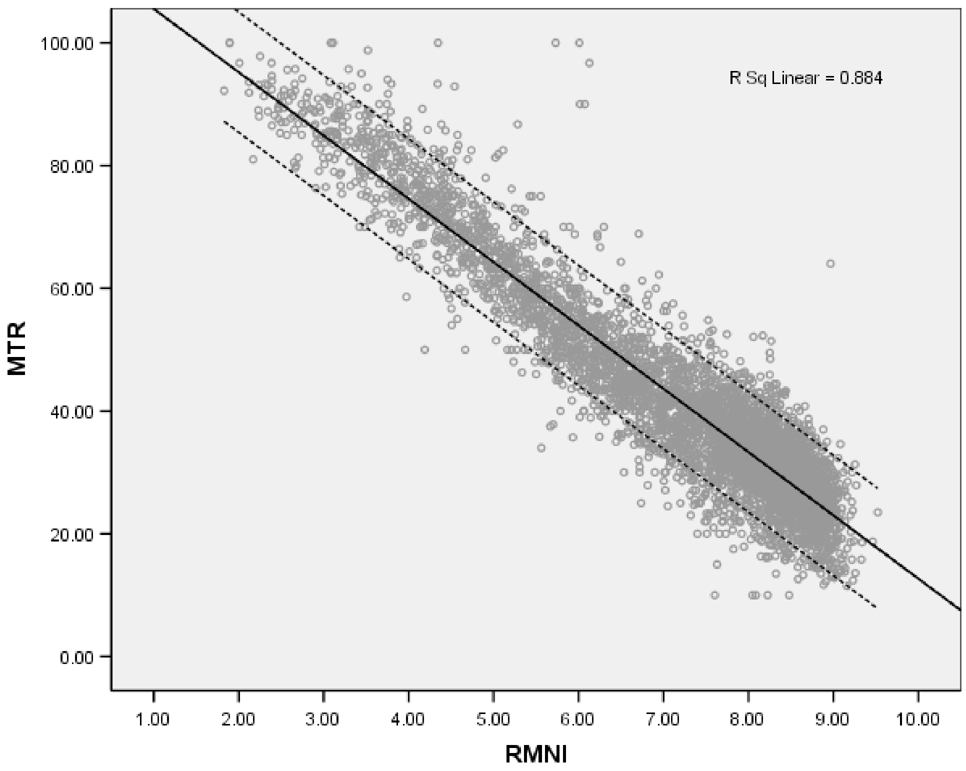


Figure 4.2 Global relationship between MTR and site RMNI scores

A comparison of the relationship between RMNI and SRP and MTR and SRP reveals that both relationships have a similar shape, although there is somewhat greater scatter in the case of MTR (r^2 = 45.7 compared to 47.5 for RMNI). The differential in favour of RMNI is greater when the correlation with TON is used as the basis for comparison (r^2 = 49.4 for MTR compared to 58.2 for RMNI). The differential between MTR and RMNI is reduced when surveys based on five or fewer taxa are excluded (r^2 = 51.4 for both MTR and RMNI versus SRP; r^2 = 57.2 and 62.7 for MTR and RMNI respectively versus TON). This suggests that MTR is more prone to the influence of species-poor surveys and that RMNI more closely approximates the true species nutrient optima. A comparison of the relationship between SRP optima and RMNI, and that with MTR, reveals considerably greater scatter at low MTR than at high RMNI scores (compare Figures 4.3 and 4.5).

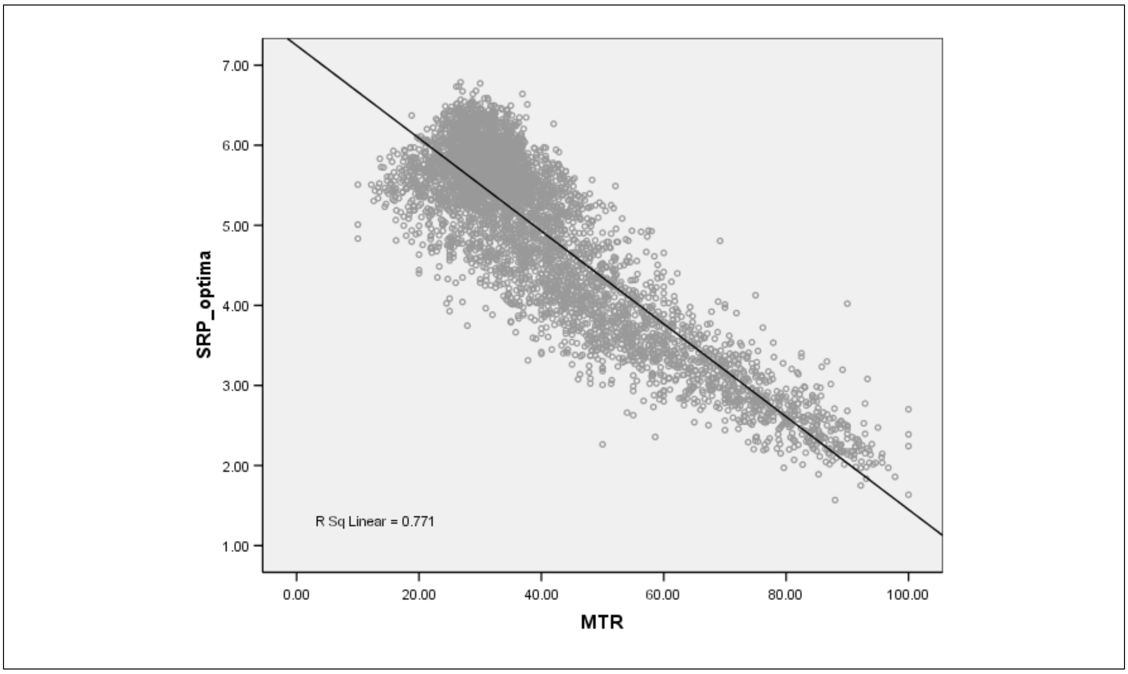


Figure 4.3 Relationship between site MTR scores and weighted average of species SRP optima

A graphical comparison between pre-existing MTR STR scores and RMNI species scores reveals, not surprisingly, that the two indices are strongly related (Figure 4.4). However, there are a number of significant outliers. These are itemised in Table 4.2, alongside the MTR values as predicted from the RMNI versus MTR relationship. At this stage it is assumed that, by integrating data from over 6,000 surveys, and by deriving scores statistically, RMNI scores provide a truer reflection of the position of river plants on a trophic gradient than do the MTR species scores derived by expert judgement. Previous assessments of the MTR system have highlighted many of these outlying species as being in need of revised scores, with the direction of change also matching recommendations (see Dawson *et al*., 1999, Szoskiewicz *et al*., 2006; Schneider, 2008). For example, Szoszkiewcz *et al.* (2006) recommended revising the MTR value for *Lemna minor* from four to three (as suggested here), although their suggested changes in MTR values for three other species (*Alisma lanceolatum*, *Fontinalis antipyretica* and *Sparganium emersum*) were not supported by our analysis. Scheidner (2008) noted that *Callitriche hamulata* had a less extreme score in French and German systems, consistent with its occurrence outside strongly oligotrophic rivers, while *Ranunculus fluitans* was more strongly associated with eutrophic rivers in continental Europe. The change in MTR value for *Callitriche hamulata* from nine to a predicted value of seven, and for *Ranunculus fluitans* from seven to a predicted value of four, is thus in line with these comments.

An analysis of the relationship between MTR values and empirical phosphate and nitrate optima in UK rivers (Dawson *et al*. 1999) also indicated that the scores assigned to *Ranunculus fluitans* and *Calitriche hamulata* were not in line with their empirical nutrient rank. Dawson *et al*. (1999) advised removing *Stigeoclonium tenue* from the MTR system since its empirical phosphate rank was so inconsistent with its MTR score (phosphate rank of six compared to MTR value of one). They did not recommend changes to the MTR scores of any other taxa but highlighted inconsistencies in the scores of 30 species compared with their empirical ranking on a phosphate or nitrate gradient. Half the species listed in Tables 4 and 5 in Dawson *et al*. (1999) also appear in Table 4.2 below and in all but one case (*Hippuris vulgaris*), the scale and direction of change is in agreement. On this evidence it seems appropriate to conclude that the RMNI system offers a better ranking of river plants on a fertility gradient.

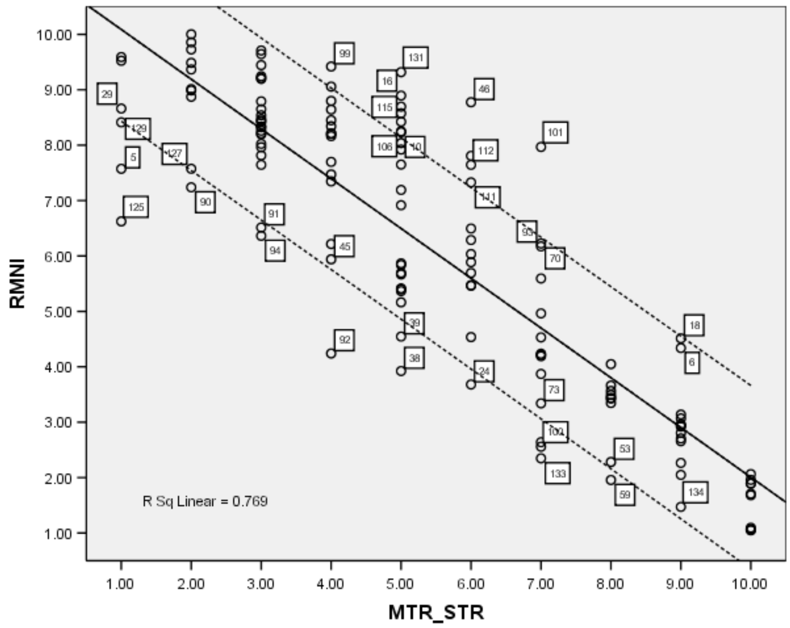


Figure 4.4 Comparison between MTR STR and RMNI species scores. Coded species symbols relate to Table 4.2.

4.2.5 Comparative value of empirical and expert metrics

Three approaches for deriving a compositional metric were examined. Their value was assessed by comparing the strength of the relationship between the site score and the annual mean SRP concentration at that site. The first approach uses the adjusted expert-based metric described above (RMNI), calculating the cover weighted average rank for each site with matching SRP data. The second approach follows an empirical route, using Canonical Correspondence Analysis with annual mean SRP as the dependent variable. The axis 1 site scores from this analysis represent the average of the species scores where these are constrained by SRP. The third approach simply derives the weighted average SRP concentration for each species based on the measured nutrient concentrations at the sites where that species occurs.

When the regression between species SRP optima and measured SRP values is used as the basis for metric validation, the coefficient fractionally outperforms that obtained between RMNI and measured SRP (r^2 = 48.1 versus 47.5). However, the RMNI versus SRP relationship is essentially an external calibration since RMNI values themselves are at no point trained upon the measured SRP values. This means that an externally validated model effectively performs as well as an internally calibrated one. A comparison of site scores based on SRP optima or RMNI indicates little material difference between these approaches. This relationship also illustrates the better performance of RMNI over MTR, as reflected in less scatter at high RMNI values.

Table 4.2 Comparison between original MTR STR and modelled scores based on RMNI species scores

Case no.	Taxa[1]	RMNI	Current MTR-STR	Modelled MTR	Current relative to modelled MTR	N
5	**Leptodyction riparium**	7.57	1	4	--	931
6	Apium inundatum	4.34	9	7	+	44
8	Azolla filiculoides	9.71	3	2	+	61
10	Berula erecta	8.24	5	4	+	254
16	**Butomus umbellatus**	8.89	5	3	++	350
18	**Callitriche hamulata**	4.51	9	7	+	480
19	Callitriche obtusangula	8.04	5	4	+	405
22	Carex riparia	9.06	4	3	+	406
23	Carex rostrata	2.64	7	8	-	211
24	Carex vesicaria	3.68	6	8	-	41
25	Catabrosa aquatica	8.70	5	3	++	35
29	**Cladophora agg.**	8.66	1	3	-	1081
38	**Equisetum fluviatile**	3.92	5	7	--	365
39	**Equisetum palustre**	4.55	5	7	-	373
42	Glyceria maxima	9.64	3	2	+	778
45	Hippuris vulgaris	5.94	4	6	-	82
46	Hydrocharis morsus-ranae	8.77	6	3	++	4
53	Jungermannia atrovirens	2.28	8	9	-	208
56	Lemna minor	8.80	4	3	+	1163
59	Littorella uniflora	1.96	8	9	-	144
70	**Oenanthe crocata**	6.22	7	5	+	961
71	Oenanthe fluviatilis	8.57	5	3	+	132
73	Pellia epiphylla	3.34	7	8	-	375
88	Potamogeton praelongus	7.81	6	4	++	28
90	Potamogeton trichoides	7.24	2	4	-	6
91	Potamogeton x fluitans	6.51	3	5	-	2
92	Potamogeton x lanceolatus	4.24	4	7	--	2
93	Potamogeton x nitens	6.17	7	5	+	1
94	Potamogeton x salicifolius	6.36	3	5	-	16
99	Ranunculus circinatus	9.42	4	3	+	53
100	Ranunculus flammula	2.56	7	8	--	543
101	**Ranunculus fluitans**	7.97	7	4	++	418
106	Ranunculus penicillatus	8.25	5	4	+	39
108	Ranunculus penicillatus subsp pseudofluitans	7.92	5	4	+	605
111	**Ranunculus (Batrachian) indet**	7.33	6	4	+	241
112	Ranunculus trichophyllus	7.65	6	4	+	44
113	Rhynchostegium riparioides	5.16	5	6	-	1604
115	**Rorippa nasturtium-aquaticum**	8.42	5	3	+	1188
125	Stigeoclonium tenue	6.62	1	5	--	74
127	Typha angustifolia	7.57	2	4	-	13
129	**Vaucheria sp.**	8.41	1	3	-	1143
131	**Veronica catenata**	9.32	5	3	++	218
133	Veronica scutellata	2.35	7	9	--	32
134	Viola palustris	1.47	9	9	-	143

[1]Species names in bold refer to species with a large number of records and a large differential between current and modelled scores.

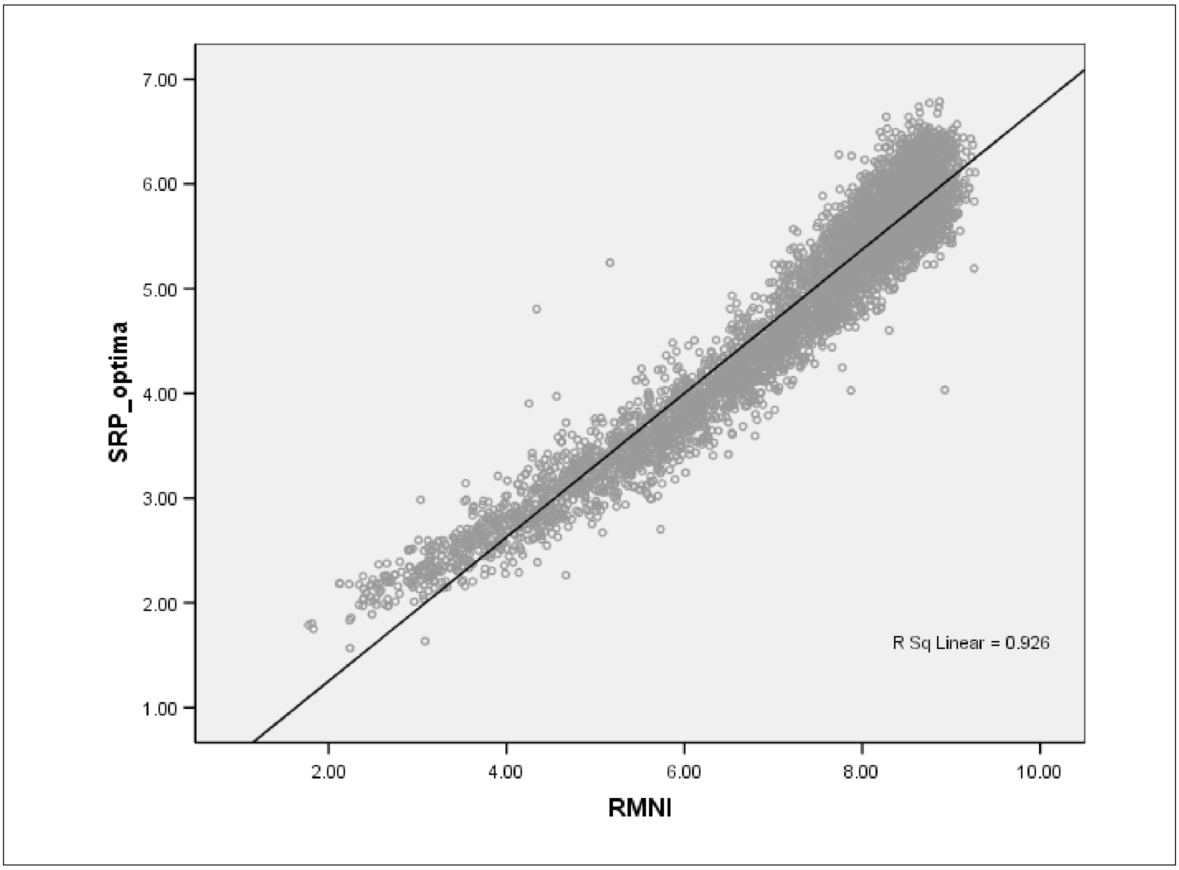


Figure 4.5 Global comparison between site RMNI and SRP optima derived from CCA

Figure 4.6 depicts the cover weighted mean SRP concentrations associated with a range of more common riverine macrophytes. The attraction in viewing data in this format is that values can be interpreted as direct concentrations. These can be viewed crudely as species-specific models which examine the distribution of each taxa independently, rather than via reciprocal averaging, in which each species has an influence on the optimum score of all other species. A comparison of the outputs of the first two approaches against the third approach (Figure 4.7) suggests that neither have a clear advantage. Evidently there is a strong and highly significant correlation between RMNI or CCA derived SRP optima and average SRP concentrations where different species occur.

Willby *et al.* (in preparation) used average total phosphorus concentrations to develop a common metric for intercalibration of classifications of lake macrophytes in Northern GIG. Warnelink *et al*. (2005) also found that this simple averaging approach was superior to a number of complex statistical models of species optima for prediction of pH in Dutch grasslands.

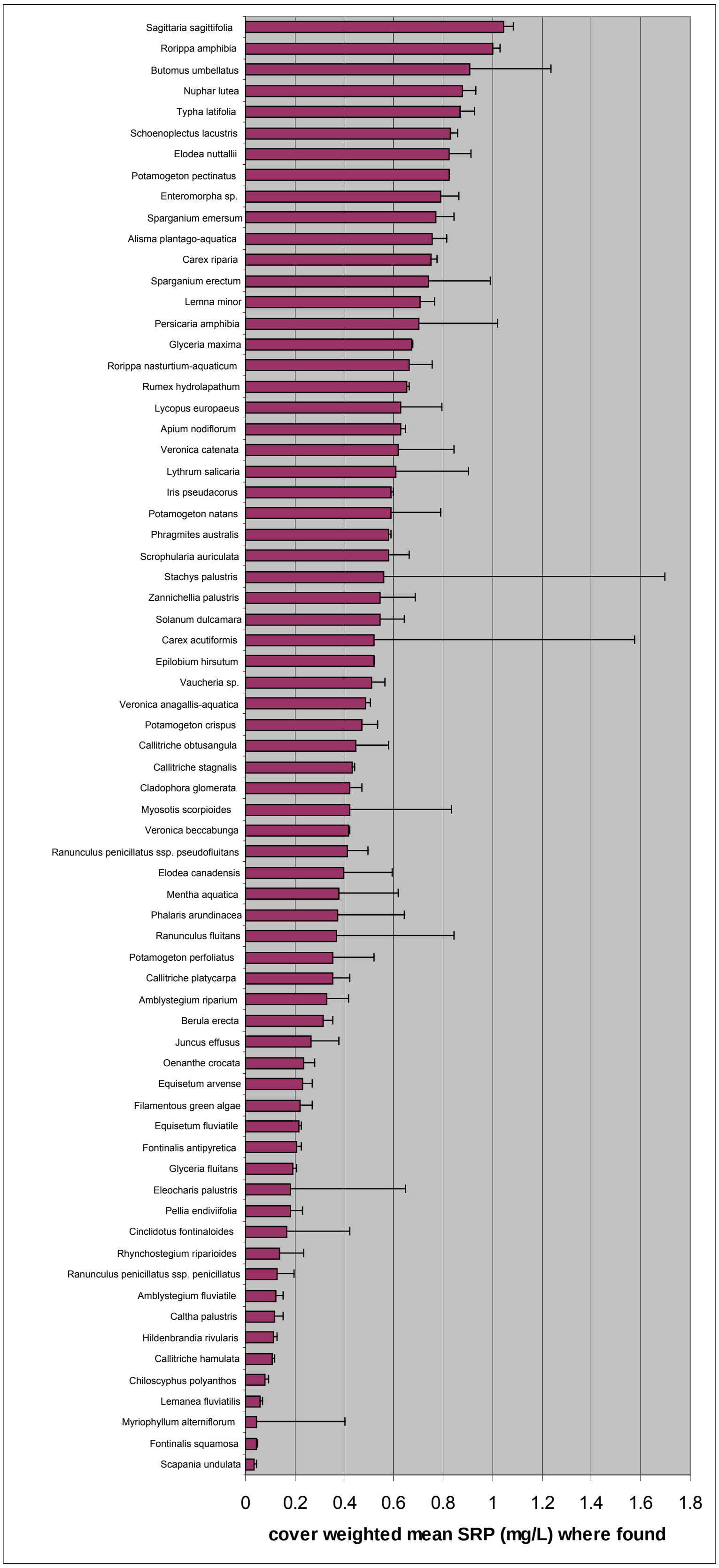


Figure 4.6 Cover weighted mean SRP concentrations associated with 70 more common riverine macrophytes

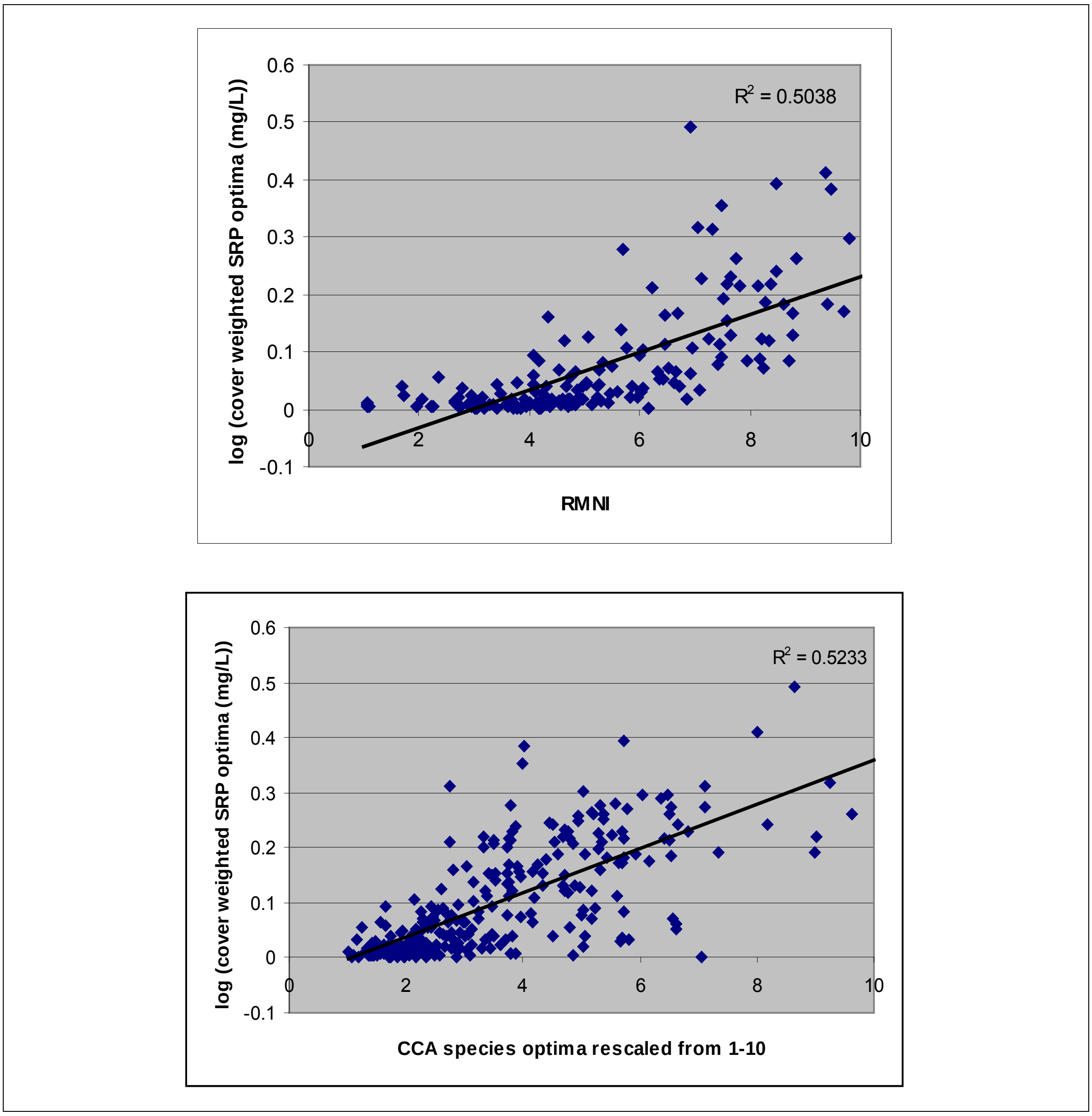


Figure 4.7 Comparison of relationship between cover weighted SRP optima derived from weighted averaging and RMNI species scores (upper panel) and CCA-based SRP optima (lower panel)

Superficially, deriving species environmental optima by weighted averaging may seem like a panacea. However, consideration of the underlying data used to derive these values indicates that they need to be treated with caution. Hence, the availability of nutrient data is strongly biased towards species occurring preferentially in fertile habitats (Figure 4.8). Comparatively little data on nutrient concentrations is available for less fertile habitats, and it seems highly likely that the data available for such habitats is biased towards locations where the potential severity of anthropogenic impact justifies frequent monitoring. Consequently, empirical nutrient optima for riverine macrophytes may present a distorted picture of the natural distribution of species in the absence of elevated nutrient inputs. Some form of weighting in favour of less fertile sites would be required to reduce this bias.

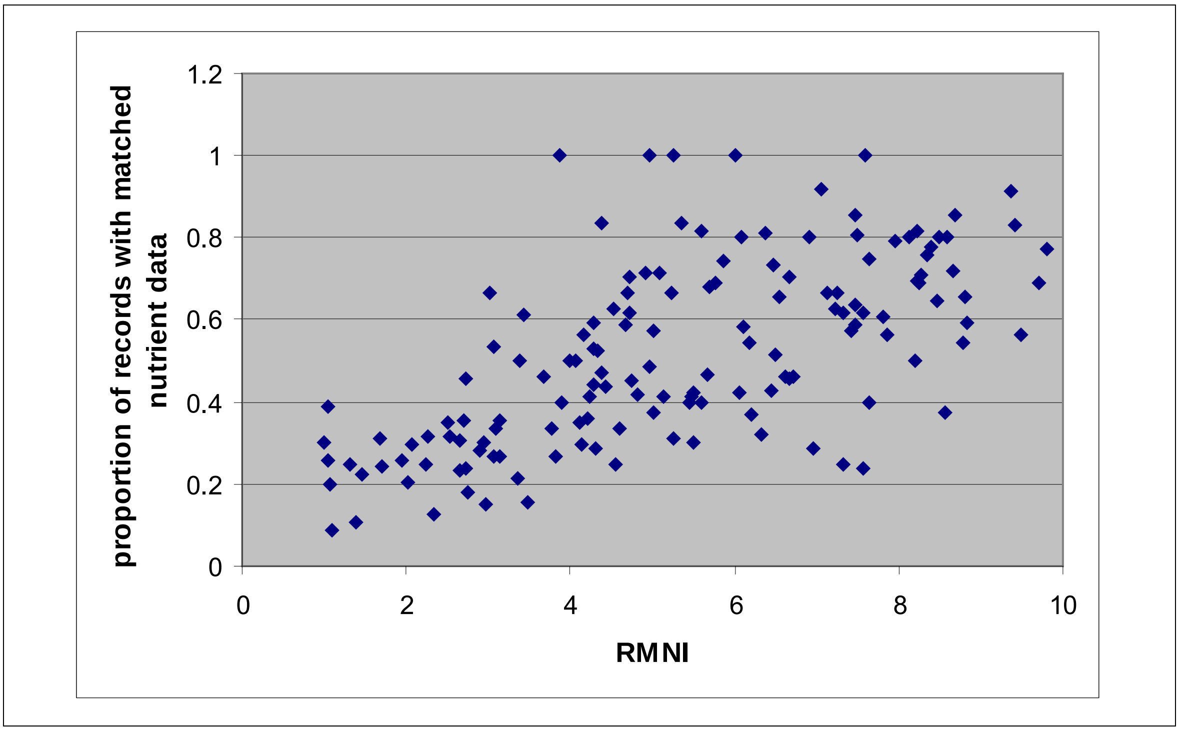


Figure 4.8 Bias in availability of data on nutrient concentrations at point of growth in relation to nutrient affinity of different riverine macrophytes

4.2.6 Difficulties in using compositional metrics

Although compositional metrics are appealingly simple to use, a number of less widely noted problems are associated with their use. The global SRP-RMNI relationship is, of course, a composite of data from different river types arranged predictably. However, it is not obvious from this the extent to which the pressure-metric relationship is type-specific. A particular problem noted in the case of the Lake Macrophyte Nutrient Index was the marked type-sensitivity of the pressure-metric relationship. A consideration of type-specific relationships between RMNI and SRP concentrations (Figure 4.9) indicates the same problem applies with rivers. There are two points of note:

i. For a given concentration of phosphorus, RMNI increases strongly with increasing alkalinity, especially when nutrient availability is low. Thus, at 60 ug/l in a high-alkalinity southern river the vegetation may be similar to that found at 20 ug/l in a high-alkalinity northern river.

ii. Sensitivity to higher concentrations of phosphorus is weak in the most base-rich river types. Thus, the RMNI versus SRP relationship is effectively saturated in the most base-rich river types once SRP exceeds 100-200 ug/l, yet continues to increase with increasing SRP in other river types.

Figure 4.9 illustrates the type-specific relationships between the site scores based on species SRP optima derived from CCA and SRP. It is clear from this that the two properties described above are not simply an artefact of the expert score approach, since they are reproduced when the SRP optima approach is followed.

The first point above is relatively easily explained. Inorganic carbon availability in the form of carbonate and bicarbonate ions increases with increasing alkalinity while macrophyte species that use inorganic carbon sources in photosynthesis naturally tend to dominate under conditions of increasing productivity. Consequently, for a given, concentration of phosphorus the productivity of vegetation will increase with increasing alkalinity which will be reflected in higher RMNI scores. In this respect, RMNI is more akin to a productivity index than a fertility index.

The second point is more difficult to explain and there could be several contributory factors. The scores for all river types tend to converge at around eight. It is probably an artefact of the statistical approach to deriving species scores that site scores will only rarely exceed nine, since the majority of species scores lie in the range four to eight. This in itself reflects the fact that most macrophytes tend to have a relatively wide ecological amplitude; few species that occur at very low fertility are strictly confined to this range and similarly few species occur exclusively at very high fertility. Moreover, those that do occur at extremes of fertility tend to be relatively rare and therefore have relatively little influence on the site score. Given the underlying tendency for RMNI to increase strongly with increasing alkalinity at low P, there is little remaining ‘room for manoeuvre’ in the highest alkalinity types with further increases in P. The relationship between RMNI and SRP would therefore be expected to saturate at progressively lower SRP with increasing alkalinity. The relationship for very high-alkalinity rivers may also be slightly distorted by the lack of contemporary sites with TP under 50 ug/l. However, it is evident that species turnover still occurs in higher alkalinity rivers at higher fertility, as captured by changing RMNI (RMNI 7-9), yet this turnover is apparently independent of water column SRP. The factors that could explain this phenomena in base-rich shallow lakes (such as an increasing influence of nitrate on composition when P is no longer limiting, sequestering of nutrients within plant tissue at high productivity, or an increasing influence of top-down controls on macrophytes in the naturally most productive lakes) are of limited relevance in rivers. Consequently, one must assume that factors associated with flow-related disturbance have an increasingly important influence on macrophytes at higher fertility in rivers, and have the potential to override the effects of fertility.

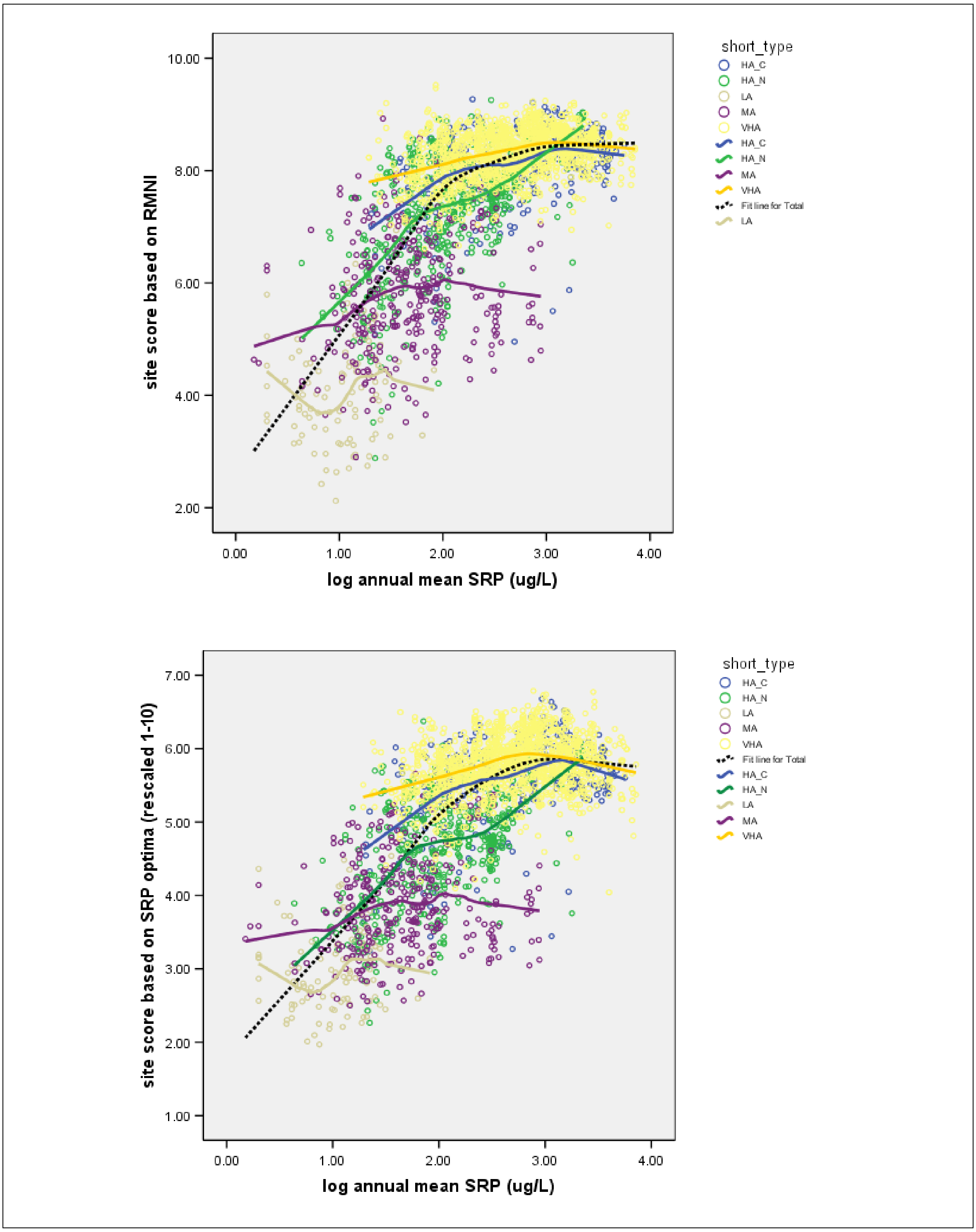


Figure 4.9 Type-specific relationships between (upper) RMNI and (lower) averaged SRP optima and measured SRP across a range of river types

4.2.7 Approaches for calculating the metric

For the purposes of this classification method, the RMNI score for a site is based on the cover-weighted average of the RMNI scores of the taxa recorded at that site.

The relative merits of different approaches to the calculation of site metric scores are discussed below. Table 4.3 presents the effects of weighting by cover, and of including or excluding species indicator (tolerance) values, on the regression between the site metric value and various nutrient determinands.

Table 4.3 Effect of different approaches to metric calculation on the strength of regressions (r^2) between a site-based metric and three chemical determinands.

	All data				Over five scoring taxa			
	pres-abs[1]	pres-abs (+ tol)[2]	cover wtd[3]	cover wtd (+ tol)[4]	pres-abs	pres-abs (+ tol)	cover wtd	cover wtd (+ tol)
SRP	46.3	45.9	45.1	44.8	49.6	49.2	48.4	48.1
TON	59.5	59.5	56.6	56.5	64.7	64.8	61.8	61.8
NH_4	21.4	21.1	21.6	21.5	21.6	21.4	21.4	21.4

[1] Unweighted, based only on presence-absence data
[2] Weighted by tolerance value only
[3] Weighted by cover value only
[4] Weighted by cover and tolerance values

The outcomes of the comparison of the effect of different approaches to calculating a site-based metric appear to be driven by several individually weak trends (Figure 4.10). Thus, low-ranking species tend to occur at lower cover where present, but are associated with higher indicator values. Hence, there is a trend for increasing variability in nutrient concentrations for higher ranking species. Taken together, this means that weighting by indicator value will have little effect because the effect of greater indicator value is offset by lower cover. The wide ecological amplitude of the majority of macrophyte taxa probably also constrains the length of the gradient in indicator values. Even when the effect of cover is removed, weighting by indicator value makes little difference, probably because assemblages tend to be composed of species covering a mixture of indicator values, rather than being composed of species with exclusively high or low values. All assemblages will contain some dominant species that are either generalists, or which use widely distributed resources and are therefore themselves widely distributed, and occur alongside some subordinate or transient species that have more specialist requirements (Grime, 1998). An *a priori* emphasis on specialist taxa should be avoided when calculating site metric scores. When the number of different taxa at a site is naturally small (say 10-15), as is frequently the case in rivers, it is preferable to base the site score on the largest number of species and to use the largest possible extent of the cover present, rather than to focus on a small number of specialist taxa with restricted cover. Our analysis also shows that an *a priori* allocation of species to useful, narrowly distributed, versus less useful, widely distributed categories (as used in the MTR system for bold species) is not supported by empirical tolerance values derived by analysis. Thus, within the MTR system, it was found that the CCA-derived tolerance values for both RMNI and phosphorus optima showed no statistical difference between bold and non-bold taxa.

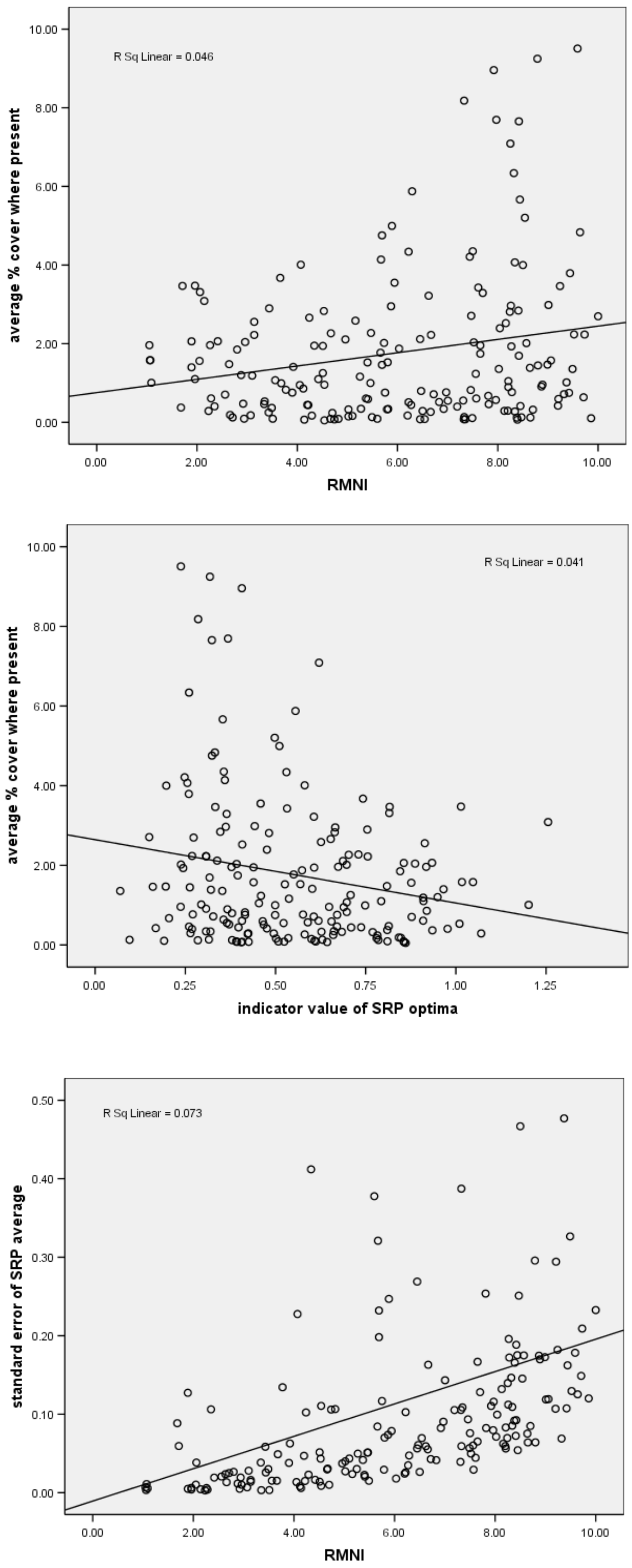


Figure 4.10. Relationships between RMNI, average percentage cover of taxa, and tolerance value or standard error of SRP values associated with scoring taxa

Inclusion of cover provides a better description of community structure. However, cover of individual taxa is probably more likely to reflect the extent and variety of physical habitat, and recent flow conditions, rather than nutrient concentrations. Species with higher cover values tend to be associated with a wider range of nutrient concentrations, therefore giving them more weight reduces the contribution of rarer species that are likely to be better indicators. Consequently, incorporating the influence of cover adds noise to the metric-pressure relationship. This result probably has little to do with variability in cover assessments between observers, since the calculation of a cover-weighted average is reliant on relative rather than absolute cover values; provided surveyors rank species in roughly the same order of cover at a site, the absolute cover that is assigned to a species is largely irrelevant. While the use of cover does not enhance the metric-pressure relationship, site scores based purely on the average of presence-absence data will be more vulnerable to the effects of surveyor variability since all species then carry equal weight. Thus, while most surveyors might report the same list of commoner taxa, detection of rarer taxa is likely to vary considerably. This will magnify the variability in the site metric score between surveyors. As far as controlling variability in river macrophyte surveys is concerned, there may be some advantage in cover-weighting.

Ecologists have debated the relative merits of deriving indicator scores from presence-absence or cover-based data. A review of different results (Diekmann, 2003) shows that the weighting method has very little influence and is therefore consistent with the analysis presented here.

4.2.8 Dependency on compositional metrics

While there is substantial empirical support for the use of compositional metrics, the above problems suggest that they cannot be used ‘blind’ for classification purposes without the risk of spurious results. Moreover, because they are effectively derivatives of structure they only address aspects of the WFD normative definitions. A number of other metrics described below have been developed to act as safeguards on the use of compositional metrics and to reflect the normative definitions more closely.

4.2.9 Criticisms of the use of river macrophytes for bioindication

Recent publications have questioned the use of indicators based on macrophyte species composition (see Paal *et al*., 2007, Demars & Thiebaut, 2008; Demars & Edwards, 2009). The most established use of macrophytes in bioindication relates to the detection of nutrient enrichment and criticisms have targeted the use of macrophytes for this purpose. These criticisms follow three main lines:

i. Due to high plasticity, most macrophytes have wide ecological limits and consequently have low indicator value.

ii. The distribution of macrophytes is correlated with numerous environmental variables and consequently macrophytes cannot be used reliably as indicators of nitrogen and phosphorus.

iii. Nutrient concentrations alone explain only a small amount of the overall macrophyte-environment relationship.

Macrophytes exhibit high plasticity which enables them to occur under a wide range of conditions. However, bioindication uses assemblage level information and the distribution of an assemblage is likely to be narrower than the distribution of its component species. The distribution of particular assemblages may still be relatively wide, while several different assemblages might occupy the same ecological niche (Paal *et al*., 2007). Even when reduced to metric level information macrophytes will present a noisier pressure-metric relationship, than will be the case for species with narrower distribution ranges, such as diatoms. Whether this should exclude macrophytes as indicators depends on the quality of the signal one expects from a bioindicator, and how this signal is to be used and interpreted.

Macrophytes will occur across a range of conditions. Rivers are strongly multivariate environments and the fact that, for example, phosphorus concentrations covary with alkalinity will always restrict the ability to assign a biological signal uniquely to a single variable. This problem applies to all riverine organisms; the difficulty of separating the influence of alkalinity and nutrients on macrophytes is as pertinent to macrophytes as it is to diatoms or invertebrates and would be best resolved by experimental manipulation under controlled conditions. While factors such as nutrient concentrations covary with alkalinity, nutrient enrichment as a pressure also covaries with other pressures and it is therefore likely to be difficult to use macrophytes in a strictly diagnostic sense. The use of macrophytes in a holistic sense, as indicators of ecological quality that are sensitive to a range of anthropogenic pressures that constitute ‘general degradation’, is encouraged by the WFD. This may be a more appropriate context in which to consider compositional metrics such as RMNI.

Nutrient concentrations tend to explain one to five per cent of the variation in river macrophyte composition depending on the dataset used. It is unavoidable that in small datasets the amount of variation in nutrient concentrations for a given set of basic environmental characteristics which arises from anthropogenic inputs is much smaller than in large datasets and consequently the importance of nutrients will appear correspondingly smaller. However, the absolute amount of variation that can be explained by nutrients is a misleading figure in a strongly multivariate environment. Major gradients in natural parameters that covary with nutrients will always dominate the analysis, while factors such as flow variability and physical habitat heterogeneity can accentuate or mitigate the macrophyte response to nutrients, thereby further weakening the signal. Indeed, under such conditions it seems naïve to expect the pure contribution of nutrients to be anything other than marginal, and is this is virtually assured through the use of variance partitioning techniques. A more realistic measure of the strength of influence of nutrients is to ask how much of the residual variation in macrophyte communities can be explained once the influence of structuring variables has been accounted for.

There is an overarching philosophical issue which relates to the degree of empiricism appropriate in auditing a biological indicator, and the degree to which simple measures of water chemistry can capture the pressure of interest. Fundamentally the prediction of or response to nutrients by macrophytes should not be confused with the detection of eutrophication. Nutrients were proposed by Grime *et al.* (1997) as ‘*the fundamental currency of vegetation processes at scales from the individual to ecosystems and landscapes’* and there are basic changes in macrophyte composition and traits, whether in lakes or rivers (Hilton *et al*., 2006), that equate to a central shift from nutrient limitation to light limitation which characterises the response to increasing productivity across global vegetation types. On these grounds, accepting or rejecting the use of macrophytes as indicators of nutrient-related pressures based purely on the strength of correlations with average ambient water column concentrations of nutrients seems a questionable test of validity.

4.3 Developing a metric to detect hydromorphological pressures

4.3.1 Background

While there is a long tradition in the use of macrophytes for water quality monitoring, the physical habitat, in terms of factors such as water depth, velocity, substrate and aspects of flow regime, also has a strong influence on the distribution of macrophytes (Baattrup-Pedersen *et al*., 2006; O'Hare *et al*., 2006). Consequently, there is a valid case to use macrophytes to infer information on the hydraulic conditions of a site. The use of macroinvertebrates to assess ecological effects of changes in flow regime is now well established through the Lotic-invertebrate Index for Flow Evaluation (LIFE) (Extence *et al*., 1999) which provides an expert ranking of velocity preferences. A broadly equivalent system of Macrophyte Flow Ranking (MFR) was devised by Holmes but has received little attention outside its intended use within the software developed to support Catchment Abstraction Management Strategies (CAMS) (Environment Agency, 2001). Despite the obvious potential, other European countries appear to have resisted the development of bioindication systems for hydromorphology based on macrophytes, perhaps believing there to be an overlapping response to nutrient enrichment and common types of hydromorphological change, or that various common species occur under a sufficiently wide range of physical habitat to be of limited indicator value.

4.3.2 Approach

An initial investigation was undertaken based on the metric Macrophyte Flow Rank (MFR) which is embedded within CAMS. MFR is based purely on expert judgement (Nigel Holmes personal communication). The intention in using MFR was to determine if this metric could be recalibrated based on the full set of species data, in the same way that had been done for MTR, using the algorithmic approach.

An identical exercise was carried out in which the site MFR was calculated for each survey in the database, using the existing expert MFR species scores. A Detrended Canonical Correspondence Analysis (DCCA) was then performed in which the site MFR forms the only explanatory variable. The species scores on the first axis of this ordination represent the weighted average of the site scores at which they occur, their true MFR optima. The species scores from this analysis were compared with the original MFR species scores (Figure 4.11).

This analysis showed that although the original MFR scores appeared to be robust at the upper end of the scale there was considerable recalibration of the lower scores, with those species originally assigned a value of one being recalibrated to values of up to 4.8. The limited range of MFR scores (one to five) also appeared to be constraining the distribution of species scores unduly.

A second analysis was performed to develop a plant hydraulic metric (River Macrophyte Hydraulic Index or RMHI) with a strong empirical basis. A Canonical Correspondence Analysis (CCA) was carried out using field data on mean substrate particle size, channel depth, channel width and stream energy (derived by assigning a numeric scoring to pools, slacks, riffles, runs and rapids) and paired plant survey data. The axis 1 species scores from this ordination were used to rank species from those associated primarily with small, shallow, coarse-bedded, high-energy channels, through to those associated with large, deep, fine bedded, low-energy channels. Species scores were normalised to a range of one to ten.

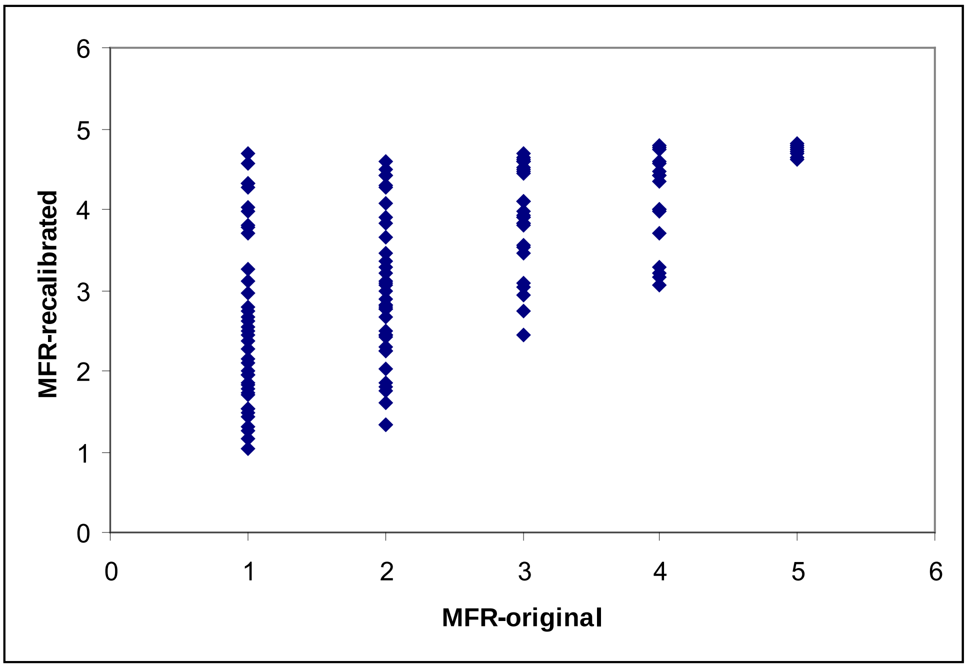


Figure 4.11 Relationship between original and recalibrated Macrophyte Flow Ranking scores

At a global scale, a close positive correlation between RMNI and RMHI scores must be expected, since fertile conditions are most likely to occur on soft, readily weathered rocks that are amenable to cultivation and which will naturally yield low-gradient, low-energy channels with fine substrates. The result of this is a close relationship between RMNI and RMHI site scores in the global dataset (Figure 4.12).

However, it is important to also consider the relationship between RMNI and RMHI scores at a species level, since it is differences at this scale that will contribute to relative differences between RMNI and RMHI at a within-type level, and relative to the values that would be expected for these metrics under reference conditions. Figure 4.13 illustrates the correlation between RMNI and RMHI species scores. While there is a strong underlying relationship (r^2 = 78), because species of fertile environments will tend to be associated with lower energy conditions, it is the scale of variation in RMHI for a given value of RMNI that is important. Thus, at a RMNI value of 7.5 the RMHI score can vary between 5.8 (species of relatively high-energy environments) and 10 (species of very low-energy environments). Likewise for an RMHI species score of seven, the equivalent RMNI score can range between five (mesotrophic) and nine (hypertrophic).

There are some clear advantages in adopting a hydraulic metric that has an unambiguous empirical basis. While it is possible to envisage an expert metric based on nutrient affinity because a variety of evidence (such as land cover, local geology, visible indicators of undesirable disturbance) can be combined to generate a set of species scores, it is much more difficult to see how the same approach could be applied to something as multidimensional as hydromorphology. A comparison between the original MFR scores and RMHI scores suggests that there is no clear underlying hydromorphological signal to the original scores. A comparison between RMHI scores and recalibrated MFR scores shows that the relationship is improved significantly compared with the original MFR scores. However, even after recalibration, MFR scores in the range four to five are unrealistically compressed (Figure 4.14).

One of the major weaknesses in the empirical approach to deriving suitable species scores is scale. When survey data is viewed at a reach scale the hydraulic habitat characteristics of different species are effectively averaged out, when, in reality, different species may occupy small patches that differ considerably in their hydraulic characteristics. Hence, *Carex rostrata* attracts a much lower RMHI score than other *Carex* species because it tends to share sites with species themselves characteristic of flashy upland, oligotrophic rivers, even though the physical microhabitat in which *Carex rostrata* occurs is characterised by the same sluggish flow and fine sediment under which other large *Carex* species tend to dominate.

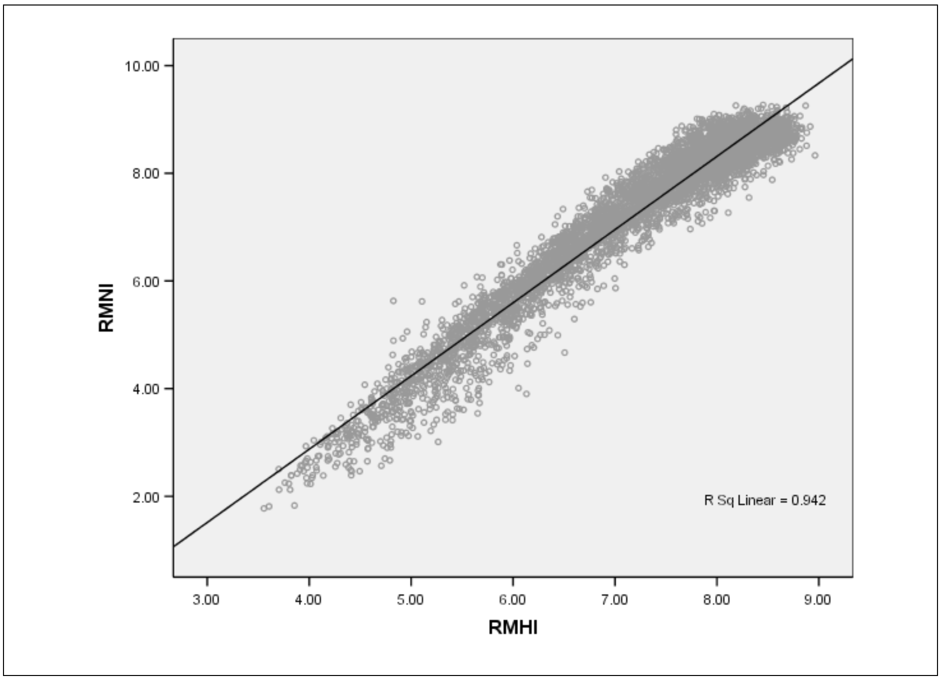


Figure 4.12 Global relationship between site scores based on the metrics RMNI and RMHI

4.3.3 Validation

The validation of a hydromorphological metric is more difficult to undertake since hydromorphology is multidimensional, and simple visual assessments of the hydromorphological condition of a reach of river, for example through River Habitat Survey, may not reflect changes in hydromorphology as perceived by macrophytes. A further difficulty is the fact that overlapping pressures may mitigate or accentuate one another. For example, sedimentation due to catchment soil erosion may be reduced in a channelized river as a result of less sediment retention, but may be accentuated in another reach overwidened by livestock poaching, or affected by flow abstraction.

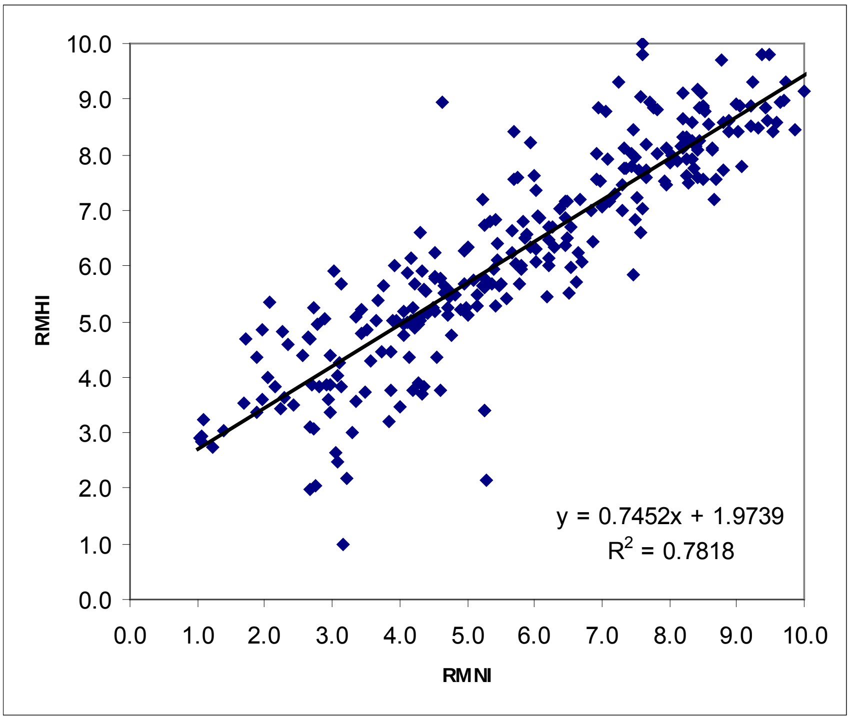
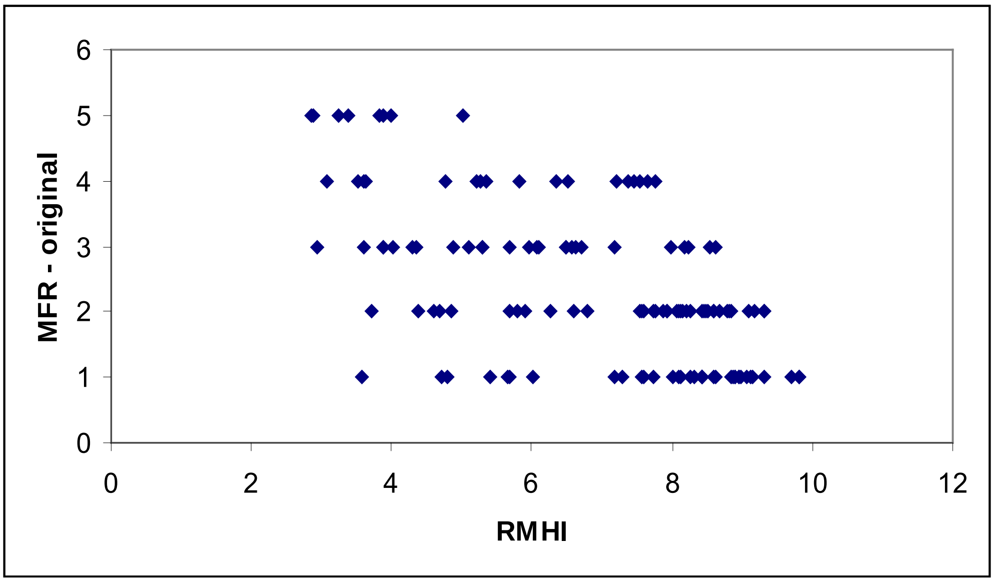


Figure 4.13 Relationship between species scores for the compositional metrics RMNI and RMHI

The principal basis for the use of the RMHI is to detect when a reach has shifted disproportionally towards a low-energy depositional environment, relative to the hydraulic environment that would be expected under reference conditions. Impacts such as sedimentation, channel widening, flow abstraction and introduction of physical structures such as weirs are all likely to result in a shift to an essentially lower energy, depositional environment. In its current form the RMHI metric operates in a unidirectional sense, only identifying increases in RMHI outside the range of variation that occurs under reference conditions. However, other hydromorphological pressures may result in an increase in stream energy, a shift to a more erosive environment, and consequently a decrease in RMHI scores relative to reference conditions. This would be expected to yield an EQR for the RMHI metric greater than unity. Currently this scenario is not identified, although various impacts (such as reduced sinuosity, flow augmentation, hydropeaking, removal of large woody debris, channel dredging or incision, sediment starvation by upstream impoundments) could all promote a shift in aquatic vegetation towards species with lower RMHI scores. Consequently, in future it may be necessary to identify absolute differences between an observed and expected RMHI score, not merely a deviation in one direction.



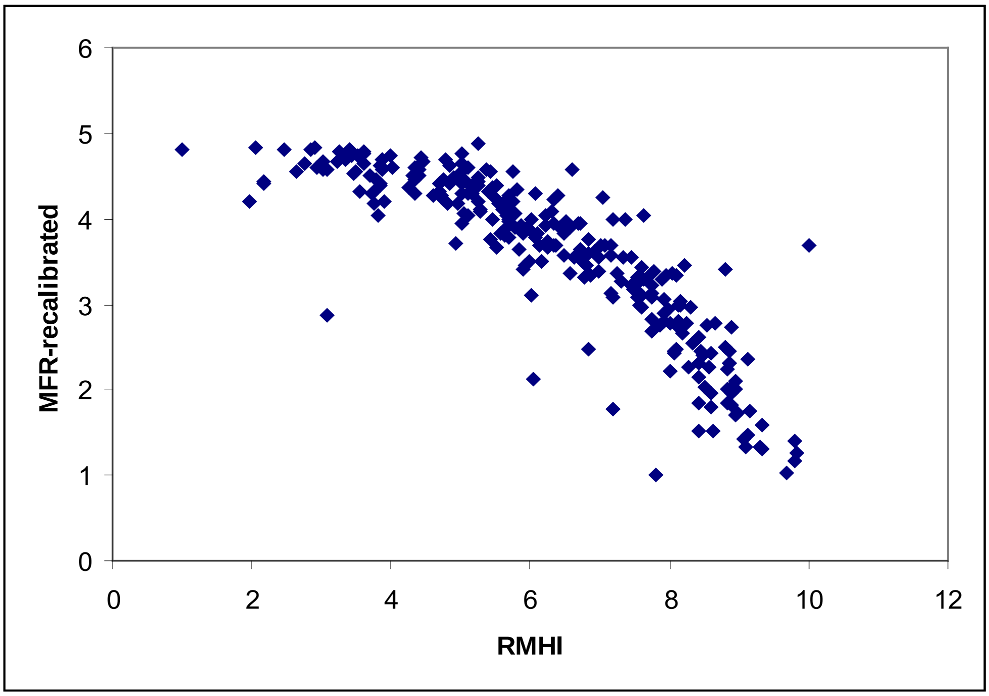


Figure 4.14 Comparison between species scores based on an empirical index, RMHI, and the original (upper) and recalibrated (lower) scores from an expert ranking (MFR) of macrophytes based on flow optima

Limited evidence to support the use of the RMHI metric is available from a comparison between metric values and Habitat Modification Scores (HMS) or Habitat Quality Assessment (HQA) scores, extracted from matched River Habitat Survey data. The Habitat Quality Assessment tends to give greater weight to geomorphological features associated with unimpacted high-energy environments. Consequently, one would nominally expect to find an inverse correlation between RMHI and HQA across a range of river types. Figure 4.15 shows that this holds for high and very high-alkalinity rivers, which tend to have lower average slope values, but does not hold for moderate and low-alkalinity stream types.

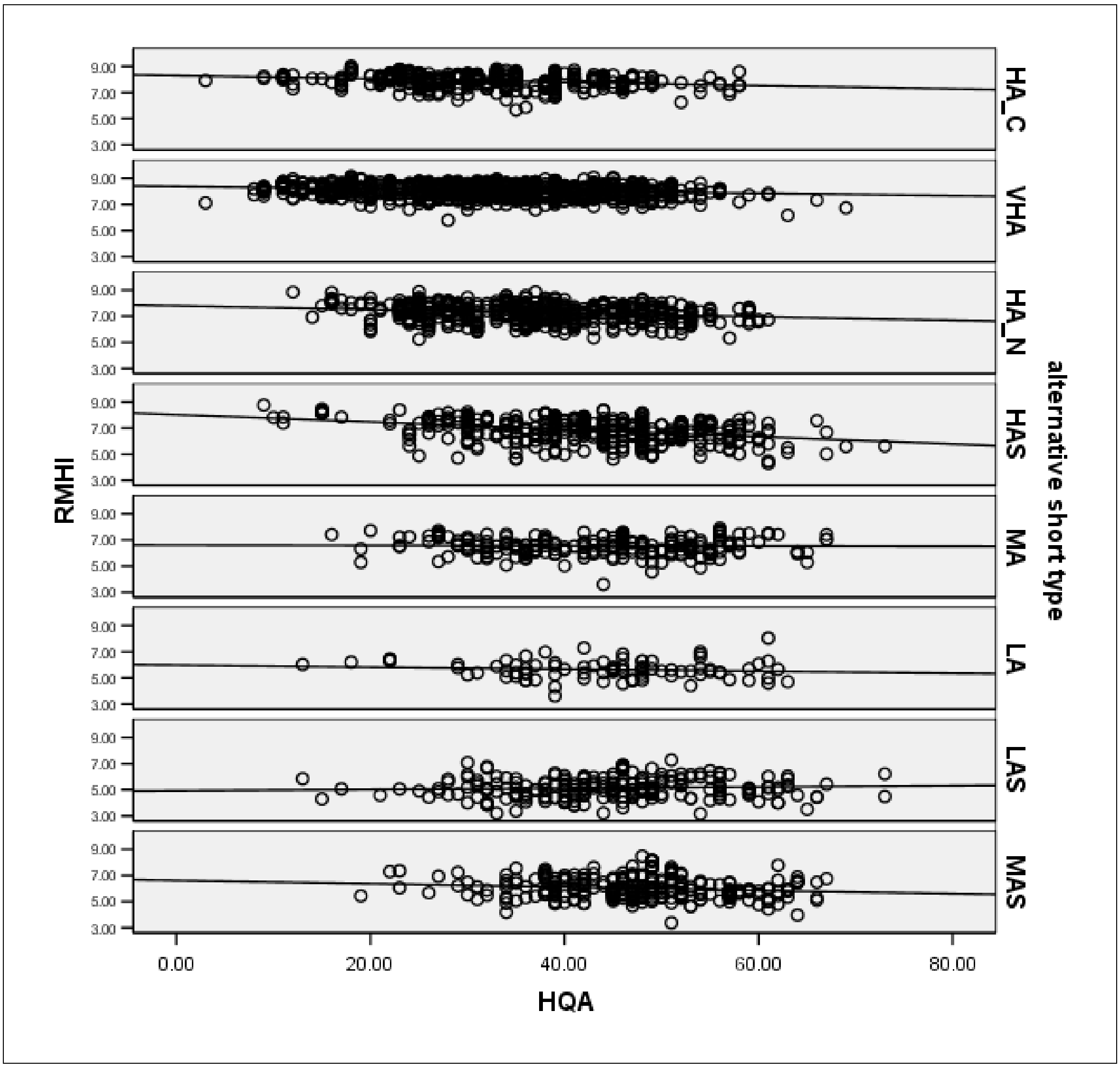


Figure 4.15 Correlations between observed RMHI site scores and Habitat Quality Assessment scores from paired River Habitat Survey data. Correlations are significant in the upper four panels (typically explaining 5-10% of the variation in RMHI) but are non-significant in the lower four river types.

In terms of the HMS, impacts that tend to lower stream energy and increase the degree of habitat modification might be expected to result in an increase in RMHI scores relative to unmodified reaches of the same river type. Arguably this trend should be more readily evident in intrinsically lower energy systems. Figure 4.16 illustrates the variation in RMHI values in different river types, stratified by Habitat Modification Class. Only in the case of the lower three panels covering three high-alkalinity subtypes (HAS, HA_C and HA_N) is there a significant positive difference between RMHI scores at HM Class 4 and 5 and those at HM Class 1. This offers tentative support for the RMHI metric in more base-rich rivers, but in other river types there is no clear direction of change in stream energy in response to hydromorphological impacts, and consequently there is no change registered in average RMHI compared to unimpacted streams of the same type. This may reflect the fact that hydromorphological alterations may both increase and decrease stream energy, thereby averaging out the macrophyte response.

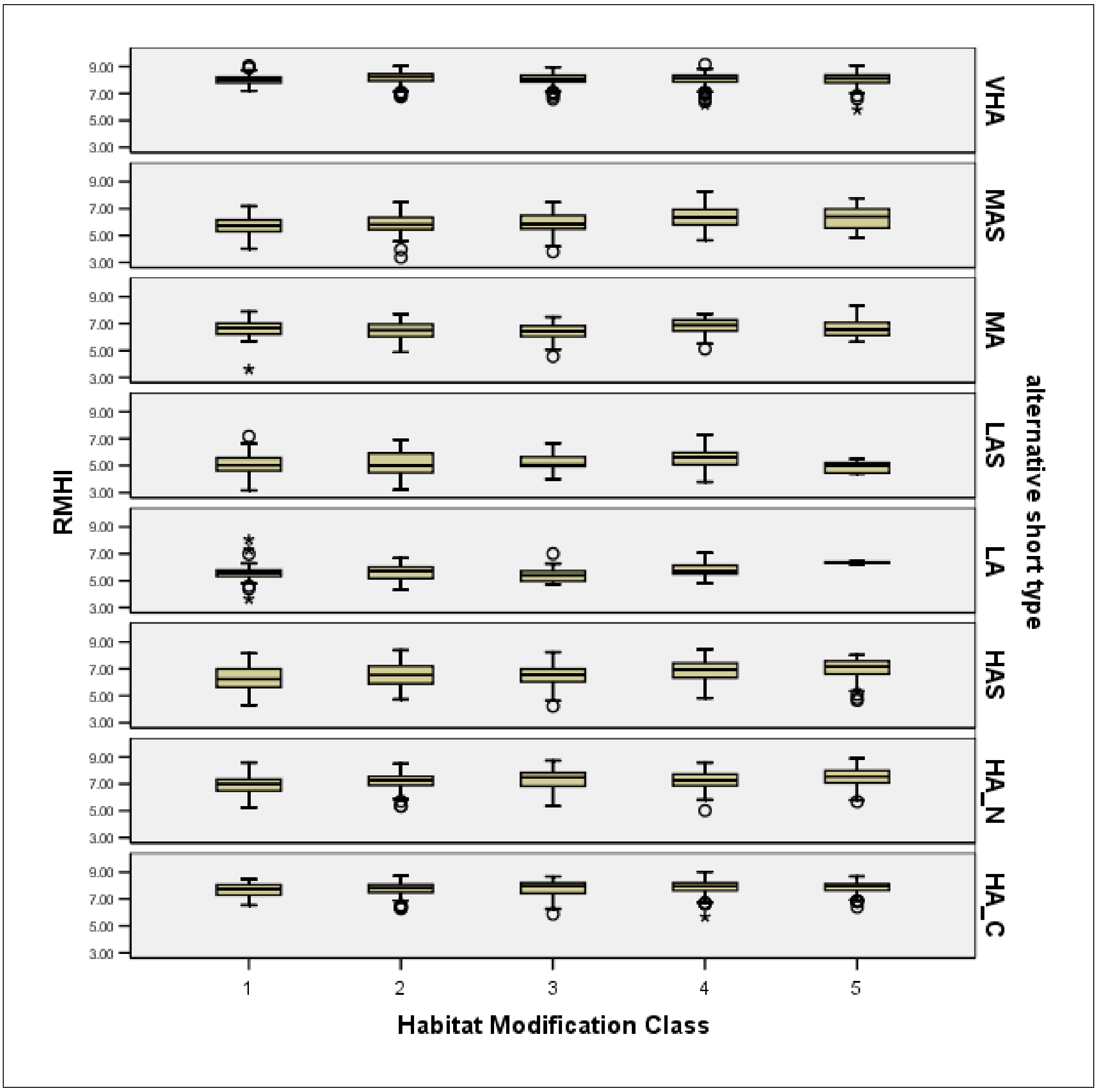


Figure 4.16 Comparison of differences mean RMHI score between different Habitat Modification Classes in relation to river type. Only the lower three panels exhibit a significant positive difference between RMHI values at HM Class 4 and 5, and those observed at HM Class 1.

The RMHI metric strongly depends on measured slope values. Thus, there is a significant inverse correlation between RMHI and slope in all stream types and across survey methods (Figure 4.17). Slope is treated as an intrinsic variable and will therefore be used to predict the value of RMHI to be expected under reference conditions. However, slope is derived automatically from the digital rivers network (technically it is derived from the difference in height between 1,500 m upstream of a survey site and 500 m downstream of the site) and it is difficult to determine without manual checking if the measured slope is the influenced slope (the product of hydromorphological modifications), or the naturalised slope (the slope that should occur in that reach in the absence of modifications). The naturalised slope should be used to predict the expected RMHI value under reference conditions, but it seems inevitable that, in some cases, the slope value that is used will be the product of an impact. This will mask the effect of any hydromorphological change on the RMHI value.

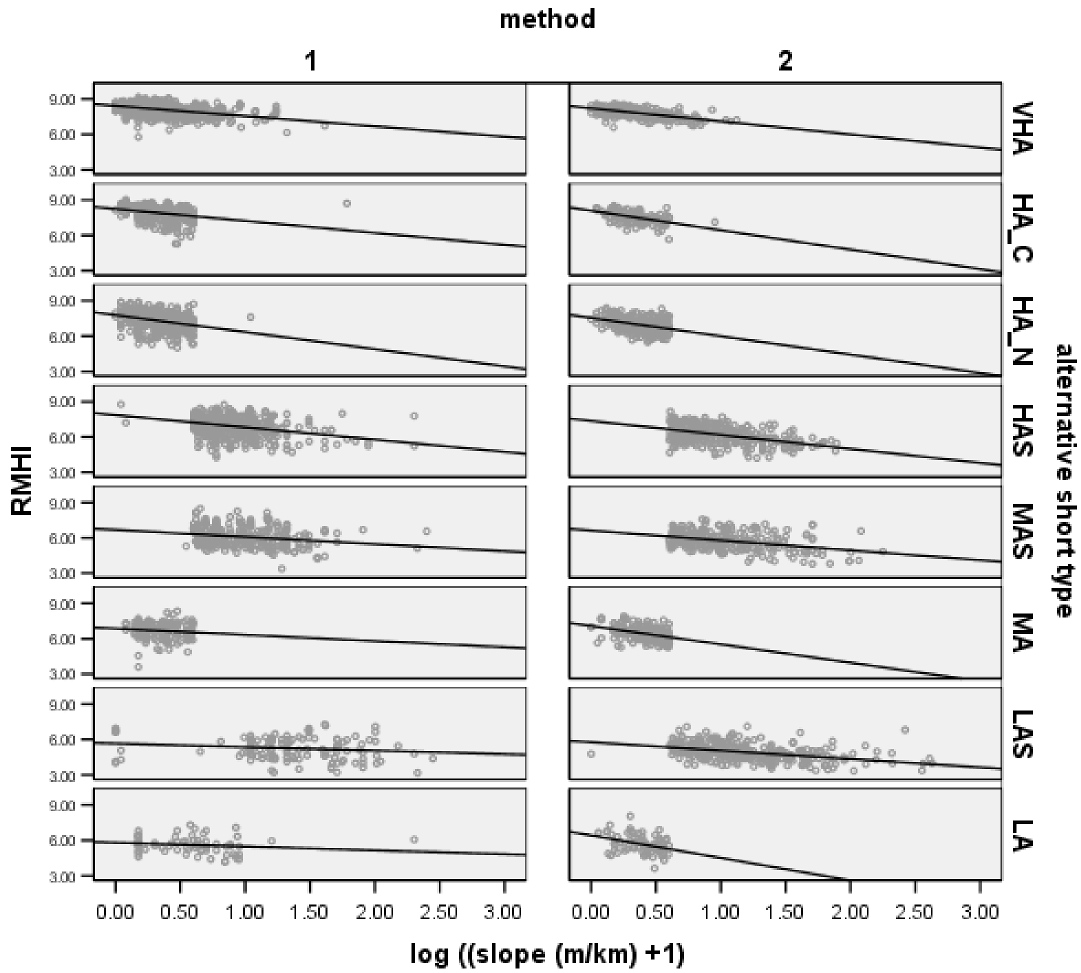


Figure 4.17 Relationship between RMHI scores and slope in different river types. Method 1 refers to MTR surveys; Method 2 refers to JNCC surveys.

4.3.4 Value of a hydraulic metric to indicate nutrient enrichment pressures

Hilton *et al*. (2006) have proposed that epiphyte growth on leaves is the main cause of macrophyte loss in rivers under increased phosphorus concentrations. They suggest that, to reduce the effects of shading associated with epiphyte growth on submerged leaves, the composition of macrophytes will shift towards floating-leaved or emergent growth forms as fertility increases. The penalty for this will be increased hydraulic drag and greater biomass losses under high flows. Species with floating-leaved or emergent foliage are, by necessity, more characteristic of lower energy environments within rivers, occupying reaches or habitats within reaches, where the effects of hydraulic drag are reduced. From the perspective of the RMHI metric these are therefore high scoring species. One obvious prediction is that the site RMHI value will increase with nutrient concentrations. This is confirmed by Figure 4.18. This figure also reveals that

RMHI exhibits proportionally more sensitivity over the SRP range 0.1-1.0 mg/l than the metric RMNI, suggesting that turnover in vegetation composition at higher nutrient concentrations in rivers is more related to accommodation of potential light limitation than it is to differences in nutrient affinity. Consequently, the metric RMHI broadens the window over which changes in aquatic vegetation can be related to nutrient concentrations.

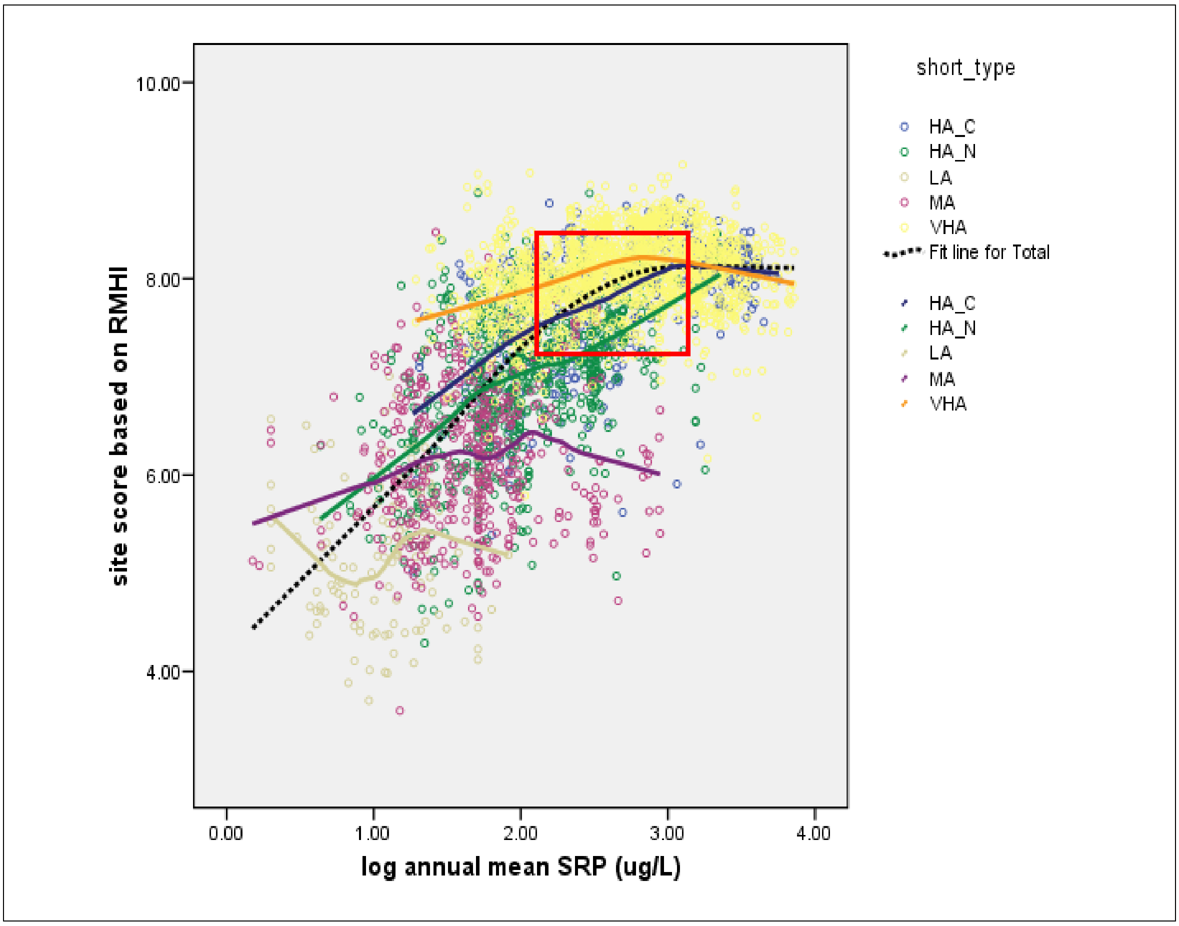


Figure 4.18 Relationship between type-specific and global RMHI scores and phosphorus concentrations. Note the greater relative change in RMHI over the SRP range 0.1-1.0 mg/l compared with saturation of the RMNI response in this range.

4.3.5 Calculating the metric

The site RMHI score is calculated in exactly the same way as the RMNI score, as a cover weighted mean of the individual species scores. RMHI scores for individual taxa are indicated in Table 4.1. A small number of species that have an RMNI score lack a RMHI score, since these taxa were not present at sites for which supporting information on depth, substrate, width and stream energy was recorded. These taxa are therefore excluded from the calculation if they happen to be present.

4.4 Richness metrics

4.4.1 Background

Richness is often viewed as an indicator of biological quality (Ricklefs & Schuter, 1993), although its use requires caution since a conservative change in richness might, for example, conceal a significant shift in composition. The WFD does not explicitly require the use of diversity metrics to assess deviation from reference condition for macrophytes, indeed diversity is only mentioned in the case of benthic invertebrates. However, given that diversity is related to composition and abundance (Huston, 1994), its use could be considered implicit. Moreover, it may offer a more sensitive indicator of pressures that reduce the species pool without strongly altering its composition. There is widespread evidence of the link between biodiversity and ecosystem function (Hooper *et al*., 2005) and various attributes of lake vegetation, such as the stability of cover, appear to be dependent on the richness of the vegetation. It is less clear how relationships between biodiversity and ecosystem function might be manifested in rivers. However, a high diversity of species is likely to be indicative of an intact flow regime, high physical habitat heterogeneity, and good levels of connectivity with upstream and lateral propagule sources. Macrophyte species richness has therefore been proposed as an effective measure for assessing the degree of hydromorphological alteration of streams (O'Hare *et al*., 2006). Given the inevitable influence of richness on some other metrics, there is an underlying statistical justification for its inclusion. The inclusion of richness can mitigate against overreliance on compositional metrics from species-poor sites.

4.4.2 Approach

Two metrics were considered. The first related to a simple count of the number of strict aquatic taxa per survey reach. The second metric was based on a count of the number of different functional groups represented in a survey, in which functional groups were composed of species exhibiting similar attributes of morphological or regenerative traits. Functional groups were defined as described in Willby *et al*. (2000), based on a matrix of morphological and regenerative traits. Twenty-three groups (Table 4.4) were defined using cluster analysis and macrophytes assigned to each group based on similarities in trait attributes (for example, 'small leaves' is an attribute of the trait 'leaf area'). Our approach to deriving functional groups is a formalised version of the manual clustering exercises carried out by previous workers in classifying plant life forms or growth forms (see den Hartog & Segal, 1964; Hutchinson, 1975; Wiegleb & Brux, 1991). Our analysis does not explicitly include ecophysiological traits (such as bicarbonate usage) due to lack of adequate information. Consequently it might be more useful for relating plants to physical habitat characteristics, although to some extent, outward growth form must be an expression of plant physiology. N_FG and N_TAXA are obviously highly positively correlated through a sampling effect. The extent of this correlation would be diminished using a cruder functional classification but this would still need to be reasonably fine-grained to be useful; thus, examples of all lowland river types, regardless of their quality, will contain submerged, floating-leaved and emergent plants, so groupings need to be at a higher resolution than this.

The parallel use of these richness metrics is threefold:

i. N_FG is likely to be less sensitive to variation in surveyor effort, level of experience and taxonomic resolution.

ii. N_FG has a more transparent link to ecosystem function because the morphological attributes of different functional groups will to some extent dictate their ability to perform a range of macrophyte-dependent functions in lakes (such as habitat support for higher trophic levels, nutrient recycling, sediment stabilisation). It could also be considered to better reflect niche diversity or occupancy at a site.

iii. High N_TAXA is likely to contribute to functional stability by insuring against FG loss that may occur through random extinction of individual taxa (the so called ‘insurance hypothesis’; Naeem & Li, 1997; Yachi & Loreau, 1999). This may be less relevant to rivers than to lakes. However, high taxonomic richness relative to functional group richness might infer a high level of connectivity with diverse upstream or lateral propagule pools

Table 4.4 Classification of river plants into 23 functional groups

FG	Descriptor	FG	Descriptor
1	lemnids and ricielids	13	vallisnerids and sagittarids
2	charophytes	14	parvopotamids
3	blue-green algae	15	magno- and parvopotamids
4	isoetids	16	parvonymphaeids and magnopotamids
5	elodeids and ceratophyllids	17	magnopotamids
6	peplids	18	batrachids
7	myriophyllids and herbids	19	red or green filamentous algae
8	umbelliferids	20	encrusting algae
9	utricularids	21	pleurocarpous mosses
10	magno and parvonymphaeids	22	acrocarpous mosses
11	herbids and elodeids	23	leafy liverworts
12	magnonymphaeids and sagittarids		

4.4.3 Validation – assembling an evidence base

General

A simple consideration of the relationship between richness, as numbers of aquatic plant taxa per reach, and the core typing variables alkalinity and slope, indicates that richness is driven principally by physical constraints rather than alkalinity, or any of its various correlates (Figure 4.19). Thus, richness is highly dependent on slope (increasing as slope declines) but shows no trend in relation to alkalinity.

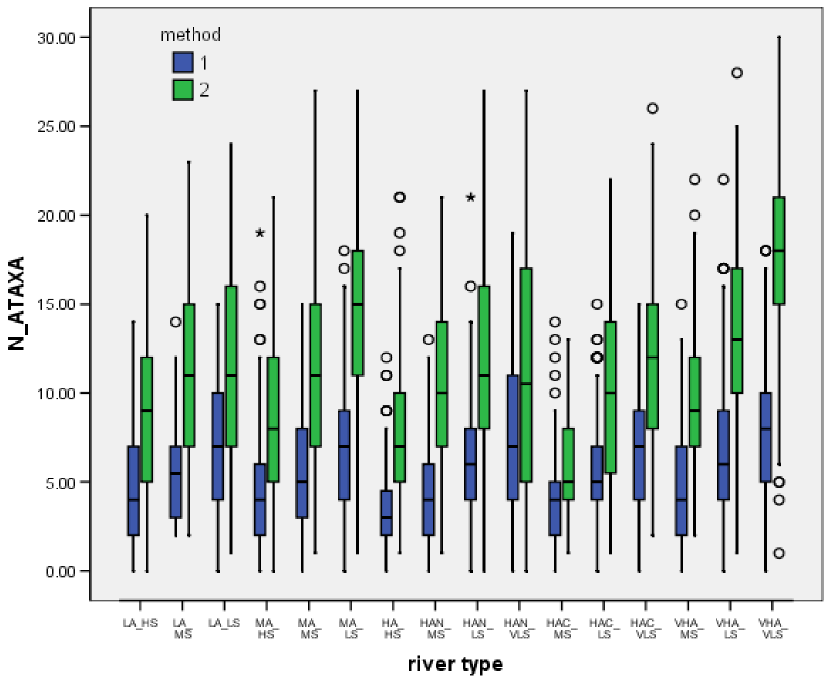


Figure 4.19 Relationship between number of hydrophyte taxa (N_TAXA) and river type using data collected by two methods (1) MTR and (2) JNCC. River types are ranked in order of increasing productivity. Note the weak signal relating to river alkalinity and the strong signal of increasing richness with decreasing slope that is a feature across all alkalinity subtypes and with both survey methods.

Nutrient-related pressures

The relationship between nutrient concentrations (TON and SRP) and the number of aquatic taxa recorded per survey was explored on a river type and survey method-specific basis. In terms of TON (Figure 4.20), there were consistent reductions in richness across both survey methods in very high-alkalinity rivers (VHA). There were significant effects of TON concentration on richness in high-alkalinity continental rivers (HA_C) but these were not consistent between methods. The JNCC survey data revealed a significant (negative) effect of TON supply on richness in high-alkalinity steep (HAS) rivers, and a significant positive effect on richness in low-alkalinity steep (LAS) rivers but these effects were not replicated by the MTR survey data. For orthophosphate, a range of relationships were evident. Richness was significantly negatively correlated with increasing phosphorus in high-alkalinity northern rivers of both slope types. Moderate alkalinity, steep rivers, high-alkalinity southern rivers and very high-alkalinity rivers showed more complex relationships in which richness tended to increase initially with enrichment but declined in the most fertile sites (Figure 4.21). When the intensity of land cover in the upstream catchment was used as a surrogate pressure indicator (Figure 4.22), there were found to be negative or unimodal effects of more intense land use on N_TAXA in very high-alkalinity, high-alkalinity, low-moderate gradient, northern rivers and high-alkalinity steep rivers.

The effects of nutrient-related pressures on macrophyte richness in rivers are somewhat opaque. This suggests that macrophyte richness is primarily a function of other factors, or that in rivers, the adverse effects of increasing fertility on richness are often mitigated by other factors such as water velocity, flow variability, shade or bed instability. Generally these factors would be expected to curb the reduction in species richness that might occur through competitive exclusion at high fertility, either due to vigorous growth of other macrophytes or shading by epiphytic algae. Moreover, in systems with long retention times increasing planktonic chlorophyll is a major influence on macrophyte growth and diversity, whereas in short retention systems these same mechanisms do not apply. These conclusions suggest that the stronger and more consistently negative effects of intensification of catchment land use are mediated through other mechanisms, such as siltation of the bed or degradation of the riparian zone, rather than directly via any increase in nutrient concentrations.

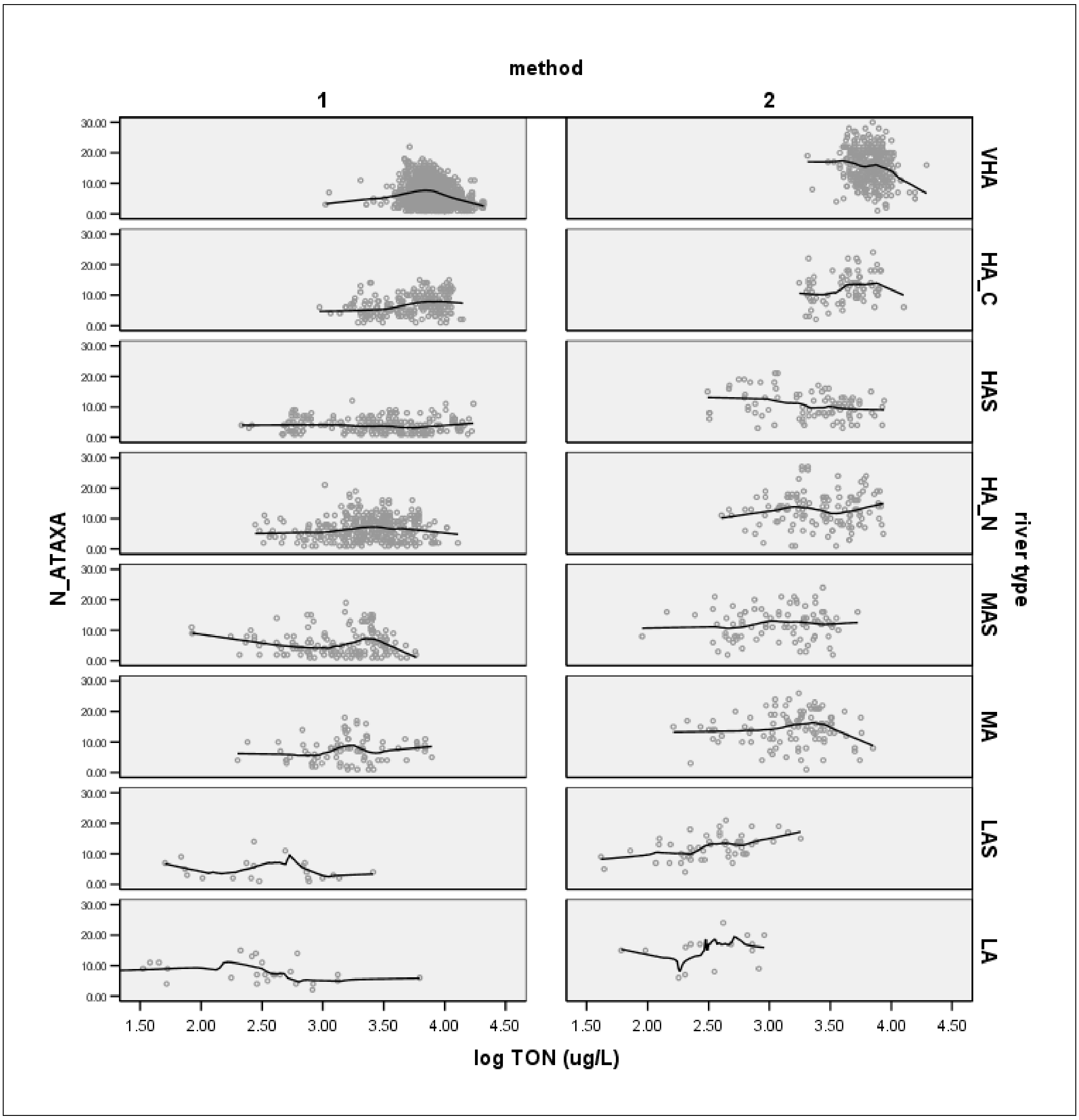


Figure 4.20 Relationships between number of aquatic plant taxa per reach and concentrations of Total Oxidised Nitrogen. Method 1 = MTR; Method 2 = JNCC. Lines shown fitted by LOWESS. Unless stated below all regressions are not significant at p = 0.05. Method 1: VHA & HA_C significant at p = 0.001. Method 2: LAS significant at p = 0.001; VHA & HAS significant at p = 0.01; HA_C significant at p = 0.05.

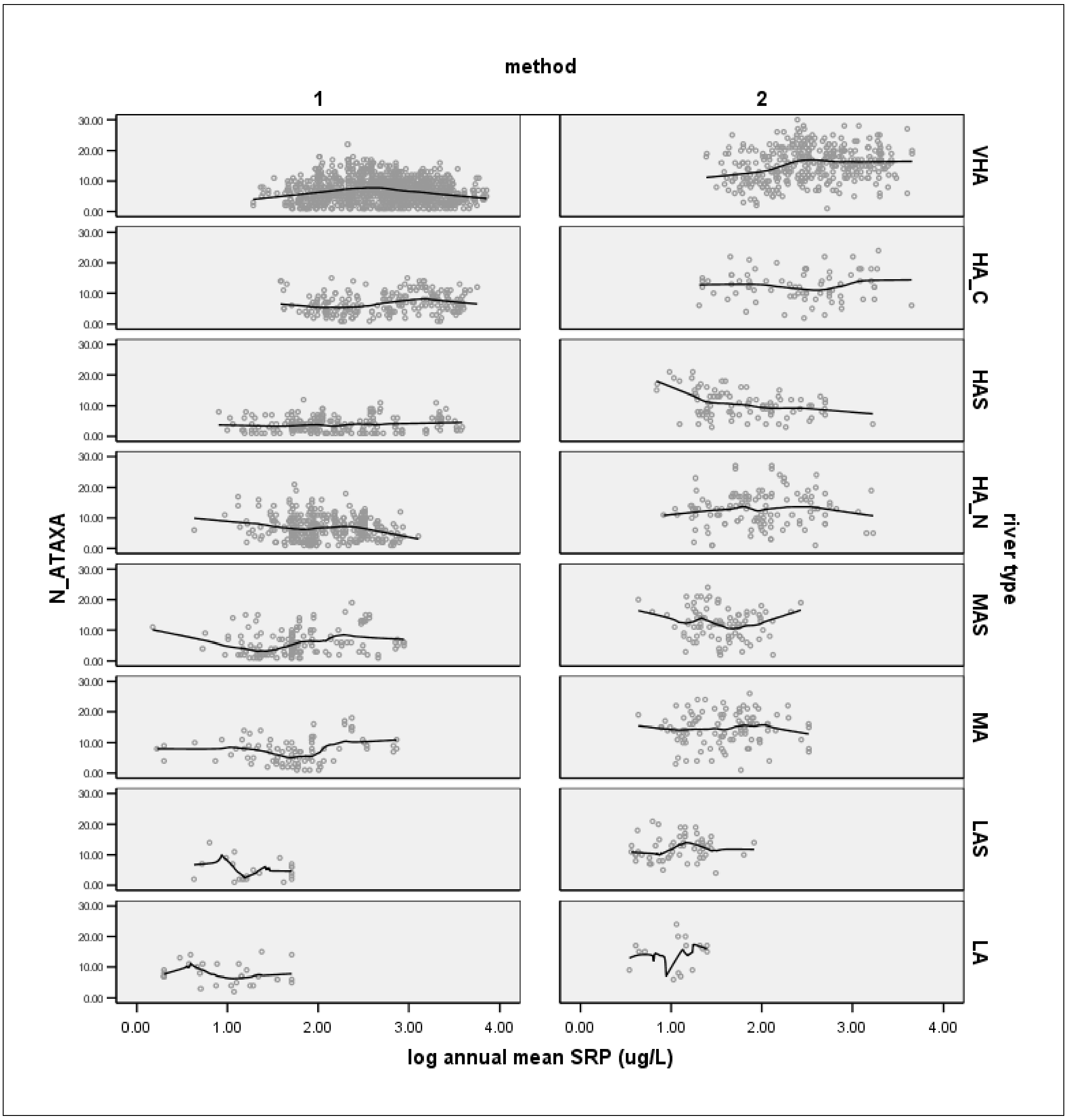


Figure 4.21 Relationship between SRP concentrations and N_ATAXA across a range of river types and two survey methods. Line shown fitted by LOWESS. Relationships are significant at p = 0.001 in type VHA (both methods); type MAS (method 1) and HAS (method 2); significant at p = 0.01 in types HA_C and HA_N (method 1); and significant at p = 0.05 in type MA (Method 1) and MAS (Method 2). Method 1 = MTR data; Method 2 = JNCC data.

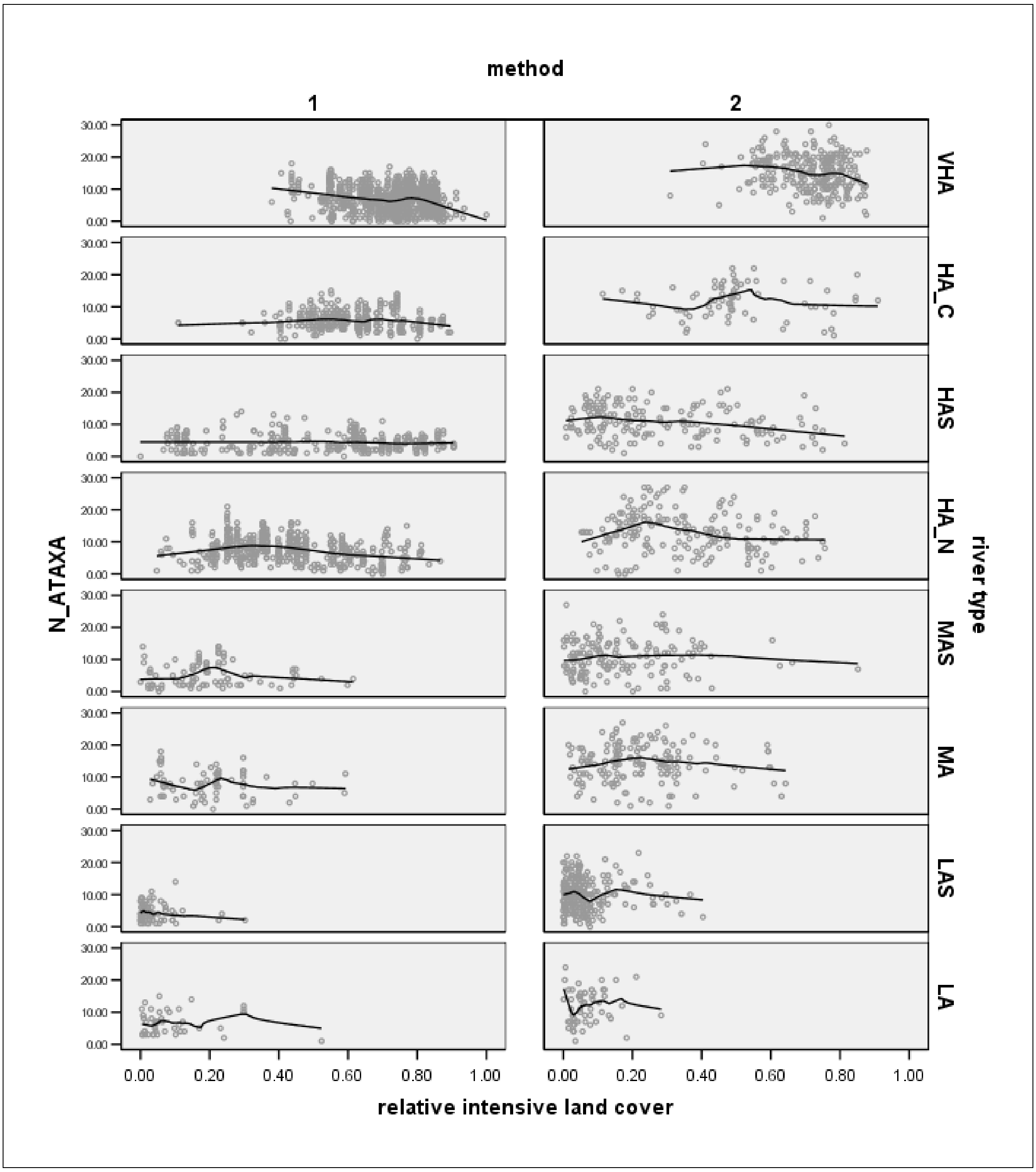


Figure 4.22 Relationships between N_ATAXA and intensity of upstream catchment land cover in a range of river types and across two survey methods. Lines shown fitted by LOWESS. Relationships non-significant except VHA (Method 1) and HA_N (both methods) significant at p = 0.001 and VHA and HAS (Method 2) significant at p = 0.01. Method 1 = MTR and Method 2 = JNCC data.

Hydromorphological pressures

Arguably macrophyte richness will be strongly influenced by physical habitat diversity within a reach. Hydromorphological alterations that lead to reduced habitat diversity should therefore translate into reduced macrophyte richness.

The primary resource for assessing the value of richness metrics in relation to hydromorphological impacts is the link to the Habitat Modification Scores (HMS) within River Habitat Survey. Figure 4.23 illustrates the relationship between HMS, in terms of a Habitat Modification Class (HMC), and taxonomic or functional group richness. The key points to note are that:

i. There is a highly significant effect of HMC on both richness metrics when either survey method is applied, although the effects are demonstrated much more clearly by N_TAXA. However, the effect is relatively subtle, typically amounting to a loss of two to three species between HMC 2 and 5 irrespective of which method is used.

ii. The effects of HMC are more evident within the MTR dataset, presumably because the longer survey length in the JNCC method increases the chance of finding small populations of those taxa adversely affected and not detectable at smaller sampling scales. It is also likely that richness is reduced by other factors at high HMC due to the nature of sites surveyed.

iii. There is a small stimulatory effect of low levels of habitat modification on richness as seen by the consistent and statistically significant increase in both metrics within either method between class 1 and 2. This indicates that small levels of habitat modification are likely to increase habitat diversity (for example, through the addition of coarse substrate)

iv. Type-specific effects are not shown but typically follow the pattern shown by the global dataset and are consistently weaker within the JNCC dataset.

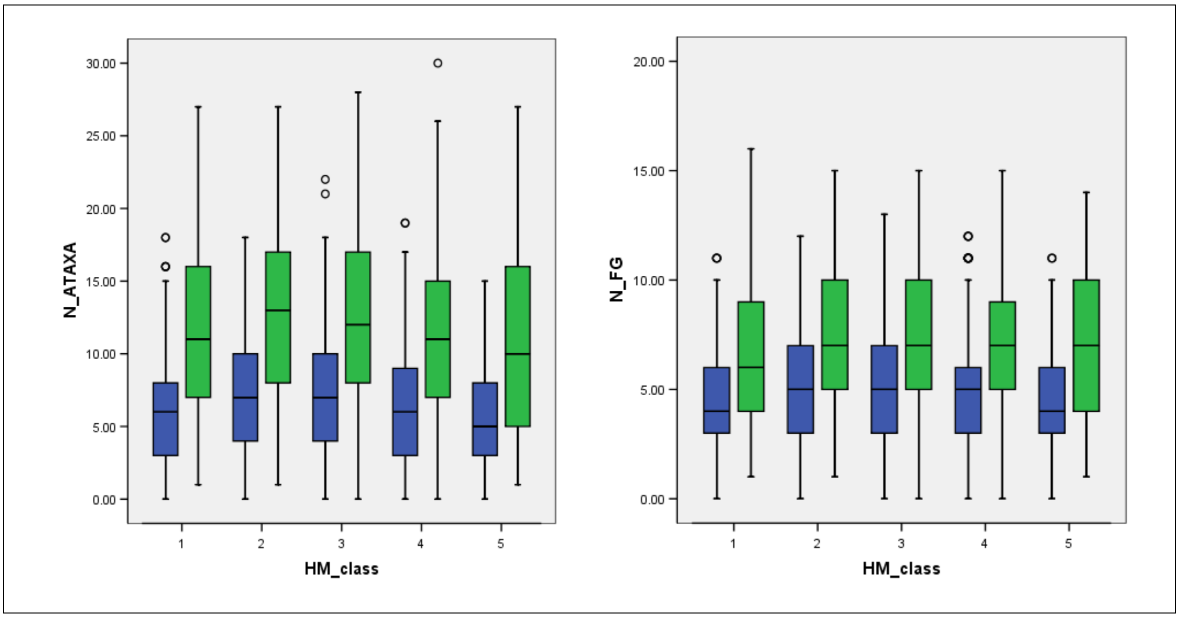


Figure 4.23 Global comparison of two richness metrics (N_TAXA and N_FG) in relation to RHS Habitat Modification Class (HMC) and two survey methods (blue = MTR; green = JNCC). In all cases the effect of HMC is significant at p = 0.001. Post hoc tests (Tukeys) indicate that in all cases there is a significant increase in richness between HMC 1 and 2.

A complementary analysis was undertaken using HMS associated with channel re-sectioning, since this was identified by CCA as the most important influence on river macrophyte species composition of the range of specific activities considered by HMS. This analysis indicated a rather different pattern to that described above, with richness being independent of the channel sectioning score in MTR surveys (p = 0.29) but strongly negatively affected in JNCC surveys ($r^2 = 0.09$, $p = <0.0001$). This is presumably a function of the scale of the impact relative to the scale of sampling reach. Thus, re-sectioning is proportionally more likely to impact on longer reaches because such reaches have a greater probability of capturing rarer microhabitats (debris dams, point bars, large boulders in lowland rivers) that contribute to richness and which are largely destroyed by re-sectioning. The periodicity with which such habitats occur means that they have a lower chance of capture within the MTR sampling unit. Consequently the impacts of re-sectioning appear to be less.

Review of the evidence base for the use of richness metrics

The environmental evidence base to support the use of richness metrics is weak. Effects demonstrated in one river type by one survey method are sometimes not reproduced by a second method, and even when a significant effect can be detected the signal to noise ratio is always poor. This is in contrast to lakes where, beyond certain thresholds, there are relatively clear and significant negative effects of increased TP and chlorophyll concentrations on macrophyte richness at a global and type-specific level (Willby *et al*., 2009). Some elements of these relationships are reproduced within rivers, with richness showing an indication of enhancement in naturally infertile sites when nutrient concentrations increase, and decreasing at much higher nutrient concentrations in naturally fertile sites. Generally the negative effects of increasing nutrient concentrations or increasing intensity of catchment land cover are demonstrated most strongly in the naturally most productive rivers, while there are some positive effects of enrichment on diversity in less fertile types. This would suggest that any use of richness metrics in classification should be weighted towards more fertile river types.

The major route for exploring links between richness and hydromorphological impacts was through linkage of River Habitats Survey and river macrophyte survey databases. This provides limited support for the value of derived indices, such as HMS and HQA. Although the significant relationships that exist can generally be accounted for ecologically, these are mostly weak and limited to a small number of river types. Consideration of primary physical environmental data gives much greater support to the use of richness metrics, revealing significant and predictable patterns across river types that can be summarised as a decrease in habitat suitability for macrophytes with increasing stream energy. Hence there are almost universally negative relationships between richness and slope (Figure 4.24), while richness is consistently positively correlated with factors such as depth and width, or a simple surrogate, such as distance downstream from source (Figure 4.25). Hydromorphological pressures which impact upon key hydraulic parameters (such as a decrease in sinuosity and increase in slope due to channelization) should translate into effects on taxa richness.

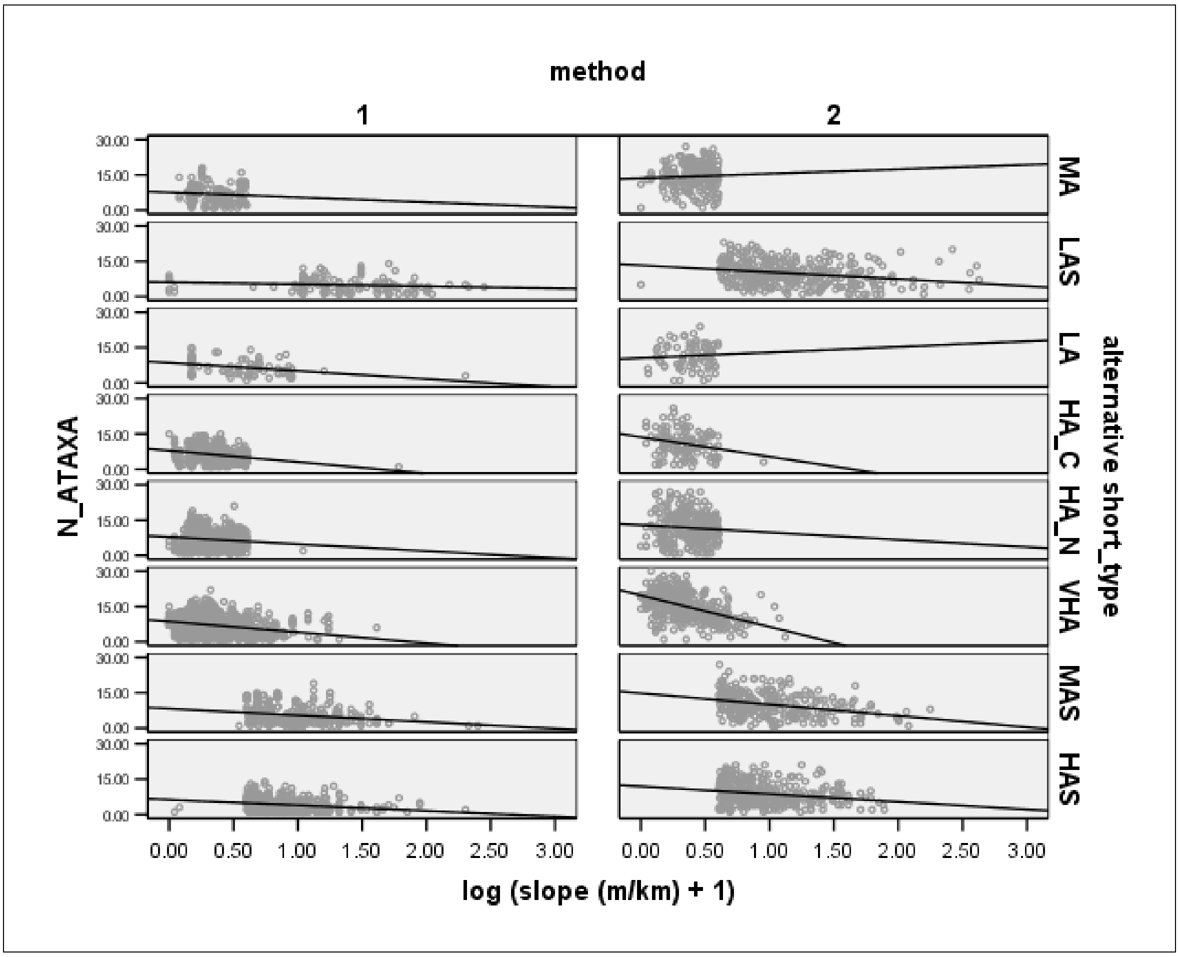


Figure 4.24 Relationship between N_TAXA and slope across a range of river types and two survey methods. All correlations significant at p = 0.01 except Method 2, types MA and LA. Method 1 = MTR; Method 2 = JNCC.

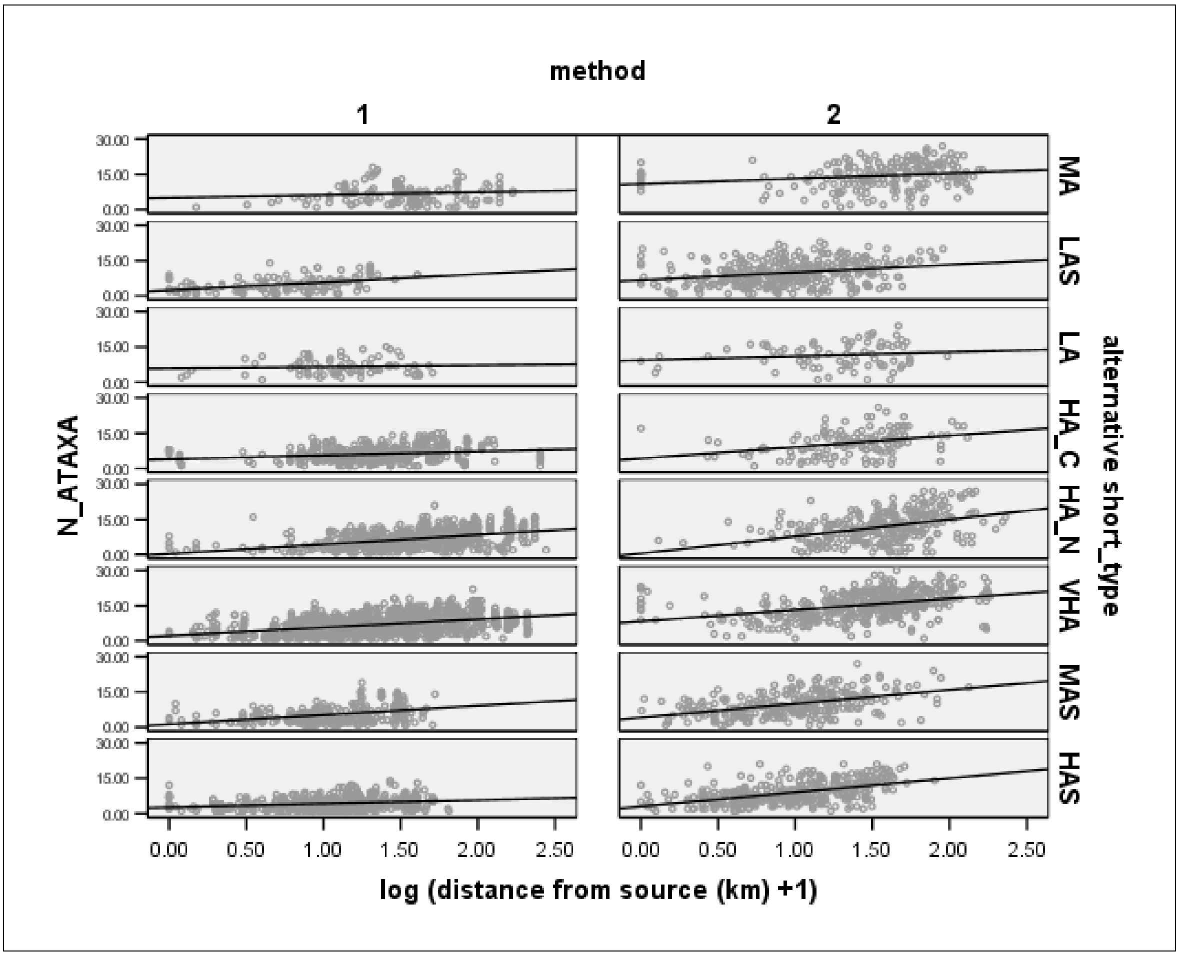


Figure 4.25 Relationship between N_ATAXA and distance from source across a range of river types and two survey methods. All correlations significant at $p = 0.01$ except type LA, Methods 1 and 2. Method 1 = MTR; Method 2 = JNCC.

Additional justifications for the use of richness metrics

Additional arguments support the use of richness metrics in classification, but these cannot be made empirically. In general, other things being equal, increased richness is likely to reflect a greater diversity of niches within a sample reach, a higher quality of upstream species pool, or a greater level of connectivity with upstream and lateral sources of propogules. All of these are likely to be features of relatively unimpacted river ecosystems with a natural flow regime. Positive correlations between richness and variables such as channel width or distance from source support the idea that lowland, low-gradient rivers offer a more hydraulically benign environment for plant growth. However, they also incorporate the effects of gathering propagules from a progressively larger upstream species pool or exchanging propagules with floodplain sources and therefore integrate some effects of connectivity on richness.

An alternative, biological approach to developing an evidence base to support the use of richness metrics in macrophyte classification involves consideration of the relationships between macrophytes and other, potentially dependent, quality elements. Figure 4.26 considers the relationship between invertebrate richness (based on the EQR of the RIVPACS metric N_TAXA) and the number of aquatic plant taxa in a reach. In 12 out of the 16 method x river type combinations, there is a highly significant positive correlation between invertebrate N_TAXA EQR and macrophyte N_ATAXA. In the remaining cases the lack of correlation is probably best explained by the availability of data or truncated nature of one or both richness gradients. The strength of these correlations greatly outweigh any of the correlations between macrophyte diversity and nutrients. The pattern is repeated if N_FG is substituted for N_ATAXA, or if invertebrate ASPT EQR is used instead of N_TAXA, but in all cases the relationships are weaker than those between N_TAXA EQR and N_ATAXA.

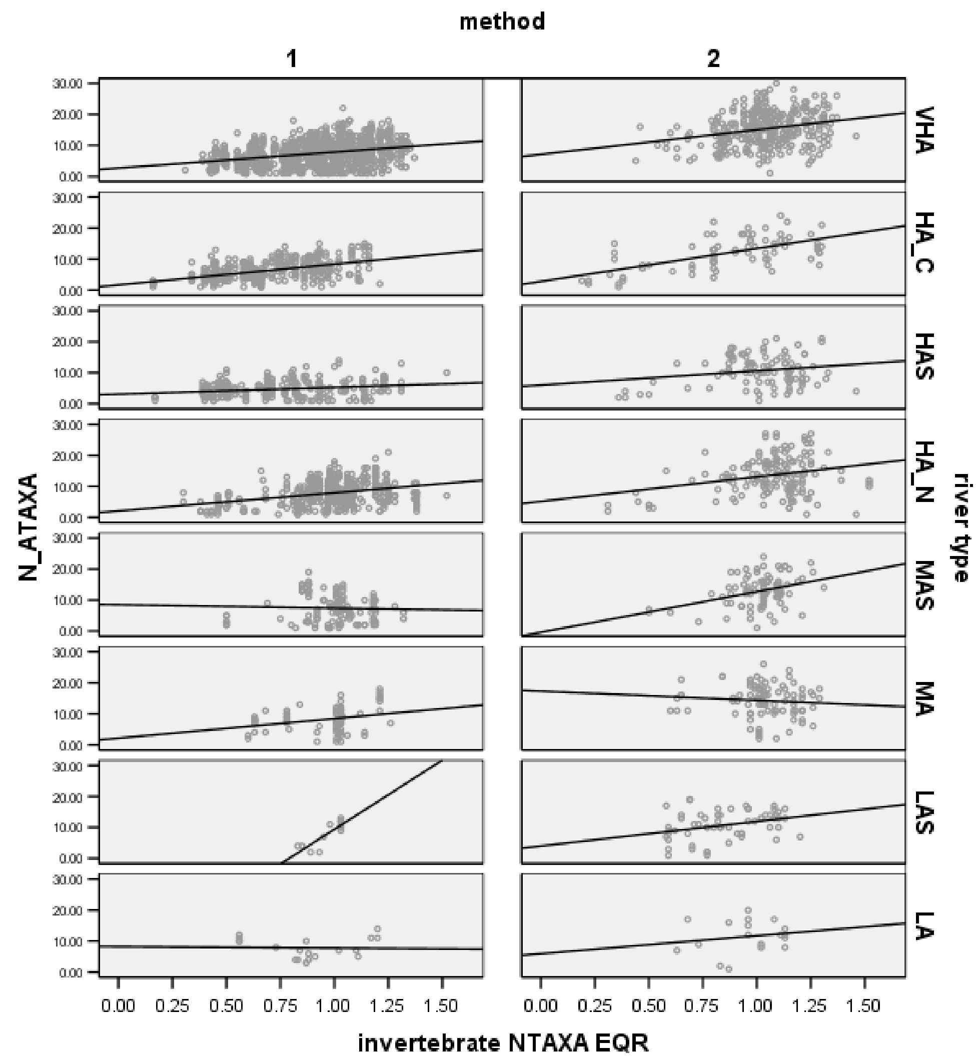


Figure 4.26 Relationship between number of aquatic plant taxa and RIVPACS invertebrate N_TAXA EQR. All relationships are significant at p = <0.001 except Type LA (Methods 1 and 2); MA (Methods 1 and 2) and MAS (Methods 1 and 2). Method 1 = MTR; Method 2 = JNCC. River types created by aggregation of some poorly populated types. VHA = very high-alkalinity (all slope categories); HA_C and HA_N = high-alkalinity rivers with slope under 3 m/km of southern (C) or northern (N) types; MAS = moderate alkalinity steep rivers; MA = moderate alkalinity rivers with slope under 3 m/km; LAS = low-alkalinity steep rivers; LA = low-alkalinity rivers.

The simplest explanation for these relationships is that both macrophytes and macroinvertebrates respond positively to greater physical habitat heterogeneity. Benthic organisms are highly sensitive to factors such as near bed velocity and substrate size, stability and permeability (Giller & Malmqvist, 1998). Consequently the fact that macrophytes mirror the response of the N_TAXA EQR offers basic support for the claim that macrophyte diversity reflects physical habitat diversity. Since macrophytes *per se* represent a major component of physical habitat diversity in rivers, and the diversity of growth forms should reflect the diversity of plant associated habitats, it is also reasonable to expect that dependent organisms, such as macroinvertebrates, might respond positively to macrophyte diversity. This seems particularly likely in higher alkalinity rivers where macrophyte cover is sufficiently high to be a major agent of physical habitat diversity in its own right. Evidently, it is in such rivers that the link between macrophyte and macroinvertebrate richness is strongest. A reduction in macrophyte diversity could thus be regarded as a form of undesirable disturbance if it resulted in deterioration in the ecological quality of a site as represented by invertebrates. Macrophytes may also benefit directly from a more diverse invertebrate fauna if this results in more grazing of attached algae and therefore less shading.

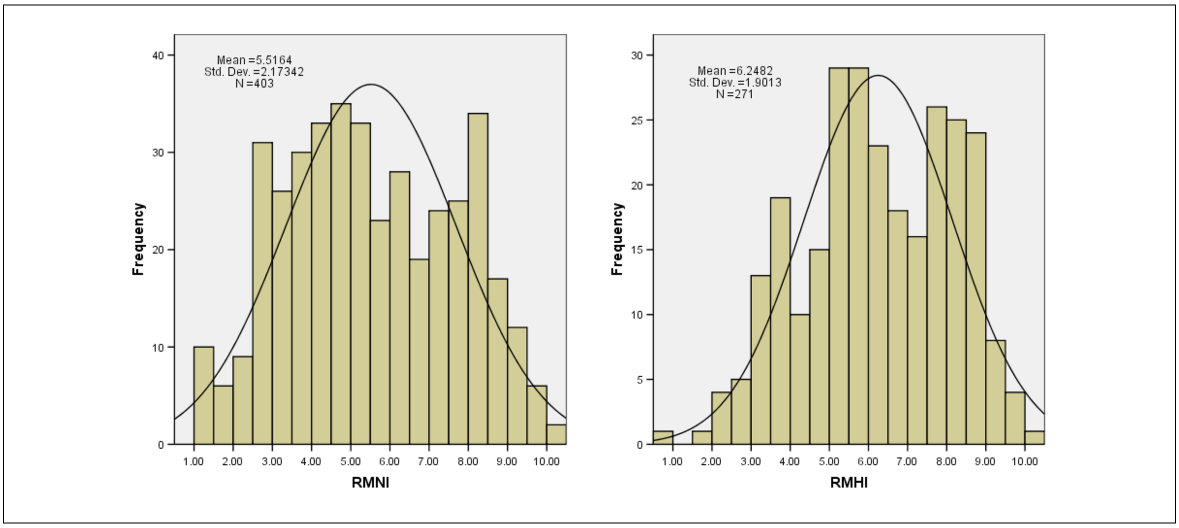


Figure 4.27 Frequency distribution of species scores for the metrics RMNI and RMHI

Finally, there are supporting statistical arguments for the use of richness metrics. Firstly, due to the distribution of taxa scores there is an inevitable positive correlation between richness and RMNI or RMHI. These correlations are consistently stronger among surveys based on the JNCC method than the MTR method, reflecting the greater survey length and correspondingly larger number of species recorded in JNCC surveys. Correlations are stronger with RMHI than RMNI and are similar to or weaker than correlations between the Lake Macrophyte Nutrient Index (LMNI) and macrophyte richness in lakes. This is a reflection of the frequency distribution of taxa scores. Hence, Figure 4.27 shows that for RMNI the frequency of taxa with scores in the range 2.5-8.5 is very even, while for RMHI the distribution of scores is more strongly unimodal, with only scores in the range 5.0-9.0 showing a relatively even distribution. In the case of the lake macrophyte metric, LMNI, the range of scores with similar frequencies of species is restricted to values from 5.0-7.5, with a pronounced tail of much rarer low-scoring species.

Secondly, use of a richness metric introduces an element of quality control and avoids overreliance on compositional metrics generated from potentially species-poor survey data. Previously, the MTR system incorporated a suffix of confidence, based partly on the number of taxa recorded, to indicate the reliability of the survey data for detecting impacts of nutrient enrichment. In our approach, richness is considered independently but can qualify a classification based only on composition. Thirdly, and following from the above, there is an ecological argument to discriminate between ecological status based only on compositional metrics when the number of taxa in a survey can vary widely without changing the compositional metric value. Thus, it is apparent from Figure 4.28, that while the dependency of a compositional metric on richness is generally quite low, the number of taxa associated with any given compositional metric value may vary by more than an order of magnitude. This level of variability in taxa richness may be a particular feature of rivers, of macrophyte surveys and of the conditions under which they are sometimes conducted. Standard sampling of invertebrates or diatoms, for example, would generally yield more species-rich and less variable numbers of taxa per sample.

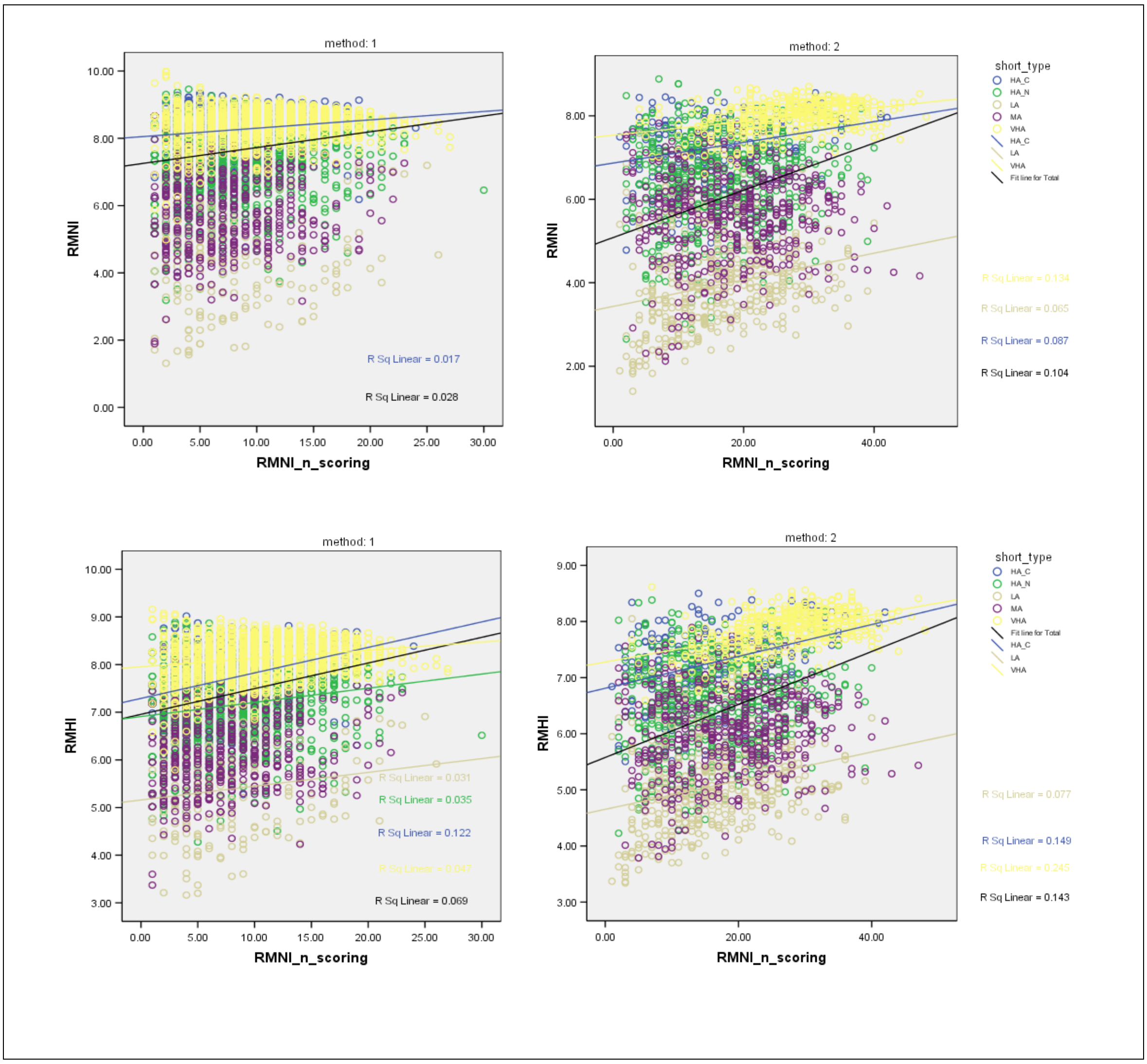


Figure 4.28 Relationships between RMNI and RMHI site scores and the number of scoring taxa based on data collected by two survey methods (1 = MTR; 2 = JNCC). Note the greater dependency of RMHI on the number of taxa, and the greater dependency of both metrics on the number of taxa when using JNCC data.

4.4.4 Comparative value of different richness metrics

The metric N_FG was conceived as a potential solution to the problem that arises when a reach supports several critical taxa whose taxonomic identity cannot be determined. This is especially likely in the case of groups such as *Ranunculus* and *Callitriche*. If the richness of the community was expressed in terms of the numbers of functional groups, rather than a higher taxonomic level, any bias against critical taxa would be reduced. Moreover, functional groups, discriminated on the basis of morphological and regenerative traits might provide a more transparent link to the diversity of physical habitat types. A comparison of the relationship between N_FG and N_TAXA in both survey methods (Figure 4.29) provides some support for the use of complementary richness metrics. Thus, for example, when the number of taxa recorded in a JNCC survey is 10, the number of functional groups may range as widely as three to 10, whereas if five functional groups are recorded the number of taxa may range between five and 15. In general, however, functional redundancy (multiple taxa per FG) is evidently much less of a feature of river than it is of lake vegetation. Even in the much longer JNCC survey reaches the mean number of taxa per FG only increases from about 1.5 to two. In both sets of survey data, there is no evidence of functional group saturation at high N_TAXA as occurs within lake vegetation.

There is sufficient variability in the relationship between N_TAXA and N_FG to suggest that both have unique value. Generally however, N_FG exhibits less significant relationships with a range of environmental variables than does N_TAXA and the evidence base for its use is not as strong.

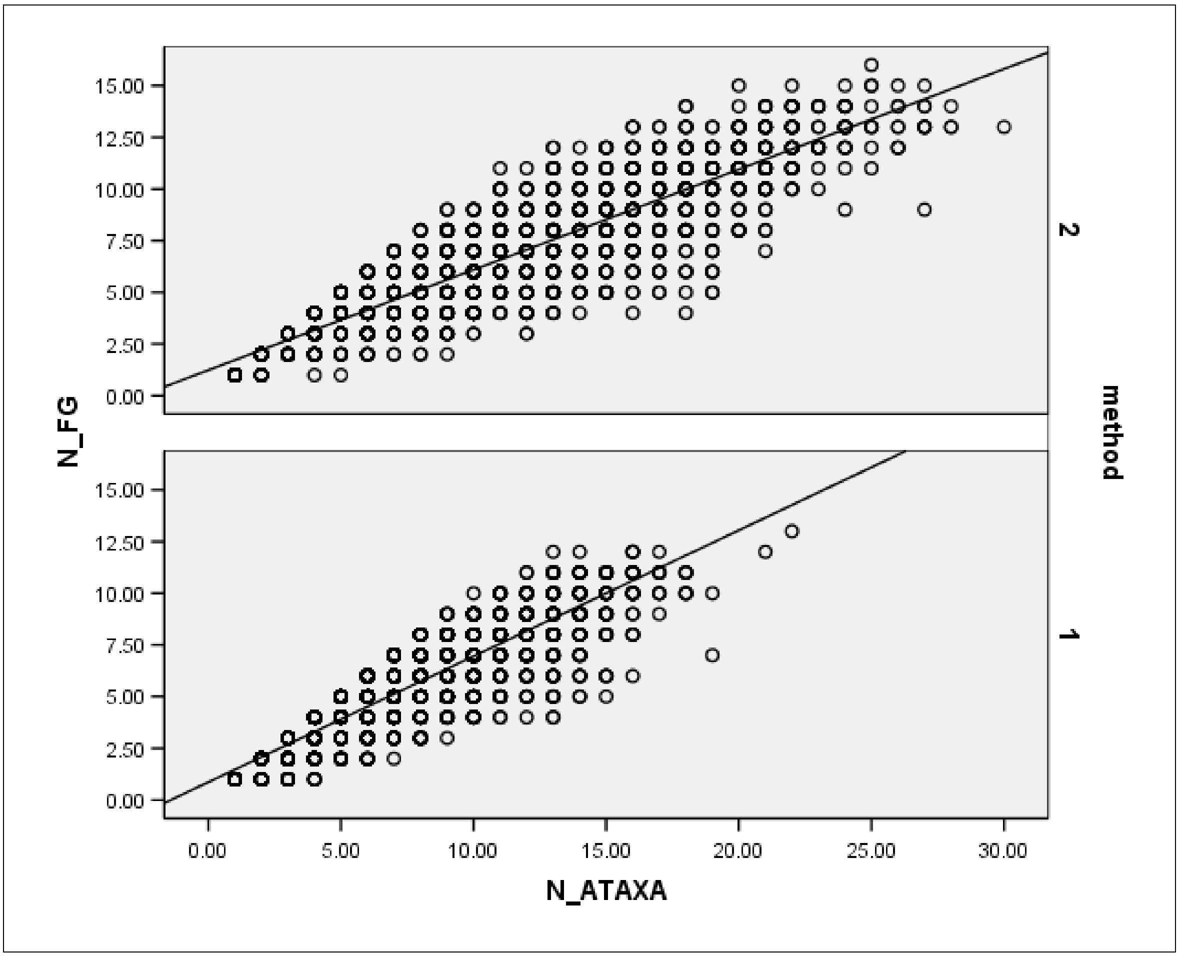


Figure 4.29 Relationship between N_TAXA and N_FG according to data collected by two different survey methods; 1: MTR (r^2 = 0.83) and 2: JNCC (r^2 = 0.79).

4.4.5 Calculating the metrics

The richness metric calculated in the LEAFPACS project is deliberately restricted to a count of the numbers of strictly aquatic taxa or functional groups per standard reach (as indicated in Table 4.1), rather than covering all taxa that potentially contribute to an RMNI or RMHI score. There are two reasons for this:

i. It is possible to generate an RMNI or RMHI score for a site based only on emergent vegetation (aquatic species are completely absent from the channel). This scenario is identified by restricting the richness metric to more or less obligately aquatic species.

ii. Richness is likely to depend on surveyor experience, survey effort and how the limits of the recording zone are interpreted. It could, for example, be enhanced considerably by recording some emergent vascular species that feature on the fixed list but technically fall outside the recording zone, or, if disproportionate survey effort is put into the detection of scoring bryophytes taxa that are growing outwith the strict low flow channel. By restricting the richness metric to aquatic taxa, the influence of additional observer-related sources of variability should be reduced.

The list given in Table 4.1 is treated as a dictionary; no distinction is made between this list and aquatic species growing as emergents, or non-aquatic species growing submerged at the time of survey, since this is likely to depend on ambient flow conditions and survey timing relative to the growing season. Therefore taxa are treated as aquatic if they commonly occur across a range of sites predominantly as floating-leaved or submerged plants, even though some (such as *Berula erecta*) have the ability to persist through the growing season at some sites in an emergent form.

The determination of number of functional groups simply relies on a count of the number of groups populated by at least one taxon.

4.5 Abundance metrics

4.5.1 Background

In contrast to lakes, abundance indices are designed to recognise *high* cover on a scale that might represent undesirable disturbance within a water body. Low macrophyte cover in lakes is frequently indicative of a pressure (such as high nutrient loading and shading by phytoplankton, or loss of littoral habitat due to artificially elevated water level fluctuations) and is likely to have a range of ecosystem effects, for example through the loss of shelter for algal grazers or reduction in sediment stability. However, in rivers, most instances of low macrophyte cover can be explained by natural causes (such as extreme flow variability, heavy tree shading, high antecedent flows, very coarse or unstable bed load, grazing by water birds), whereas high cover may signify a range of pressures, including increased nutrient loading, sedimentation, reduced flow variability or water abstraction, or reduction in stream power associated with decrease in slope caused by weirs or impoundments.

Until recently, total areal coverage has not been routinely assessed as part of macrophyte surveys. However, a variety of measures related to abundance can easily be derived from normal survey data. These can include:

i. Total or absolute cover of all taxa or a subgroup of taxa (such as filamentous algae, or invasive non-native species) based on summation of cover scores, previously back transformed into percentages. As a result of multilayering of plants within the water column, this can generate values exceeding 100 percent. However, this may be advantageous if it conveys extra information about the vertical development of vegetation within the water column. Such a value is more analogous to the PVI system (percent volume infested or inhabited) used routinely in assessments of abundance of lake vegetation, although direct summation of values produces a value somewhat dependent on the total number of taxa recorded.

ii. Mean cover measured across all taxa or a subgroup of taxa. This requires back transformation of cover values to percentages followed by averaging of these values. It eliminates dependency of a total cover value on the number of taxa recorded and may therefore be useful for comparing between sites or methods. This approach was used to generate an abundance-based metric for lake macrophytes, the rationale being that abnormally high or low mean cover per taxa is a distortion of the typically log-normal abundance frequency distribution.

iii. Relative cover of individual taxa or subgroups of taxa, based on summation of transformed cover scores, and expressed as a proportion of the summed transformed cover scores of all species. Relative cover values may be considered useful if the total extent of vegetation varies greatly between sites or types, for example due to differences in the availability of suitable habitat. However, they require quantification of the overall vegetation, including components in which one may not be interested, and are potentially misleading. Consider, for example, a river with extensive macrophytic vegetation extending over 50 percent of a reach, in which there is 10 percent cover of green filamentous algae (a relative cover of 20 percent). A second river with only five percent total plant cover has two percent cover of filamentous algae (a relative cover of 40 percent). Although this represents only one-fifth of the absolute cover of filamentous algae at the first site, it still constitutes twice the relative cover.

4.5.2 Approach

All data was first converted to percent cover values. In the case of the nine-point cover scoring system used in the MTR method, values were recoded to the mid-point of the range of cover values associated with each point on the scale (for example, for a cover score of five, which equates to a percent cover range of five to ten, a value of 7.5 per cent was used). A more sophisticated approach was required to identify suitable percent cover values in data acquired using the JNCC method, in which taxa are assigned a score from one to three. Surveys conducted using the MTR method were assessed separately to examine the total distribution of cover scores in all surveys across the nine possible classes. This confirmed an approximately exponential decline in the frequency distribution of cover scores, with the cover score of one (under 0.1 per cent) attracting 45 per cent of all records. The under representation of the lowest cover values noted by Willby *et al*. (2009) in the case of lake macrophyte surveys clearly does not extend to rivers. The scores used in the MTR method were then partitioned into ranges appropriate to the scoring system used in the JNCC method (Table 2.2). A cover score of one in the JNCC method was assumed to equate to a cover of one in the MTR method. For a JNCC score of two (covering scores of two, three and four in the MTR method), weighted averaging applied to the mid-point of the cover scores indicated that a cover of 1.4 percent would be appropriate. Since this fell within the range of cover values for an MTR cover score of three, all JNCC cover values of two were transformed to a percent cover of 1.7 to match the value applied to the MTR data.

For JNCC cover values of three (corresponding to an MTR range of five to nine) weighted averaging indicated that a cover of 24.9 percent would be appropriate. Since this fell at the border of two MTR cover classes (six and seven), and was also felt intuitively to be rather high, the data was reassessed using surveys distributed across the full range of river types, rather than the predominantly lowland rivers in which most

MTR surveys had been conducted. This exercise confirmed the suitability of the score for JNCC Class 2 but indicated that a percent cover of 18.6 was more appropriate for JNCC Class 3. Since this fell squarely into MTR Class 6 (10-25 per cent) a value of 17.5 per cent was used for transformation purposes, to be consistent with the approach for transforming MTR cover values to percent. The match between JNCC scores of two and three and MTR scores of three and six respectively is consistent with an independent harmonisation of the two scoring systems (N. Holmes, personal communication).

A series of different abundance metrics were calculated based on summation of harmonised cover scores transformed into percent cover, as described in section 2.4.1. The following metrics were considered:

- mean percentage cover per taxa;
- total cover of all hydrophyte taxa;
- absolute cover of green filamentous algae;
- cover of green filamentous algae relative to total hydrophyte cover.

Each metric was assessed in terms of the strength of its relationship with a suite of potential pressure indicators (mean SRP, TON, NH_4-N and percentage intensive upstream land cover). The datasets used here were large compared to most studies and there was no opportunity to control (by measurement or choice of sites) for the wide range of factors that contribute to variation in plant cover within streams (shading, substrate availability, flow variability, accrual period, water colour, inter observer variability, and so on). Consequently, a high level of noise was to be expected and the occurrence of a statistically significant signal and large F ratio was rated above the actual amount of variation explained when judging the utility of different metrics.

4.5.3 Validation and comparative value of basic measures of abundance

An initial assessment of the relationship between total and mean cover per taxa across the river typology using the two survey methods highlights some basic patterns but also reveals important differences between survey methods. Relationships between total cover or mean cover per species and different environmental measures were consistently much weaker using data obtained from JNCC surveys. This is almost certainly due to the considerable uncertainty attached to the interpretation of JNCC covers scores of three, which could technically translate to anything within the range five to 100 per cent. The mean cover per taxa is much lower with data from JNCC surveys which is partly a function of the percent interpretation of the highest cover score, but probably also reflects the tendency to record larger numbers of species with very low cover scores within such surveys.

Of the abundance metrics considered, mean cover per hydrophyte taxa was more strongly correlated with all the pressure indicators than total hydrophyte cover. An alternative mean cover value calculated using number of functional groups rather than number of aquatic taxa to derive the mean was expected to improve the strength of relationships with pressure variables but this proved not the case, with all global correlations proving slightly weaker than mean cover based on the number of taxa.

Figures 4.30 and 4.31 consider the relationships between mean cover per taxa or total hydrophyte cover, and a range of pressure indicators. In all cases there is a significant global relationship between the abundance metric and the pressure indicator but the variation explained is small (typically one to five per cent) and the relationship appears to consist of two phases. In the first phase, mean or total cover decreases from five or 25-30 per cent respectively to one to two and 10-15 per cent respectively until values of around100 ug/l SRP, 2-3 mg/l TON and 100ug/l NH_4-N. This pattern is shown by the moderate and lower alkalinity river types. The relationships are consistently negative across these types although the significance of these relationships is often marginal. Thereafter, in the second phase, mean or total cover increases from one to two and five to 15 per cent respectively up to 8-10 and 40-50 per cent respectively. This pattern is shown by most or all of the higher alkalinity river types.

The second phase in these relationships can be explained relatively easily in terms of stimulation of macrophyte growth by increased nutrient supply, or perhaps more rapid replacement of biomass lost during high flows. It is generally accepted that macrophyte cover and biomass will respond positively to increased nutrient supply, other factors being equal (Carr and Chambers, 1998), and Hilton *et al*. (2006) used this principle in developing their model of vegetation response in rivers to changing nutrient load. What is less clear is why macrophyte cover should decrease in less productive rivers as nutrient supply increases, since increased nutrient supply should stimulate production. Possible explanations are that in streams not hydraulically favourable for macrophyte growth, increased phytobenthic production due to nutrient enrichment precludes growth of bryophytes which normally dominate suitable microhabitats. Alternatively, higher stream nutrient concentrations might promote growth of tall riparian herbs and trees leading to greater shading of the channel. There may be secondary such as greater mineral turbidity and sedimentation, or reduced particle size and increased bed mobility, which may have disproportionately adverse effects on attached bryophytes. Such changes in substrate may, at the same time, be too local, small scale, or short-lived to support significant growth of rooted vascular plants.

One downside of focusing on excessive macrophyte growth is that low macrophyte cover in less productive rivers, which appears to be related to increasing enrichment, will not be registered, unless it is covariable with another metric such as taxa richness. Unfortunately, there are probably numerous natural causes of low macrophyte cover in less productive rivers (such as shading, low bed stability, extensive bed rock exposures, strongly elevated antecedent flows), whereas it is more difficult to account for excessive growth of vegetation in more productive rivers other than by invoking anthropogenic influences.

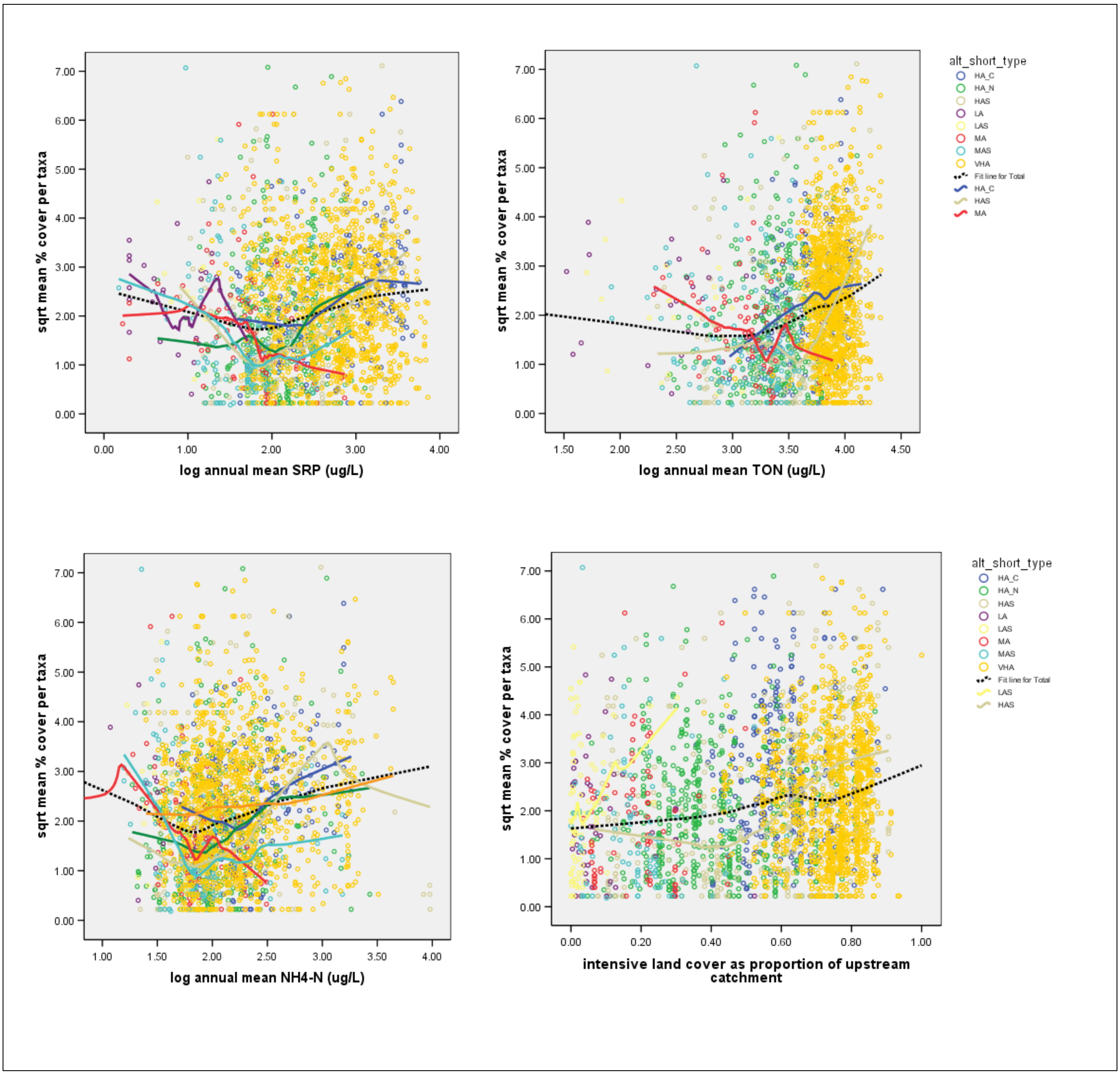


Figure 4.30 Relationships between mean percentage cover per taxa and a selection of pressure indicators. Plotted are the global relationship plus type-specific relationships (as LOWESS) where linear regression significant at p = 0.05. Note two phase responses for SRP and NH_4-N with mean cover increasing above values of 100-150 ug/l.

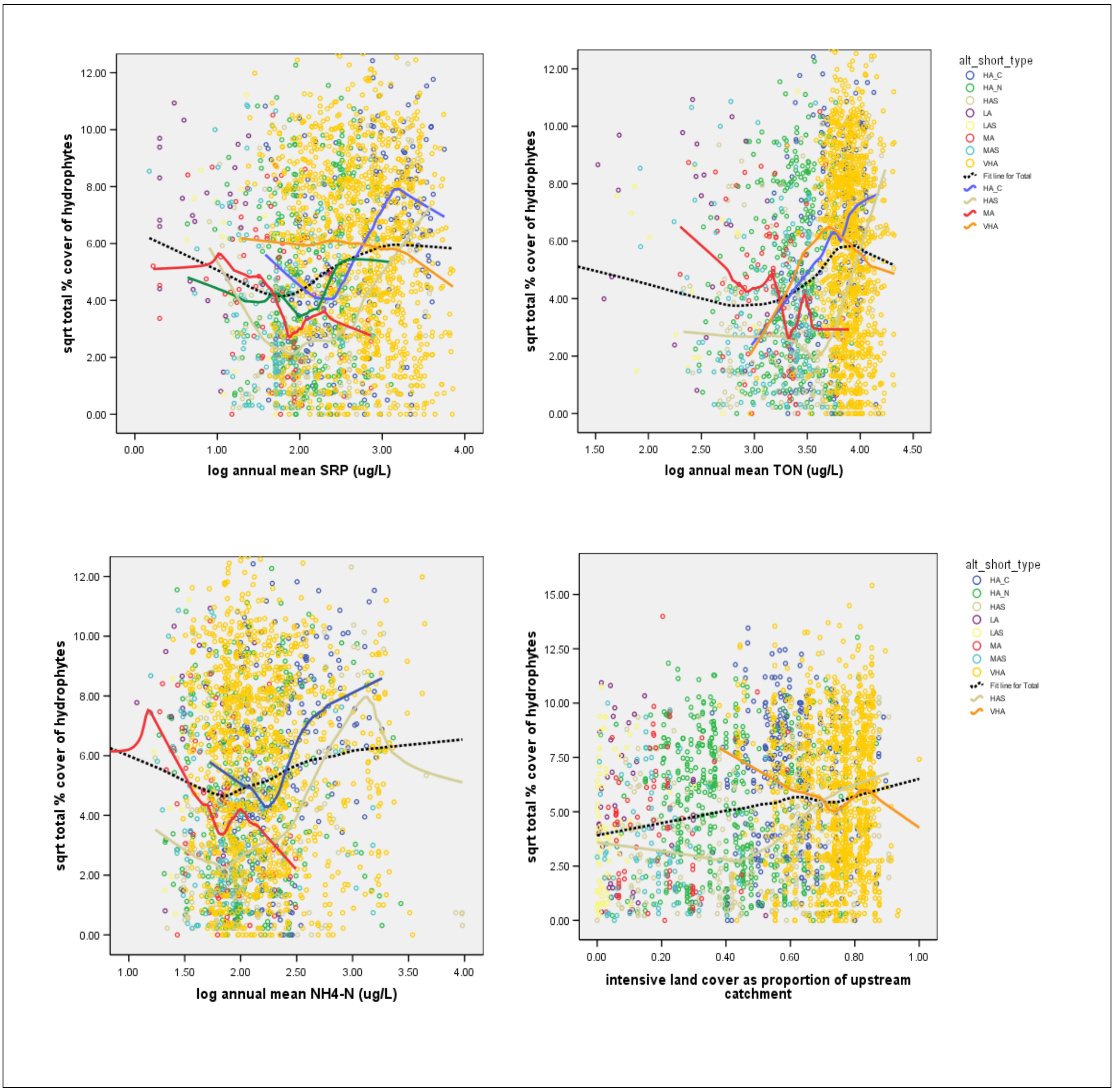


Figure 4.31 Relationships between total hydrophyte cover and a selection of pressure indicators. Plotted are the global relationship plus type-specific relationships (as LOWESS) where linear regression significant at p = 0.05. Note two phase responses for SRP and NH_4-N with mean cover increasing above values of 100-150 ug/l and cover increasing in higher alkalinity rivers when TON exceeds five mg/l.

4.5.4 Algal cover as an abundance metric

Rather than measuring abundance across all aquatic species, it may be preferable to consider the abundance of subgroups of taxa. Increased cover of benthic or epiphytic filamentous algae is widely regarded as being indicative of nutrient enrichment (Hynes, 1960; Dodds, 2006) and drives a number of undesirable disturbances associated with eutrophication (ECOSTAT, 2005). This interpretation is supported by empirical evidence of measurements of algal standing crop or chlorophyll in relation to nutrient availability, whether enhanced by anthropogenic activity (Biggs, 2000), or experimentally, such as through the use of nutrient diffusing substrata (Bowes *et al*., 2007).

Absolute and relative algal cover was determined as described in Section 4.2.2. Algal cover refers here solely to green filamentous algae and covers all species, taxa or aggregates that fall under this term (*Cladophora* spp, *Cladophora glomerata*, *Enteromorpha*, filamentous green algae, *Hydrodyction*, *Spirogyra*, *Stigeoclonium*, *Vaucheria*). Specific records of *Cladophora aegagrophila* were excluded from the total since this species commonly forms an extensive short benthic turf in streams with low to moderate nutrient concentrations and on the basis of both its RMNI score (5.56) and low SRP optima relative to other algal taxa (0.16 mg/l), was considered unlikely to contribute to a general algal response to nutrient enrichment. Other non-green filamentous algae, such as *Lemnaea* spp, were also excluded. The advantage of this approach is that all green filamentous algae can be considered, irrespective of how they have been documented by different surveyors. Thus, it would appear that some surveyors have used the term 'filamentous green algae' specifically for unidentified taxa that do not fall within the other commonly recognised groups, whilst others have used this terms as a catch-all to cover any green filamentous algae. It is also highly likely from the data assembled that the taxon *Cladophora* spp has been (ab)used to incorporate large filamentous types belonging to other genera.

MTR data shows that absolute filamentous algal cover displays a clear global signal in relation to nutrient concentration and upstream land use (Figure 4.32). Thus, for total filamentous algal cover, the global relationship indicates a rapid increase from a baseline of one to two per cent cover up to 10 per cent cover once phosphate and ammonia concentrations exceed 100-150 ug/l and TON exceeds 1-2 mg/l. At a type-specific level relationships only tend to be significant for the naturally more fertile high and very high-alkalinity types and confirm the placement of the aforementioned thresholds. In general, thresholds appear to lie somewhat higher for high-alkalinity rivers on soft geologies than those with similar alkalinity but draining sandstone or hard limestone catchments. In all cases relationships are noisy and the measured environmental data only explains three to ten per cent of the variation in significant relationships. Relationships for relative algal cover (Figure 4.33) are somewhat weaker, although still significant, and show less evidence of threshold effects. However, there is some indication that relative algal cover in high-alkalinity streams increases strongly above 100 ug/l SRP, five mg/l TON and 100 ug/l NH_4-N. Increases in relative algal cover from a baseline below 20 per cent up to 50 per cent tend to be indicative of increased enrichment.

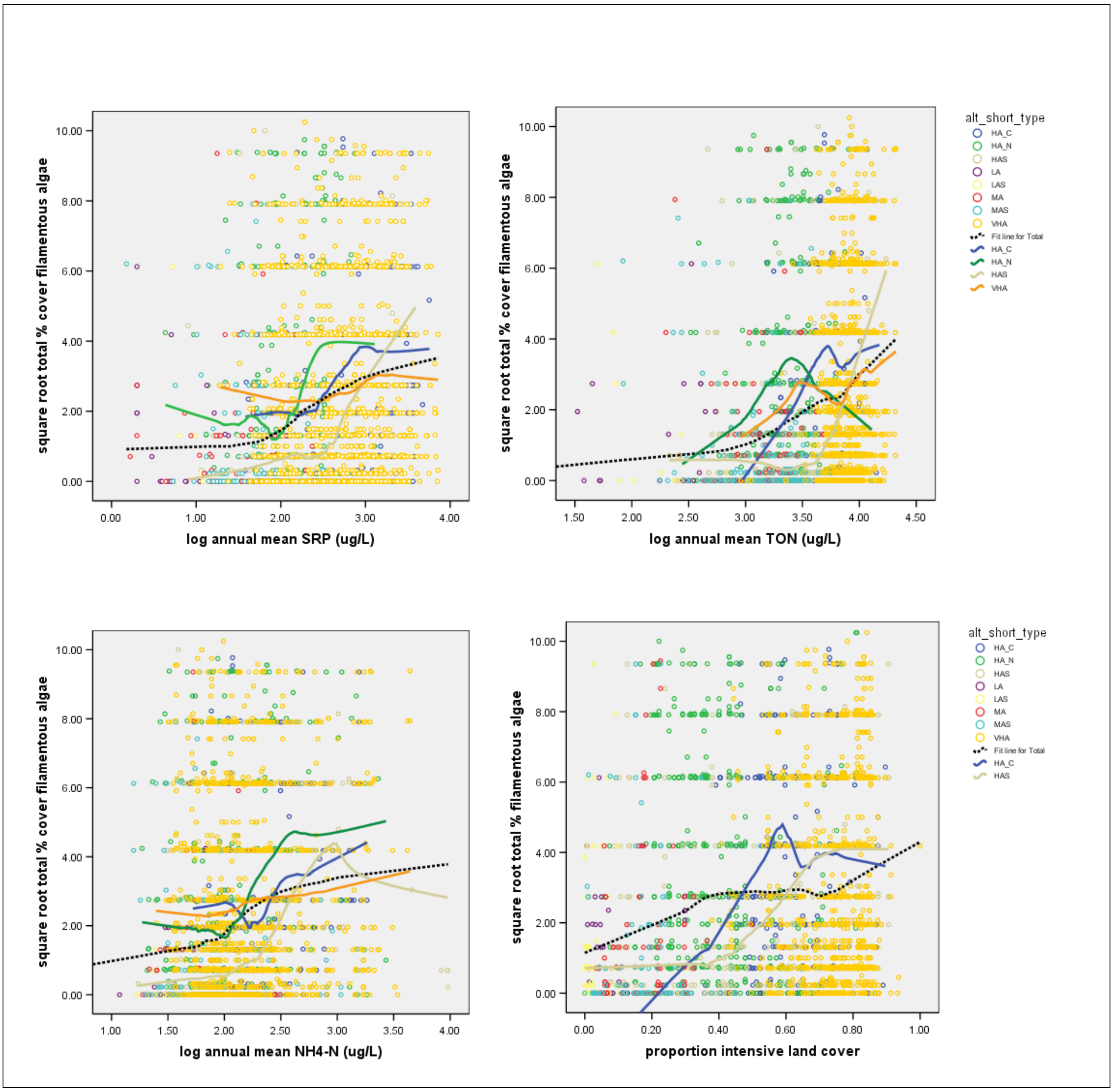


Figure 4.32 Relationships between total cover of green filamentous algae and a selection of pressure indicators. Plotted are the global relationship plus type-specific relationships (as LOWESS) where linear regression significant at p = 0.05. Note general threshold responses at 100-150 ug/l SRP, 1-2 mg/l TON, 100-150 ug/l NH_4-N, and intensive land cover of around 40 per cent.

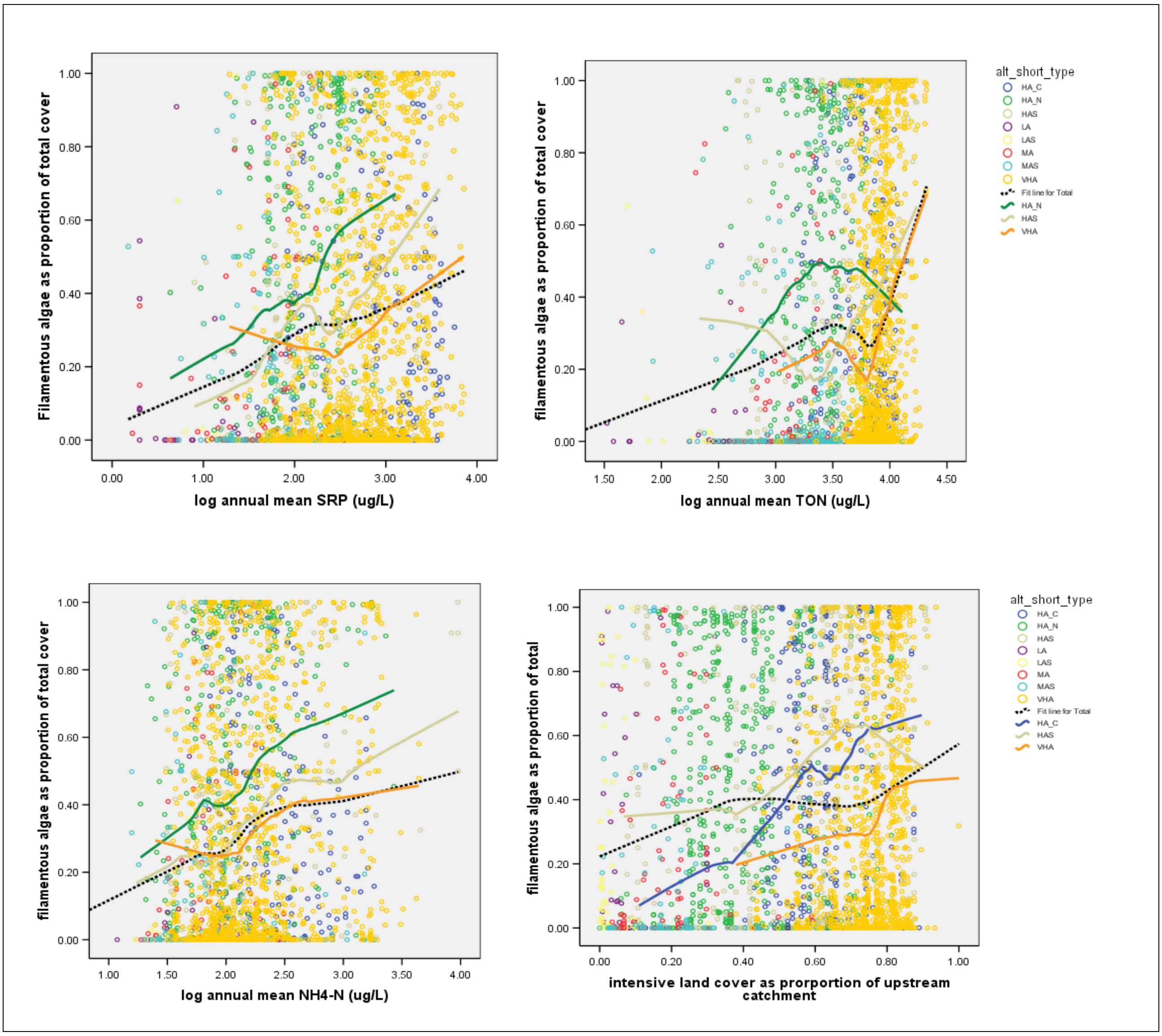


Figure 4.33 Relationships between relative cover of green filamentous algae and a range of pressure indicators

4.5.5 An abundance metric for invasive species

The lake classification tool uses an abundance metric based on the relative cover of aquatic non-native invasive species at a site. The range of invasive species found in standing water sites is quite large and includes several that do not occur in rivers (such as *Crassula helmsii*, *Myriophyllum aquaticum* and *Lagarosiphon major*) or are much rarer in flowing than in standing waters (such as *Azolla filiculoides*). In the available data for rivers, *Elodea* species account for 98 per cent of the total cover of non-native invasive species. One new addition, *Hydrocotyle ranunculoides*, colonised Britain in the early 1990s and is currently confined mainly to slow-flowing rivers and drains in south east Britain. This species was absent from the river macrophyte survey database that was used to develop the classification tool here. The overall extent and relative cover of invasive hydrophytes is also greater in standing waters (global mean absolute cover of 0.9 and 3.0 per cent in rivers and lakes respectively; global mean relative cover of 2.2 and 3.5 per cent in rivers and lakes respectively). This suggests that rivers are less readily invaded, perhaps because hydraulic forces result in a generally less favourable habitat and because excessive growth of invasive species is usually curbed by the effects of flow variability. Among the options to deal with invasive species in rivers would be to have a dedicated metric or to adopt a rule-based approach that deals independently with a range of invasive species that belong to different quality elements. Thus, UKTAG (2004) have proposed an approach to deal with invasive species that prevents a water body being classified at good ecological status if high risk invasive species are present as established populations.

Although invasive species are a pressure in their own right, it is nevertheless useful to understand whether there is an underlying environmental basis to their distribution and cover. Data show that a high absolute cover of invasive hydrophytes only occurs in rivers with high nutrient concentrations or with catchments dominated by intensive agriculture. Generally the relationships with pressures are much weaker than those observed with the other abundance metrics and significant relationships are confined to naturally fertile rivers. This is largely because, even under the most impacted conditions, the cover of invasive hydrophytes in rivers can range from zero to more than 50 per cent. Whether these relationships are causal is unclear; it may simply be that lower energy rivers, which are intrinsically more fertile, are more amenable to colonisation or have lower rates of extinction.

Absolute cover of invasive species is not recommended as a metric to assess riverine vegetation, since high values only occur when high values of a superior metric such as total hydrophyte cover also occur, thus rendering the invasive metric redundant. In theory, the absolute cover could be substituted for relative cover of invasive species, which is more independent of the other abundance metrics tested. However, relative cover of invasive species was found to be related even less strongly to pressure indicators than absolute cover. It is also, arguably, misleading measure for streams since small cover of an invasive species in a site with small total cover of aquatic vegetation might constitute a rather high relative cover, yet it might be difficult to justify treating a small population of an invasive species as established and potentially damaging.

A metric based on the cover of invasive non-native species is not recommended for the classification of rivers based on macrophytes. A rule-based approach should be implemented to deal with those rare instances in which demonstrably high risk species, such as *Hydrocotyle ranunculoides*, occur.

4.5.6 Selecting an appropriate abundance metric

The approach used to identify the best abundance metrics for classification purposes was based on joint consideration of the strength of relationship between each candidate metric and various pressure indicators (as described above) and the extent of intercorrelation between the metrics. Table 4.5 summarises the linear correlations between transformed abundance metrics and the various pressure indicators. Of all the metrics, mean cover per taxa displays the strongest overall global correlation across the range of pressure indicators while the cover of filamentous green algae is the next best independent metric. Across all abundance metrics, phosphate and nitrate tend to be more influential than ammonia, and urban land use is more important than agricultural land cover. In the case of general macrophyte abundance metrics, nitrate and agricultural land cover are more important whereas for algal metrics, phosphate and urban land cover are more significant. Point sources of enrichment may have a stronger bearing on algal cover, while diffuse agricultural sources of nutrients may be more important in terms of overall cover of all hydrophyte taxa.

Table 4.5 Non-parametric correlations between abundance metrics and pressure indicators

Variable		Algal cover	Relative algal cover	Mean cover/ taxa	Total hydrophyte cover	Mean cover/ FG	Invasive spp cover
SRP	Coefficient	0.242	0.190	0.168	0.180	0.141	0.110
	Prob	***	***	***	***	***	***
	N	2297	2261	2261	2297	2261	2297
TON	Coefficient	0.232	0.180	0.201	0.189	0.169	0.077
	Prob	***	***	***	***	***	***
	N	2294	2258	2258	2294	2258	2294
NH_4-N	Coefficient	0.197	0.190	0.156	0.112	0.124	0.041
	Prob	***	***	***	***	***	*
	N	2300	2264	2264	2300	2264	2300
% agriculture	Coefficient	0.159	0.121	0.170	0.135	0.149	0.038
	Prob	***	***	***	***	***	ns
	N	2679	2653	2653	2679	2653	2679
% urban	Coefficient	0.280	0.260	0.206	0.169	0.191	0.075
	Prob	***	***	***	***	***	***
	N	2679	2653	2653	2679	2653	2679

Correlations based on Spearmans rank test. *** significant at p = 0.001; * p = 0.05.

Table 4.6 shows that some metrics are strongly intercorrelated and there would thus be little value in retaining two or more such metrics. Based on the overall strength of their relationship with nutrients and land cover and their intermediate pair-wise correlation, absolute cover of green filamentous algae and mean cover per taxa of hydrophytes would appear to be the two most useful abundance metrics. These two metrics exhibit somewhat different profiles in terms of correlations with pressure indicators. They also address subtly different aspects of abundance, one related to the distribution of cover and dominance, the second to the absolute cover of a group of taxa that appear especially responsive to nutrient enrichment. Neither compositional nor richness metrics directly capture these aspects of the aquatic vegetation.

All the abundance metrics assembled here were derived from taxa level cover values rather than measured independently. Thus, a measure of mean hydrophyte cover, for example, had to be obtained from summation of the cover values of individual taxa. However, in the future macrophyte abundance metrics could be measured as part of an abbreviated assessment of macrophytes and phytobenthos without the need for a full macrophyte survey on each occasion or at every sample site. To achieve this, it would be necessary to calibrate single measures of total hydrophyte and filamentous algal cover against measures based on summation of individual taxa. In the case of algae, the difference is probably minor as the number of green filamentous algal genera reported at most sites is small and compared to macrophytes, the distribution of algae is fairly two-dimensional. The calibration for hydrophyte cover is likely to be more critical, although it could be achieved fairly easily since total hydrophyte cover has been assessed routinely as part of WFD macrophyte surveys carried out over the last four years and most surveys carried out for UWWTD purposes using the MTR method were instructed to include measures of total channel cover.

Table 4.6 Non-parametric intercorrelations between abundance metrics derived from macrophyte survey collected using the MTR method. Metrics are ranked from left to right in descending order of strength of relationship with pressure indicators.

metric	Algal cover	Relative algal cover	Mean cover/ taxa	Total hydrophyte cover	Mean cover/ FG
Relative algal cover	0.838				
Mean cover/taxa	0.586	0.277			
Total hydrophyte cover	0.665	0.258	0.900		
Mean cover/FG	0.603	0.280	0.985	0.917	
Invasive spp cover	0.179	0.016[a]	0.080	0.260	0.077

All correlations significant at $p = 0.001$, except a, which is not significant at $p = 0.05$. N = 4138.

Under the UWWTD definition of eutrophication, both abundance metrics could be considered undesirable in that they reflect accelerated growth of plants and a shift to algal dominance. Consequently they might form a useful intermediary between macrophyte and diatom-based assessment. These abundance metrics could be measured in the field by non-specialists and combined with diatom sampling.

5 Establishing reference conditions

5.1 Introduction

The Water Framework Directive requires the identification of reference conditions against which deviation is measured. Therefore reference condition is a concept of overarching and critical importance. While it is generally understood that reference condition implies a ‘pristine’ or ‘near pristine’ state, the Directive is light on detail in terms of defining the term reference condition. Guidance from the REFCOND project (Wallin *et al*., 2005) has sought to clarify this position by defining reference conditions as a state in the present or past corresponding to very low pressure, without the effects of major industrialisation, urbanisation and intensification of agriculture, and with only minor modification of physico-chemistry, hydromorphology and biology. Nevertheless, the problems remain of, firstly, defining levels of anthropogenic pressure that lead to insignificant impacts on the biology, and secondly, where necessary, of finding an historical date that can form a baseline for reference state that is sufficiently ambitious, yet not totally incompatible with present day European human population densities.

The UK database of macrophyte surveys of rivers is large relative to that available for macrophytes in many other EU countries, or for other biological quality elements in the UK (excluding benthic river macroinvertebrates). Given this weight of evidence, an approach to establishing reference conditions was prioritised in this project since its bearing may extend beyond macrophytes and could influence the classification of a large number of water bodies.

We interpret reference conditions as the ecological conditions that existed in water bodies in the late pre-industrial era when anthropogenic impacts were minor and localised relative to today. This comparison is also made in the light of the most degraded conditions found today and not relative to some notional pre-human landscape. Thus, ‘worst available’ could be regarded as component of the definition of reference status since aspects of the normative definitions, such as ‘minimal distortion’, can only be assessed given an understanding of what constitutes ‘severe alteration’. Thus, the setting of class boundaries should be a logical progression of the same framework used to separate reference and non-reference sites.

5.2 A conceptual framework for defining ecological status

Ecological quality is described in terms of deviation from reference conditions using an Ecological Quality Ratio (EQR). While this operates on a continuous scale, classification requires subdividing the EQR into status bands. One could stratify the EQR gradient into bands of equal width, or base the high/good and subsequent boundaries on a small percentile of the distribution of a metric within the population of reference sites. However, the normative definitions for high, good and moderate ecological status demand a more considered approach that takes account of ecological changes that occur across gradients of specific pressures or general degradation.

This project developed a conceptual framework for the placement of class boundaries. This was first discussed in Phillips *et al*. (2003) and has been refined at various stages since, but the underlying framework has been subsequently adopted by other tools and was supported in the guidance by ECOSTAT (2005) on the setting of class boundaries. The framework is described in the following diagram (Figure 5.1). This concept envisages that taxa can be assigned to different functional response groups (such as pressure-sensitive and pressure-tolerant species) that characterise their broad response to a pressure. Consequently, a metric sensitive to that pressure can be stratified, according to the relative proportions of these response groups, in a manner that reflects the normative definitions for different classes of ecological status (Table 5.1). There are some parallels in this approach to the concept of macrophyte Ecological State Groups (ESGs) and integration of the changing proportions of ESGs into systems for the ecological evaluation of coastal lagoons (Orfanidis *et al*., 2003).

A logical ecological interpretation of high status is that the most tolerant taxa (which persist in the most degraded sites) are absent or rare (or account for a small proportion of species or cover), while the most sensitive taxa (absent from the most degraded sites) are common (or dominate the cover). This framework envisages that the crossover between tolerant and sensitive taxa forms the mid-point of moderate status. This class is thus a transition zone between two states. In the first state, macrophyte-mediated ecosystem functions are unaltered and there is a subtle shift in taxonomic composition from high to good. In the second state, macrophyte-mediated functions are severely degraded or fundamentally altered from those existing at high or good status and there is a subtle shift in taxonomic composition from poor to bad. Thus the taxonomic shifts within states are small compared to the shift between states, while poor and bad status are the exact inverse, in terms of the representation of different response groups, of good and high status respectively. The boundaries between good and moderate status (G/M), set at a ratio of sensitive to tolerant species of 65:35, and moderate and poor status (M/P), set at a ratio of sensitive to tolerant species of 35:65, reflect the average standard error (15) in logistic regressions between the major response groups and the metric to which they are related[1]. Thus, if dominance by the most tolerant species is associated with, or contributes directly to, undesirable disturbances (such as loss of increased sedimentation or flood risk) these disturbances can be considered to have a high probability of occurrence at the M/P boundary but a low probability of occurrence at the G/M boundary. The concept of functional response groups, defined in terms of sensitivity to disturbance, does not feature in the normative definitions for macrophytes but the principal is supported by the normative definitions for other biological quality elements. Thus, for example, at good status, for benthic invertebrate fauna in lakes, it is specified that the ratio of disturbance-sensitive to insensitive taxa shows slight signs of alteration from type-specific levels.

The relative positions of high-good and poor-bad boundaries are symmetrical, with sensitive species overwhelmingly dominant at one and tolerant species overwhelmingly dominant at the other. Using the same standard error from logistic regressions, a ratio of sensitive to tolerant species of 85:15 is used as the high-good boundary, since this represents the upper error when tolerant species are predicted to be absent. These ratios are reversed at the poor-bad boundary with 15 per cent sensitive species representing the lower error when sensitive species are predicted to be absent.

[1] This represents a change in approach from that reported previously in which the GM and MP boundaries varied according to the standard error of the relationships on a type-by-type basis, since this led to a contraction of M class when SE was small and an expansion when SE was large. The SE is also set by a linear regression between the relative cover of each response group and the pressure gradient over the linear phase of each response.

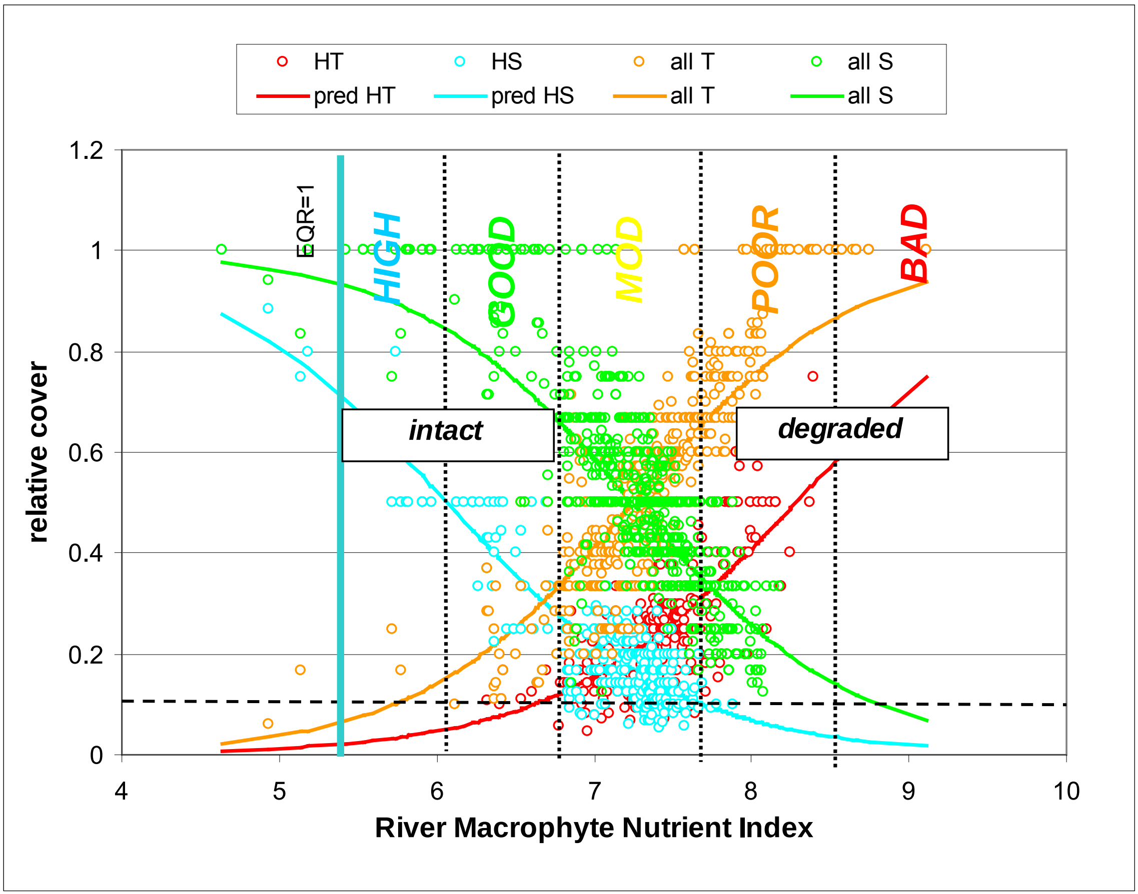


Figure 5.1 Conceptual framework relating structural changes in macrophyte assemblages to normative definitions which was used to provide an ecological interpretation of class boundaries. The River Macrophyte Nutrient Index (RMNI) is shown as an example of an inferred pressure gradient that can be stratified based on the relative abundance of different response groups.

Table 5.1 Interpretation of normative definitions for river macrophytes in the conceptual framework developed for this project

Status	Normative definition	Conceptual framework	
		Structure	Function
High	Taxonomic composition corresponds totally or nearly totally to undisturbed conditions; only minor evidence of distortion. No detectable changes in average macrophytic abundance.	HS taxa dominate, T taxa if present are strongly subordinate, HT taxa occur only as transients and are never established.	Typical macrophyte mediated functions (e.g. habitat support, bed and bank stabilisation, biogeochemical cycling, aesthetics) all intact. No undesirable disturbances.
Good	Slight changes in composition and abundance compared to high status but these should not indicate accelerated growth leading to undesirable disturbances to ecosystem or physicochemical environment	S taxa dominate. HS taxa are scarcer and account for about half the contribution of S taxa. T taxa present but remain subordinate. HT taxa if present, are rare.	Functions delivered at H all intact. Undesirable disturbances rare. Macrophyte cover stable.
Moderate	Composition differs moderately from type-specific communities and is significantly more distorted than the changes observed at good status. Moderate changes in average macrophyte abundance are evident.	A clear transition zone within which T taxa increase significantly but without displacing S taxa. HT taxa present and established but coexisting with HS taxa.	Functions delivered at H performed with reduced efficiency due to shifts in morphological and regenerative trait attributes of macrophyte taxa Undesirable disturbances associated with greater plant biomass or cover of filamentous algae regular but not dominating.
Poor	Major alterations relevant to type-specific conditions including substantial deviation in community composition.	T taxa dominate, of which about half are HT taxa. S remain present but are clearly subordinate. HS taxa, if present, are rare. Essentially the inverse of good.	Functions delivered at H/G significantly impaired, contributing to greater incidence and persistence of undesirable disturbances. Macrophyte cover potentially high and dominated by small number of taxa, often filamentous algae.
Bad	Severe alterations relevant to type-specific conditions including absence of large portions of biological communities associated with undisturbed conditions.	HT taxa dominate, S taxa if present are rare, HS taxa occur only as transients and are never established. Essentially the inverse of high.	Few, if any, elements of original function survive. Undesirable disturbances linked to macrophytes (e.g. fish kills, algal blooms, flooding, biological invasions) frequent and dominating. Macrophyte cover large or unstable.

HS = highly sensitive; S = sensitive; HT = highly tolerant; T = tolerant.
H = High; G =Good

5.3 Identifying reference sites – theory and practice

The WFD (European Union, 2000) requires that "*type-specific biological reference conditions shall be established, representing the values of the biological quality elements specified…for that surface water body type at high ecological status*". There are a number of options, described below and in Figure 5.2, to meet this requirement.

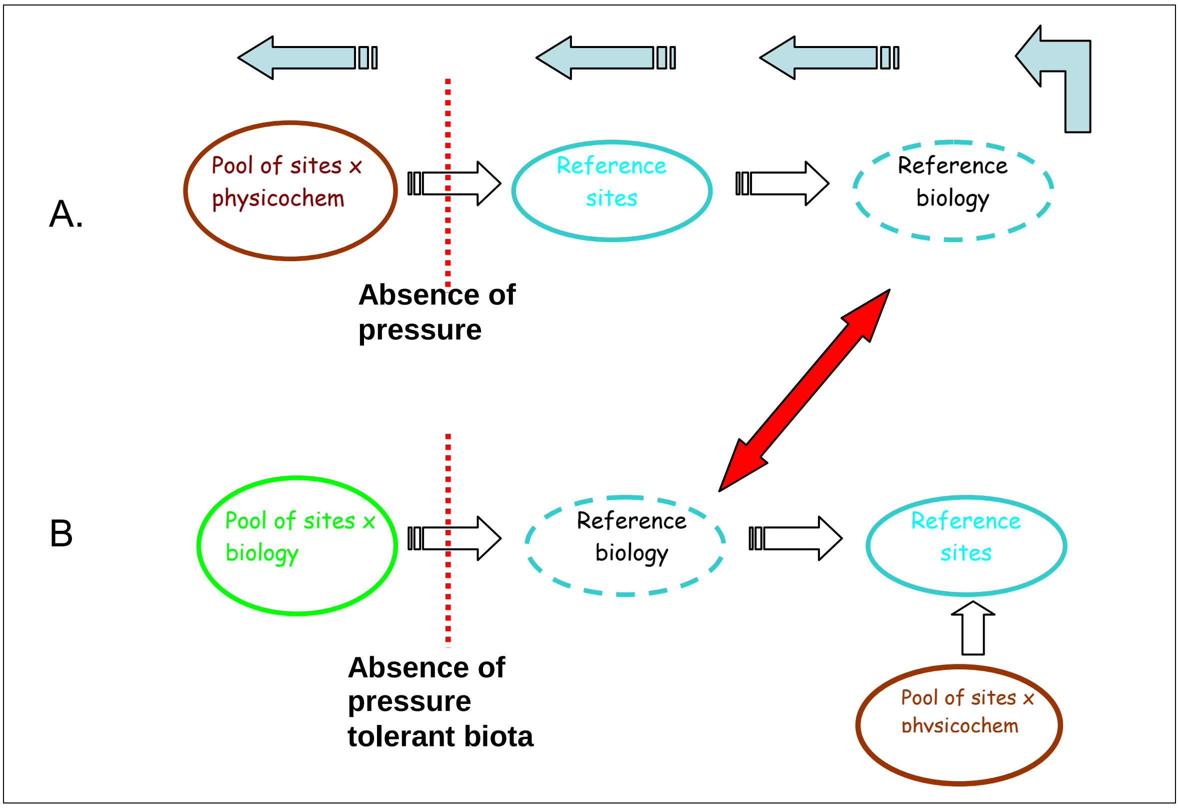


Figure 5.2 Alternative protocols for the identification of biological reference condition sites

Approach A (Figure 5.2) is based on the selection of reference sites where physicochemistry and hydromorphology are considered to be minimally distorted. It is assumed that biological assemblages at sites passing this test constitute reference conditions. REFCOND guidance (Wallin *et al*., 2005) encourages the use of this approach. It relies (i) on the existence of a large and high quality pressure dataset that is spatially and temporarily contemporaneous with the biological dataset and (ii) an ability to define, *a priori*, baseline conditions for pressure indicators. This approach is likely to involve expert opinion, cross-referencing to biological information to identify response thresholds, or access to palaeoecological data. There is an underlying risk of circularity in this approach if screening of environmental data is used to identify reference sites and thus reference biology when the same environmental data is then used to set standards for supporting variables. A further difficulty is that failure to find sites free from pressures – a virtual certainty in some lake and river types – means that this approach may offer little progress in the establishment of biological reference conditions. Moreover, the guiding image of what constitutes reference biology may be distorted by a conservation ethic of what represents the most 'desirable' biology. There are also practical difficulties with this approach because screening thresholds are liable to be set on the basis of the resolution of available data, or limits of detection (such as 20 ug/l orthophosphate), which may have little biological relevance. A generic standard for minimal distortion for physicochemical elements (such as nutrient concentrations) will inevitably lie above or below the values associated with minimal distortion at an individual biological quality element level. This approach is most closely aligned to a 'global' or holistic reference state concept in which values for all quality elements can be considered minimally distorted. However, it is unlikely that the necessary empirical data, or understanding, exists to fully support this concept. Therefore the results of screening via this approach must be considered to represent a population of *potential* reference sites that require subsequent confirmation through expert consideration of the biology that they support. Depending on the outcome of this inspection the original screening thresholds may require revision. Hence, Approach A must be seen as an iterative process.

Approach B (Figure 5.2) is based on compiling a large biological dataset and screening these sites on a type-by-type basis to identify those where indicators of particular pressures are rare or absent. This approach depends on the identification of biological indicators for a suite of pressures, which is likely to rely on expert opinion, or the validation of these indicators using environmental data from a subset of sites. This approach is perhaps best suited to large biological datasets with an incomplete and variable match to pressure data. This approach has several attractions. Firstly it can be applied to archived historical data at sites for which environmental data may be almost totally lacking. Secondly, standards for supporting variables can be set using independent data (not used in the initial identification of reference sites). Thirdly, it is straightforward to model metric values associated with the absence or near absence of pressure indicators, even if no such sites exist, without the need for any *a priori* judgement of values for supporting variables. Such modelling can be undertaken either at a type-specific level (for example, RMNI value associated with a maximum cover of highly sensitive taxa can be predicted by back projection even when no sites meet this criteria), or at a generic level (for instance, using information from all types where reference conditions exist and using a model built on such data to 'fill in the blanks' for 'unpopulated' types). In this approach, reference condition is quality element-specific; the sites or conditions identified capture minimal distortion from the perspective of the quality element and are therefore suitable for the construction of a classification system for that element, but there is no guarantee that they embrace minimal distortion as far as the full range of quality elements are concerned. Approach B relies partly on a space-time substitution to reconstruct temporal changes associated with increasing pressure. However, it is calibrated against archived historical data or information from large-scale biological recording networks in which the 'end members' of the available species pool for particular regions are known.

In reality, these approaches should not be viewed independently but should be seen instead as part of a bilateral approach to identify reference conditions. Cross-comparison of the reference biology generated by each approach is integral to defining reference conditions. This is compatible with the general view that reference sites should be derived through a combination of palaeolimnological approaches, expert judgement, hindcast modelling and interpretation of contemporary data (Moss *et al*., 2003), rather than by prescription. The need for biological screening to help identify reference sites reflects the imperfections and inadequacies of environmental data and the tenuous relationships between biology and environmental indicators of different pressures. Thus, Wallin *et al*. (2005) include the option of screening for reference sites on the basis of saprobity indicators, such as benthic macroinvertebrates. In our study, the volume and quality of biological data is the greatest asset in terms of tool development, while the quality of, and match to, directly measured environmental data is more restrictive. Thus, for example, only one-third of the 404 reference sites used in this project have a comprehensive match to data on SRP, $N\text{-}NH_4$ and TON concentrations, and in many cases the match is of poor quality (for example, chemistry data collected up to 10 years before or after biology data and up to 10 km upstream or downstream of biology data). Routine chemical monitoring of rivers has tended to focus on rivers in densely populated areas with numerous discharge consents and where contemporary reference conditions are unlikely to be found. Consequently, in this project the second approach formed the main route to establishing biological reference conditions, supported by cross-referencing to environmental data (such as land cover) where available and of sufficient quality. This is a pragmatic measure necessary to build a tool from a large pre-existing set of data, rather than from data provided by bespoke sampling designed to identify reference sites. Although there are some inherent weaknesses in this approach it has passed the test of intercalibration across several Geographical Intercalibration Groups (GIGs) and a range of river and lake types, and consequently can be considered fit for purpose.

5.4 Applying the conceptual framework to the selection of reference sites

5.4.1 Identifying functional responses groups

Application of the conceptual framework described above depends initially on the separation of taxa into different ‘functional response groups’. A number of national classification systems developed by other European countries promote the notion of sensitive and tolerant species. Although this distinction is appealingly simple, sensitivity is a relative concept, referable to specific pressure gradients, and forms a continuum. Its use in classification therefore has the risk of introducing another tier of discontinuity into classification. Moreover, ‘sensitive’ and ‘tolerant’ species have often been defined purely from expert opinion into which value judgements associated with conservation value and rarity are often set. Thus, it would be difficult to achieve consensus from a set of experts asked to assign species to different response groups without first setting strict definitions. Consequently, this approach requires consistently applied rules for classifying taxa.

The approach used here to define threshold values of RMNI to delimit different response groups uses a type-specific ordination of survey data constrained by the site RMNI score. The expert view is essentially still embedded in this approach but any bias is reduced by the recalibration of species along the pressure-response gradient in Section 4.1. On the basis of this analysis each species acquires an axis 1 score reflecting the centroid of its occurrence, plus a tolerance value, reflecting the range of site RMNI scores over which that species occurs. Note that a type-specific approach is required since a generic set of groupings would not be appropriate to all river types. For example, if macrophyte species ranks lie on a continuous scale from one to ten, regarding all species with ranks of one to three as always being strongly negatively responding and those with ranks above seven as always positively responding may be suitable for base-poor, upland rivers, but would be totally inappropriate for lowland base-rich sites where site scores typically exceed seven and species characteristic of nutrient-poor conditions are naturally absent.

The basis for classifying species into different response groups is that when species ordination scores switch from negative to positive, the vegetation changes from dominance by overall negative to positive responders on the inferred pressure gradient. This point coincides with the centroid of the site scores included in the analysis. The most strongly negatively scoring species are considered to be the most reliably sensitive indicators of a pressure and are therefore termed ‘highly sensitive species’ (HS). These species are separated from other negatively responding species that ultimately decline along a pressure gradient but are stimulated by a low level of pressure (henceforth referred to as ‘sensitive’ or S species), by using the species score plus its indicator value. The indicator value can be interpreted as a measure of the width of response of each species with respect to an environmental variable, with narrowly distributed taxa having small indicator values. When the sum of the species and indicator scores exceeds zero, a species is no longer considered HS since its statistical indicator value 'carries' it into potentially impacted sites (Figure 5.3). The same approach is used to separate tolerant (T) and highly tolerant (HT) species. Thus, the most positively scoring species are considered the most reliably tolerant indicators of a pressure and are referred to as highly tolerant species (HT). These species are separated from other positively responding species that generally increase along a pressure gradient but decline at the highest levels of pressure (henceforth referred to as tolerant or T species), by using the value of species score minus indicator score. When this value falls below zero, species are no longer considered HT since their indicator value carries them into a zone of less impacted sites.

In adopting this approach, the terms 'sensitive' and 'tolerant' should not be used interchangeably with 'reference' and 'impact'. The assignment of species to response groups is based on their *relative* importance in the vegetation and highly tolerant species, for example, can only be interpreted as representing an impact where they dominate the flora. A second point relates to species lying on the boundary of tolerance and sensitivity. The terms 'ubiquitous' or 'indifferent' are used in some European classification systems for river macrophytes (see Schaumburg *et al*., 2004) and could be applied in this context to a small number of widespread and often abundant taxa located close to the centroid of the axis. Rather than disregarding such taxa, our approach considers that the greater the relative number of such taxa at a site the more likely it is to lie at the interface of tolerance and sensitivity, and thus to represent an intermediate level of pressure. Hence all species are assigned to one of four categories.

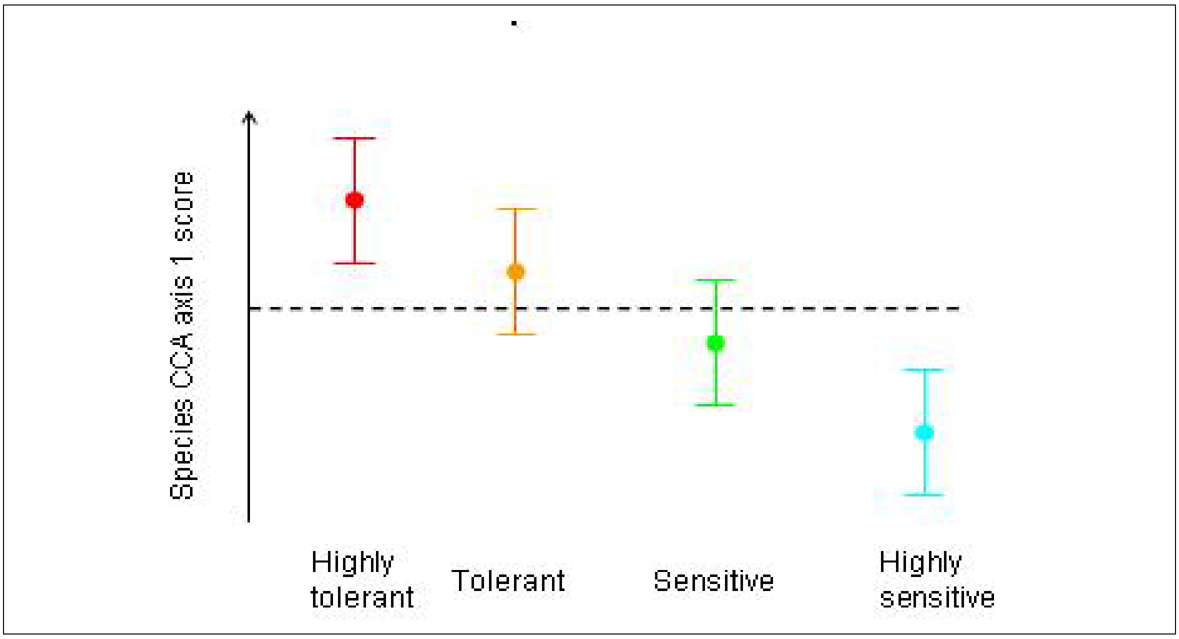


Figure 5.3 Basis for assignment of taxa to response groups. Left axis represents first axis scores of a Canonical Correspondence Analysis (CCA) in which presence-absence species data for that water body type represents dependent variable and site pressure index is explanatory variable. Solid dots represent optima and bars show indicator value. Standardisation of functional response groups.

The above scheme would provide an arbitrary but satisfactory generic basis for separating taxa into different response groups. However, to make comparisons between types it would be necessary to assume that the sample sizes in terms of sites or surveys per type were similar and that the population of surveys in each type spanned a similar gradient of impact. In reality, neither of these assumptions hold true. Any ordination approach would generate scores from which species could be assigned to different response groups but the members of these groups would not be comparable between adjacent types if gradient lengths differed. For example, it is reasonable to suppose from their species composition that the least impacted conditions in low-alkalinity upland streams in north-west Britain mark the true end of a gradient, yet is unclear where the most degraded end of the gradient should be anchored for streams of this type. Conversely, due to a long history of degradation, it is unlikely that the best available high-alkalinity shallow rivers mark the true unimpacted end of a gradient, while it is likely that the worst available sites are correctly anchored.

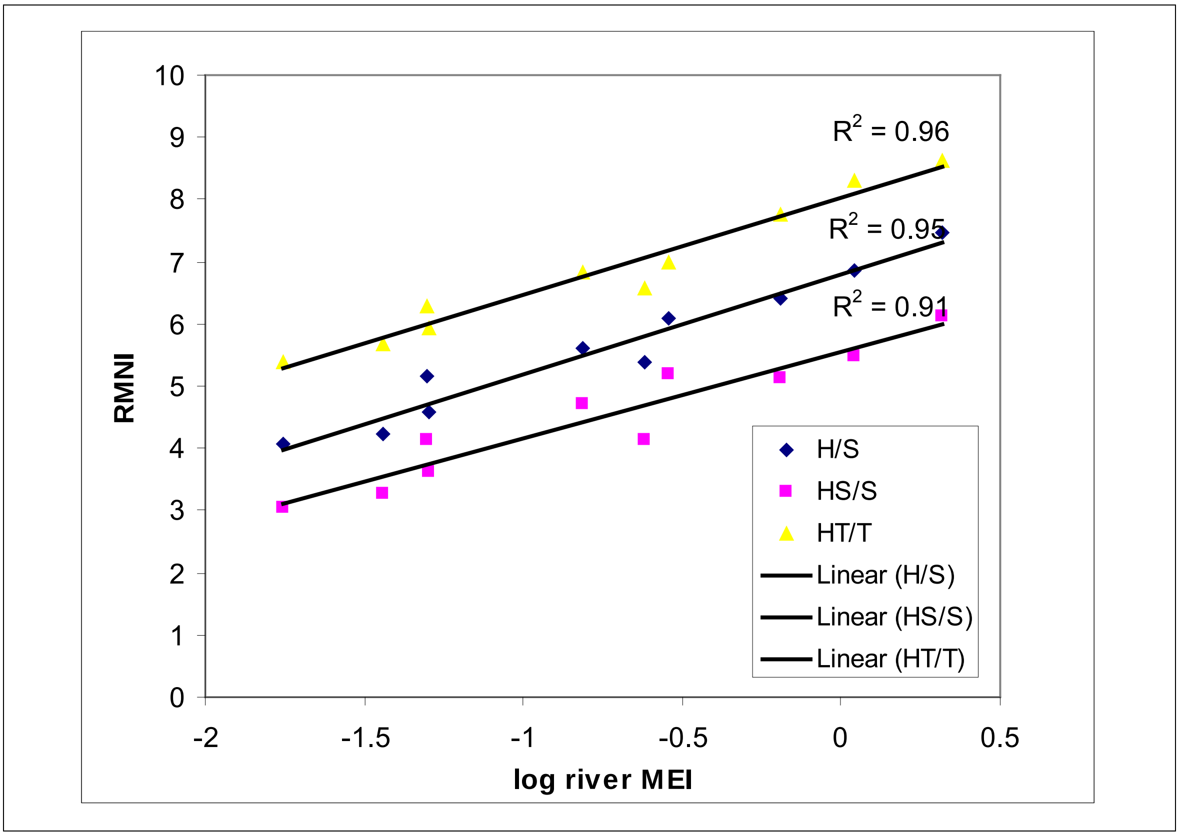


Figure 5.4 Models used to standardise threshold values of RMNI delimiting different response groups in the different river types. The plotted values of RMNI are the thresholds obtained from the CCA approach described above.

Table 5.2 RMNI upper thresholds for different functional response groups pre- and post standardisation

River type	Unstandardised – from CCA			Post-standardisation – from linear model		
	HS	S	T	HS	S	T
Low alk, low gradient	3.63	4.57	5.93	3.74	4.73	5.99
Low alk, mod gradient	3.27	4.24	5.67	3.54	4.49	5.76
Low alk, high gradient	3.06	4.06	5.38	3.10	3.99	5.27
Mod alk, low gradient	5.18	6.08	6.98	4.80	5.94	7.17
Mod alk, mod gradient	4.7	5.6	6.83	4.41	5.50	6.74
Mod alk, high gradient	4.15	5.17	6.28	3.73	4.71	5.98
High alk, high gradient	4.15	5.38	6.57	4.69	5.81	7.05
High alk, mod gradient, NA	5.14	6.42	7.75	5.29	6.50	7.72
High alk, mod gradient, SC	6.31	7.4	8.27	5.84	6.85	7.85
High alk, low gradient, NA	5.48	6.86	8.29	5.61	6.87	8.08
High alk, low gradient, SC	6.78	7.92	8.86	6.24	7.27	8.20
High alk, very low gradient, NA	6.13	7.46	8.62	6.00	7.32	8.52
High alk, very low gradient, SC	6.87	7.89	8.78	6.67	7.71	8.58
Very high alk, mod gradient, SC	6.72	7.68	8.3	6.64	7.68	8.55
Very high alk, low gradient, SC	6.72	7.78	8.91	7.05	8.11	8.92
Very high alk, very low gradient, SC	6.78	7.85	8.64	7.37	8.44	9.20

To standardise the RMNI scores which delimit response groups from highly sensitive to highly tolerant, the CCA scores obtained in Section 5.4.1 for each river type were regressed against a Morpho Edaphic Index (MEI) value for the range of river types (Figure 5.4). MEI is normally used to define potential productivity in lakes based on alkalinity and depth, and was developed originally to assess potential fisheries production (Ryder, 1965). In rivers it was envisaged that slope would operate in a similar role to depth in lakes, but by constraining productivity due to high shear stress. The river MEI (MEIr) is therefore a synthetic index of potential productivity. The river MEI was calculated as the log of (alkalinity (as meq/l)/slope (as m/km)) in which values for alkalinity and slope are the median of values of sites in each river type. This step also has the benefit of downweighting the influence of the small number of surveys that may have been assigned to the wrong water body type due to erroneous environmental data. Standardised scores are given in Table 5.2. Values are the upper thresholds of RMNI for each response group in each river type. Thus, HT taxa are represented by RMNI values exceeding the upper threshold shown for T taxa.

Allocation of species to different response groups was based only on the RMNI index. A separate analysis based on RMHI was considered but dismissed due to the close correlation between RMNI and RMHI scores. Thus, an exploratory analysis using RMHI and based on a few river types indicated that the allocation of species to response groups using RMHI was normally the same as that achieved using RMNI, and was never different by more than one group (for example, species defined as highly sensitive according to one index might be defined as sensitive using the other index).

The full membership of the different response groups is given in Table 5.3 based on the RMNI thresholds presented in Table 5.2. Some qualification is needed of the terms highly tolerant through to highly sensitive to interpret this table. Thus:

1. Highly tolerant taxa are not confined to the most impacted sites. They are likely to be present across the pressure gradient. Highly tolerant species are designated based on the *relative* proportion of cover at a site reflecting the fact that their share of the vegetation increases with impact as more sensitive species are progressively 'deleted'. Highly tolerant species are likely to be present as a subordinate component of the vegetation in reference sites when their share is reduced by the high diversity of other species. Theoretically, the absolute cover of highly tolerant species could in fact be highest in reference sites and decrease with increasing impact provided that relative cover was lowest in reference sites and increased with increasing impact.

2. Highly sensitive taxa are indicative of the lowest level of impact and should therefore occur in reference sites, but will not be the *only* species to occur in reference sites.

3. The classification of species into response groups is specific to a given pressure. Thus it ignores the possibilities that species with very low RMNI scores may, if dominant, be indicative of acidification, or that invasive species may be classed as sensitive species with regard to the nutrient enrichment pressure, yet are indicators of an impact in their own right. The other metrics proposed here will correct for this effect. Guidance on the role of alien species in classification (UKTAG, 2004) will also compensate for the risk of awarding sites with established populations of high risk species an inappropriate level of ecological status.

4. Certain species (with an RMNI score above 9.2, such as *Ceratophyllum demersum, Potamogeton pectinatus*) are considered highly tolerant in all river types. There is good evidence that these species increase with nutrient enrichment and that they can dominate the most enriched sites. However, such species must have always had a niche in the landscape, and are not unique to impacted sites, as indicated in Point 1. These tend to be predominantly species of standing waters that may naturally have had a niche within lower energy habitats of larger rivers (such as connected backwaters) or their floodplain, but would not normally occur in abundance in the main channel due to the effect of scouring flows. Greater nutrient concentrations have probably opened up higher energy environments to some of these species, allowing them to compensate for biomass losses due to scouring flows which would not have been possible under natural nutrient regimes.

5. The overall classification of rivers based on macrophytes is achieved through the combination of information provided by a range of metrics, not just RMNI. This table therefore cannot be used as a guide in its own right to the species that will be found in the best and worst sites within a river type. Hence a site that contains only highly sensitive species will not be classed as high status if the number of such taxa is low. Conversely a site that contains only highly tolerant taxa will not be classed as bad if the diversity of such taxa is high.

Table 5.3 Response group members based on type-specific stratification of the RMNI metric

Taxa	RMNI	Low1 Low2 NA3	Low Mod NA	Low High NA	Mod Low NA	Mod Mod NA	Mod High NA	High High NA	High Mod NA	High Mod SC	High Low NA	High Low SC	High Vlow NA	High Vlow SC	Vhigh Mod SC	Vhigh Low SC	Vhigh Vlow SC
Acorus calamus	9.49				4						4	4	4	4	4	4	4
Alisma lanceolatum	8.47				4	4			4				3	3	3	3	3
Alisma plantago-aquatica	7.82	4	4	4	4	4	4		4	3	3	3	3	3	3	2	2
Anthelia julacea	2.7			1													
Apium inundatum	4.34	2	2	3	1	1	2				1	1	1		1	1	1
Apium nodiflorum	8.64	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	3
Azolla filiculoides	9.71										4	4	4	4	4	4	4
Baldellia ranunculoides	4.34		2	3													
Batrachospermum sp(p)	5.46	3	3	4	2	2	3	2	2	1	1	1	1		1	1	1
Berula erecta	8.24			4		4	4	4	4	4	4	4	3	3	3	3	2
Bidens cernua	8.13				4	4			4		4	3		3	3	3	2
Bidens tripartita	8.39				4				4		4		3	3	3	3	2
Blindia acuta	1.09	1	1	1		1	1		1			1		1			
Blue-green algal scum/pelts	5.1	3	3	3	2	2	3	2	1	1	1	1	1	1	1	1	1
Bolboschoenus maritimus	7.65	4									3		3	2	2		2
Brachythecium plumosum	2.92	1	1	1	1	1	1	1	1	1	1	1	1				
Brachythecium rivulare	3.56	1	2	2	1	1	1	1	1	1	1	1	1		1	1	1
Bryum alpinum	3.83		2	2	1	1	2	1			1						
Bryum dixonii	5.22		3		2	2	3	2		1	1	1					
Bryum pseudotriquetrum	2.71	1	1	1	1	1	1	1	1		1	1					
Butomus umbellatus	8.89		4	4	4		4	4	4		4	4	4	4	4	3	3
Calliergon cuspidatum	3.49	1	1	2	1	1	1	1	1	1	1				1	1	
Callitriche brutia var hamulata	4.51	2	3	3	1	2	2	1	1	1	1	1	1	1		1	1
Callitriche hermaphroditica	5.75		3	4	2		3		2		2	1	1			1	1
Callitriche obtusangula	8.04		4	4	4	4	4	4	4	4	3	3	3	3	3	2	2
Callitriche platycarpa	7.56	4	4	4	4	4	4	4	3	3	3	3	3	2	2	2	2
Callitriche spp.	6.67	4	4	4	3	3	4	3	3	2	2	2	2	1	2	1	1
Callitriche stagnalis	6.47	4	4	4	3	3	4	3	2	2	2	2	2	1	1	1	1
Callitriche stagnalis/platycarpa	6.21								2		2		2				
Callitriche truncata	6.47			4	3		4				2		2				1
Caltha palustris	4.2	2	2	3	1	1	2	1	1	1	1	1	1	1	1	1	1

Carex acuta	7.19	4			4	4	4	4	3		3	2	2	2	2	2	1
Carex acutiformis	8.21	4		4	4	4	4	4	4	4	4	4	3	3	3	3	2
Carex aquatilis	3.9	2	2		1	1		1					1				
Carex elata	4.54	2	3	3	1	2										1	1
Carex lasiocarpa	3.41		1														
Carex paniculata	7.49			4	4					3	3	3		2	2	2	2
Carex recta	5.42												1				
Carex riparia	9.06		4		4	4	4	4	4	4	4	4	4	4	4	4	3
Carex rostrata	2.64	1	1	1	1	1	1	1	1		1	1	1	1			1
Carex vesicaria	3.68	1	2	2	1	1	1	1	1		1		1		1		1
Catabrosa aquatica	8.7					4			4	4		4	4	4	4	3	3
Ceratophyllum demersum	9.73				4	4				4	4	4	4	4	4	4	4
Chara globularis	3.3		1				1										
Chara sp.	3.85		2	2				1									1
Chara vulgaris	3.77	2	2	2		1		1	1		1						1
Chiloscyphus pallescens	4.78			3					1								
Chiloscyphus polyanthos	4.05	2	2	3	1	1	2	1	1	1	1	1	1	1		1	
Cinclidotus fontinaloides	5.37	3	3	4	2	2	3	2	2	1	1	1	1	1	1	1	1
Cladophora aegagropila	5.66		3	4	2	3	3	2	2	1	2	1	1	1	1	1	1
Cladophora glomerata	7.5	4	4	4	4	4	4	4	3	3	3	3	3	2	2	2	2
Cladophora glomerata/Rhizoclonium hieroglyphicum	8.66	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	3
Collema dichotomum	4.42		2	3	1	2	2		1		1		1				
Cratoneuron filicinum	5.02		3	3	2	2	3	2	1	1	1	1	1	1	1	1	1
Dermatocarpon spp. (aggregated)	3.51	1	1	2	1	1	1	1	1		1		1				
Dichodontium flavescens	2.94	1	1	1	1	1	1	1	1	1	1						
Dichodontium palustris	1.68	1	1	1	1	1	1	1	1		1						
Dichodontium pellucidum	3.07	1	1	1	1	1	1	1	1		1	1	1		1		
Draparnaldia	3.04		1				1		1								
Drepanocladus fluitans	3.73		2	2					1			1					
Elatine hexandra	4.17	2															
Eleocharis acicularis	5.35	3							2				1				1
Eleocharis palustris	4.54	2	3	3	1	2	2	1	1		1	1	1	1	1	1	1
Eleogiton fluitans	2.06	1	1	1	1	1	1	1	1		1				1		1
Elodea canadensis	7.65	4	4	4	4	4	4	4	3	3	3	3	3	2	2	2	2
Elodea nuttallii	9.44	4			4	4	4		4	4	4	4	4	4	4	4	4
Equisetum fluviatile	3.92	2	2	2	1	1	2	1	1	1	1	1	1	1	1	1	1
Filamentous green algae	7.61	4	4	4	4	4	4	4	3	3	3	3	3	2	2	2	2

Fisidens polyphyllus	3.84	2															
Fissidens crassipes	6.2	4	4	4	3	3	4	3	2	2	2	1	2	1	1	1	1
Fissidens curnovii	3.94		2	2													
Fissidens osmundoides	3.06			1													
Fissidens rivularis	5.95				3	3			2								
Fissidens rufulus	4.7		3		1	2	2	2			1		1				
Fissidens serrulatus	5.27			3													
Fissidens sp. (aggregated)	5.8		4	4		3	3	2	2	1	2	1	1			1	1
Fissidens viridulus	4.66		3	3	1	2	2	1	1	1	1	1	1			1	1
Fontinalis antipyretica	5.4	3	3	4	2	2	3	2	2	1	1	1	1	1	1	1	1
Fontinalis squamosa	3.66	1	2	2	1	1	1	1	1	1	1	1	1			1	1
Glyceria declinata	6.66			4	3	3	4	3	3	2	2	2	2	1	2	1	1
Glyceria fluitans	5.25	3	3	3	2	2	3	2	1	1	1	1	1	1	1	1	1
Glyceria fluitans agg	5.81				2	3		2	2	1	2	1	1		1	1	
Glyceria maxima	9.64	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Glyceria notata	8.28			4	4	4	4	4	4	4	4	4	3	3	3	3	2
Glyceria x pedicillata	7.12								3	3		2	2	2	2	2	1
Gongrosira incrustans	7.46				4	4		4	3	3	3	3	3	2	2	2	2
Groenlandia densa	7.96				4				4	4	3	3	3	3	3	2	2
Heribaudiella fluviatilis	5.49		3	4	2	2	3	2	2	1	1	1	1		1	1	1
Hildenbrandia rivularis	6.03	4	4	4	3	3	4	3	2	2	2	1	2	1	1	1	1
Hippuris vulgaris	5.94	3	4	4	2	3		3	2	2	2	1	1	1	1	1	1
Hottonia palustris	6.93																1
Hydrocharis morsus-ranae	8.77														4		3
Hydrodictyon reticulatum	8.79				4	4	4				4	4	4	4		3	3
Hygroamblystegium fluviatile	5.41	3	3	4	2	2	3	2	2	1	1	1	1	1	1	1	1
Hygroamblystegium sp.	6.55						4					2					
Hygroamblystegium tenax	5.27			3	2	2	3	2	1	1	1	1	1		1	1	
Hygrobiella laxifolia	2.76	1	1	1													
Hygrohypnum duriusculum	3.33			2													
Hygrohypnum eugyrium	4.28	2			1		2										
Hygrohypnum luridum	2.8	1	1	1	1	1	1	1	1	1	1	1	1				
Hygrohypnum ochraceum	2.96	1	1	1	1	1	1	1	1	1	1	1	1	1		1	
Hyocomium armoricum	1.96	1	1	1	1	1	1	1	1	1	1		1				
Hypericum elodes	2.66		1	1													
Iris pseudacorus	6.92	4	4	4	3	4	4	3	3	3	3	2	2	2	2	1	1
Isoetes lacustris	3.02	1		1	1												

Juncus articulatus	3.1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Juncus bulbosus	1.89	1	1	1	1	1	1	1	1	1	1	1	1			1	1
Jungermannia atrovirens	2.28	1	1	1	1	1	1	1	1		1						
Jungermannia exsertifolia	3.87			2				1	1								
Jungermannia obovata	2.97	1	1	1													
Jungermannia paroica	4		2														
Jungermannia pumila	3.29		1				1										
Jungermannia sp.	2.41	1	1	1	1		1	1	1								
Jungermannia sphaerocarpa	3.08	1	1	1		1	1										
Lemanea fluviatilis	4.51	2	3	3	1	2	2	1	1	1	1	1	1	1	1	1	1
Lemanea sp(p.)	4.53	2	3	3	1	2	2	1	1	1	1	1	1	1	1	1	
Lemna gibba	10								4		4	4	4	4	4	4	4
Lemna minor	8.8	4		4	4	4	4	4	4	4	4	4	4	4	4	3	3
Lemna minuta	9.21				4	4						4	4	4	4	4	4
Lemna trisulca	8.21				4				4	4		4	3	3	3	3	2
Leptodictyon riparium	7.57	4	4	4	4	4	4	4	3	3	3	3	3	2	2	2	2
Littorella uniflora	1.96	1	1	1	1	1	1		1								
Lobelia dortmanna	2.72	1	1	1													
Luronium natans	4.37	2		3	1												
Lythrum salicaria	7.33	4	4	4	4	4	4	4	3	3	3	3	3	2	2	2	1
Marsupella aquatica	3.17			2			1										
Marsupella emarginata	1.06	1	1	1		1	1	1			1		1				
Mentha aquatica	6.27	4	4	4	3	3	4	3	2	2	2	2	2	1	1	1	1
Menyanthes trifoliata	3.14	1	1	2	1	1	1	1	1		1		1				1
Mimulus guttatus	5.79	3	4	4	2	3	3	2	2	1	2	1	1	1	1	1	1
Mimulus sp./hybrid	5.6	3	3	4	2	3	3	2	2	1	1		1			1	
Monostroma sp.	6.86				3				3		2			2		1	
Montia fontana	3.35	1	1	2	1	1	1	1	1	1	1					1	
Myosotis laxa	4.82	3			2	2	3	2	1		1		1	1	1	1	
Myosotis scorpioides	6.83	4	4	4	3	4	4	3	3	2	2	2	2	2	2	1	1
Myosotis secunda	4.74	3	3	3	1	2	3	2	1							1	1
Myriophyllum alterniflorum	3.44	1	1	2	1	1	1	1	1	1	1	1	1	1		1	1
Myriophyllum spicatum	8.26	4	4	4	4	4	4	4	4	4	4	4	3	3	3	3	2
Myriophyllum verticillatum	7.53													2			
Nardia compressa	1.05	1	1	1	1	1	1	1	1								
Nardia scalaris	2.73	1		1													
Nardia sp.	1.4			1			1	1									

Nitella flexilis (agg.)	4.39	2	2	3	1	1			1		1				1		
Nitella opaca	4.31	2	2		1	1	2		1								
Nitella sp	4.59	2	3	3					1			1	1				
Nitella translucens	4.17	2															
Nostoc commune	5.14	3		3	2			2	1								
Nostoc parmelioides	4.12		2	3	1	1	2	1	1		1		1				
Nostoc verrucosum	4.71		3	3	1				1		1						
Nuphar lutea	8.42	4	4	4	4	4	4		4	4	4	4	3	3	3	3	2
Nymphaea alba	5.69	3	3			3					2	1		1		1	1
Nymphoides peltata	9.37										4	4		4	4	4	4
Octodiceras fontanum	6.54				3	3					2	2	2		1	1	1
Oenanthe aquatica	6.06		4		3	3	4		2		2		2	1			1
Oenanthe crocata	6.22	4	4	4	3	3	4	3	2	2	2	1	2	1	1	1	1
Oenanthe fistulosa	8.27					4						4		3	3	3	2
Oenanthe fluviatilis	8.57			4	4		4	4	4		4	4	4	3	4	3	3
Orthotrichum rivulare	4.71	2		3	1	2	2	2			1						
Palustriella commutata	4.61		3	3			2	1	1		1	1					
Pellia endiviifolia	6.5	4	4	4	3	3	4	3	2	2	2	2	2	1	1	1	1
Pellia epiphylla	3.34	1	1	2	1	1	1	1	1	1	1	1	1	1		1	1
Pellia sp.	4.67	2		3	1	2	2	1	1	1					1		
Persicaria amphibia	8.2	4	4	4	4	4	4	4	4	4	4	3	3	3	3	3	2
Persicaria hydropiper	6.97	4	4	4	3	4	4	3	3		3	2	2	2	2	1	1
Phalaris arundinacea	7.52	4	4	4	4	4	4	4	3	3	3	3	3	2	2	2	2
Philonotis caespitosa	2.74	1		1													
Philonotis fontana	2.66	1	1	1	1	1	1	1	1	1	1						
Phragmites australis	7.7	4	4	4	4	4	4	4	3	3	3	3	3	2	3	2	2
Platyhypnidium alopecuroides	4.35						2										
Platyhypnidium riparioides	5.16	3	3	3	2	2	3	2	1	1	1	1	1	1	1	1	1
Porella cordaeana	4.95							2									
Porella pinnata	4.91	3			2	2	3										
Potamogeton alpinus	4.96	3	3	3	2	2			1				1	1		1	1
Potamogeton berchtoldii	7.35	4	4		4	4		4			3	3	3	2	2	2	1
Potamogeton crispus	8.02	4	4	4	4	4	4	4	4	4	3	3	3	3	3	2	2
Potamogeton filiformis	6				3						2		2				
Potamogeton friesii	8.19												3				2
Potamogeton gramineus	4.24	2	2	3	1	1					1		1		1	1	
Potamogeton lucens	8.54		4		4						4	4	4	3	3	3	3

Potamogeton natans	5.69	3	3	4	2	3	3	2	2	1	2	1	1	1	1	1	1
Potamogeton nodosus	7.05				3									2			1
Potamogeton obtusifolius	5.84	3									2						1
Potamogeton pectinatus	9.59	4			4	4			4	4	4	4	4	4	4	4	4
Potamogeton perfoliatus	8.16	4		4	4	4	4	4	4		4	3	3	3	3	3	2
Potamogeton polygonifolius	1.71	1	1	1	1	1	1	1	1	1	1		1	1			1
Potamogeton praelongus	7.81				4				4		3		3	3	3		2
Potamogeton pusillus	7.47	4			4	4					3	3	3	2		2	2
Potamogeton trichoides	7.24										3						1
Potamogeton x bottnicus	6.41												2				
Potamogeton x cooperi	6.07				3						2		2				
Potamogeton x fluitans	6.51								3								
Potamogeton x lanceolatus	4.24							1									
Potamogeton x nitens	6.17					3											
Potamogeton x olivaceus	5.44				2	2					1		1				
Potamogeton x salicifolius	6.36				3						2		2			1	1
Potamogeton x sparganifolius	3.87	2		2	1												
Potamogeton x suecicus	6.02										2		2				
Potamogeton x zizzii	4.19	2	2		1												
Potentilla palustris	2.88	1	1	1	1	1		1	1				1				
Racomitrium aciculare	1.89	1	1	1	1	1	1	1	1		1						
Ranunculus (sect Batrachian) sp or hybrid	7.33	4	4	4	4	4	4	4	3	3	3	3	3	2	2	2	1
Ranunculus aquatilis var aquatilis	5.67	3			2	3	3	2	2	1	2	1	1	1		1	1
Ranunculus aquatilis var diffusus	7.65	4		4			4	4	3		3	3	3		2	2	2
Ranunculus circinatus	9.42											4			4	4	4
Ranunculus flammula	2.56	1	1	1	1	1	1	1	1	1	1	1	1		1	1	1
Ranunculus fluitans	7.97			4	4	4	4	4	4	4	3	3	3	3	3	2	2
Ranunculus hederaceus	5.47	3	3	4	2	2	3	2	2	1	1	1			1		
Ranunculus omiophyllus	3.43	1	1	2	1	1	1	1	1				1				1
Ranunculus peltatus var baudotii	9.06														4		
Ranunculus peltatus var peltatus	6.22	4	4	4	3	3	4	3	2	2	2	1	2	1	1	1	1
Ranunculus penicillatus	8.25				4				4	4			3		3	3	2
Ranunculus penicillatus ssp. penicillatus	6.29	4	4	4	3	3	4	3	2	2	2	2	2	1	1	1	1
Ranunculus penicillatus ssp. pseudofluitans	7.92	4	4	4	4	4	4	4	4	4	3	3	3	3	3	2	2
Ranunculus penicillatus subsp vertumnus	5.87	3	4		2	3		3	2		2	1	1			1	1
Ranunculus sceleratus	9.86			4		4	4		4	4	4	4	4	4	4	4	4
Rhodochorton violaceum	4.14	2	2	3	1	1	2	1	1		1						

Riccardia chamaedryfolia	4.91	3		3		2	3	2									
Riccardia multifida	5.25				2	2			1								
Riccia sp.	4.86		3		2	2	3	2	1	1			1				
Rivularia	4.77							2									
Rorippa amphibia	9.2				4	4	4	4	4	4	4	4	4	4	4	4	4
Rorippa nasturtium-aquaticum	8.42	4	4	4	4	4	4	4	4	4	4	4	3	3	3	3	2
Rorippa palustris	7.32	4			4	4	4		3	3	3	3	2	2		2	1
Rumex hydrolapathum	8.65			4	4	4	4	4	4	4	4	4	4	4	4	3	3
Sagittaria sagittifolia	9.24				4					4	4	4	4	4	4	4	4
Scapania sp. (aggregated)	2.14		1	1		1	1	1	1	1							
Scapania subalpina	3.21		1	2				1									
Scapania uliginosa	2.66			1													
Scapania undulata	2.05	1	1	1	1	1	1	1	1	1	1						
Schistidium agassizii	2.23	1	1	1	1	1			1		1						
Schistidium rivulare	5.16					2		2									
Schoenoplectus lacustris	8.44	4	4	4	4			4	4	4	4	4	3	3	3	3	2
Schoenoplectus tabernaemontani	7.43	4			4						3		3			2	
Scirpus sylvaticus	6.45	4		4	3	3	4		2	2	2	2	2	1	1	1	1
Scorpidium revolvens	4.29	2	2				2		1		1						
Sium latifolium	7.08												2			2	
Sparganium angustifolium	2.26	1	1	1	1	1					1						
Sparganium emersum	8.32	4	4	4	4	4	4	4	4	4	4	4	3	3	3	3	2
Sparganium erectum	8.34	4	4	4	4	4	4	4	4	4	4	4	3	3	3	3	2
Sparganium natans	3.59	1				1											
Sphagnum denticulatum	4.84	3															
Sphagnum sp(p)	1.07	1	1	1	1	1	1	1	1		1						
Spirodela polyrhiza	8.99				4				4	4	4	4	4	4	4	4	3
Spirogyra	6.45	4	4	4	3	3	4		2		2			1	1	1	1
Stigeoclonium tenue	6.62	4	4	4	3	3	4	3	3		2	2	2		1	1	1
Stigonema sp	4.32	2		3												1	
Tetraspora lubrica/gelatinosa	6.72				3	3		3	3		2		2		2	1	1
Thamnobryum alopecurum	4.22	2	2	3	1	1	2	1	1	1	1	1	1		1		1
Tolypothrix penicillata	2.96	1	1	1					1								
Triglochin palustris	4.07	2	2	3			2		1				1			1	
Typha angustifolia	7.57				4					3				2		2	2
Typha latifolia	8.87	4	4		4	4	4		4	4	4	4	4	4	4	3	3
Ulva flexuosa	9.52		4	4	4	4	4	4	4	4	4	4	4	4	4	4	4

Utricularia intermedia	2.74	1	1														
Utricularia minor	3.77	2						1									
Utricularia sp	3.23	1	1														
Utricularia vulgaris s.l.	3.72	1															
Vaucheria sp(p)	8.41	4	4	4	4	4	4	4	4	4	4	4	3	3	3	3	2
Veronica anagallis-aquatica	8.45	4	4	4	4	4	4	4	4	4	4	4	3	3	3	3	3
Veronica beccabunga	7.31	4	4	4	4	4	4	4	3	3	3	3	2	2	2	2	1
Veronica catenata	9.32			4	4			4	4	4	4	4	4	4	4	4	4
Veronica catenata x anagallis-aquatica	8.34			4									3		3	3	2
Veronica scutellata	2.35	1	1	1	1		1		1			1					1
Zannichellia palustris	9.01					4		4	4	4	4	4	4	4	4	4	3

5.5 Screening survey databases for reference sites

Having assigned all sites to a river type and produced a standardised classification of taxa into different response groups on a type-specific basis, it is possible to extract surveys from the database that meet a set of pre-defined criteria. These criteria are summarised below. In general, biological indicators provide an adequate screen in their own right (less than 10 per cent of sites that passed these criteria were subsequently removed by the imposition of the pressure indicator criteria). On this basis, it is assumed that when screening by pressure indicators is not possible, or is incomplete due to lack of data, sites which meet the biological criteria should be admitted to the population of reference sites.

Table 5.4 Biological and physical criteria used in the screening of reference sites on UK rivers

Biological indicators

- Less than 15 per cent of total cover composed of ‘pressure-tolerant’ taxa.
- Highly pressure-sensitive species present.
- Cover of highly tolerant species less than 10 per cent of total cover.
- Number of aquatic taxa and number of aquatic plant functional groups over 25th percentile of type-specific richness (with separate standards for different methods).
- Total hydrophyte cover and mean cover score per species within type and method specific 10-90 percentile range.
- No taxa with individual cover scores of six or greater (10-25 per cent cover).
- No established invasive alien or translocated native species (cover scores of two or less, maximum one per cent cover).
- Documented acid-tolerant taxa (*Juncus bulbosus* and aquatic sphagna and leafy liverworts) comprise less than 50 per cent of total hydrophyte cover.
- Absolute cover of green filamentous algae less than 2.5 per cent.
- Relative cover of green filamentous algae less than 20 per cent.

Pressure indicators (based on interpretation of REFCOND guidance)

- Concentrations of major nutrients under thresholds proposed by REFCOND (mean N-NH_4 = 0.05 -0.1 mg/l depending on type; mean SRP = 20-40 ug/l depending on type; mean N-NO_3 = 2-4 mg/l depending on type).
- River Habitat Survey HMS Class 1 or 2.
- No resectioning of reaches recorded by RHS.
- Influenced flows within 10 per cent of naturalised flows.
- Impacted land cover (tilled land, permanent pasture, verges, amenity grassland, urban and suburban) less than 20 per cent of catchment area.

In terms of biological criteria, the significance of 15 per cent of cover being represented by pressure-tolerant species is explained in Section 5.2. In effect, this value represents the upper prediction error when tolerant species are completely absent.

The use of land cover data and population data has been trialled as surrogate pressure measurements when no directly measured concentrations of nutrients are available. These generally support the biologically based definition of reference conditions based on option B (Figure 5.2) and the rules defined above. To permit the inclusion of screening by land cover a putative set of reference sites was identified based on all criteria other than land cover. This confirmed that almost all reference sites identified by these criteria alone had catchments in which impacted land cover (essentially tilled land, permanent pasture or urban and suburban land use) accounted for under 30 per cent of the area, and that impacted land cover was usually well below 20 per cent. This therefore supported the use of the 20 per cent threshold recommended by REFCOND (Wallin *et al*., 2005). The distribution of surveys by final class in relation to land cover is illustrated in Figure 5.5 which confirms the concentration of high status biology in catchments with predominantly low intensity land use.

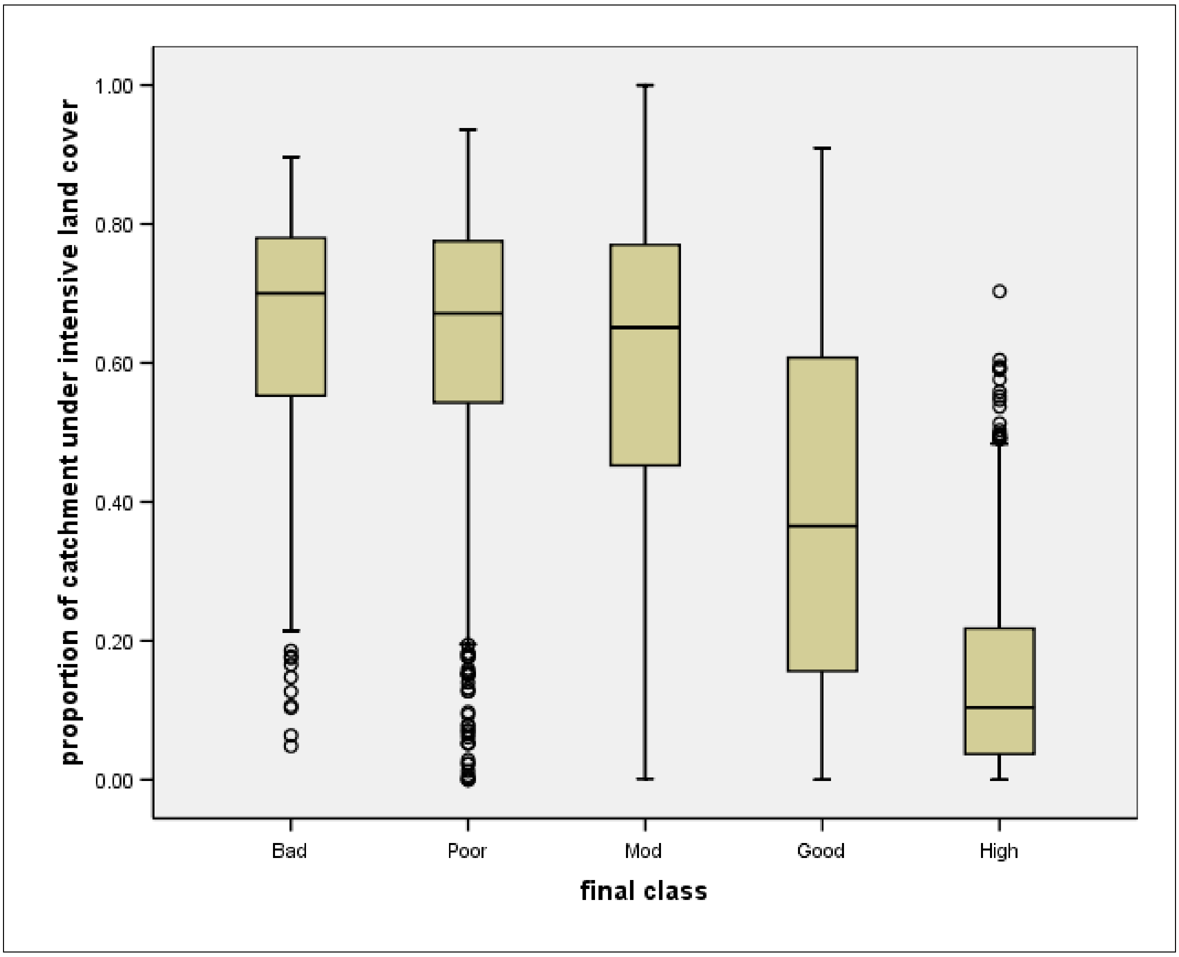


Figure 5.5 Relationship between impacted land cover in the catchment and final ecological status of sites based on their biology. Note that high status sites retain the below 20 per cent threshold set for reference sites although high status biology can occasionally occur in more intensive catchments. Conversely, moderate-bad classes typically occur in catchments with over 50 per cent intensive land cover (although can be found occasionally when intensity of land cover is low). Good status occupies an intermediate position.

6 Predicting the expected flora of rivers under reference conditions

6.1 Background

The derivation of ecological status relies on comparison between the observed value of a metric or set of metrics with values expected under reference conditions. This comparison is made in the form of an Ecological Quality Ratio (EQR). Various approaches have been used to 'predict' the expected values for metrics. At a European level the use of type-specific classifications is widespread. Under this approach, the median value of a metric found in a set of reference sites belonging to a single type is used as the expected value. However, following the site-specific approach developed for invertebrate classification through RIVPACS, most UK classification tools have attempted to provide site-specific expected values for use in classification.

There are essentially three routes to site-specific predictions. The first, followed by RIVPACS, undertakes a series of mapping steps in which environmental predictor data is linked to site type, site type is linked through TWINSPAN to biological community structure, and metrics are predicted from community structure using multiple discriminant analysis. This is a relatively intensive process, lacks flexibility (for example the addition or removal of a site from the reference network requires the re-clustering of community structure and calculation of new algorithms to predict reference metric values), and is constrained by the range of sites which offer reference biological assemblages. A second option would be to develop species-specific models (see Barendregt & Bio, 2003) based only on reference site biology, and to use these models to predict the probability of each species occurring given the combination of values for environmental predictors at a test site. The metric values required for classification could then be generated from the predicted assemblage. This approach is statistically robust and has the advantage of providing a guiding image for an impacted site in terms of a list of taxa expected in the absence of impacts. However, it is computationally demanding and is a convoluted route to achieving reference metric values.

A third and functionally simpler process is to predict the metric values directly from a set of linked environmental data. Walley & Fontama (1997) affected this step for river macroinvertebrates using a back-propagation neural network and showed that this offered comparable or superior predictive ability to the standard RIVPACS approach, with less bias in predictions and via a simpler overall route. The direct approach to metric prediction is followed here, although in this instance General Linear Modelling is used as the basis for prediction. Kelly *et al.* (2008) reported that this gave acceptable performance for the prediction of the diatom metric TDI, while prediction of TDI via a back-propagation neural network yielded a model with similar prediction errors to the simpler regression models. A major advantage of the direct approach is that, by using a population of metric values, it is possible to predict the values of metrics when reference sites are lacking.

One consequence of this approach to predicting reference metric values is that the tool does not predict the actual composition of the assemblage expected under reference conditions, in terms of a list of taxa with their probabilities of occurrence. In the first phase of the PLANTPACS project, Maberly *et al*. (2001) suggested that deviation from the expected plant assemblage could be used as the basis for a disturbance index for assessing ecological status, thus following the approach used in the early versions of RIVPACS. Although the option to generate this type of information remains, it has been excluded here for a number of reasons, several of which are specific to macrophytes:

i. There is a marked paucity of reference condition sites in some river types which means that predictions of the flora in some rivers will be outside the envelope of reference conditions on which the model is built, and consequently will not be reliable.

ii. Compared to generally mobile invertebrates, dispersal limitation is a constraint on the occupancy of suitable sites by macrophytes. This results in the majority of species having a comparatively low probability of occurrence. Conversely, there is a high risk of failing to find a taxa whose occurrence is 'expected'. Under such circumstances the utility of direct predictions of taxonomic composition seems questionable. As examples, Willby & Eaton (2001) used Multivariate Discriminant Analysis (MDA) to predict changes in the vegetation of the Montgomery Canal with increases in boat traffic, while Willby & Birk (submitted) explored high status plant assemblages of different intercalibration river types in developing a common metric. In both cases, the number of species with an expected probability of occurrence exceeding 50 per cent (species more likely to be present than absent) was low (four to six species, 5-10 per cent of the potential species pool). Even by lowering the threshold for probability of occurrence to 20 per cent (species five times more likely to be absent than present), the number of expected species only increased to 15-20 (25-35 per cent of the species pool). By comparison, RIVPACS would typically predict 30-40 species or 15-20 families of macroinvertebrates to occur with over 50 per cent probability in comparable lowland river types. Hawkins *et al*. (2000) found that predictions of invertebrate species models were significantly more robust when they were restricted to species with a probability of capture above 0.5. Given that this threshold would exclude all but the commonest and most widely distributed species of macrophytes, such comparisons of observed and expected assemblages would not be useful.

iii. A further problem deriving from point ii is that macrophyte species with the highest probability of occurring are invariably common and widespread, distributed over much of the quality gradient and consequently have low indicator value. Thus, in the examples cited in ii these species included *Elodea canadensis*, *Lemna minor* and *Sparganium emersum.*

iv. The use of expected taxa lists as a benchmark for comparison means that any observations of taxa expected to be absent from reference sites (such as most invasive alien species and some highly tolerant species) are redundant.

v. Assemblages composed of species that regularly co-occur plus species distributed more or less independently along environmental gradients will be poorly served by the types of shortcut assemblage models employed within RIVPACS (Olden *et al*., 2006).

vi. Assemblage models are constructed on imperfect survey data in which detection bias within and between observers has the potential to influence the results of the clustering process and subsequent predictions.

vii. Even the most likely taxa at a site have a rather low probability of occurrence (point ii above). Therefore, predictions could be misleading if they are used by surveyors as a guide to which species they might encounter at sites they are not familiar with.

viii. In contrast to metric-based approaches, the results of classification are likely to be more prone to observer-based sources of variability when the classification of a site relies on finding particular species.

6.2 Type- versus site-specific classifications

The simplest approach to deriving type-specific reference metric values is to merely calculate the median value of a metric for the reference sites in each river type. EQRs for all non-reference site members of that type can then be expressed relative to this median value. The difficulty with this approach is that each river type is treated in isolation when, in reality, even discrete types comprise a gradient of productivity. Moreover, the populations of some reference sites in a given type are very small (under five) and are therefore dubiously representative of the true reference condition for that type (for example, probably situated closer to the high/good boundary than middle of reference). A superior approach might therefore be to first standardise type-specific values *across* a gradient rather than treat them independently. Thus, the type-specific value for a given type becomes influenced to some degree by values established by adjacent types on a productivity gradient, rather than being defined in isolation. This is consistent with the type-specific screening approach in which RMNI thresholds for the different functional response groups were standardised prior to screening. The standardisation of type-specific reference values introduces some of the attributes of site-specific prediction and could be seen as a hybrid approach.

Several detailed standardisation procedures were discussed in depth when developing a lake classification based on macrophytes, but the end result was always inferior to that derived using a site-specific classification. Among the weaknesses encountered was a tendency for a type-specific approach to severely underpredict the expected values for high status sites compared with a site-specific approach. This reflects the inability of a type-specific approach to incorporate additional biologically relevant environmental variables or to accommodate sites that lie naturally on type boundaries and which type-specific screening would normally reject.

Once site-specific models are derived it is highly unlikely that one would return to a type-specific approach, unless insufficient environmental data was available for site-specific models to be applied, or if a type-specific approach was necessary for intercalibration purposes. However, in such cases it is easy to derive type-specific values for different metrics simply by determining the median of the population of site-specific predicted metric values for all the sites in that type. It is scarcely conceivable that one would attempt type-specific predictions of some metrics, such as richness, given that they are likely to depend on large-scale spatial variables, such as distance from source or catchment area, that do not contribute directly to the typology. On the basis of experience with using lake macrophytes for water body classification, we only consider site-specific classifications from this point onwards.

The process of developing a biologically relevant typology followed by screening at a type-specific level is the means to an end and not the end in itself. These processes generate a population of reference sites and it is therefore possible, using the environmental data linked to individual surveys, to predict reference values for any given metric at a site-specific level (Figure 6.1). Although the type-specific approach has been much favoured by those developing tools for a variety of biological quality elements (including macrophytes) in continental Europe, the site-specific approach has some theoretical and practical advantages:

i. Ability to incorporate environmental variables that cannot be accommodated in a simple typology but which contribute to biological variation.

ii. Allowance for continuous variation rather than reducing within-type variability to a single value. Thus, artificial discontinuities and under-representation of sites at type boundaries is avoided.

iii. Inclusion of all data in a single model circumvents problems caused by small numbers of reference sites in a single type and reduces prediction error.

iv. Ease of modification. Models are easily refined as new environmental data becomes available or reference sites are added or removed.

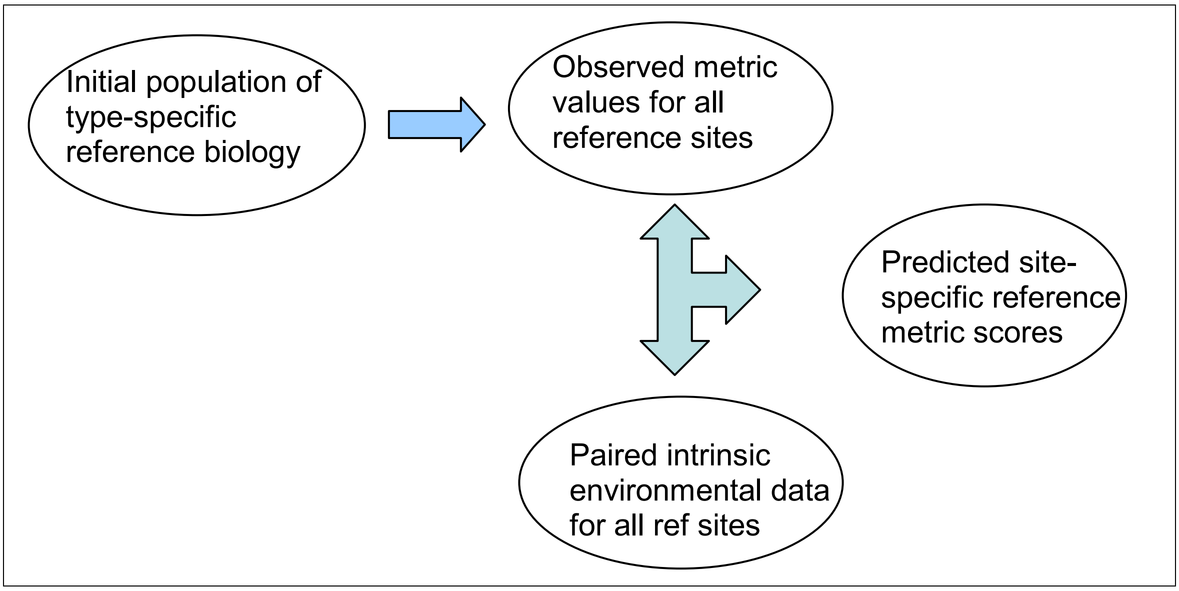


Figure 6.1 Underlying process behind site-specific prediction of reference metric values

The following section outlines the process of developing models of site-specific reference conditions for each of the metrics discussed in Section 4, followed by information on the calculation of the associated EQR and basis for the derivation of class boundaries. The reference value for a water body is considered fixed and is not free to vary with variations in environmental predictors. Thus, the environmental data used should represent the best available, long-term view of that water body (based on the long-term average for parameters such as alkalinity) rather than face value measurements taken at the time of biological data collection. It is assumed that geographically-based parameters (such as altitude, distance from source) can be measured with negligible error.

The models developed here are based on the best available environmental data. For some metrics a minimum common subset of predictors was assumed to exist but for others, various scenarios of environmental data availability were considered. Note, however, that only a single model can be used per water body to predict reference values for a particular metric. As additional data becomes available, and the values for some predictive variables change or estimates are improved, it may be necessary to revisit these models to adjust the coefficients.

6.3 River Macrophyte Nutrient Index

6.3.1 Prediction

RMNI values associated with the population of 404 reference sites were used as the basis for developing models to predict site-specific reference values. The set of intrinsic environmental data associated with these sites was used as predictors. RMNI was significantly correlated with a range of variables, most notably alkalinity, slope and distance from source (Figure 6.2). Since the scatter in the relationship between RMNI and these variables was generally quite small (Figure 6.2), a stepwise multiple regression procedure was used to identify the most parsimonious models for predicting RMNI. To accommodate various scenarios of environmental data availability (such as no available measures of alkalinity) a number of models were developed. Model performance was assessed based on the percentage variance in RMNI explained by the combination of environmental variables (the squared correlation between observed and predicted values), the standard error of the prediction, and the slope and intercept of the relationship between expected and observed values.

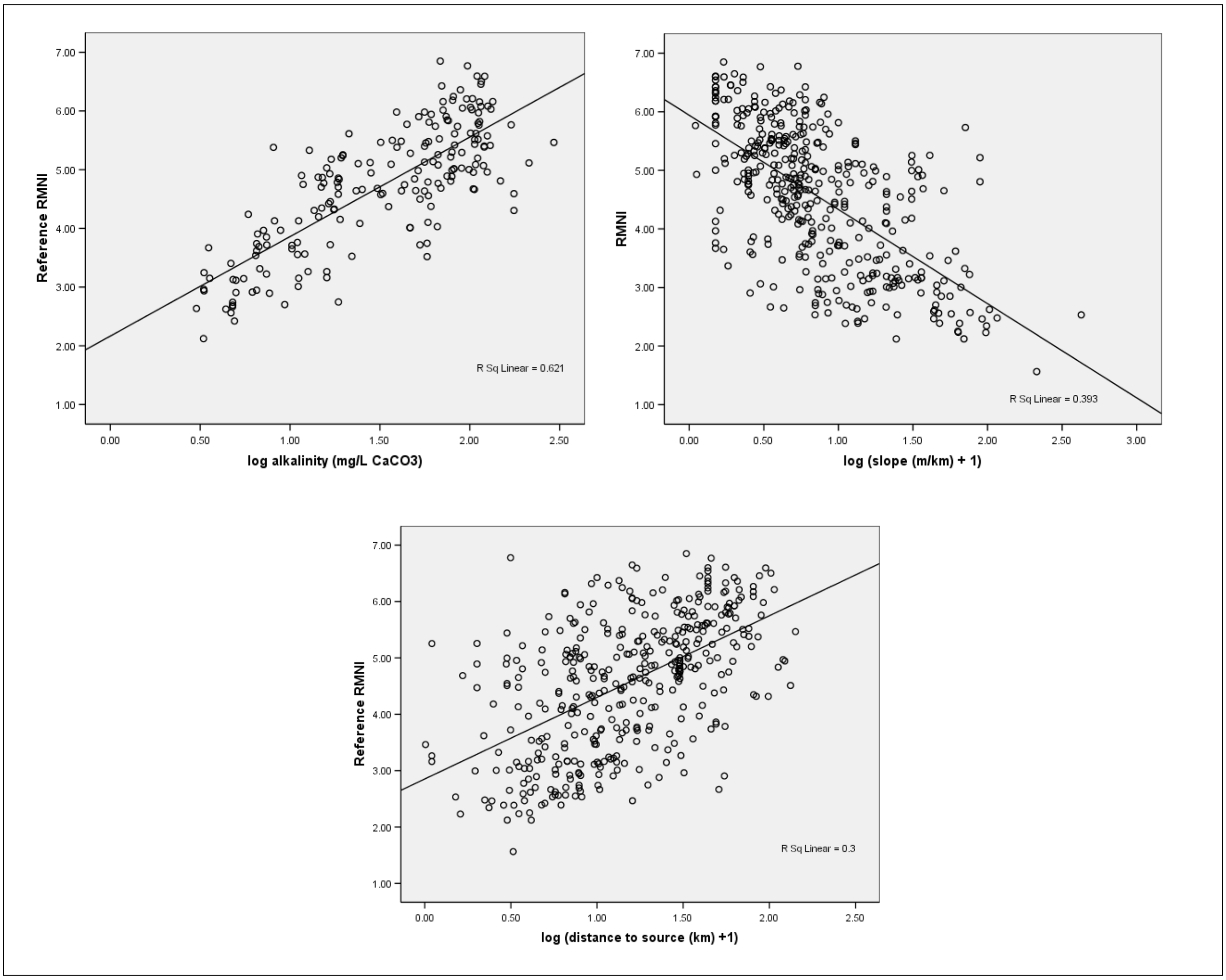


Figure 6.2 Scatter plots of relationship between reference site RMNI and three intrinsic environmental variables: alkalinity (upper left), slope (upper right) and distance to source (lower)

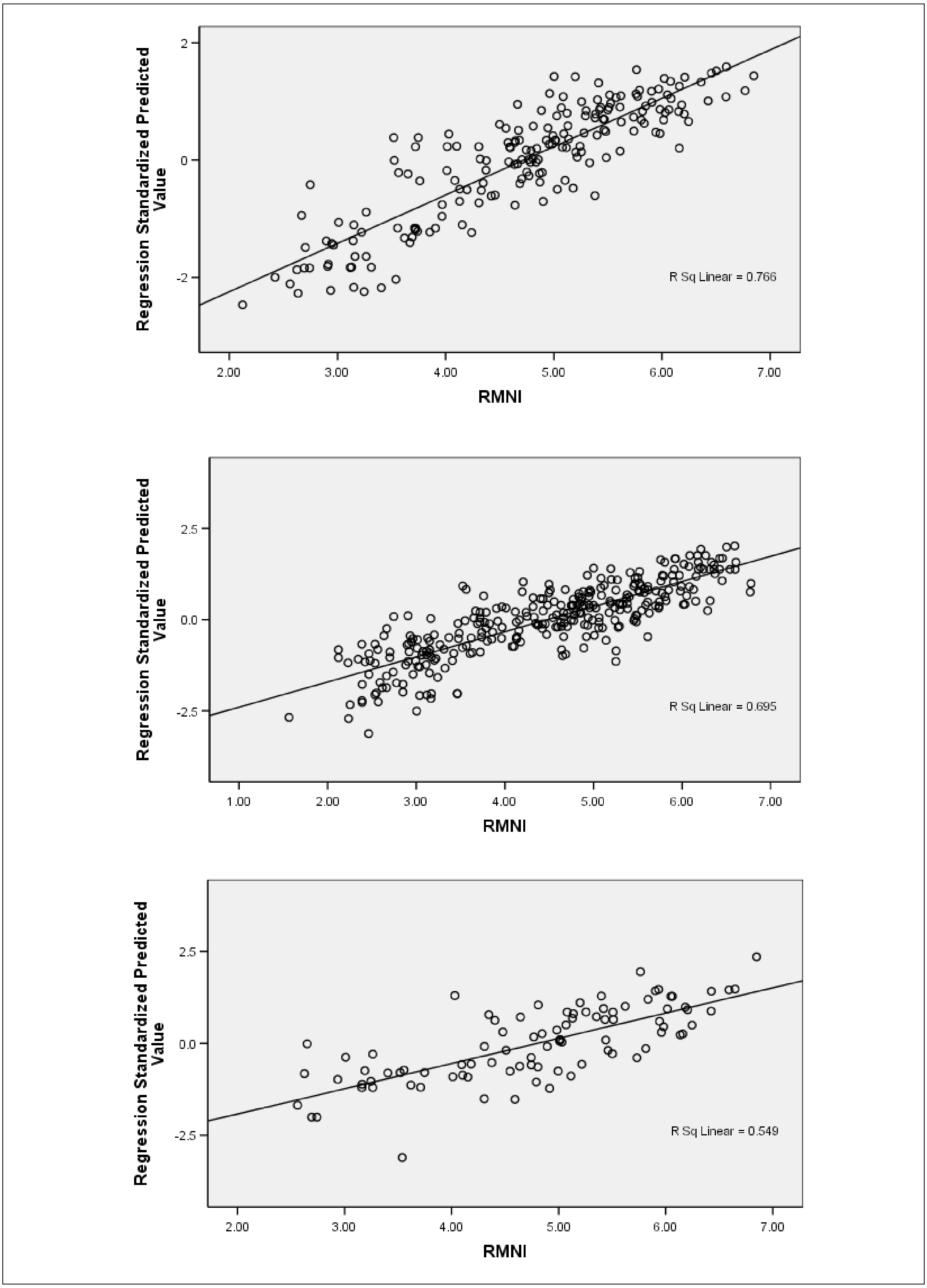


Figure 6.3 Application of predictive models for RMNI at reference sites. Upper: Preferred model for UK wherever alkalinity data available. Middle: Model for GB when no alkalinity data available. Lower: Model for IR when no alkalinity data available.

Table 6.1 Predictors and coefficients for models of site-specific reference RMNI values

Variable	Preferred model	Order	GB - missing alkalinity	Order	IR - missing alkalinity	Order
(Constant)	3.4272762		-10.66494765		-5.127209963	
Alkalinity (mg/l $CaCO_3$)	-0.004697663	4				
Alkalinity (mg/l $CaCO_3$) (Log_{10})	1.812963063	1				
Distance_to_source (km)			0.009342555	4		
Distance_to_source (km) ((Log_{10}x +1))	0.595211713	3				
Distance_to_source (km) (Log_{10})			0.640551548	7		
GB Easting (Log_{10})			1.615259357	2		
GB Northing			-3.5345E-06	3		
GB Northing (Log_{10})			2.031789861	6		
Height of source (m)					-0.001345665	3
IR Easting (Log_{10})					2.246652169	2
Site altitude (m)					-0.004142721	4
Slope (m/km)	0.010299229	6	0.003568621	8	0.015828875	5
Slope (m/km) (Log_{10}(x +1))	-0.80436531	2	-0.949883077	1	-1.386973944	1
Source altitude (m) (Log_{10})	-0.525017678	5	-1.394077868	5		
R	0.88		0.83		0.74	
R Square	0.77		0.69		0.55	
Adjusted R Square	0.76		0.69		0.52	
Std. Error of the Estimate	0.52		0.68		0.75	

All model terms significant at p = 0.01 after stepwise selection. The adj R^2 refers to the coefficient of determination and is equivalent to the variance in observed RMNI explained by RMNI predicted from environmental data. Models are ranked from left to right in decreasing order of desirability (decreasing model strength and increasing prediction error). Order refers to order of entry of each term into a stepwise regression model. IR = Northern Ireland/Ireland (Ecoregion 17).

The three models developed are summarised in Table 6.1 and illustrated in Figure 6.3. The preferred model requires data on alkalinity and should always be used for WFD classification purposes. Two alternative models were developed for GB and Ireland to enable classification of sites for which no reliable alkalinity value was available. The preferred model and alternative GB model perform well (and are comparable with the performance of the reference LMNI model for lakes), explaining 76 and 69 per cent respectively of the variation in reference RMNI. The preferred river model also requires fewer terms (six versus eight) than the preferred lake model. Prediction error compares less favourably with prediction of LMNI (0.52 versus 0.38 for preferred models for RMNI and LMNI respectively) and is noticeably poorer for alternative models.

RMNI is strongly dependent on alkalinity. Strong model performance is therefore guaranteed whenever alkalinity is used as a predictor of RMNI at reference sites. From a metric perspective, the advantage here is that covariation with alkalinity is largely removed at this stage because observed metric values are compared with the metric value expected for a given alkalinity. Consequently the deviation between observed and expected values should be purely a consequence of nutrient enrichment (or other covariable pressures). This point is discussed in detail in Section 7.8.

6.3.2 EQR calculation

The RMNI EQR is calculated as:

$$EQR = (O_s - O_m)/(E - O_m)$$

O_s = Observed site score
O_m= Maximum (worst achievable) score on scale
E = Expected score under reference conditions

In this case $O_m = 10$ since this is the maximum possible species score and would be the score observed at a site in which only the highest scoring taxa was present. However, basic geological constraints mean that this degree of degradation would not be appropriate to all river types (see Section 6.3.3 below). The value O_m is therefore calculated using the function:

IF $E \geq 4.765$ $O_m = 10$

IF $E < 4.765$ $O_m = 1.26 * E + 4.0$

Subtracting the theoretical maximum (worst) RMNI site score ensures that low RMNI scores achieve a high EQR. Therefore if the observed RMNI value for a site is six and the value expected at reference conditions, as predicted by a site-specific model is five, the EQR is:

$$EQR = (6 - 10)/(5 - 10) = 0.8$$

6.3.3 Placement of class boundaries for RMNI

The process followed to establish class boundaries for RMNI is simply an extension of the conceptual framework used to define biological reference conditions in Section 6. For each river type and using logistic regression, the RMNI score was determined equivalent to the flora being composed of 15, 35, 65 and 90 per cent tolerant responders. The basis for these thresholds is illustrated in Figure 5.1 and the rationale is explained in Table 5.1. At a value of 50 per cent tolerant responders, the vegetation is in equilibrium between positive and negative responders, a position considered to equate to the middle of moderate status. The prediction error between the relative proportions of tolerant species and RMNI is typically close to 0.15 in the linear phase of the relationship and fixed values of 35 and 65 per cent respectively are therefore used to define the good-moderate and moderate-poor boundaries. Essentially at these thresholds there is a low or high probability respectively that the cover of tolerant species will exceed that of sensitive species. The RMNI score associated with these thresholds was then related to the modelled RMNI score associated with a vegetation composed of seven per cent tolerant species, this being taken as the mid-point of the population of reference sites when the high-good boundary lies at 15 per cent. In several cases the type-specific reference RMNI was derived using this approach when no or very few sites were available within a type that met the required standard.

On the basis of the above analysis, provisional class boundaries for the RMNI metric emerge as 0.86, 0.73, 0.6 and 0.47 based on a weighted average of class boundary EQR values across all river types, where the value for each type is weighted by the number of contributing sites. The lowering of EQRs at class boundaries relative to a draft single metric classification of UK rivers reflects the re-scoring of some species, the splitting of several river types to discriminate between northern and southern variants, and the inclusion of a standardisation step for classifying the functional response groups, as discussed in Section 5.4.1. This more ecologically-focused approach to deriving boundaries is somewhat in contrast to the more commonly adopted approach of taking some lower percentile of the distribution of reference site EQR values and using the difference between this and one and the basis for subsequent class boundaries. Thus, in RIVPACS, a fifth percentile EQR of 0.89 in reference sites translates to class boundaries at 0.11 intervals (0.78, 0.67 and 0.56).

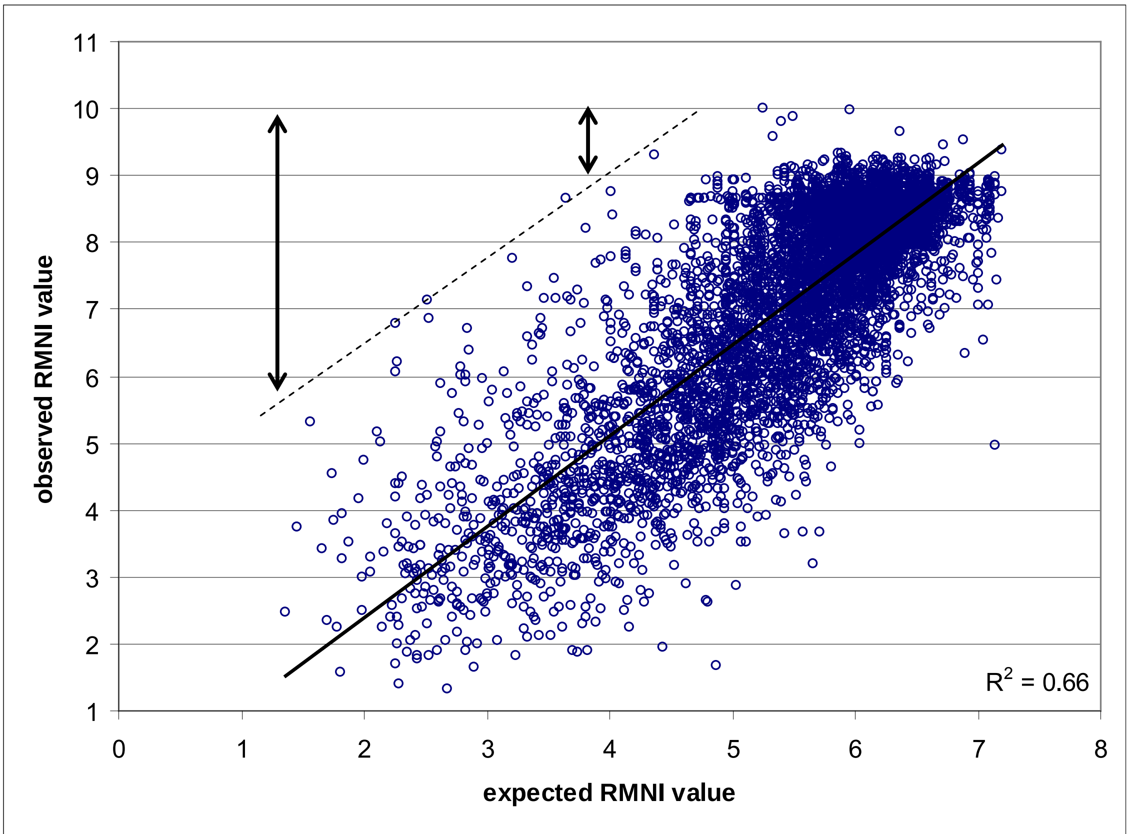


Figure 6.4 Relationship between observed and expected values for RMNI. Arrows indicate difference in 'height to ceiling' values at different points on the expected RMNI gradient. This difference will influence the EQR calculated. The dashed line tracks the worst practicably achievable observed RMNI values for a given expected value.

Closer inspection of the approach used to calculate EQR values suggests this simple method is flawed when the compositional metric is constrained to a fixed range (one to 10 in this instance). This is because the distance between the 'ceiling' value and the observed value changes along the gradient of the metric, providing greater 'room for manoeuvre' at low metric values. A simple plot (Figure 6.4) of observed versus expected metric values for the metric RMNI makes this apparent. One consequence of this is that the EQR value at a class boundary, when set on the basis of change in the relative proportion of different response groups, can vary quite widely (from 0.95 in low-alkalinity, high-gradient streams, to 0.85 in very high-alkalinity, low-gradient rivers for example). Any kind of average of these values would result in a level of deviation too relaxed for some types and too stringent for others to equate to normative definitions. Similarly, deriving the high-good class boundary value based on the percentile distribution of reference EQR values calculated using the simple approach, may be flawed since any population of contemporary reference sites is likely to be heavily biased towards less disturbed low-alkalinity upland streams. An alternative method for calculating the EQR which circumvents this problem is to use the worst practicably achievable metric value for a given site, rather than the overall maximum on the scale. Thus, as is evident from Figure 6.4, a value of 10 remains suitable when the expected RMNI value exceeds 4.8. However, when the expected value is as low as three it is virtually impossible for the observed value to exceed eight, even in the most degraded site, due to basic geological constraints on the level of degradation possible in low-alkalinity upland rivers, for which an expected RMNI of three is likely to be appropriate.

Fitting a line to enclose the uppermost observed RMNI values for a given expected value provides a function for calculating O_m when E is known. This indicates that the maximum possible value of 10 is valid when E exceeds 4.76 but at lower values of E the value of O_m decreases from 10 to around six, following the line $O_m = 1.26 * E + 4.0$. Table 6.2 shows the effect of this adjustment on the calculation of EQRs. Thus, when E is three and six and O_s is 4.18 and seven the associated EQR values are 0.83 and 0.75 respectively when a fixed upper value of 10 is used for O_m. Although both sites would be classified as good status on the basis of this metric the difference of 0.08 EQR units is equivalent to more than half a class. When a calculated value of O_m is used based on the maximum achievable, the EQR for both sites is the same (EQR = 0.75)

Table 6.2 Illustration of effect of adjusting maximum achievable value of RMNI on the calculated EQR

E	O_s	O_m	EQR	O_m (adj)	EQR
3.00	4.18	10.00	0.83	7.72	0.75
6.00	7.00	10.00	0.75	10.00	0.75

E = Expected score under reference conditions; O_s = Observed site score; Om = Maximum (worst achievable) score on scale; O_m (adj) = worst achievable score relative to expected value.

By following the same ecological rationale for class boundary placement, and repeating the calculation of the type-specific EQR value associated with each class boundary when the adjusted O_m is used in the calculation, a revised set of boundaries of 0.85, 0.7, 0.55 and 0.4 are obtained (Table 6.3). The main effect of these boundaries compared to the provisional version will be to slightly reduce the proportion of high status sites when expected RMNI values are low (offering more discrimination in lower alkalinity upland rivers), and to slightly reduce the proportion of lower status sites when expected RMNI values are high (providing a more equitable distribution of sites over mid and lower classes). By examining the distribution of site-specific EQR values for RMNI (Figure 6.5) one can put our approach into context. The fifth and tenth percentile of this distribution corresponds to 0.80 and 0.83 respectively. An EQR of 0.85 corresponds to the 12th percentile and represents a relatively precautionary position.

Table 6.3 Summary of class boundary values for river metrics

Class/ Boundary	Metric					normalised scale
	RMNI	HYDR	NFG	NTAXA	ALG_COV	
Ref	1	1	1	1	1	1
H-G	0.85	0.85	0.8	0.8	0.99	0.8
G-M	0.70	0.70	0.6	0.6	0.95	0.6
M-P	0.55	0.55	0.4	0.4	0.91	0.4
P-B	0.40	0.40	0.2	0.2	0.82	0.2

Reference refers to an EQR of one. H-G = high-good boundary, etc.

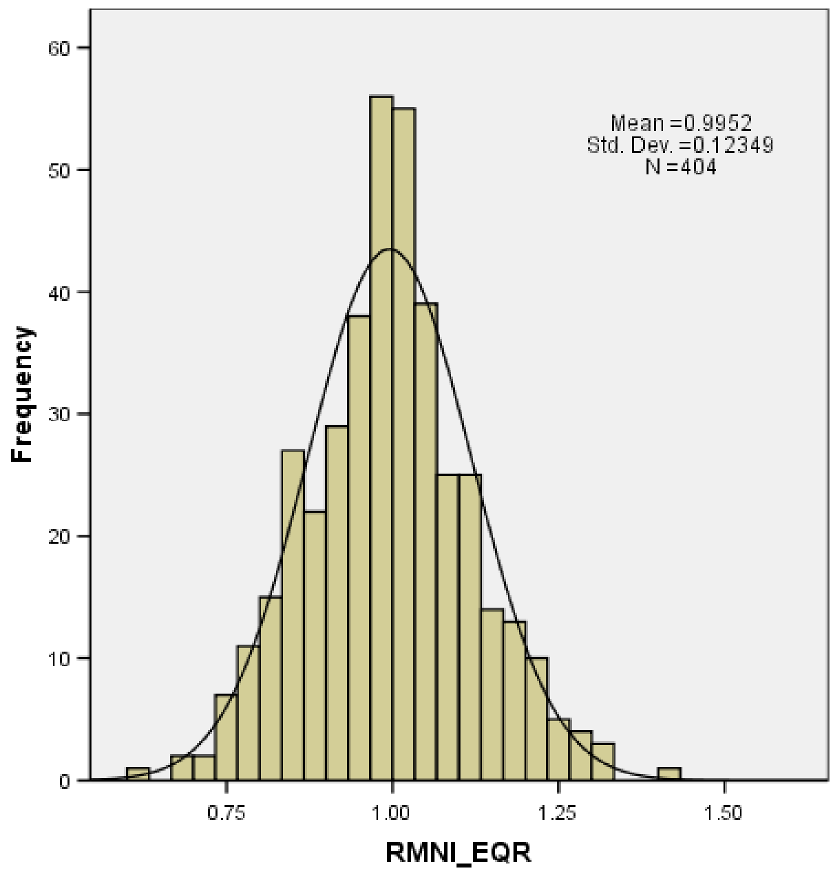


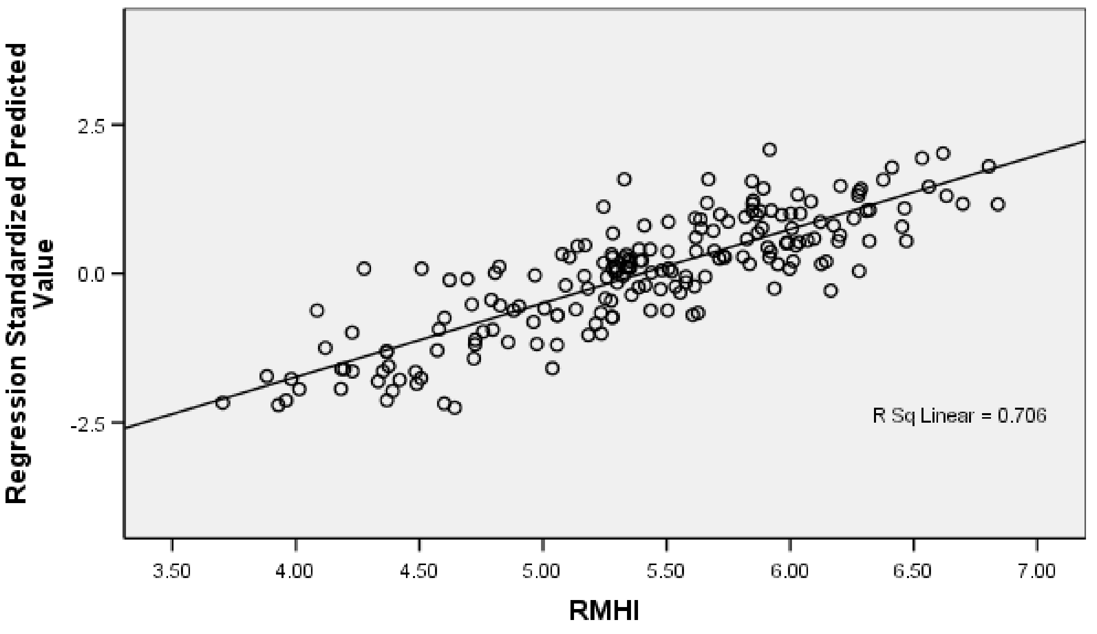
Figure 6.5 Distribution of site-specific reference EQR values for the RMNI. The fifth and tenth percentiles of the EQR distribution lie at 0.80 and 0.83 respectively.

6.4 River Macrophyte Hydraulic Index

6.4.1 Prediction

RMHI values associated with the population of 404 reference sites were used as the basis for developing models to predict site-specific reference values. The set of intrinsic environmental data associated with these sites was used as predictors. As with RMNI, RMHI was significantly correlated with alkalinity, slope and distance from source. The same stepwise multiple regression procedure was used to identify the most parsimonious models for predicting RMHI. To accommodate the different scenarios of environmental data availability, a number of models were developed. Model performance was assessed based on the percentage variance in RMHI explained by the combination of environmental variables (squared correlation between observed and predicted values), standard error of prediction, and slope and intercept of the relationship between expected and observed values.

The three models developed are summarised in Table 6.4 and illustrated in Figure 6.6. The preferred model requires data on alkalinity and should always be used for WFD classification purposes. Two alternative models were developed for GB and Ireland to enable classification of sites for which no reliable alkalinity was available. The preferred model and alternative GB model perform well (and are comparable with performance of the reference RMNI model), both explaining 70 per cent of the variation in reference RMHI. The prediction error for all RMHI models is superior to that for RMNI (about ±0.4).



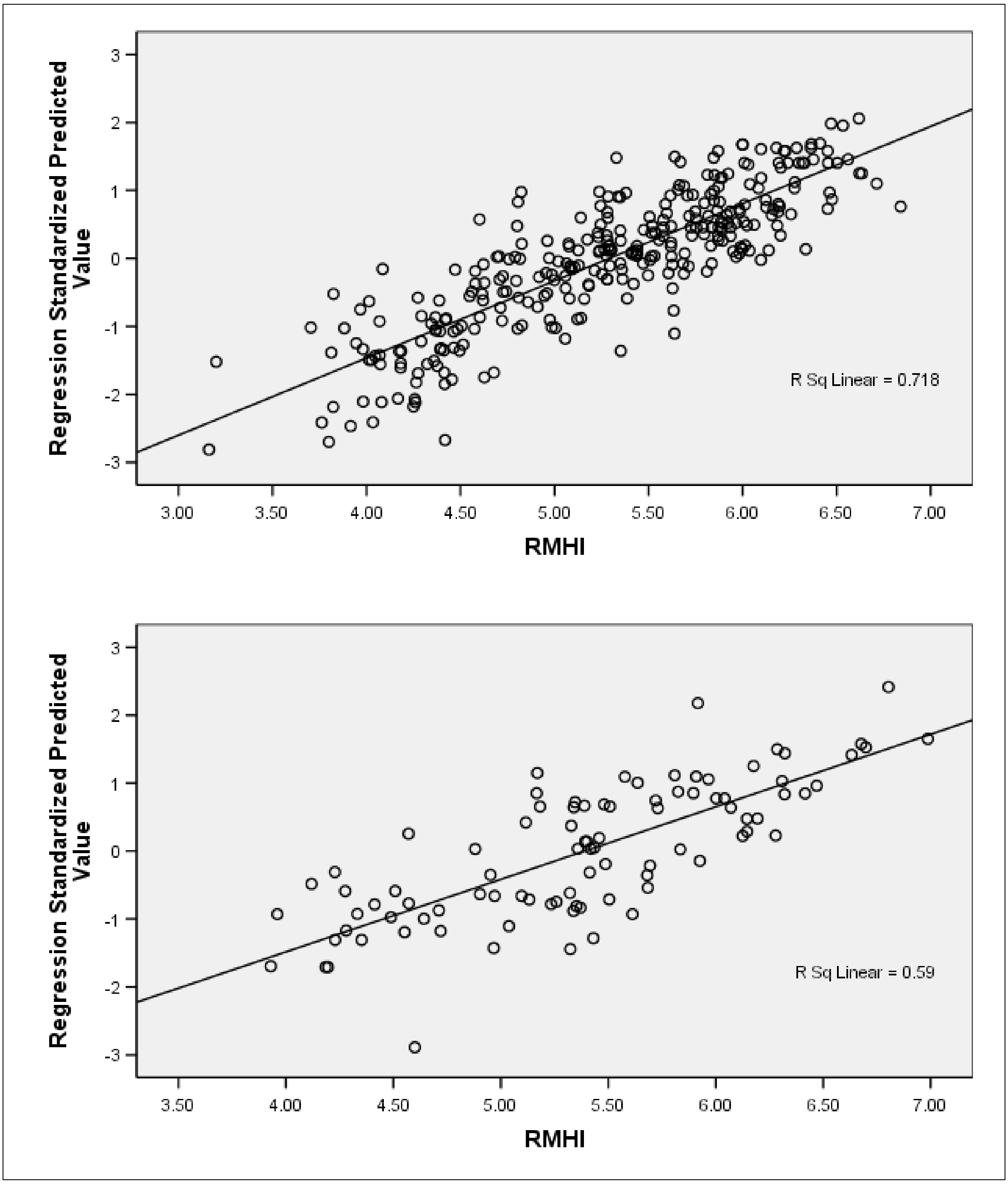


Figure 6.6 Application of predictive models for RMHI at reference sites. Upper: Preferred model for UK wherever alkalinity data available (n=192). Middle: Model for GB when no alkalinity data available (n=312). Lower: Model for IR when no alkalinity data available (n=92).

Table 6.4 Predictors and coefficients for models of site-specific reference RMHI values

Variable	Preferred model	Order	GB - missing alkalinity	Order	IR - missing alkalinity	Order
(Constant)	6.558330198		3.449711415		0.228522195	
Alkalinity (mg/l $CaCO_3$) (Log_{10})	0.664861516	1				
Distance_to_source (km)	0.005868304	3	0.008669079	4		
GB Easting			7.50176E-07	2		
GB Northing			0.957343121	6		
GB Northing (Log_{10})			-1.60217E-06	3		
Height_of_Source (m)					-0.000902232	2
Irish Easting (Log_{10})					1.255430324	3
Site altitude (m)					-0.00234372	4
Slope (m/km)	0.006709381	5	0.002760384	7	0.011873231	5
Slope (m/km) (Log_{10}(x +1))	-0.85111236	2	-1.012019662	1	-1.141762054	1
Source altitude (m) (Log_{10})	-0.640378719	4	-0.913341586	5		
R	0.840198085		0.847164605		0.767908276	
R Square	0.705932822		0.717687868		0.589683121	
Adjusted R Square	0.698027791		0.711187259		0.565827489	
Std. Error of the Estimate	0.372009942		0.400769171		0.473808419	

All model terms significant at p = 0.01 after stepwise selection. The adj R^2 refers to the coefficient of determination and is equivalent to the variance in observed RMHI explained by RMHI predicted from environmental data. Models are ranked from left to right in decreasing order of desirability (decreasing model strength and increasing prediction error). Order refers to order of entry of each term into a stepwise regression. IR = Northern Ireland/Ireland (Ecoregion 17).

6.4.2 EQR calculation

The RMNI EQR is calculated using the same method as for RMNI. Therefore:

$EQR = (O_s - O_m)/(E - O_m)$
O_s = Observed site score
O_m= Maximum (worst achievable) score on scale
E = Expected score under reference conditions

In this case $O_m = 10$ since this is the maximum possible species score and would be the score observed at a site in which only the highest scoring taxa was present. However, basic geological constraints mean that this degree of degradation would not be appropriate to all river types (see Section 6.3.3). The value O_m is therefore calculated using the function:

IF $E \geq 6.7$ $O_m = 10$

IF $E < 6.7$ $O_m = 1.18 * E + 2.1$

Subtracting the theoretical maximum (worst) RMHI site score ensures that low RMHI scores achieve a high EQR. Therefore, if the observed RMHI value for a site is six and the value expected at reference conditions, as predicted by a site-specific model is five, the EQR is:

$EQR = (6 -10)/(5 -10) = 0.8$

6.4.3 Placement of class boundaries for RMHI

The process for establishing class boundaries based on RMNI is computationally laborious. Since the percentile distribution of RMHI EQR values in reference sites matched, almost perfectly, that of RMNI, and because RMNI and RMHI are strongly correlated, the high-good boundary for RMHI was simply set at the 12th percentile of the reference EQR distribution (Figure 6.7). This was found to equate to the high-good EQR for RMNI when established via the type-specific logistic regression model route. Thus, the high-good boundary for RMHI is 0.85 with subsequent class boundaries following those already employed for RMNI (0.7, 0.55 and 0.40, Table 6.3).

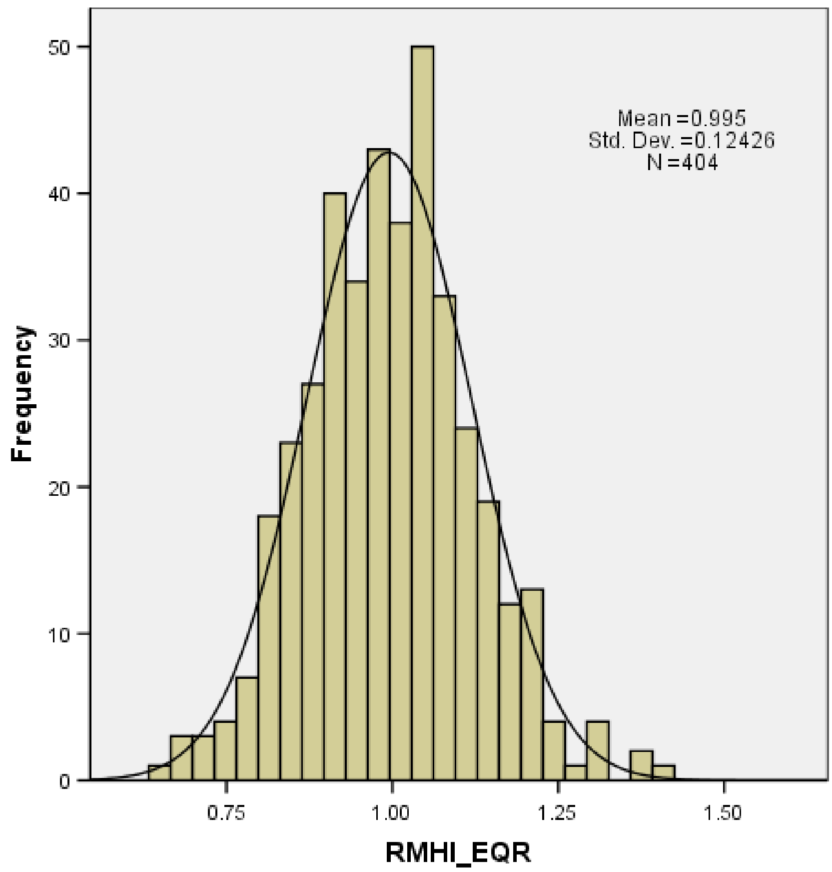


Figure 6.7 Distribution of site-specific reference EQR values for the metric RMHI. The fifth and tenth percentiles lie at 0.80 and 0.83 respectively.

6.5 Richness metrics

6.5.1 Prediction

Both richness metrics were modelled by stepwise linear regression from the values of N_TAXA or N_FG in the population of 404 reference condition surveys, applying the same suite of environmental variables as were used to model compositional metrics. To allow for the fact that reference condition surveys were contributed by two different survey methods that would have an obvious effect on the richness recorded, the population of reference sites was split between those contributed by MTR surveys (100-m reach) and those contributed by JNCC surveys (1,000 m reach). The following account includes the necessary models to use if applying a classification to JNCC survey data, since these were used to enable a full classification of surveys in the project database. For future classification purposes, only the 100-m specific model must be used.

Because N_TAXA and N_FG are unconstrained or vary over a much wider range than metrics such as RMNI, a small number of reference sites can attain very high EQR values (2-2.5). This is not desirable since it amplifies the variability in reference EQR and therefore restricts the utility of the metric for classification purposes. The quality of models based on untransformed values are also weak, meaning that the distribution of resulting EQR values would imply setting the H-G boundary so low as to render these metrics useless for classification purposes. Modelling of log transformed N_TAXA and N_FG, while only marginally improving model performance, was considered preferable, since it resulted in a reduced standard deviation, and effectively reduced the influence of unusually taxa- or FG-rich sites, thereby keeping EQR values within the range of those found for other metrics.

Models for expected N_TAXA and N_FG at reference sites are detailed in Table 6.5. Generally richness metrics could be modelled slightly more effectively in rivers than in lakes, while data from JNCC surveys could be modelled more effectively than that contributed by MTR surveys. Although a rationale exists for the use of richness metrics, in common with other biota there are major difficulties associated with the prediction of richness, which suggest there are fundamental problems with the prediction methods or the supporting datasets. For example, within RIVPACS the prediction of numbers of families is substantially poorer than for ASPT (Moss *et al*., 1999), leading to correspondingly wide class boundaries for this metric (Clarke *et al*., 1996). Walley & Fontama (1998) found that the use of a back propagation neural network hardly improved the prediction of the number of families compared to RIVPACS (although the bias was reduced) and advised that it would be unwise to build a system in which prediction of richness was an integral component. They suggested that variation in sample effort contributed significant noise to the reference dataset. Surprisingly, in our analysis there is little evidence that using a substitute metric, such as N_FG which should be less sensitive to survey effort, actually reduces prediction error.

Table 6.5 Predictors and coefficients for site-specific models of richness metrics

Variable	N_FG								N_TAXA			
	MTR-IR		JNCC-IR		MTR_GB		JNCC_GB		MTR - global		JNCC - global	
	Coefficient	Order	Coefficient	Order	Coefficient	Order	Coefficient	Order	Coefficient	Order	Coefficient	Order
(Constant)	0.813705		0.9731348		0.7850636		1.5030752		0.5485448		0.7180155	
Distance to source (km)			0.0008793	2			0.0008144	5				
Distance to source (km) (Log_{10} (x+1))											0.2274409	1
GB Easting							-4.25E-07	2				
GB Northing					1.434E-07	2						
GB Northing (Log_{10})							-0.066129	4				
Site altitude (m) (Log_{10}(x + 1))											0.056194	2
Slope (m/km)			0.0006427	3			0.0006196	3				
Slope (m/km) (Log_{10} (x + 1))	-0.1226553	1	-0.1645161	1	-0.1624955	1	-0.179607	1	-0.1586006	1		
Source altitude (Log_{10})									0.1724771	2		
R	0.43		0.59		0.54		0.65		0.51		0.61	
R Square	0.18		0.35		0.29		0.42		0.26		0.37	
Adjusted R Square	0.18		0.34		0.27		0.41		0.25		0.36	
Std. Error of the Estimate	0.13		0.11		0.12		0.10		0.15		0.13	

All model terms significant at $p = 0.01$ after stepwise selection. The adj R^2 refers to the coefficient of determination and is equivalent to the variance in the observed reference metric value explained by the reference metric values predicted from environmental data. Models refer to different survey methods and regions of application. MTR = 100-m survey reach. JNCC = 1,000-m survey reach. GB = Great Britain IR = Ireland.

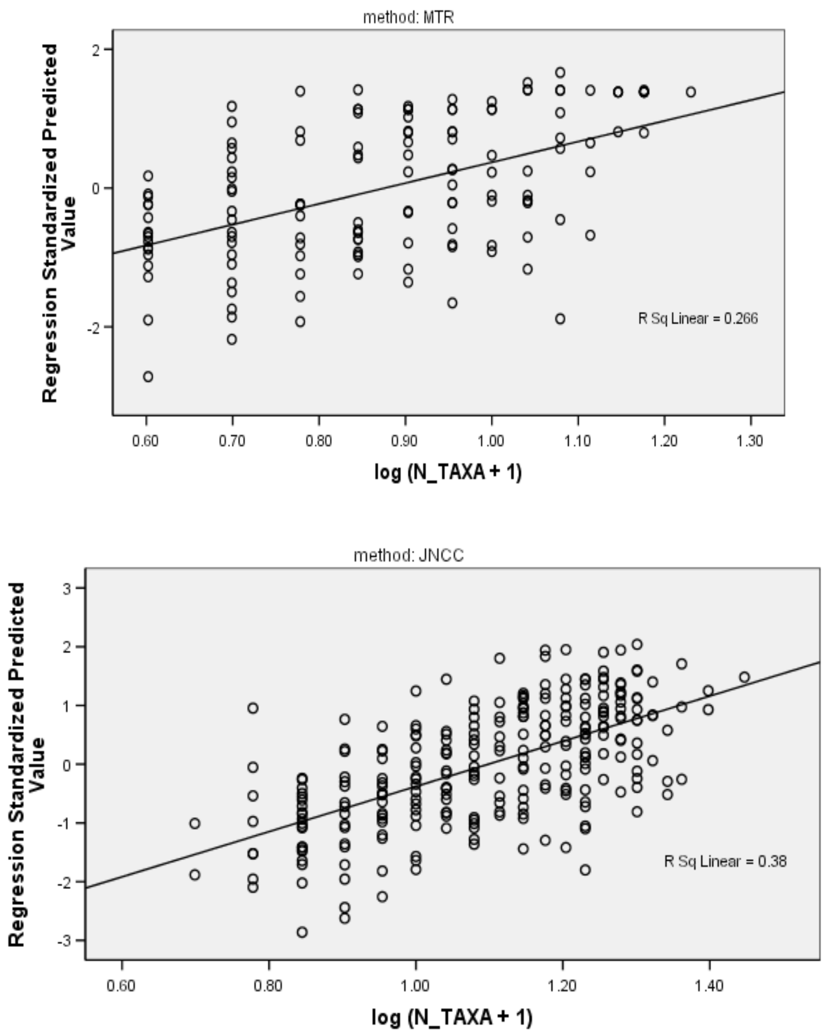


Figure 6.8 Predicted versus observed values for log transformed N_TAXA in reference sites. Left panel, model to use with MTR data. Right panel; model to use with JNCC data.

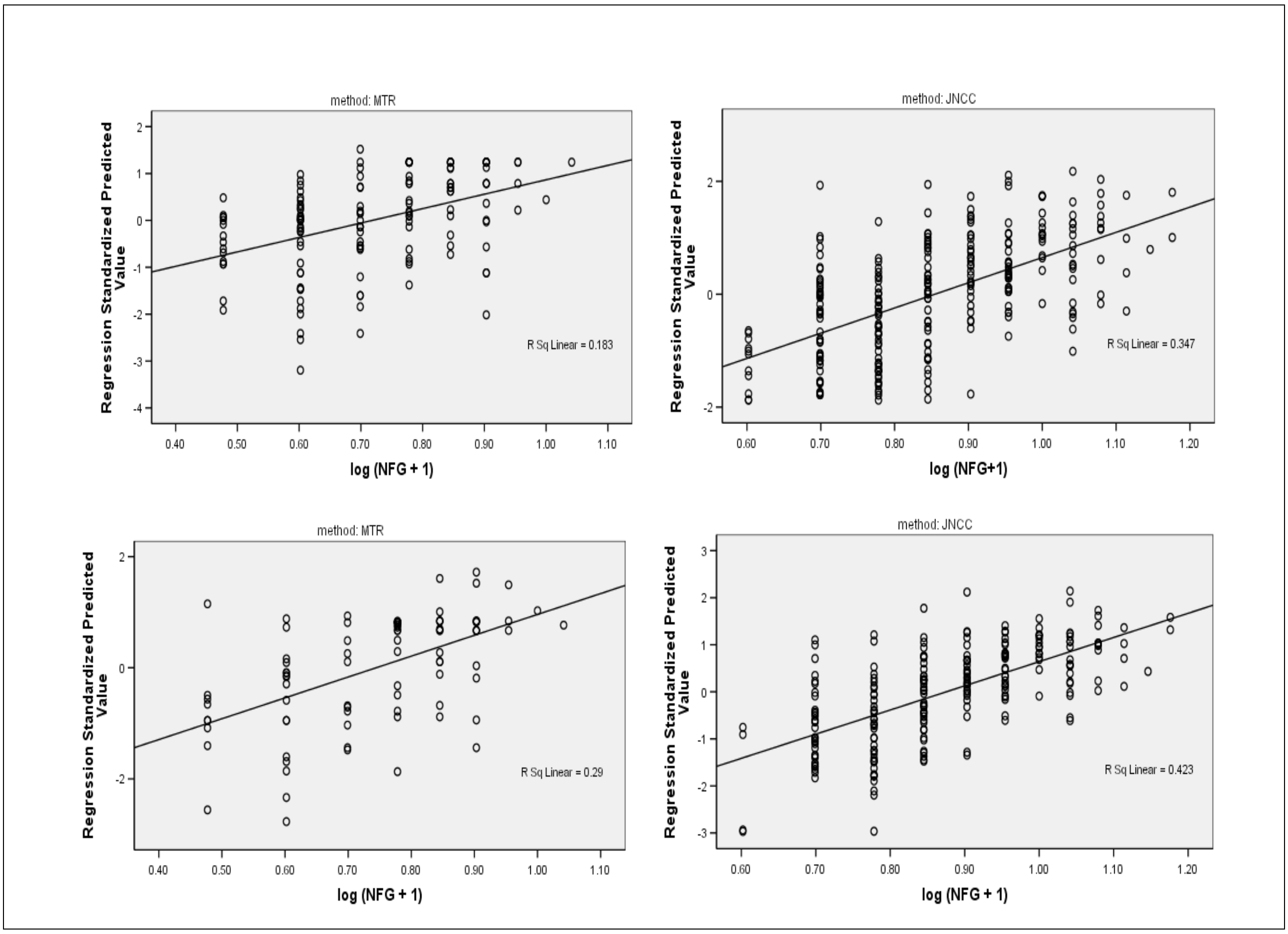


Figure 6.9 Predicted versus observed values for log transformed N_FG in reference sites. Upper panels, model to use in IR with MTR data (left) and JNCC data (right). Lower panels, model to use in GB with MTR data (left) and JNCC data (right).

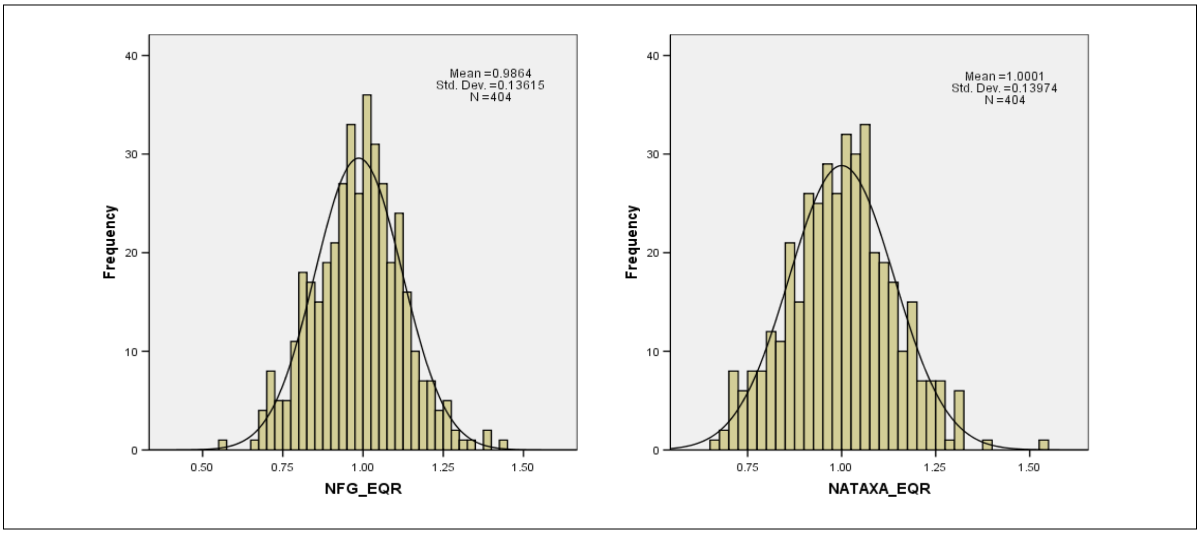


Figure 6.10 Frequency distribution for reference site EQRs for NFG (left) and N_TAXA (right). The 15th percentile of each distribution lies at around 0.85.

6.5.2 Comparative value of different richness metrics

There is relatively little to chose between N_TAXA and N_FG as a richness metric. N_FG requires allocation of species to functional groups but has a number of advantages, ranging from a significantly lower skew in the distribution of EQRs (as illustrated in the frequency distribution histograms), to less bias due to variation in sample effort and taxonomic resolution in recording, plus a clearer overall ecological rationale. On these grounds N_FG might be the preferred richness metric. High N_FG EQR relative to N_TAXA EQR might be interpreted as evidence of high physical habitat heterogeneity in more dynamic sites or efficient resource partitioning in more fertile sites. Meanwhile high N_TAXA EQR relative to N_FG EQR might be interpreted as evidence of high functional redundancy (more taxa per FG) which may be associated with less physically diverse sites with high connectivity to upstream or floodplain propagule sources.

6.5.3 EQR calculation

The models for both richness metrics were generated from $\log_{10}$ x +1 transformed values and the EQR is therefore calculated as $\log_{10}$ (O+1) /E, where O is the observed value of N_TAXA or N_FG and E is the expected value expressed in $\log_{10}$ x +1 terms. The (x+1) term is introduced simply to accommodate sites supporting only one species for which the log value is zero.

Thus, if the observed N_TAXA is eight and the predicted metric value (in $\log_{10}$ (x + 1) terms) is 1.04 the N_TAXA EQR is calculated as:

$$EQR = \log_{10} (8+1)/1.04 = 0.92$$

If the absolute value of the Expected value is required for comparative purposes, this can be obtained simply by calculating the exponent of the value predicted from the environmental variables in Table 6.5, and then subtracting one.

6.5.4 Placement of class boundaries for richness metrics

The H-G boundary for both metrics, based on the 15th percentile of the EQR frequency distribution was 0.85 (Figure 6.10). The rationale for the placement of class boundaries for these metrics is statistical rather than ecological. Thus, the boundaries are imposed using the percentile distribution of EQRs in reference sites (0.85 for H/G with subsequent boundaries at 0.15 intervals). The class boundaries or weight subsequently given to this metric must reflect the underlying variability in the reference site model and there are clearly many factors unaccounted for (such as catchment isolation, size of catchment species pool, degree of lateral connectivity with floodplain habitats, presence of upstream online lakes, survey effort or surveyor expertise) that could contribute to variation in either metric. Indeed, there is no evidence that the practical arguments in favour of N_FG as a metric lead to superior prediction. Translating the EQR thresholds applied, sites at the H-G boundary sites will support on average 70 per cent of the expected N_FG or N_TAXA and 50 per cent at the G-M boundary. It is reasonable to suppose that macrophyte-dependent functions in rivers will be impaired if N_FG or N_TAXA falls below these thresholds, or there will be strong evidence for constrained variation in physical habitat. Alternatively, if there are no grounds for suspecting an impact when richness EQRs are returned below the G-M boundary (less than half the expected N_FG or N_TAXA), it is possible that a site has been under-sampled and the quality of the survey or surveyors should be scrutinised.

6.6 Abundance metrics

6.6.1 Prediction

Modelling abundance metrics presents considerable problems because there is naturally high variation in abundance measures that cannot be accounted for by the suite of available environmental predictors. Using the same environmental variables as the models for composition and richness metrics, it was possible to explain only five to ten per cent of the variation in selected abundance metrics at reference sites. This could be improved marginally by introducing highly site-specific measures such as depth, shading and substrate size, but these create a need for additional data collection, as well as raising the question of whether such local variables can be regarded as being genuinely unimpacted. Explanation of variation in macrophyte cover in rivers is largely dependent on information on flow variability (Riis *et al*., 2008; Franklin *et al*., 2008), especially factors such as the accrual period (time between sampling and most recent flow event of a sufficient magnitude to reduce cover through scouring). Technically, it would be possible to retrieve this information for most rivers from gauging station data and thence derive accrual periods from the date of macrophyte survey. However, this would be an onerous task and would greatly increase the effort needed in future to model metric values under reference conditions.

An alternative approach may be required which takes account of how likely abundance metrics will contribute to classification. Firstly, the evidence suggests that abundance metrics are likely to be of limited use in rivers of low to moderate fertility. Secondly, the focus in abundance metrics is on recognising *excessive* growth of macrophytes that may contribute to undesirable disturbances, rather than *explaining* variation in macrophyte and algal cover. Consequently it is more important to identify the upper capacity of abundance metrics in different rivers at reference conditions under scenarios that are most favourable to macrophyte growth (such as long accrual period, stable, well lit bed). It is then not necessary to explain all the deviations in abundance *below* these values since these will occur naturally due to a range of environmental factors, such as high water velocities, scouring, shading, low bed stability, lack of suitable rooting medium. However, metric values increasingly in *excess* of the carrying capacity under reference conditions should contribute to a drop in ecological status.

Several potential abundance metrics were identified in Chapter 4. Of these, absolute cover of filamentous algae had the strongest relationship with pressure variables and was less strongly intercorrelated with other metrics. Various pressures might lead to an increase in the overall cover of hydrophytes or average cover per species but these appear to already be picked up existing compositional and richness metrics. Algal cover is different since modest amounts may not adversely affect the diversity of other species, while its influence on compositional metrics will only be significant when the cover of other species is small. This may occur whether the absolute cover of filamentous algae is high or low. Consequently, the scenario of a large absolute cover of filamentous algae may not be adequately registered by the compositional and richness metrics. Since filamentous algal cover increases with nutrient enrichment, sites with a large algal cover may be overclassified.

Having been unable to model algal cover satisfactorily using environmental variables, the median cover of algal cover in reference sites (0.05 per cent) was used as the basis for the calculation of the EQR. The HG boundary value of 2.5 per cent reflects the fact that this appears, from empirical data (Section 4), to be the highest baseline cover that it is not obviously elevated as a result of greater nutrient inputs.

6.6.2 EQR calculation

The EQR for ALG COV is calculated as:

EQR = (Obs ALG COV-100)/(0.05-100)

Subtracting 100 ensures that low algal cover achieves a high EQR. In the unlikely event that derived algal cover exceeds 100, the value is automatically capped at 100, thus producing an EQR of zero.

6.6.3 Placement of class boundaries for algal cover

The rationale for the placement of class boundaries for this metric is ecological rather than statistical and takes direct account of changes in percentage algal cover in relation to nutrient concentrations (Figure 4.30). The boundaries used are given in Table 6.6. In deriving a value for the cover of filamentous algae, unless this has been assessed independently on a percentage scale, the value obtained depends on transforming cover score data to a percentage scale. Discontinuities within the cover scoring system will impose discontinuities on the distribution of inferred percentage algal cover values. The class boundaries take these discontinuities into account. Thus, for example, MTR cover scores of six and seven have percentage cover equivalents of 17.5 and 37.5 per cent.

Table 6.6 Absolute percent cover of green filamentous algae in rivers at class boundaries

	ref	H-G	G-M	M-P	P-B	max
% cover	0.05	2.5	7.5	17.5	37.5	100

Reference refers to an EQR of one. H-G = high-good boundary, etc.

6.7 Harmonising metric EQRs

The EQR for each of five metrics (RMNI, RMHI, N_TAXA, N_FG, and ALG_COV) is determined for each survey based on the approach described above. A full worked example is given in Section 6.8. Although each metric is expressed on a numeric scale, and the EQR represents the level of distortion for that metric from reference condition, the different EQRs are not all themselves directly comparable in this format because they may be scaled over different ranges (such as differences in absolute minimum EQR) and because class boundaries lie in different places. Thus, for example, a value of 0.85 on the RMNI EQR scale (H-G boundary) cannot be compared directly with a value of 0.85 on the ALG_COV EQR (approximate middle of moderate status). Consequently a translation scheme is required to map all EQRs onto a common class boundary system (Table 6.7) before combining metrics based on their EQR, or identifying metrics in which the EQR is high or low relative to other metrics. (Note that this becomes redundant once metrics are expressed in terms of percentage confidence of class). The only simple alternative to this approach would be to assign each metric to a status band, recode these to a simple numeric scale (such as 1=bad to 5=high) and take the average or minima of these numbers. However, this approach is crude and would result in the loss of information (such as whether an EQR is above or below a class boundary).

In the approach followed here all metrics are harmonised to a standardised EQR range of zero to one with class boundaries at intervals of 0.2 EQR units. For compositional and richness metrics, class boundaries are synchronous and a single transformation is adequate. This can be achieved via a simple equation. In the case of the metric ALG, a piece wise linear transformation was applied which effectively stretches or compresses each class over the range of 0.2. EQR units. The advantage of harmonising metrics prior to their combination into a final EQR for the site is that it avoids the need for any subsequent transformation of the final EQR value since this is already scaled from zero to one with class boundaries at intervals of 0.2.

Selecting approaches to combine harmonised metrics is discussed in Section 7.

Table 6.7 Equations for harmonising metrics to a common class boundary system based on intervals of 0.2 units. After use of this rescaling it is legitimate to average across metric EQR values and so on.

Metric	Rescaling function
RMNI, RMHI, N_FG or N_TAXA	=IF(EQR>1,1,IF(EQR<0.251,0,(1.333*EQR-0.333)))
ALG_COV	=IF(EQR>=1,1,IF(EQR>=0.990495,((EQR-0.990495)/(1-0.990495))*0.2+0.8,IF(EQR>=0.950475,((EQR-0.950475)/(0.990495-0.950475))*0.2+0.6,IF(EQR>=0.910455,((EQR-0.910455)/(0.950475-0.910455))*0.2+0.4,IF(EQR>=0.82041,((EQR-0.82041)/(0.910455-0.82041))*0.2+0.2,IF(EQR<0.82041,(EQR/0.82041)*0.2))))))

6.8 Worked example

This section considers survey data for a fictional site and shows how metric values, EQRs and adjusted (harmonised) EQRs would be calculated. The process of achieving a final EQR for the water body based on macrophytes is discussed in Section 7.

Table 6.8 shows the macrophyte data collected during a standard 100-m survey of a very high-alkalinity, lowland river in southern England. Table 6.9 shows environmental data for the site required to predict reference metric values.

Table 6.8 Summary of macrophyte survey data used in worked example

Taxa	Cover score	RMNI	RMHI	A_TAXA	FG	ALG_COV
Apium nodiflorum	3	8.64	8.08	1	8	
Callitriche obtusangula	1	8.04	7.98	1	6	
Cladophora glomerata	4	7.50	6.84	1	19	3.8
Fontinalis antipyretica	2	5.40	5.95	1	21	
Hildenbrandia rivularis	4	6.03	6.07	1	20	
Lemna minor	2	8.80	8.59	1	1	
Oenanthe crocata	1	6.22	6.48	1	8	
Phalaris arundinacea	6	7.52	7.24			
Phragmites australis	3	7.70	8.94			
Ranunculus fluitans	8	7.97	7.44	1	18	
Rumex hydrolapathum	1	8.65	8.11			
Site		**7.52**	**7.35**	**8**	**7**	**3.8**

Grey cells indicate primary survey data. White cells indicate data used to calculate metric values

Table 6.9 Environmental data for site used as basis for worked example

Variable	unit	value
Site altitude	m	45
Slope	m/km	0.9
Distance to source	km	58.5
Altitude of source	m	140
Alkalinity	mg/l $CaCO_3$	217
GB Northing	m	128728

6.8.1 Determine the RMNI EQR using the RMNI index

The **observed RMNI** score for the river is the cover-weighted average of the ranks for the individual species (see Table 6.8):

Observed RMNI =

$$\frac{(3*8.64+1*8.04+4*7.50+2*5.4+4*6.03+2*8.8+1*6.22+6*7.52+3*7.7+8*7.97+1*8.65)}{3+1+4+2+4+2+1+6+3+8+1} = 7.52$$

The **expected RMNI** score for this site under reference condition is calculated using an equation derived from multiple regression with variables from UK reference rivers including: alkalinity, altitude, altitude of source, slope and distance of site from source.

Thus:

Expected RMNI = 3.427276+1.812963* Log_{10} (alkalinity+1) + -0.80437* Log_{10} (slope +1) +0.595212* Log_{10} (distance from source +1)+ -0.0047*alkalinity+ -0.52502* Log_{10}(source altitude+1)+0.010299*slope

This results in an expected RMNI score of 6.36.

The EQR for this river using the RMNI metric alone is therefore:

$$\text{RMNI EQR} = \frac{(7.52-10)}{(6.36-10)} = 0.68$$

Both the observed and the expected values have 10 subtracted from them. This reflects the theoretical maximum (worst) RMNI site score that can be achieved in a given river type (in this case 10), and ensures that low RMNI scores achieve a high EQR. The river status based on this metric alone would be moderate (RMNI EQR of between 0.55 and 0.70).

Since RMNI EQR lies between 0.25 and one the adjusted RMNI EQR is obtained from:

adj RMNI EQR= 1.333*RMNI EQR -0.333 = 0.57

6.8.2 Determine the RMHI EQR using the RMHI index

The **observed RMHI** score for the river is the cover-weighted average of the ranks for the individual species (see Table 6.8):

Observed RMHI =

$$\frac{(3*8.08+1*7.98+4*6.84+2*5.95+4*6.07+2*8.59+1*6.48+6*7.24+3*8.94+8*7.44+1*8.11)}{3+1+4+2+4+2+1+6+3+8+1}=7.35$$

The **expected RMHI** score for this site under reference condition is again calculated using an equation derived from multiple regression with a set of variables from UK reference rivers including: alkalinity, altitude, altitude of source, slope and distance of site from source.

Thus:
Expected RMHI = 6.55833+0.664862* Log_{10} (alkalinity+1)+-0.85111* Log_{10} (slope +1) +0.005868* distance to source +-0.64038* Log_{10} (source altitude+1)+0.006709*slope

This results in an expected RMHI score of 6.85.

The EQR for this river using the RMHI metric alone is therefore:

$$\text{RMHI EQR} = \frac{(7.35-10)}{(6.85-10)} = 0.84$$

Note that both the observed and expected values have 10 subtracted from them. This reflects the theoretical maximum (worst) RMHI site score that can be achieved in a given river type (in this case 10), and ensures that low RMHI scores achieve a high EQR. The river status based on this metric alone would be good (RMHI EQR of between 0.70 and 0.85).

Since RMHI EQR lies between 0.25 and one the adjusted RMHI EQR is obtained from:

adj RMHI EQR= 1.333*RMHI EQR-0.333 = 0.79

Based only on a pairwise comparison of RMNI and RMHI, the vegetation indicates that this site is proportionally more impacted by nutrient-related pressures than by hydromorphological pressures.

6.8.3 Determine functional diversity (N_FG) EQR based on number of functional groups

The observed number of functional groups (N_FG) for this river is seven (see Table 6.8). There are eight aquatic plant taxa because the helophytes *Phalaris arundinacea*, *Phragmites australis* and *Rumex hydrolapathum* are excluded. The remaining species belong to different FG except *Oenanthe crocata* and *Apium nodiflorum* which share the same FG. Consequently there are seven FG.

The expected number of functional groups is derived from an equation based on the relationship between NFG, slope and Northing. The equation predicts Log_{10}(NFG+1).

Thus:
Expected Log_{10}(N_FG + 1) = 0.785064+-0.1625* Log_{10} (slope + 1) +0.000000143*Northing

The expected number of functional groups would be 0.758 (as $Log_{10}(x +1))$ or 4.7 in real terms.

$$\text{The N\_FG EQR} = \frac{Log.(ObservedN\_FG + 1)}{Log.(PredictedN\_FG + 1)} = \frac{0.903}{0.758}$$

This gives a N_FG EQR of 1.19.

The river status based on this metric alone would be high (N_FG EQR above 0.85).

Since the N_FG EQR lies above one the adjusted RMHI EQR is obtained by capping the individual metric EQR at one. Effectively this signifies that a site contains at least as many FG as might be expected in a comparable river under reference conditions.

Prediction of the expected N_FG is highly dependent on the survey method employed. The same function cannot be used if data is collected as 500 or 1,000 m units.

6.8.4 Determine taxonomic diversity as number of taxa (N_TAXA) EQR

The observed number of aquatic plant taxa (N_TAXA) is eight as detailed above. The expected number of taxa is predicted by a function based on slope and source altitude as predictors and this predicts Log_{10} (N_TAXA+1).

Thus:
Expected Log_{10}(N_TAXA + 1) = (0.548545+-0.1586* Log_{10}(slope +1) +0.172477* Log_{10} (source altitude+1)

In this case the expected number of taxa would be 0.875 (as $Log_{10}(x+1)$ or 6.5 taxa in real numbers.

$$\text{The N\_TAXA EQR} = \frac{Log.(ObservedN\_TAXA + 1)}{Log.(PredictedN\_TAXA + 1)} = \frac{0.954}{0.875}$$

This gives an N_TAXA EQR of 1.09.

The river status based on this metric alone would be high (N_TAXA EQR above 0.85).

Since the N_TAXA EQR lies above one the adjusted EQR is obtained by capping the individual metric EQR at one. Effectively this signifies that a site contains at least as many taxa as might be expected in a comparable river under reference conditions.

As with N_FG, prediction of expected N_TAXA depends on the survey method used. The same function cannot be used if data is collected as 500 or 1,000 m units.

6.8.5 Determine algal cover EQR based on percent cover of green filamentous algae

A single taxon of green filamentous algae (*Cladophora glomerata*) was recorded in this survey. The cover score assigned to this taxon (four) should be converted to the mid-point of the percentage range associated with this score (see Table 2.2). In this case the conversion is to a value of 3.8 per cent.

The algal cover in reference sites could not be modelled satisfactorily using the same suite of predictors available for the composition and richness metrics. Consequently a global reference value of 0.05 per cent cover is used based on the median of the population of reference sites.

Provided observed ALG_ COV is under 100 per cent the ALG_COV EQR is given by

$$\frac{(3.8-100)}{(0.05-100)} = 0.96$$

Note that both the observed and expected values have 100 subtracted from them. This reflects the theoretical maximum (worst) ALG_COV site score of 100, and ensures that high algal cover achieves a low EQR. The river status based on this metric alone would be good (ALG_COV EQR of between 0.97 and 0.92).

Rescaling of the ALG_COV EQR to a harmonized range and class boundary system is accomplished by a slightly different approach due to the widely varying width of the classes in the unharmonised system. This approach relies on stretching each unharmonised class to a standard width of 0.2 units and mapping the observed EQR within the appropriate range based on a simple linear transformation.

Thus:

= IF(x>=0.975,((x-0.975)/(1-0.975))*0.2+0.8,IF(x>=0.925,((x-0.925)/(0.975-0.925))*0.2+0.6,IF(x>=0.825,((x-0.825)/(0.925-0.825))*0.2+0.4,IF(x>=0.625,((x-0.625)/(0.825-0.625))*0.2+0.2,IF(x<0.625,(x/0.625)*0.2)))))

Where x = the untransformed ALG_COV_EQR.

This generates an adjusted ALG_COV_EQR for the survey of 0.75.

6.8.6 Combining individual metrics into overall EQR for site

The approach used to combine individual metrics to obtain an overall EQR for the site is discussed in Chapter 7. Section 7.5 takes the worked example provided above to demonstrate the approach.

7 Achieving an overall classification based on river macrophyte metrics

7.1 Introduction

Any classification can be secured from the EQR based on a single metric. However, independent of the uncertainty associated with that metric, the fewer the metrics on which a classification is based, the greater the risk of classifying a site as impacted (moderate or worse status) or unimpacted (good or better) when the weight of evidence from a broader spectrum assessment would suggest otherwise. Conversely, the more metrics considered within the classification, the greater the probability that a site will fail on at least one. Multimetric assessments of individual quality elements should in principle also bring the assessment based on different quality elements more closely into line, since they will reflect pressures to which other quality elements are most responsive, as well as reflecting the mechanisms underlying secondary effects of one quality element upon another. Consequently, how the information from different metrics is combined will have an important influence on the final classification of a site.

It is implicit in the normative definitions that more than one metric is required per quality element to assess deviation from reference condition. This may be because different attributes of a quality element feature in the definition (abundance, composition, diversity) or because several different metrics must be employed simultaneously to cover a range of different pressures. Generally the more metrics that are used, the more likely it is that a pressure will be successfully detected.

Confidence in some metrics is influenced by the value of other metrics. For example, indices that use composition of vegetation to infer different pressures have less confidence associated with them when vegetation is sparse or species-poor. By contrast, when the compositional metrics use scores expressed on a constrained scale (one to 10 in this case), there is a weak tendency in high or good status sites for the metric value (RMNI in this case) to increase with increasing number of species. In this project five metrics were developed, RMNI, RMHI, N_FG, N_TAXA and ALG_COV, to collectively address the major anthropogenic and biological pressures to which rivers are exposed (nutrient enrichment, hydromorphological modification and acidification). This, however, does not preclude the addition of other compositional metrics. For example, acidification and the more severe hydromorphological impacts are probably adequately addressed by the suite of metrics already in use, but extra compositional metrics could improve sensitivity to these pressures when present at lower intensity.

7.2 A rationale for combining metrics

7.2.1 Multimetric approaches

The most common rationale for multimetric approaches is to provide sensitivity to a range of different pressures, with different compositional metrics used to reflect these pressures, without necessarily seeking to distinguish between them. For example, classifications based on river invertebrates rely increasingly on the combined use of the metrics Average Score Per Taxon (ASPT), Lotic-invertebrate Index for Flow Evaluation (LIFE) and Acid Water Indicator Community (AWIC) to assess pressures due to organic pollution, flow modification and acidification, respectively. In the case of macrophytes, Dodkins *et al*. (2005) proposed a multimetric system for assessing multivariate pressures on water courses in Northern Ireland using species optima for the variables silt content, dissolved oxygen, nitrate and pH. Other macrophyte-based classification systems have used a wider spectrum of metrics to reflect the overall integrity of the vegetation in relation to a wide cross-section of pressures including, for example, biological invasions. This is more in line with the use of multimetric systems to assess plant biotic integrity in lakes and wetlands developed in North America (see Miller *et al*., 2006; Mack, 2007; Rothrock *et al*. 2008). For example, in Pond PSYM (Pond Action, 2002) the metrics, number of submerged and marginal species, Trophic Ranking Score (from Palmer *et al*., 1991) and number of uncommon plant species were found to be the most useful three metrics to reflect environmental degradation using macrophytes. Willby *et al*. (2008) used a combination of metrics, including number of aquatic plant species, aquatic plant biomass, emergent plant cover, emergent plant richness and a compositional metric based on nutrient sensitivity to assess the ecological status of canals based on their macrophytes. Meanwhile, some macrophyte-based assessment systems developed in other European countries for WFD purposes (such as Schaumburg *et al*., 2004), do not use multiple metrics in a strictly integrated sense, but introduce other metrics, such as richness or cover, as 'bolt-ons' to override assessments based on traditional compositional metrics whenever expert opinion indicates this to be necessary. At present, there has been rather limited application of the multimetric approach to vegetation in rivers (such as Ferreira *et al*., 2002), the majority of examples coming from wetlands and lakes.

The LEAFPACS project adopted a multimetric approach, partly to reflect the impacts of different types of pressures, but primarily to explore the basis for the use of individual metrics. The principle of this approach is shown in Figure 7.1. Only two compositional metrics (RMNI and RMHI) are used in LEAFPACS, partly because these reflect the dominant pressures on European rivers. It is also likely to prove difficult to disentangle the effects of interrelated pressures at a finer level. Thus there is a general gradient from erosional, high-energy, nutrient-poor sites to depositional, low-energy fertile sites. This might be propagated by a range of pressures such as eutrophication, sedimentation, channel or bank realignment and abstraction, and these might act collectively or in isolation. Opportunities might remain to incorporate other compositional metrics sensitive, for example, to abstraction *per se*, as opposed to direct geomorphological impacts. The additional metrics used in LEAFPACS have a dual basis. Firstly, reliance on compositional metrics is unwise when metric values may derive from sites that are abnormally species-poor or sparsely vegetated. This may be a feature of data on macrophytes which tends to be species-poor and/or dominated by low cover values, compared to, for example, data for diatoms or macroinvertebrates, which tends to be taxa-rich and based on a large number of individuals. Secondly, different types of metrics are required to provide complementary sensitivity across a full pressure gradient, as well as across a full range of river types. Thus, richness and cover-based metrics are increasingly important at higher levels of enrichment and in more naturally fertile rivers where compositional response to enrichment is quickly saturated.

The multimetric approach of LEAFPACS is in marked contrast to classification tools for lakes and rivers based on diatoms (Kelly *et al*, 2008), which have adopted a unimetric approach, based on the Trophic Diatom Index (TDI). Although this may seem at odds with the need for holistic ecological assessments a recent study (Reavies *et al*., 2008) has indicated that, in the case of diatoms, single metric assessments using traditional weighted average indices deliver superior sensitivity to multimetric approaches. Multimetric approaches may therefore not have universal applicability.

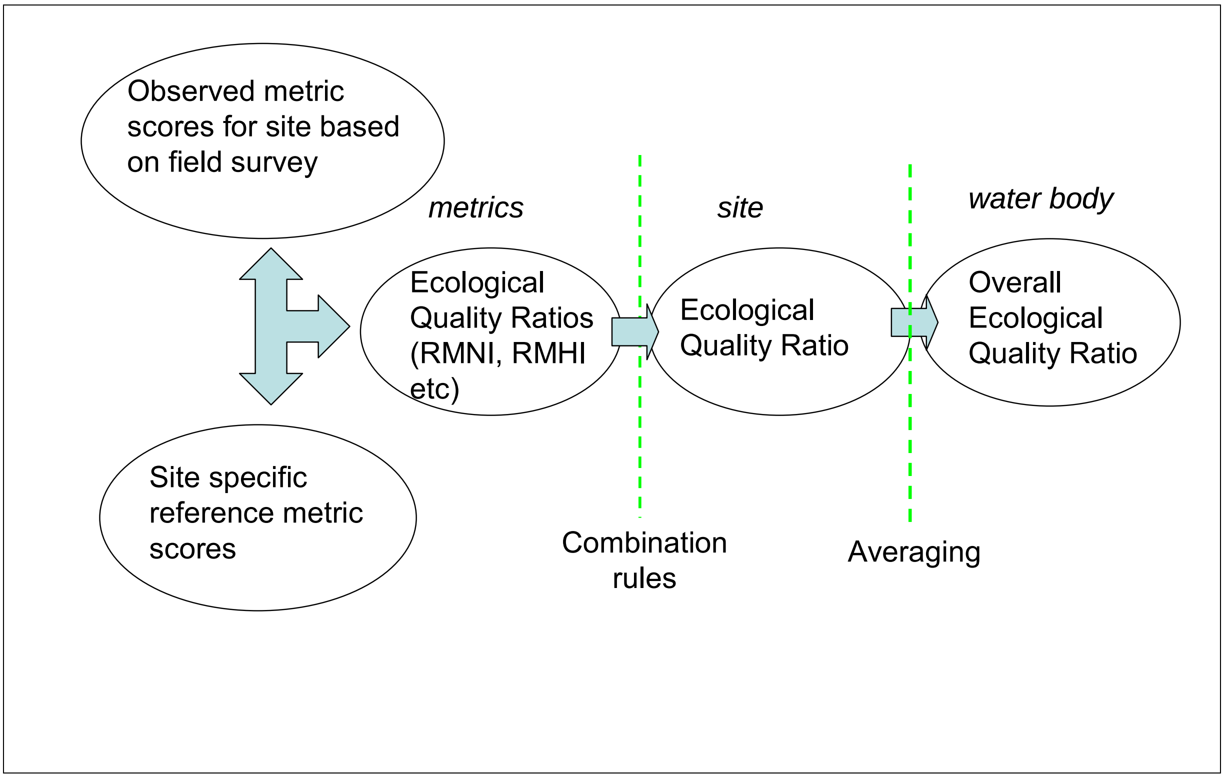


Figure 7.1 Summary of multimetric classification procedure in LEAFPACS

7.2.2 Combining and weighting metrics

A range of approaches are available for combining metrics to achieve a classification. Multimetric systems normally examine the variability of metrics over space and time within reference sites, their responsiveness to different pressures and their degree of intercorrelation, before assigning weights to each metric. Hughes *et al*. (1998), Dodkins *et al*. (2005) and Reavies *et al*. (2008) provide examples for fish, macrophytes and diatoms respectively.

The initial approach taken in this project was to base the final class for a water body on the metric with the lowest EQR, since this is closest to the 'one out, all out' approach advocated in the WFD to achieve an overall site classification using a range of quality elements (there is an additional caveat here, since the metric that effectively dictates the classification of a site is not necessarily the one that gives the highest confidence of class for a given status). It is unclear, however, if this approach should also be applied at the within-quality element level when dealing with different attributes of an element, such as composition, abundance and richness.

There is a rationale to using the metric with the lowest associated class since the specific pressure or attribute of the quality element which deviates most from reference condition is clearly identified. Under this approach the class of a site cannot be redeemed by other metrics with higher EQR values. Thus, for example, the status of a site with a low RMNI EQR cannot be raised by the presence of diverse flora and low algal cover (high N_FG and ALG_COV EQRs).

Taking the minimum EQR is a conservative approach. Its disadvantage is that it does not adequately discriminate between sites with general failure across a suite of metrics, from sites where one metric is impacted while others are largely 'healthy'. Thus, in the example above, a site with a low EQR for RMNI and a high EQR for ALG_COV and N_FG would not be discriminated from a site with a similarly low EQR for RMNI but also low EQRs for COV and N_FG. This is significant because, *inter alia*, resources to support Programme of Measures (PoMs) may be allocated differently between these cases. Moreover, conservation of water body ecosystem function lies at the heart of the WFD. Arguably, a vegetation which approaches the natural richness and abundance for that water body, yet is altered in composition (for example through anthropogenic pressure or biological invasion), will still retain and support more of the ecosystem functions to which macrophytes contribute than will a similar water body with a vegetation that is largely unaltered compositionally, yet is species-impoverished or has unnaturally high cover of filamentous green algae. These concepts are summarised in Table 7.1. A more detailed consideration suggests there would be virtue in basing the ecological status for a site on information from more than just the lowest metric.

7.2.3 Weighting metrics

Although metrics are typically assigned equal weight in terms of their contribution to the final classification of a site, the decision to weight equally should have the same statistical or ecological underpinning as unequal weighting. There are a number of reasons why metrics should carry unequal weight including: (i) high intercorrelation among some subsets of metrics, (ii) differences in the strength of correlations between metrics and pressures, (iii) differences in inherent variability of measurements of some variables required to predict a metric value under reference conditions (although this should translate to an effect on the class boundaries for that metric), or (iv) because some metrics behave differently across a pressure gradient.

In the case of the two compositional metrics, RMNI and RMHI, both are strongly inter-correlated at a site scale so one could not use both metrics in an additive sense. However, the EQR for each metric should indicate the relative degree of impact on the vegetation of nutrient enrichment, or hydromorphological alteration.

In the case of richness metrics (N_TAXA and N_FG), consideration of the relationship between richness and pressure (Section 4.2.3) suggests that these metrics should be variably weighted depending on the position of the site on a productivity gradient. Thus, at low baseline productivity, when species richness should be constrained by nutrient limitation, the presence of a relatively diverse flora (N_FG or N_TAXA EQR above one) may be suggestive of nutrient enrichment. High richness EQR values should therefore be neutral or even negatively weighted. This effect is largely achieved by capping the richness EQRs at one. Conversely, at high baseline productivity, unless there is high instream habitat heterogeneity, conditions might be expected to lead to loss of species through competitive exclusion by dominant, canopy-forming, tolerant taxa. Hence, the presence of a relatively diverse flora should be seen as a positive indicator which would enhance the ecological status of that site relative to a similar water body with fewer taxa. Consequently the weight given to high values of richness metrics relative to compositional metrics should increase with increasing productivity. Low richness metric EQRs would always carry a negative weight (especially if the EQR was lower than that returned for other metrics) since this would indicate that the assemblage was less diverse than would be expected under reference conditions. A range of pressures, including acidification, modification of water level regime or establishment of high risk invasive species might then be suspected, depending on the river type.

In our classification, the metric ALG_COV is associated with a higher measurement error and a weaker correlation with pressure data than the core metrics. However, the class boundaries are set sufficiently wide to reflect this. From consideration of naturally productive rivers using only composition and richness metrics as the basis for classification, a significant number of sites are reported at good ecological status when the level of filamentous algal cover would strongly suggest otherwise (for example, cover values exceed 10 per cent). To reflect this in the final class for a water body and acknowledge the role of algal cover as an indicator of undesirable disturbance, it proved necessary to give ALG_COV an equivalent weight to the remaining metrics.

Table 7.1 Conceptual basis for combining different macrophyte-based metrics to classify water body ecological status based on contributions to different deliverables

Macrophyte state variables			Deliverables				Condition	Status
Structure	Diversity	Abundance	Stability	Ecosystem-dependent functions	Biodiversity support	Cultural value		
+	+	+	✓✓	✓✓	✓✓	✓✓	Unaltered	High
-	+	+	✓✓	✓✓	✗	✗	Altered - recoverable	G/M
+	-	+	✗	✓	✓	✓✓	Altered - recoverable	G/M
+	+	-	✓	✗✗	✓✓	✓	Altered	M
-	-	+	✗	✓	✗✗	✗	Altered	M
+	-	-	✗✗	✗✗	✓	✓	Altered - unrecoverable	M/P
-	+	-	✓	✗	✗	✗✗	Altered - unrecoverable	M/P
-	-	-	✗✗	✗✗	✗✗	✗✗	Destroyed	Bad

Altered recoverable condition is considered able to achieve full recovery to more or less unaltered state through internal processes or minimal intervention. Altered unrecoverable requires a PoM to achieve recovery. Where ecological status has been destroyed restoration to a set of objectives other than GES may be required.

7.2.4 Intercalibration and the combining of metrics

At the intercalibration stage of tool refinement, it is preferable to see the metrics of the national method as fixed 'ingredients' and intercalibration as the 'recipe' that governs how these ingredients are best combined to achieve an outcome compatible with the view of other intercalibrating Member States (MS). Rules for combining metrics have been developed iteratively through intercalibration of the UK lake classification method at both Northern/Atlantic and Central-Baltic Geographical Intercalibration Groups (GIGs). These rules have been adopted and where necessary refined for the process of intercalibrating national methods for river macrophytes within Central Baltic-GIG. This process will now not be completed until late 2010.

7.3 Application to macrophyte-based classification of rivers

7.3.1 Approaches considered

A number of approaches were considered for combining metrics to achieve an overall class. These are listed below.

i. the metric with the lowest EQR across all metrics;
ii. average EQR across all five metrics;
iii. average of a subset of metrics (such as RMNI, ALG_COV and N_FG);
iv. average of the two lowest EQRs;
v. a complex rule-based approach for combining metrics.

A sixth approach based on averaging across the three lowest EQRs was trialled but this did not give materially different results to Method 3.

7.3.2 Results

Methods 2 and 3 above resulted in the lumping of large numbers of surveys in high or good classes and were clearly not sufficiently sensitive. The evidence suggests that even when several metrics exhibit significant distortion, their low EQR is 'rescued' through averaging with metrics that show little impact. Generally, few sites exist where all metrics are degraded to a similarly high degree. The only way that Methods 2 or 3 would give better resolution would be to change the class boundaries. If one examines the distribution of the combined EQR for the reference sites, the 10^{th} percentile of the EQR distribution is 0.94 for both methods which would suggest a need to adjust the class boundaries accordingly (upwards). However, even after making this adjustment averaging approaches do not compare favourably with Methods 1 or 4. Method 4 has the distinct advantage of discriminating between surveys where all metrics are impacted and sites where one metric is significantly impacted but there is a large differential to the next lowest EQR. This discriminatory power is important because it takes information from a range of attributes of a quality element prescribed by the WFD as well as being able to indicate whether aspects of ecosystem function that depend on macrophytes are slightly or significantly degraded.

This is illustrated in the examples in Table 7.2 below. Consider that a simple class boundary system of 0.2 units per class is in place (high status runs from 1.0 to 0.8). Site B would be classified as poor by all approaches for combining metrics. However Site A would be classified as poor by the minimum metric approach (Method 1), moderate by the averaging over all metrics approach (Method 2) and moderate by the averaging across the worst two metrics approach (Method 4). Method 2 may give too optimistic a view while Method 1 fails to discriminate between two sites which are arguably different in the level of impairment of ecosystem function (as suggested by the concepts in Table 7.1). On this basis, averaging across the two lowest EQRs was used provisionally in this project to define the final class. An additional advantage of this approach is that it is amenable to the incorporation at a later date of additional metrics sensitive to specific pressures (such as shoreline modification or changes to water level regime). This approach was subsequently refined to a more complex rule-based approach as a result of intercalibration, and to reflect a shift in the relationship between productivity and richness with increasing pressure. A simple example of such an approach is illustrated in Table 7.3, whereby Metric 3 carries twice the weight of Metrics 1 and 2.

Table 7.2 Simplified examples illustrating alternative methods for combining metrics from two sites

Metric	Site A	Class	Site B	Class
1	0.3		0.3	
2	0.6		0.3	
3	0.75		0.3	
Minimum	0.3	Poor	0.3	Poor
Average	0.55	Mod	0.3	Poor
Average worst two	0.45	Mod	0.3	Poor
Rule-based average	0.6	G/M	0.3	Poor

7.4 Final classification rules

The final rules for classification are based on multiple permutations of rules for combining metrics, some of which are described above. They are informed by the successful intercalibration of the UK lake macrophyte classification method at NGIG/AGIG (Option 2 based) and CBGIG (Option 3 based) levels and take account of the interim outcomes of intercalibration of river macrophyte classification methods at CGIG. If additional metrics are developed to reflect other pressures or refine pressure-response relationships, these would need to be integrated rather than changing the rules for combining the existing suite of metrics. The following rules are used to combine metrics developed in this project and yield a classification of the same or similar rivers that is likely to be compatible with the view of other GIG MS. It is possible that further minor amendments will be needed to the combination rules to ensure successful intercalibration following recent adjustments to national methods by a number of member states. However, any such modifications are unlikely to change the overall EQR of any site by more than 0.05 EQR units (one quarter of a class).

Use of rescaling rules described in Section 6.8 will ensure that all metrics are scaled from zero to one with class boundaries at 0.2 unit intervals. No subsequent rescaling of the final site EQR is required.

STEP 1:

Combine composition and richness EQRs to give an interim EQR

i. Identify the lowest EQR of each pair of metrics, composition and diversity.

ii. If the values of the lowest of the two diversity metric EQRs is *less* than the lowest of the two composition EQRs, then the mean of the lowest composition metric EQR and the lowest diversity EQR is calculated to obtain the interim EQR for the site, with the lowest diversity metric receiving 50 per cent of the weight of the lowest composition metric. Thus

$$\frac{\left(\min(RMNI\_EQR, RMHI\_EQR)\right) + 0.5 * (\min(N\_FG\_EQR, N\_TAXA\_EQR)\right)}{1.5}$$

iii. If the value of the lowest diversity EQR is *greater* than the minimum composition EQR, then the lowest diversity EQR is multiplied by a weighting factor and added to the lowest composition EQR. This product is then divided by the weighting factor plus unity. Thus

$$\frac{\left(\min(RMNI\_EQR, RMHI\_EQR)\right) + weighting * (\min(N\_FG\_EQR, N\_TAXA\_EQR)\right)}{1 + weighting}$$

The weighting factor is defined by a simple logistic regression based on the expected RMNI value and is given by:

Weight D = 0.25 + (1/(EXP(LN(1500) +expected RMNI*LN (0.31))+1/0.5))

The weighting factor is designed to compensate for the attenuated change in composition at high fertility compared to the potential change in diversity, and to reflect the fact that high richness is likely to reflect high instream physical habitat heterogeneity which is often greatly compromised in naturally more productive systems. The weighting factor is defined by a simple logistic regression based on the expected RMNI value. This gives the richness metrics a maximum of 70 percent of the weight of the composition metrics when productivity is naturally high (RMNI around seven), dropping to a weight of 25 percent when productivity is naturally low (expected RMNI of under four). This step increases the EQR of diverse relative to impoverished high-alkalinity rivers with the same composition EQR. In moderate and low-alkalinity rivers (RMNI under five) the lower weighting factor given to the diversity metric EQRs ensures that composition is the dominant determinant of composition. This ensures that a diverse site on a low or moderate alkalinity river with a distorted composition is not elevated in its status.

STEP 2:

Combine the abundance EQR with the interim EQR

i. If the interim EQR based on composition and richness is less than the ALG COV EQR, the interim EQR forms the overall EQR. This means that a site with a low EQR based on composition and richness cannot be 'improved' by having a low algal cover.

ii. If the interim EQR, as described in Step 1, is greater than the ALG COV EQR, the final EQR is determined from the following weighted average. This has the consequence of lowering the final EQR of sites that support unnaturally elevated growths of filamentous algae.

$$\frac{(inerimEQR + weighting * (ALG\_COV\_EQR))}{1 + weighting}$$

The weighting factor is defined by a simple logistic regression based on the expected RMNI value and is given by:

= 2*(1/(EXP(LN(2624653085.79034)+expected RMNI *LN(0.0165738290871162))+1/0.5001)))

This weighting factor is designed to reflect the weak link between algal cover and nutrients at low-moderate alkalinity, compared to the strong link at high-alkalinity, especially in southern-continental rivers. The weighting factor is defined by a simple logistic regression based on the expected RMNI. This gives the richness metrics a maximum of 100 percent of the weight of composition metrics when productivity is naturally high (RMNI around 6.5), decreasing to a weight of zero when productivity is naturally low (expected RMNI below four). This step decreases the overall EQR of moderate to high-alkalinity rivers with large filamentous algal cover relative to sites with sparse algal cover but with the same interim EQR (based on composition and richness). In moderate and low-alkalinity rivers (RMNI under five) the lower weighting given to ALG_COV EQR allows for the fact that an intermittently high cover of small, probably nutrient sensitive filamentous algal taxa are a natural feature of such streams.

These classification rules are summarised in Figure 7. 2.

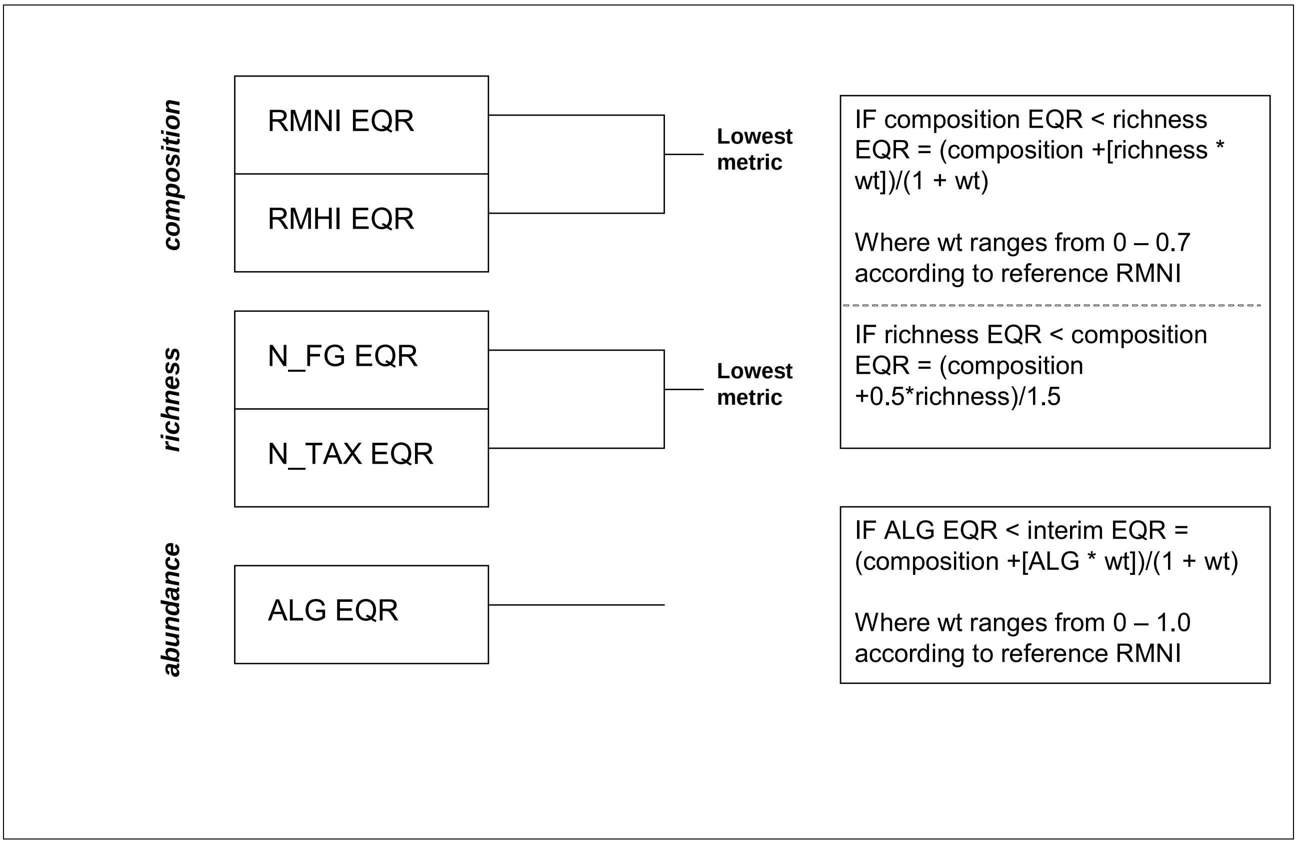


Figure 7.2 Summary of combination rules used in LEAFPACS

7.5 Examples of classification

7.5.1 Reference to worked example

The following example is used to illustrate the process of calculating expected values, metric EQRs and the approach for combining EQRs to reach a final classification. The worked example used in Section 6.8 forms the starting point.

Consider the worked example. The EQRs and their adjusted equivalent for the various metrics (based on the equations in Table 6.7) are summarized in Table 7.3 below. In this example the minimum of the composition EQRs (RMNI and RMHI) is 0.57, while the minimum of diversity EQRs (N_FG EQR and N_TAXA) is one. Since the diversity EQR (1.00) is greater than the composition EQR (0.57), the interim EQR is given by

$$0.73 = \frac{\left(0.57 \;\; + 0.60 * 1.00\right)}{1 + 0.60}$$

Where 0.60 is the weighting factor given by:

Weight D = 0.25 + (1/(EXP(LN(1500) +expected RMNI*LN (0.31))+1/0.5))

In which the expected RMNI = 6.36

This interim EQR (0.73) is less than the ALG COV EQR (0.75) and therefore no further refinement of the interim EQR is needed to give the overall EQR for the site.

This gives the site an overall EQR of 0.73, placing it towards the middle of good status. In this case, despite some evidence of an impacted composition in terms of RMNI, high diversity is sufficient to elevate the status of the site from the upper end of moderate to good. In this example the ALG COV EQR closely supports the interim EQR and therefore has no impact on the overall EQR. If the interim EQR was 'unsafe' the ALG COV EQR would have lowered this sufficiently to reduce the overall class to moderate. Hence, for example, if the observed filamentous algal cover had carried a score of six, which transforms to a cover score of 17.5 per cent, this would have lowered the adj_ALG COV EQR to 0.40, which would in turn have reduce the overall EQR to 0.57.

Thus:

$$0.57 = \frac{\left(0.73 \;\; + (0.40 * 0.99)\right)}{1 + 0.99}$$

Where 0.99 is the weighting factor for the ALG COV metric given by:

Weight A = 2*(1/(EXP(LN(2600000000)+expected RMNI *LN(0.0166))+1/0.5001)))

In which the expected RMNI = 6.36.

Table 7.3 Summary of metric EQRs, adjusted EQRs and process of combining EQRs in worked example

<table>
<tr><th></th><th></th><th>OBSERVED</th><th>EXPECTED</th><th>RAW EQR</th><th>ADJ EQR</th><th>MINIMUM</th><th>WEIGHT_D</th><th>INTERIM</th><th>WEIGHT_A</th><th>OVERALL</th><th>CLASS</th></tr>
<tr><td rowspan="2">Composition</td><td>RMNI</td><td>7.52</td><td>6.36</td><td>0.68</td><td>0.57</td><td rowspan="2">0.57</td><td rowspan="2">1</td><td rowspan="4">0.73</td><td rowspan="4">1</td><td rowspan="5">0.73</td><td rowspan="5">Good</td></tr>
<tr><td>RMHI</td><td>7.35</td><td>6.85</td><td>0.84</td><td>0.79</td></tr>
<tr><td rowspan="2">Diversity</td><td>N_FG</td><td>0.9</td><td>0.76</td><td>1.19</td><td>1</td><td rowspan="2">1</td><td rowspan="2">0.6</td></tr>
<tr><td>N_TAXA</td><td>0.95</td><td>0.87</td><td>1.09</td><td>1</td></tr>
<tr><td>Abundance</td><td>ALG_COV</td><td>3.8</td><td>0.05</td><td>0.96</td><td>0.75</td><td></td><td></td><td></td><td>0.99</td></tr>
</table>

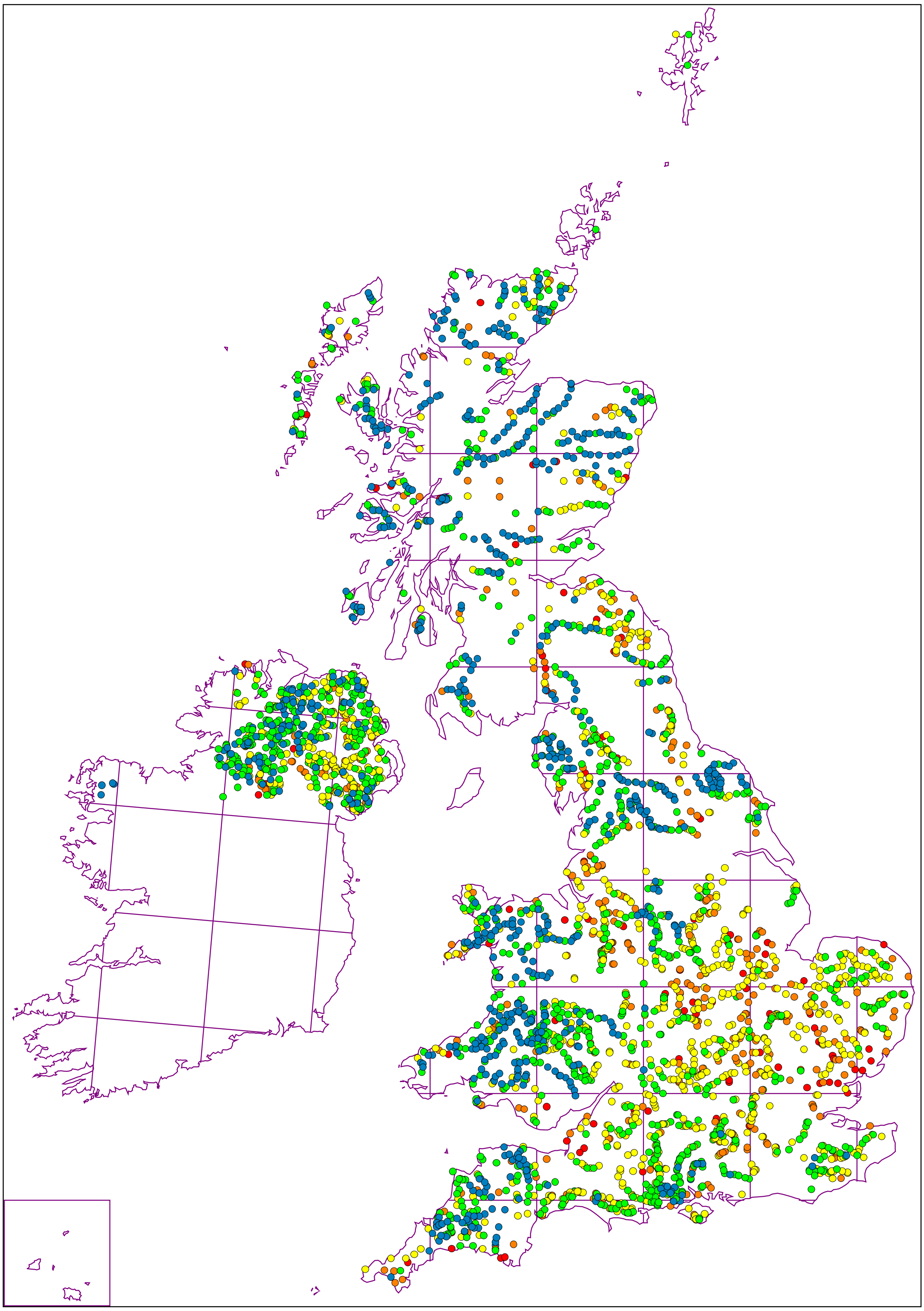


Figure 7.3 Final classification of UK rivers based on average site EQR using data from MTR and JNCC surveys. Colour coding follows WFD convention (blue = high status, green = good, yellow = moderate, orange = poor and red = bad).

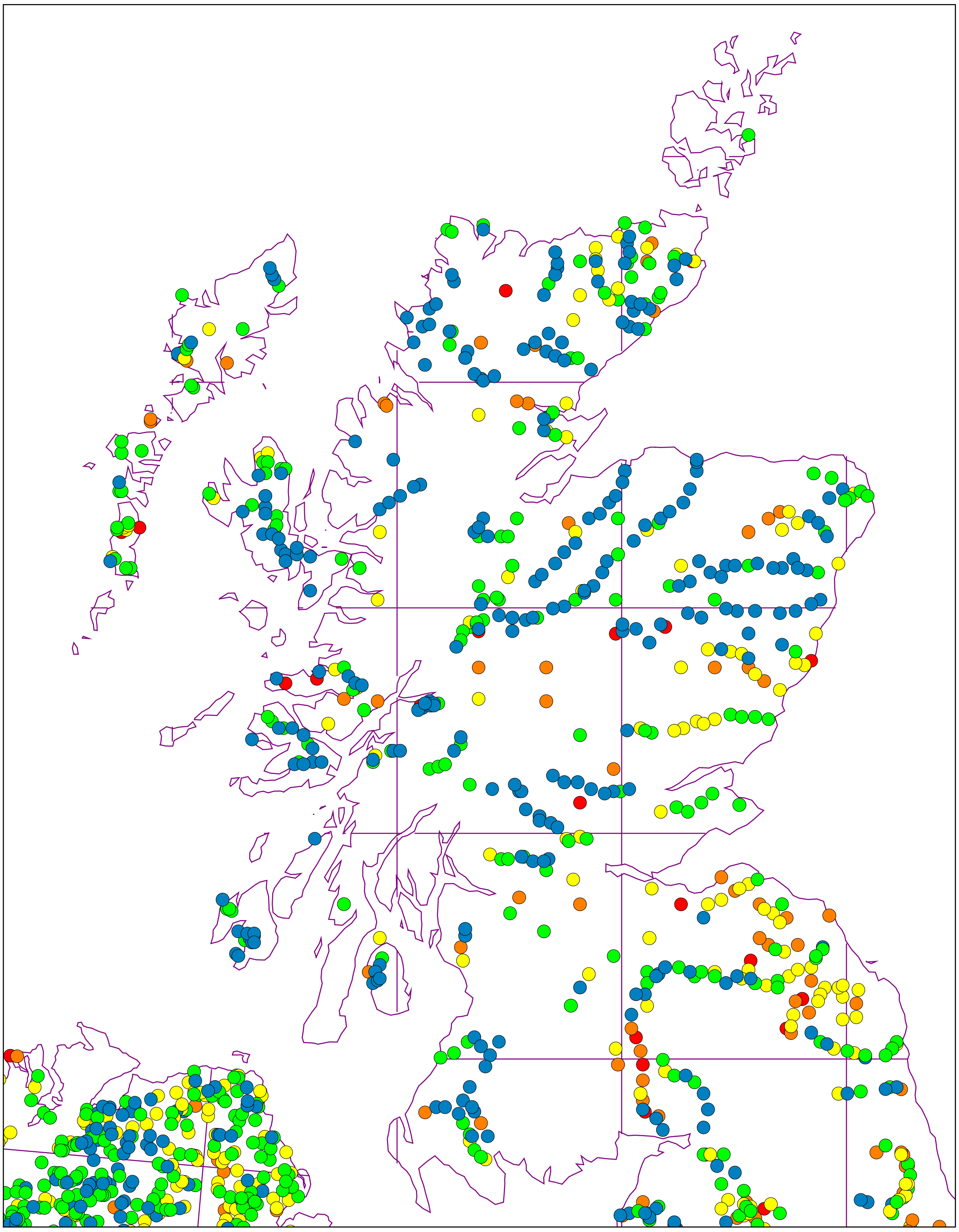


Figure 7.4 Distribution of river macrophyte survey sites in Scotland by mean class. Six sites in the Northern Isles are not shown.

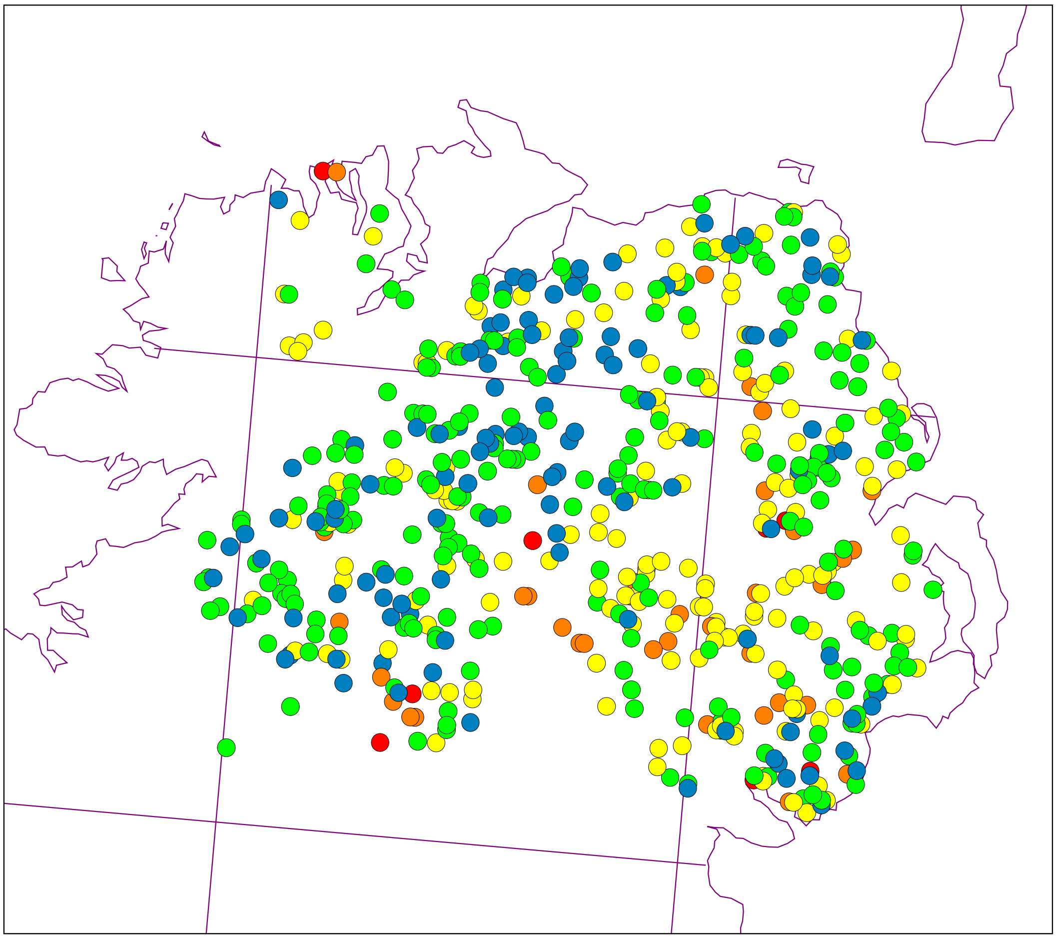


Figure 7.5 Distribution of river macrophyte survey sites in Northern Ireland by mean class

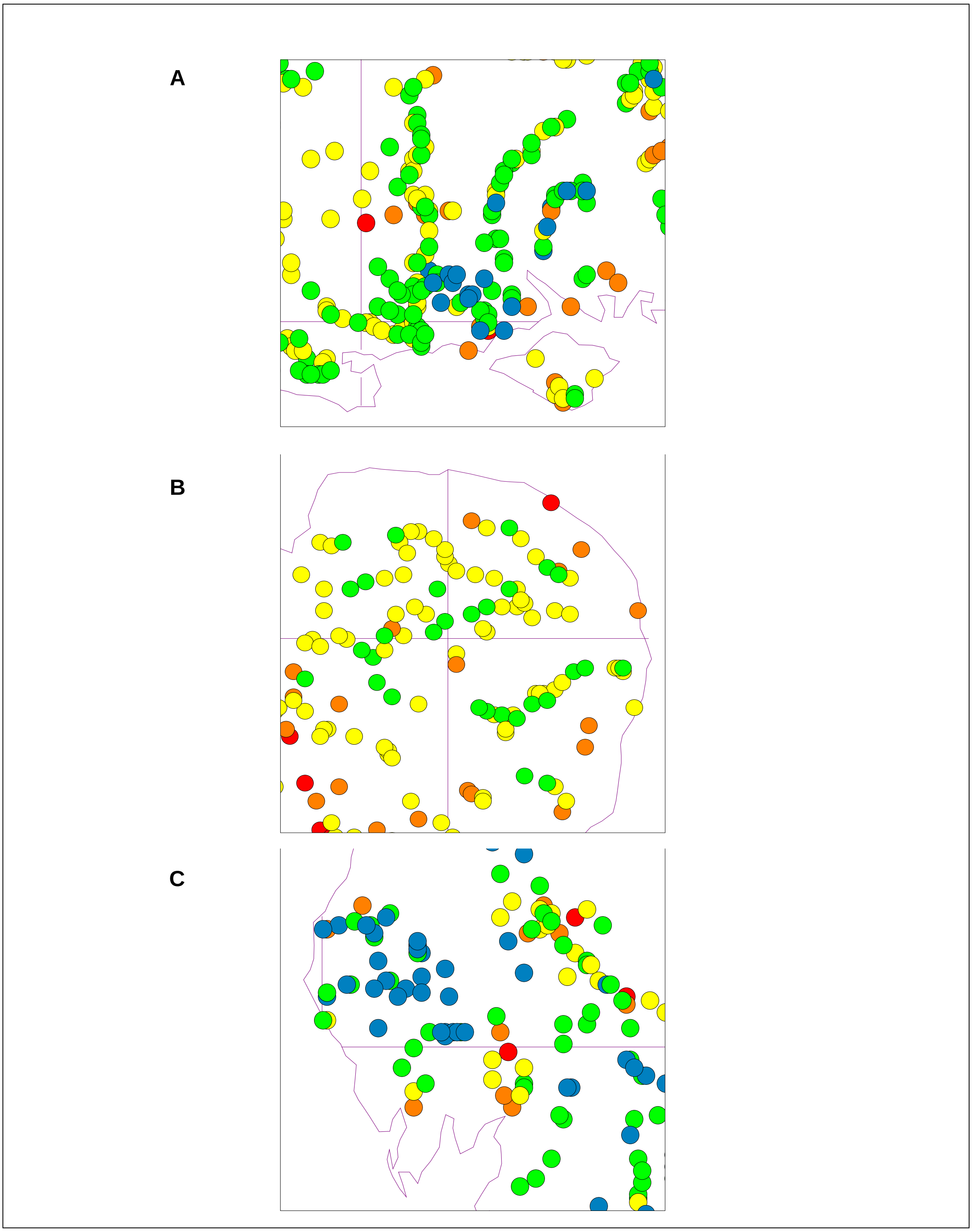


Figure 7.6 Distribution of river macrophyte survey sites by mean class in A: Dorset/Hampshire/Wiltshire, B: East Anglia, C: NW England.

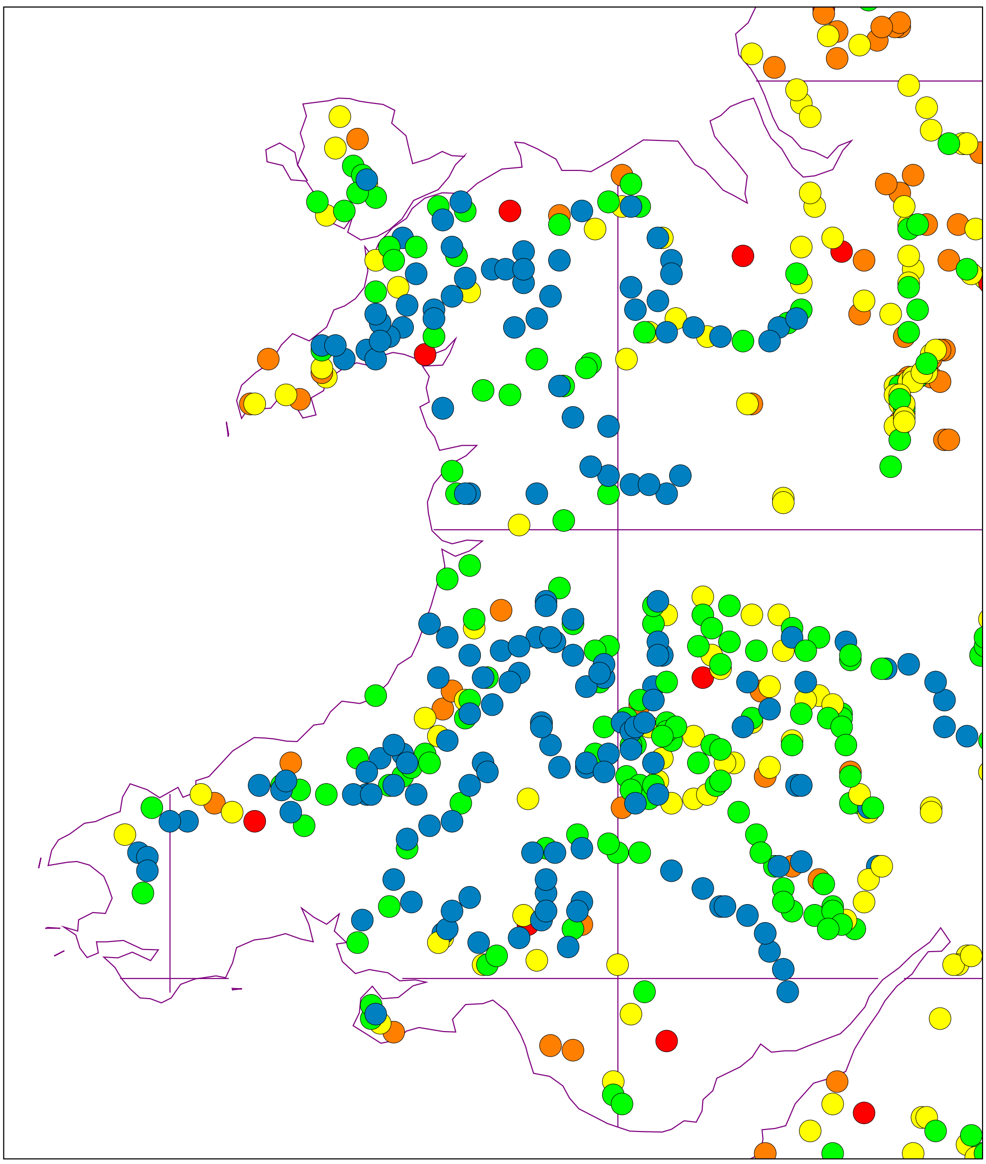


Figure 7.7 Distribution of river macrophyte survey sites in Wales by mean class

7.6 Overall implications for classification

7.6.1 Comparison on a geographical basis

On a geographical scale this classification translates as shown in Figure 7.3. Immediately obvious is the concentration of high and good status sites on rivers in the more sparsely populated areas of the north and west of Britain and the high incidence of moderate and poor status sites on rivers in central and eastern England and central Northern Ireland. However, individual sites of moderate or poor status can occur almost anywhere, even on rivers where high or good status sites otherwise dominate. This may reflect local pressures, idiosyncracies in some survey data, or use of inappropriate values for the environmental variables used to predict site-specific reference conditions. The associated figures (Figures 7.4-7.7) illustrate countries or regions with a high density of rivers that have formed the focus of previous interest.

The output of the final classification is summarised on a country-by-country basis in Table 7.4 based on the mean class at every site surveyed. This highlights the relatively impacted nature of rivers in England and Wales and, to a lesser extent, Northern Ireland, where about half of sites surveyed would fail to achieve good or better status (although this figure will be lower when confined to water bodies with 95 per cent confidence of class being less than good). In England and Wales, 23 per cent of sites surveyed would be classified as poor or bad. This partly reflects the intensity of pressures on rivers in England and Wales driven by agricultural intensification and urbanisation but it will also reflect the distribution of sampling sites and emphasis on comparing conditions above and below major point sources such as sewage treatment works. Consequently the extent of poor and bad status sites may be slightly exaggerated. In Scotland, the percentage of impacted rivers is low due to the vast numerical dominance of minimally impacted base-poor streams in sparsely populated catchments in the north and west of the country. Thus 72 per cent of macrophyte survey sites in Scotland were classified as good or better and only 12 per cent as poor or bad.

Table 7.4 Summary of final classification of river macrophyte survey sites in the UK by number and percentage

		High	Good	Moderate	Poor	Bad	Total
England & Wales	Count	393	670	864	478	82	2,487
	%	15.80	26.94	34.74	19.22	3.30	
Scotland	Count	257	162	93	53	19	584
	%	44.01	27.74	15.92	9.08	3.25	
N. Ireland	Count	89	244	227	69	12	641
	%	13.88	38.07	35.41	10.76	1.87	
Total	Count	739	1,076	1,184	600	113	3,712
	%	19.91	28.99	31.90	16.16	3.04	

7.6.2 Comparison on a type-by-type basis

Figure 7.8 illustrates at a type-specific level the average final site-specific EQR of all UK river macrophyte survey sites. Sites are ranked from left to right in approximate order of increasing baseline productivity and data includes all surveys in each river type, across all dates. The most striking feature is the strong reduction in mean EQR with increasing productivity with the majority of sites on the most productive lowland rivers falling below the global mean EQR. At a more subtle level the same trend applies when comparing the northern and southern variants of different high-alkalinity river types. Only in low-alkalinity rivers would the majority of sites in each type be classified as being at high ecological status.

The distribution of sites by type and class is summarised in Table 7.5.

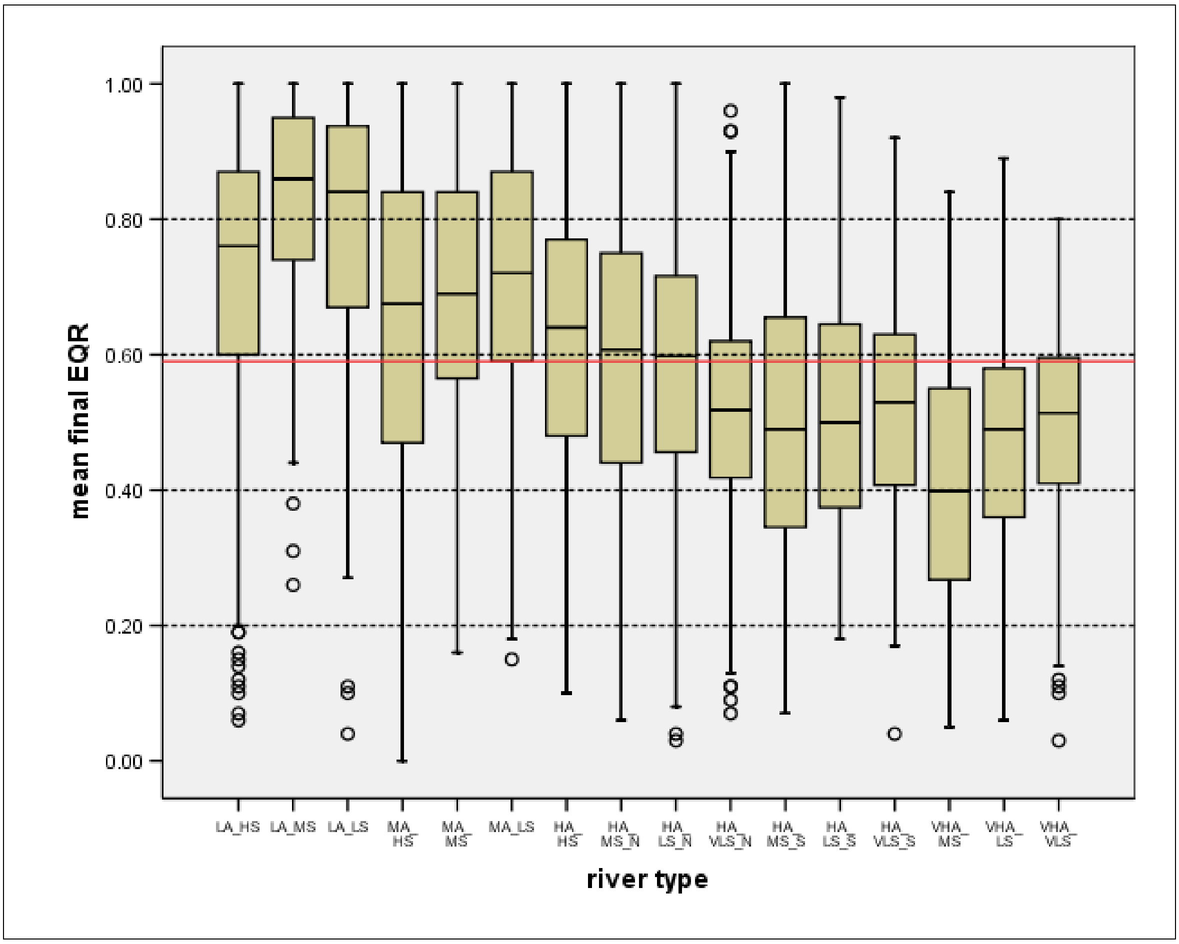


Figure 7.8 Distribution of mean final EQRs for individual survey sites based on all surveys undertaken. Class boundaries are indicated by dashed lines. The global average EQR across all sites (0.59) is shown as a solid red line. Types are ranked in approximate order of increasing productivity.

Table 7.5 Summary of distribution of sites by class and river type

River type		Bad	Poor	Mod	Good	High	Total
Low Alk_Low Gr	n	3	2	7	30	65	107
	%	*2.8*	*1.9*	*6.5*	*28.0*	*60.7*	
Low Alk_Mod Gr	n		3	10	36	95	144
	%	*0.0*	*2.1*	*6.9*	*25.0*	*66.0*	
Low Alk_Hgh Gr	n	10	28	39	103	143	323
	%	*3.1*	*8.7*	*12.1*	*31.9*	*44.3*	
Mod Alk_Low Gr	n	2	15	51	100	100	268
	%	*0.7*	*5.6*	*19.0*	*37.3*	*37.3*	
Mod Alk_Mod Gr	n	1	17	44	77	81	220
	%	*0.5*	*7.7*	*20.0*	*35.0*	*36.8*	
Mod Alk_Hgh Gr	n	16	24	48	67	80	235
	%	*6.8*	*10.2*	*20.4*	*28.5*	*34.0*	
Hgh Alk _Vlow Gr_N	n	7	49	141	66	18	281
	%	*2.5*	*17.4*	*50.2*	*23.5*	*6.4*	
Hgh Alk_Low Gr_N	n	9	41	121	120	60	351
	%	*2.6*	*11.7*	*34.5*	*34.2*	*17.1*	
Hgh Alk_Mod Gr_N	n	9	47	73	90	60	279
	%	*3.2*	*16.8*	*26.2*	*32.3*	*21.5*	
Hgh Alk_Hgh Gr	n	5	26	60	73	47	211
	%	*2.4*	*12.3*	*28.4*	*34.6*	*22.3*	
Hgh Alk _Vlow Gr_S	n	4	25	54	33	8	124
	%	*3.2*	*20.2*	*43.5*	*26.6*	*6.5*	
Hgh Alk_Low Gr_S	n	1	53	53	56	10	173
	%	*0.6*	*30.6*	*30.6*	*32.4*	*5.8*	
Hgh Alk_Mod Gr_S	n	5	41	45	31	16	138
	%	*3.6*	*29.7*	*32.6*	*22.5*	*11.6*	
VHgh Alk_VLow Gr	n	11	90	232	110	7	450
	%	*2.4*	*20.0*	*51.6*	*24.4*	*1.6*	
VHgh Alk_Low Gr	n	15	96	169	67	11	358
	%	*4.2*	*26.8*	*47.2*	*18.7*	*3.1*	
VHgh Alk_Mod Gr	n	16	43	38	18	5	120
	%	*13.3*	*35.8*	*31.7*	*15.0*	*4.2*	

Grey cells indicate modal class for each river type.

7.7 Case studies

7.7.1 SAC rivers

A number of rivers in the UK are designated Special Areas of Conservation (SAC) based on the quality of their aquatic vegetation which fits the description of habitat '3260: Water courses of plain to montane levels with the *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation'. Figure 7.9 examines the variability and mean EQR of macrophyte survey sites on each of the designated rivers. This confirms that in most cases, SAC rivers would generally be classed as being at high or good status. However in a small number of cases (Axe, Lambourne, Wensum), while the vegetation may qualify as a feature of interest, this may be the product of underlying pressures. Alternatively, even in its impacted state, a river may merit selection because it represents the best surviving example of the habitat type across a larger region (Mainstone, 2008). This emphasises the need to place ecological classifications in their context; some lowland rivers situated within SSSIs or SACs may be the best contemporary examples of their vegetation type, even though, with an EQR close to the good-moderate boundary, they exhibit relatively large deviation from reference conditions.

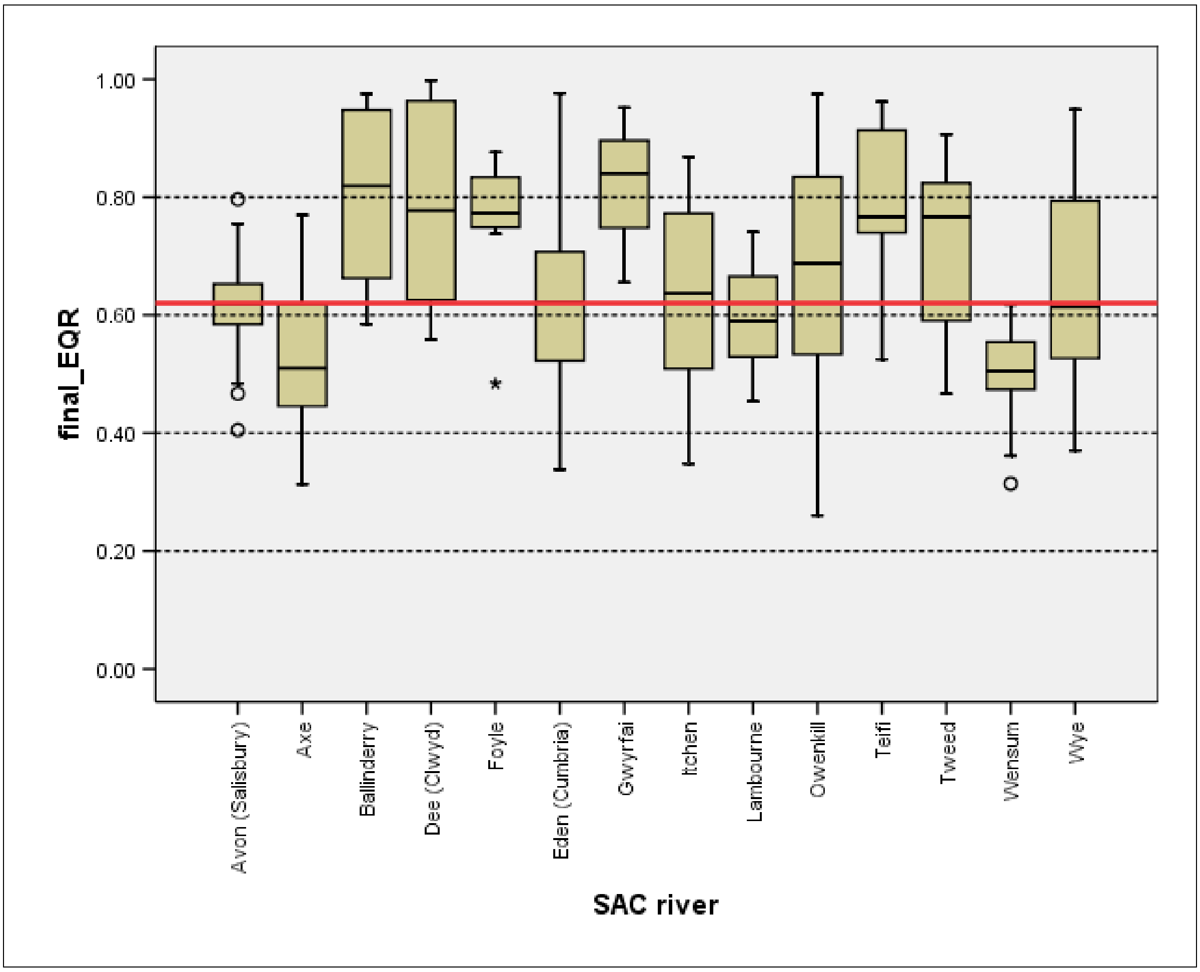


Figure 7.9 Comparison of mean final EQR of 14 UK rivers designated as Special Areas of Conservation based on the quality of *Callitricho-Batrachion* vegetation. Dashed lines indicate class boundaries and red line indicated global mean EQR of all river macrophyte survey sites. Means based on 5-25 sites per river.

7.7.2 Longitudinal change in status along major rivers

A consideration of some larger rivers allows a comparison of change in status from source to mouth. The rivers selected here for analysis have been subject to intensive survey effort mainly for conservation inventory and purposes of compliance monitoring. Figures 7.10 to 7.15 enable the average status of a river to be assessed relative to the global dataset, and for any longitudinal trends in status to be identified.

When resources for monitoring are limited, a survey at the downstream end of a water body should provide a reasonable assessment of the water body as a whole since it will integrate the effects of the main upstream pressures. Based on an analysis of trends in ecological status in 44 large and well-monitored rivers, the data indicate that ecological status decreases downstream in 32 per cent of cases, increases downstream in 20 per cent and is independent of distance downstream in 48 per cent. This suggests that, in general, the location of a survey point within a water body matters relatively little in relation to distance downstream because there is no evidence that a site located near the downstream end will tend to have a higher or lower EQR than a site at any other point in the water body. Evidently in some cases, a site located at the downstream end will give an unduly optimistic or pessimistic view of the status of the upstream water body, if this is the only site surveyed. Rivers in northern England (Figure 7.15) reveal the clearest trend of decreasing status from source to mouth. These rivers generally rise at high elevations (above 300 m) where anthropogenic pressures are likely to be low and ecological quality therefore high, but pass through industrial towns in their middle and lower reaches with an associated rise in anthropogenic pressures from point sources and channel modifications.

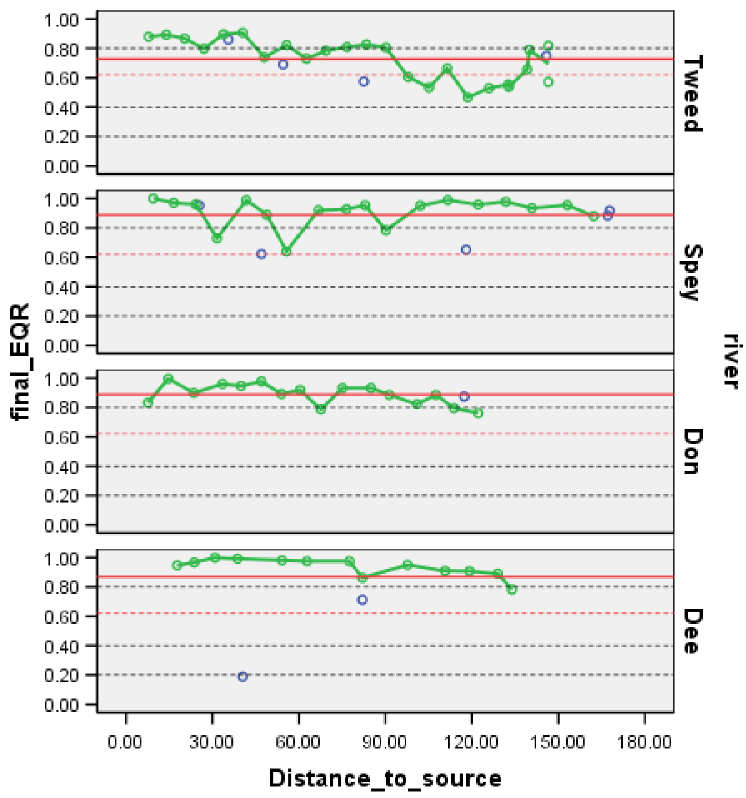


Figure 7.10 Source to mouth change in mean site EQR of four major Scottish rivers. Solid red line shows overall mean EQR for river, dashed red line shows global mean EQR of all river macrophyte survey sites. Blue dots and line = MTR surveys; green dots and line = JNCC surveys. Note the generally high status of all these rivers and substantially higher than average status of almost all sites. Only the Tweed displays evidence of a decline in quality in its downstream reaches. MTR sites are shown but not connected by an interpolation line due to the scarcity of sites.

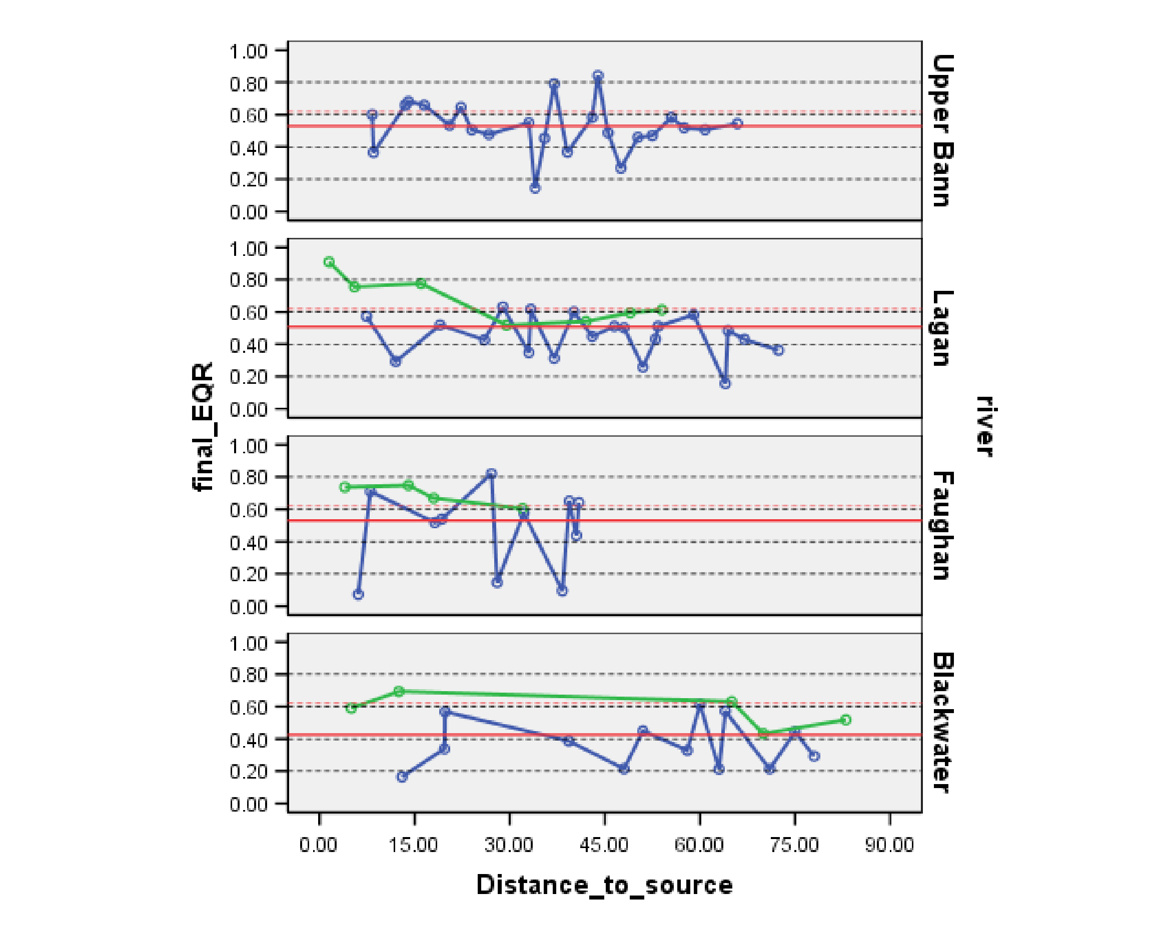


Figure 7.11 Source to mouth change in mean site EQR of four major Northern Irish rivers. Solid red line shows overall mean EQR for river, dashed red line shows global mean EQR of all river macrophyte survey sites. Blue dots and line = MTR surveys; green dots and line = JNCC surveys. Note the generally lower than average status of these rivers. Only the Lagan displays evidence of a decline in quality in its downstream reaches. JNCC surveys located away from point sources reveal higher quality in upper reaches.

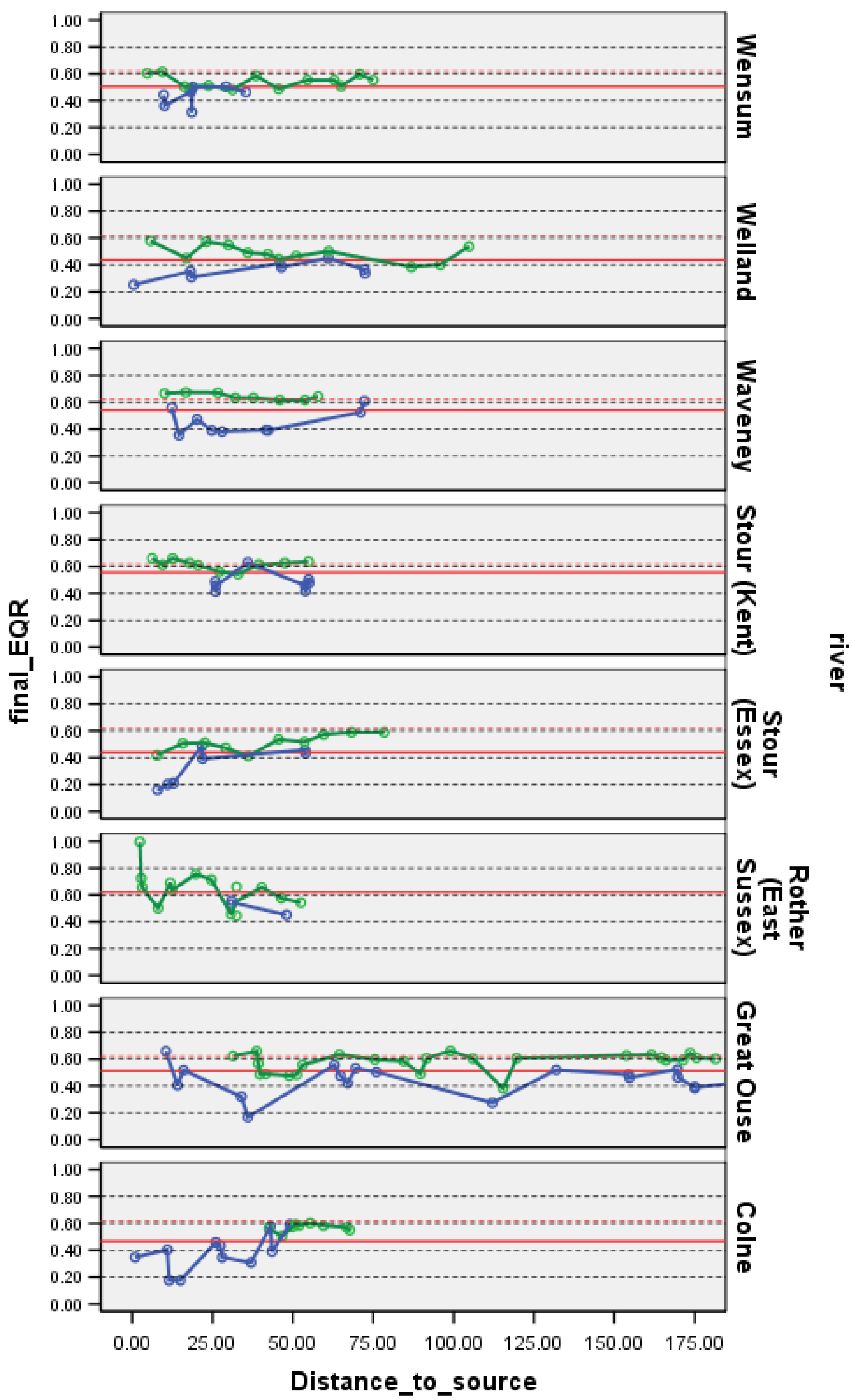


Figure 7.12 Source to mouth change in mean site EQR of major rivers in eastern England. Solid red line shows overall mean EQR for river, dashed red line shows global mean EQR of all river macrophyte survey sites. Blue dots and line = MTR surveys; green dots and line = JNCC surveys. Note the lower than average status of most of these rivers. In the Rother quality decreases downstream and increases downstream in the Colne and Essex Stour.

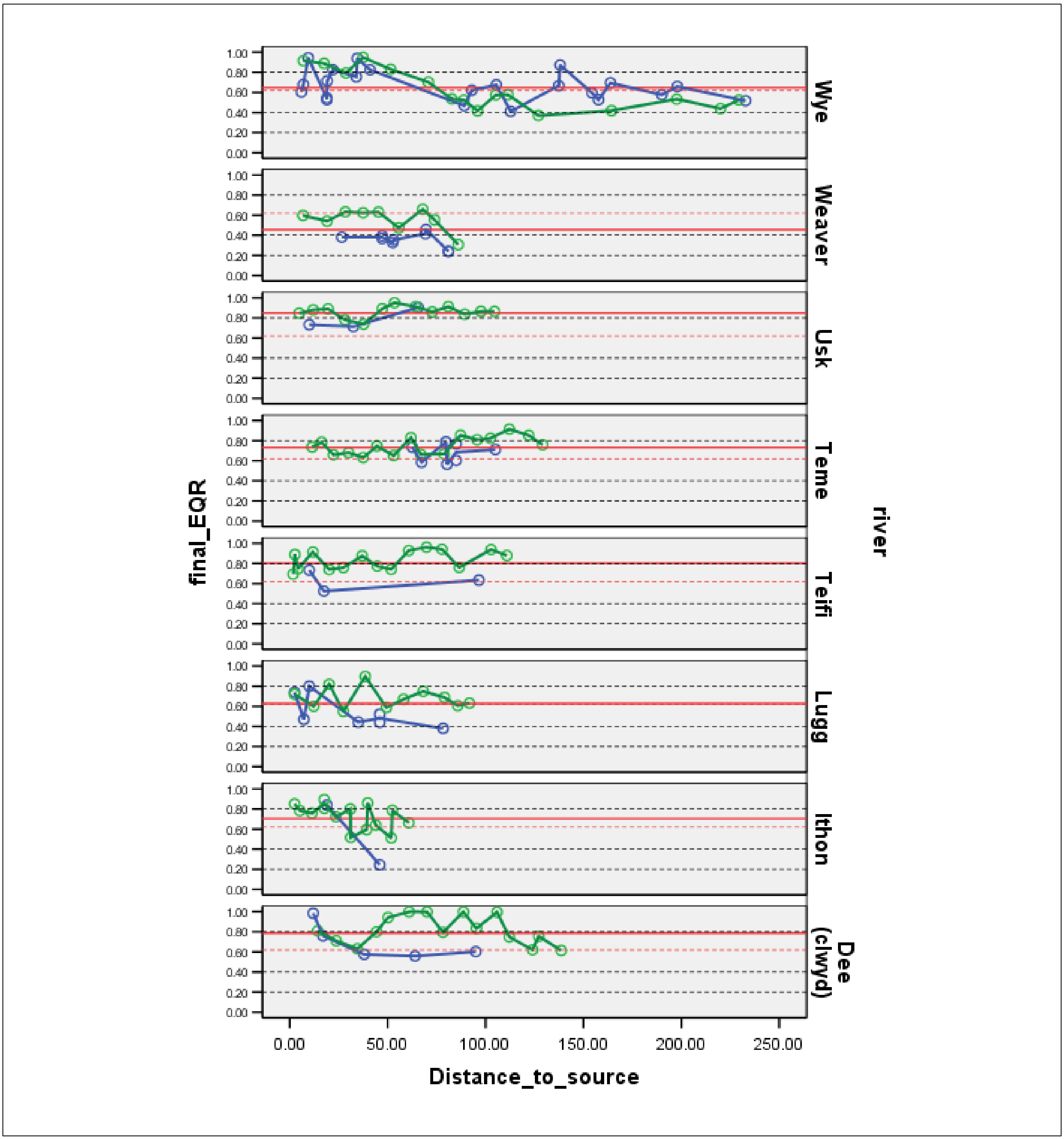


Figure 7.13 Source to mouth change in mean site EQR of major rivers in Wales, Welsh borders and west Midlands. Solid red line shows overall mean EQR for river, dashed red line shows global mean EQR of all river macrophyte survey sites. Blue dots and line = MTR surveys; green dots and line = JNCC surveys. Note the above average status of most of these rivers. The Wye, Weaver, Lugg and Ithon reveal declining ecological status downstream while the Usk, Teme and Teifi show evidence of slightly increasing status.

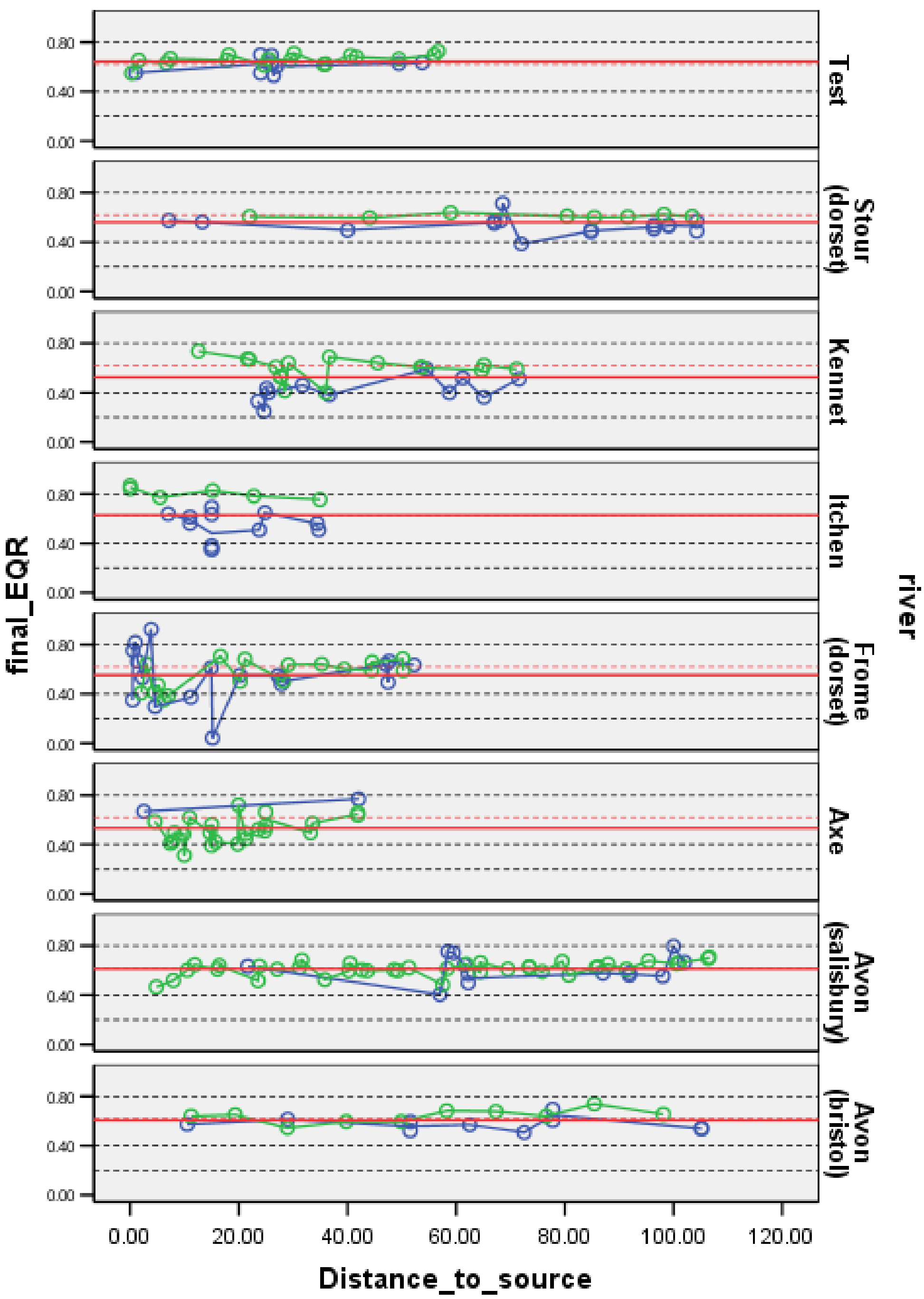


Figure 7.14 Source to mouth change in mean site EQR of major rivers in southern England. Solid red line shows overall mean EQR for river, dashed red line shows global mean EQR of all river macrophyte survey sites. Blue dots and line = MTR surveys; green dots and line = JNCC surveys. Note the average status of most of these rivers. The Itchen reveals declining status downstream, whilst the Axe and Frome indicate increasing status.

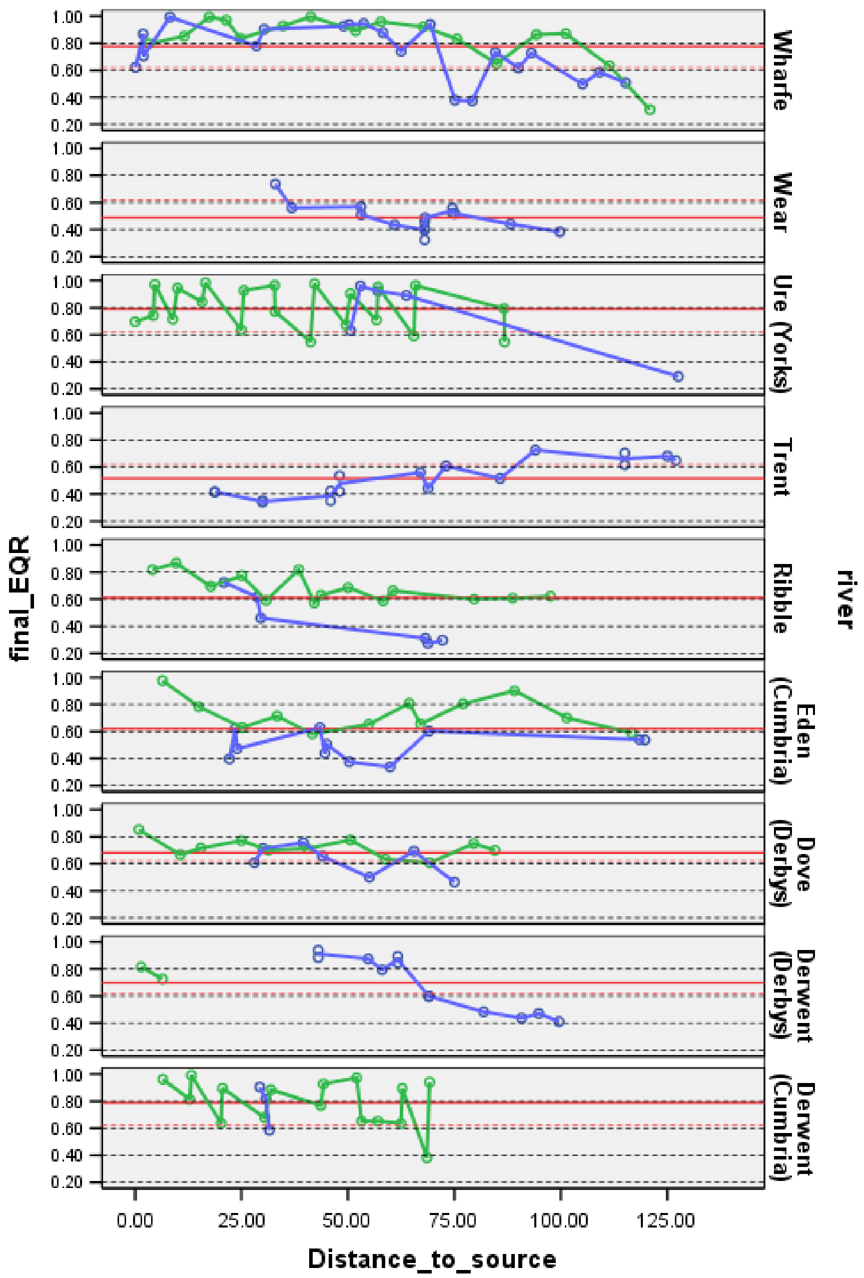


Figure 7.15 Source to mouth change in mean site EQR of major rivers in northern England. Solid red line shows overall mean EQR for river, dashed red line shows global mean EQR of all river macrophyte survey sites. Blue dots and line = MTR surveys; green dots and line = JNCC surveys. Note the above average status of these rivers excluding the industrialised Trent and Wear. With the exception of the Trent and Eden all these rivers indicate declining ecological status from source to mouth reflecting the high elevation of the source and the concentration of anthropogenic pressures in the lower catchment.

Two points to note when comparing longitudinal change in status of these rivers are the clear tendency for the data from MTR surveys to show greater spatial variability and provide lower EQRs than those provided by JNCC survey data. At a 1,000-m scale local spatial variability attributable to habitat variability will be largely subdued, while impacts apparent at a 100-m scale will be buffered by the inclusion of unaffected reaches. Because MTR survey data was often obtained to assess changes in vegetation upstream and downstream of specific point sources, a higher spatial variability and lower overall EQR is to be expected. Figure 7.16 shows that this effect is most severe at the lowest EQR values, with rivers that would be classified as poor on the basis of MTR survey data being classified as moderate using JNCC survey data. In general, sampling of a river water body directly downstream of a major point source would not be a good strategy for sampling river macrophytes for WFD purposes, unless the water body was characterised by a high density of point sources. In this sense the data from JNCC surveys may offer a more reliable guide to the likely ecological status of some rivers than data contributed by MTR surveys.

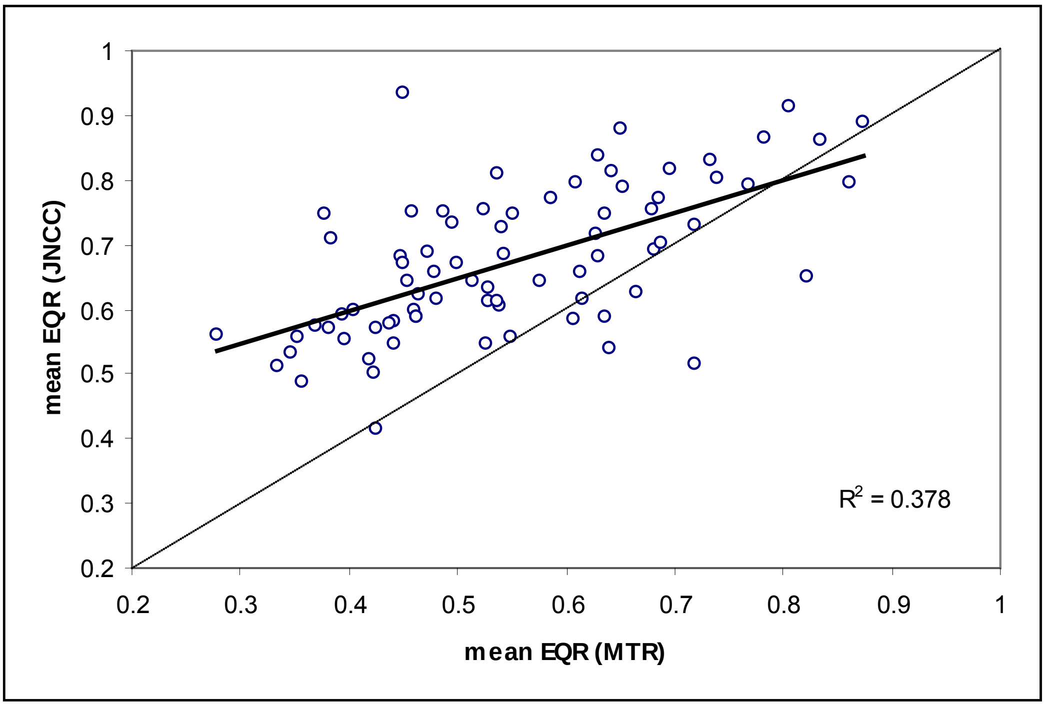


Figure 7.16 Mean EQR of 78 rivers as indicated by MTR and JNCC survey methods. Solid line indicates the fitted relationship. The dashed line indicates the distribution points would follow in a one-to-one relationship. Each river was assessed at a minimum of 10 sites.

7.8 Application to environmental standards for nutrients in rivers

7.8.1 Pressure-response relationships

The relationship between RMNI and either SRP or TON was presented in Figure 4.1. While this looks like a pressure-response relationship, it is in fact deceptive. The relationship is driven primarily by the natural gradient in macrophyte composition found in rivers distributed across the full natural productivity gradient (where productivity is determined by supply of phosphorus from weathering of the underlying geology), rather than being driven by nutrient enrichment *pressure* relative to a background level, as would be caused by anthropogenic inputs. Hence the relationship is dependent on a large degree of covariation between alkalinity and phosphorus. In effect this type of relationship views change in vegetation across a spatial gradient of phosphorus concentration as a proxy for the change that might occur over time at a site through enrichment, yet in reality, changes on this type of scale (three orders of magnitude of SRP) could never occur at an individual site owing to basic environmental constraints. Consequently the true biological response is a much dampened version of that which might be expected from Figure 4.1. As an analogy one could consider plotting phosphorus concentration of a large number of rivers against the proportion of tilled land in the upstream catchment. Although the concentration of phosphorus will undoubtedly increase with increasing tillage, greater tillage will largely reflect an underlying gradient from upland, base-poor, infertile catchments to lowland, base-rich, naturally fertile ones. If this underlying pattern was removed, the SRP signal attributable exclusively to cultivation would be substantially reduced.

To extract the strict pressure-response relationship from covariation with determinants of the natural fertility gradient, some form of variance partitioning is necessary. In effect this analysis tests whether there is a significant relationship between a pressure and the biological response once covariation between the response and driving variables, such as alkalinity, has been removed. It is somewhat arbitrary whether this covariation is removed when first developing metrics (for example, by minimising the covariation between a metric, such as RMNI, and alkalinity) or at a later stage (by incorporating alkalinity as a predictor of values of metrics expected at reference condition, and then expressing the observed value as a ratio of the expected, for example). If there is not a unique biological response to phosphorus this means that the response to an increase in phosphorus at a given level of alkalinity caused by anthropogenic loading is too weak to be of use. Some caution is required in interpreting variance portioning tests since they are a relatively blunt instrument and there may an element of 'throwing the baby out with the bath water'. As an example, some of the biological response to phosphorus will be shared with alkalinity because some anthropogenic enrichment by P will involve application of rock phosphate to agricultural land. This will elevate both the alkalinity and SRP of a site background levels. Variance partitioning would, however, exclude this shared element of the response.

Relationships between metrics and pressures are useful for diagnostic purposes. For example, RMNI is presented as a metric that uses composition of macrophytes as a measure of likely nutrient status. However, LEAFPACS is a multimetric system which uses a suite of metrics to provide complementary sensitivity to various pressures, most notably eutrophication. This multimetric approach appears to be necessary to deliver sensitivity across the full width of a pressure gradient as well as in the full range of river types, thereby overcoming the weaknesses inherent within single metric-based classification. Although RMNI might be seen as the primary measure of macrophyte response to nutrient enrichment in rivers it is clear that other metrics, such as richness or algal cover, can modify the signal provided by composition alone. Thus, species-poor, low-alkalinity rivers with little indication of enrichment may actually be acidified, while high species richness of fertile lowland rivers will to some extent mitigate against compositional changes. Consequently, it is instructive to explore the relationship between a pressure and resulting biology based on the overall biological response, as reflected in the final EQR for a water body, rather than on the basis of single metrics. Depending on the outcome it may be possible to use such an analysis to derive environmental standards for variables that support high or good ecological status.

When examining raw metric values (such as RMNI or N_TAXA) it is simple to evaluate these by plotting them directly against a pressure. However, a different approach is needed when dealing with the multimetric response expressed in the form of an EQR since, at a global level, the EQR will not necessarily decrease with increasing pressure. Thus, a final EQR of 0.8 may occur at, say, 100 ug/l SRP in a high-alkalinity lowland river in southern England but at the same nutrient concentration a low-alkalinity upland river would be grossly impacted in terms of its macrophytes and would probably have an EQR below 0.2. Even at a type-specific level the relationship between EQR and pressure is likely to be obscured by the variation in background nutrient concentrations between the upper and lower limits of the alkalinity band which defines each river type.

Figure 7.17 attempts to represent the eutrophication pressure-macrophyte response relationship by plotting river SRP concentrations relative to the alkalinity of the river with macrophyte classes overlain. As a basic minimum, if macrophytes reflect the anthropogenic nutrient signal one would expect a worsening of macrophyte class with increasing P for a given alkalinity and that macrophyte classes should show some separation and be arranged in a consistent manner. Figure 7.18 largely confirms these patterns, especially at higher alkalinity, and is a simple graphic indication of a likely underlying pressure-response relationship. Poor separation of the classes moderate to bad, however, indicates that other factors may lead to failure to achieve good ecological status when nutrient concentrations are not especially high.

To determine statistically whether there was a pressure-response relationship, the final EQR was used to predict river SRP having first fit the effects of background variables. To accommodate the potential influence of slope on productivity a Morpho Edaphic Index (MEI) for rivers was devised in which slope in rivers takes the same role as depth in lakes (increasing slope reduces potential productivity for a given alkalinity). The river MEI (MEIr) was calculated as Log_{10}(alkalinity (meq/l)/slope (m/km). Alkalinity, slope, MEI and GIG type (a dummy variable used to discriminate between high-alkalinity rivers in CB-GIG on soft calcareous geologies from those in N-GIG or A-GIG on hard limestones) were used as predictive variables. Prior to fitting, model checks were carried out to identify any major SRP outliers by correlating SRP and alkalinity on a class-specific basis. At this point those values falling above the individual 95 per cent confidence interval were removed and the regression was refitted. Slope was never a significant predictor, and somewhat surprisingly, MEI was consistently inferior to alkalinity alone as a predictor of SRP concentrations.

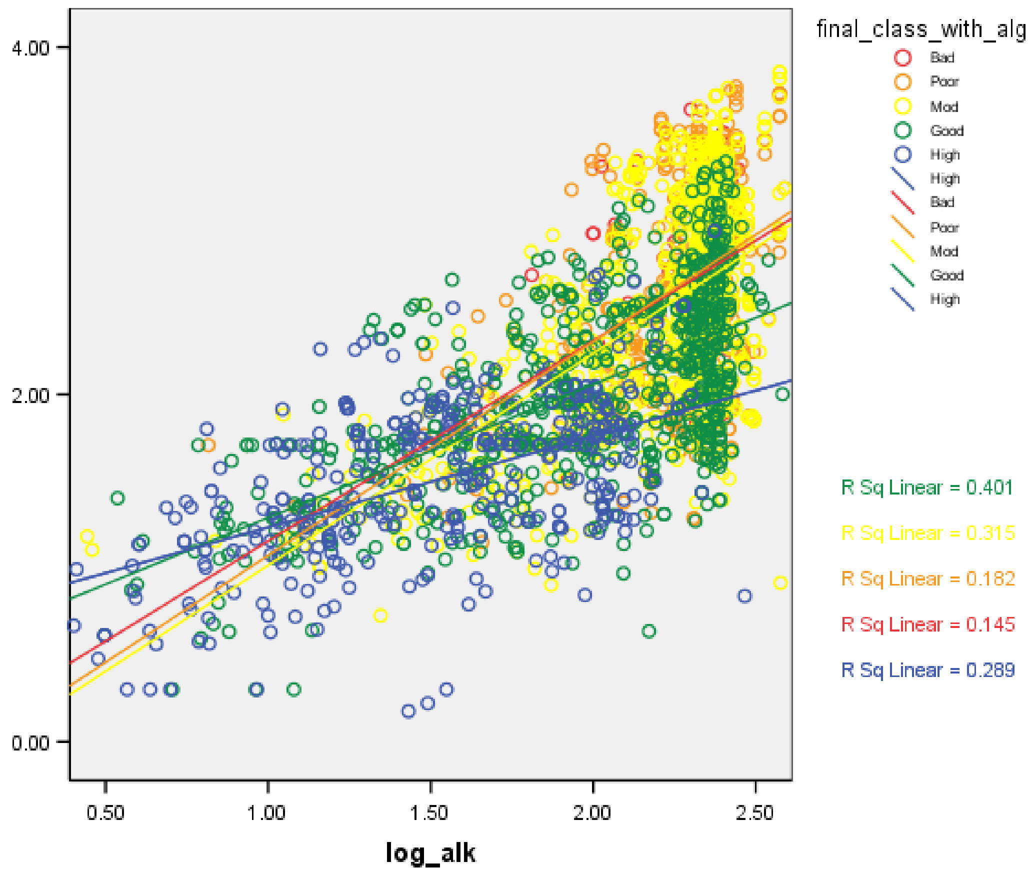


Figure 7.17 Annual mean SRP versus $\log_{10}$ alkalinity stratified by macrophyte-based ecological status. All plotted lines are significant at p = 0.01.

The results of the analysis are summarised in Table 7.6. As would be expected alkalinity is a highly efficient predictor of SRP, with alkalinity, MEIr and GIG type together explaining 53 per cent of the variation. Macrophyte EQR explains an additional four per cent of variation, although this is still highly significant (p = <0.0001). These figures compare closely with the same analysis for lakes, although for rivers the prediction error is about one-third higher. While the amount of variation in SRP that can be linked exclusively to macrophyte response appears small, this value should not be thought of in absolute terms but rather as a fraction of the variation left unexplained after fitting alkalinity, MEI and GIG-type, which will always be the main drivers of the concentration of SRP in a river.

Given that, at best, 65-75 per cent of total variation in SRP could be explained by a full suite of predictors, the four per cent ‘explained’ by macrophyte ecological quality represents perhaps 20-30 per cent (four per cent of the unaccounted for 12-22 per cent) of the total that could be explained after fitting alkalinity, MEI and GIG-type. The four per cent variation in SRP explained by macrophytes also needs to be put into context. For example, the extent of intensive land cover in the catchment should be a measure of anthropogenic nutrient loading to a river but this variable only explains 2.2 per cent of the variation in SRP after fitting alkalinity, MEI and GIG type. Interestingly, the component of variation in SRP that can be explained by macrophytes is largely undiminished (reduced from four to three per cent) and remains highly significant *after* fitting the land cover variable. This shows macrophytes have unique indicative value.

Macrophytes are not especially poor predictors of nutrient status (or vice versa), as has sometimes been asserted (see Demars & Thiebaut, 2008), rather variance partitioning is a conservative tool for extracting a signal. The small pure effect of phosphorus on macrophytes should not be attributed to their limited bioindication potential or the wide ecological amplitude of many individual species. It is purely a reflection of the extent of covariation in the global relationship between alkalinity and SRP. A note of caution should also be added about testing for links between macrophyte-based ecological status and water chemistry using datasets that will inevitably be much smaller than that used here. While it is possible that such analyses will reveal a stronger link than found here, it may equally be the case that no relationship is found if, for example, the range of river types, MEI, phosphorus gradient or EQR is rather narrow. More specialised studies (such as Demars & Edwards, 2009) will have difficulty in achieving the same range of variation in SRP for a given value of alkalinity or MEIr.

Having demonstrated statistically the unique and significant relationship between SRP and river macrophytes, the global relationship was refined further to inferr environmental standards. Two separate models were developed, one based on Northern-Atlantic rivers, and the second on southern-continental type rivers in which alkalinity, MEIr, and final EQR were used as model terms. In the second model, low- and moderate-alkalinity rivers across the UK were included to maximise the gradient of SRP and macrophyte EQR, but Northern-Atlantic high-alkalinity rivers were excluded. Both models were first screened for outliers as described above. In contrast to lakes, the macrophyte-SRP relationship differs little in strength between the generally less fertile pool of Northern-Atlantic rivers (pure effect of 4.7 per cent) and the more fertile southern-continental lakes (pure effect of four per cent). Any difference presumably reflects the greater potential for nutrient limitation in the less fertile Northern-Atlantic rivers and the greater relative influence of physical factors such as flow disturbance and physical habitat variability on macrophytes in more productive rivers.

The same analyses were repeated for TON. Nitrogen can be modelled more effectively than phosphorus using alkalinity, MEI and GIG type as predictors (r^2 = 73 per cent). The additional variation in TON explained by the river macrophyte overall EQR was much smaller than for SRP (one to two per cent) although it was always significant. There was no suggestion that nitrogen is proportionally more important in rivers from one particular GIG type. A class-specific analysis (Figure 7.18) indicates that high status sites tend to have lower N concentrations for a given alkalinity, but that the separation of other classes is rather poor.

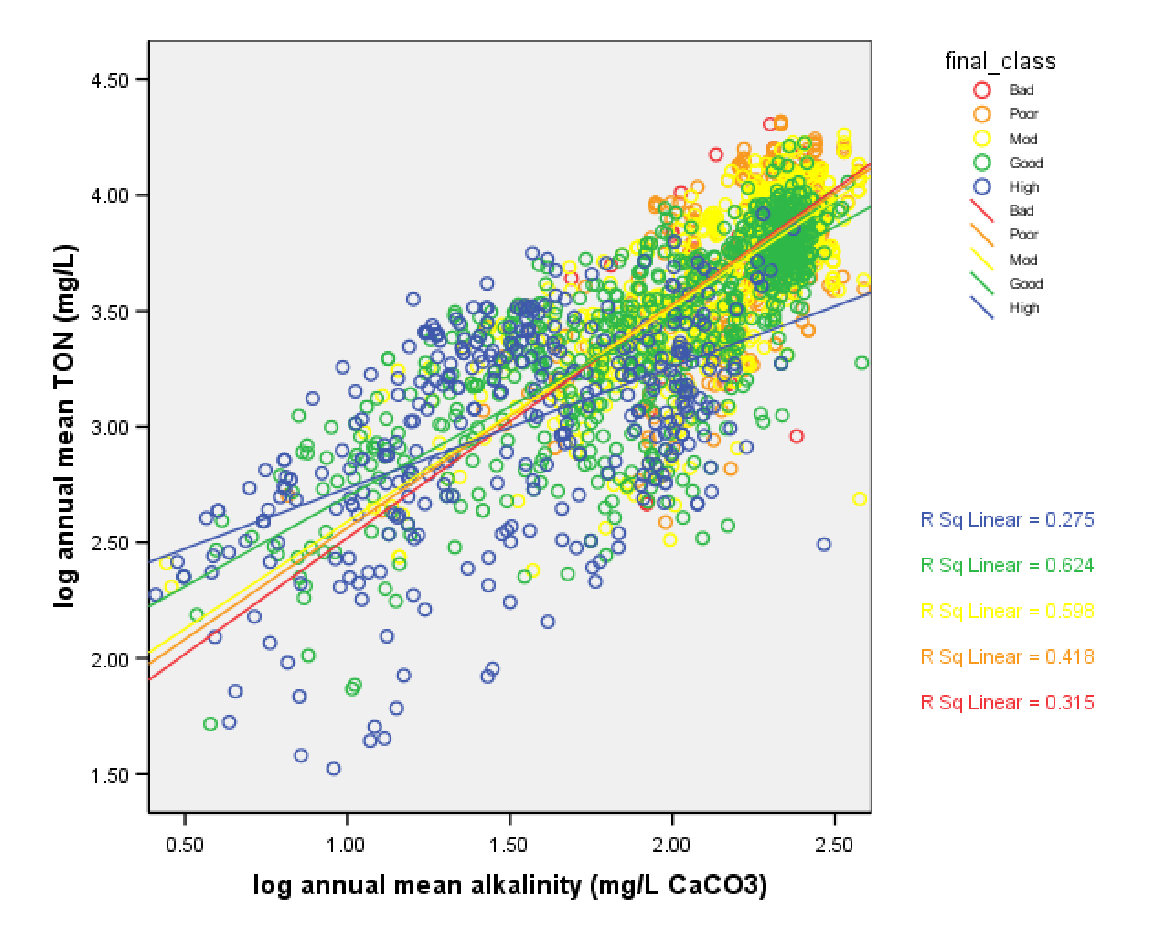


Figure 7.18 Annual mean TON versus $\log_{10}$ alkalinity stratified by macrophyte-based ecological status. All plotted lines are significant at p = 0.01.

7.8.2 Deriving environmental standards for nutrients in rivers

Having established models relating river SRP to core typing variables (alkalinity and slope, via MEIr), and shown that macrophyte EQR is an additional term in these models, such models can be used to predict SRP values associated with a particular EQR for any value of alkalinity and MEIr. Consequently, these models can be used to predict the values of SRP associated with class boundaries in each river type. For example, the median values of alkalinity and slope found across the UK resource of high-alkalinity, low-gradient southern-continental rivers could be used to derive an MEI value and then used, in conjunction with an EQR of 0.8, to estimate the average SRP at the high-good boundary in this particular river type. Following the rationale employed by UKTAG (2006) this value could be treated as a standard to support high ecological status of macrophytes in this river type since it represents the highest predicted value of SRP that a river could have whilst supporting macrophytes at high ecological status. Similarly an EQR value of 0.6 could be employed in the model to predict the standard for good ecological status since this EQR represents the good-moderate boundary. Hence, for a typical high-alkalinity, low-gradient southern-continental river SRP concentrations not exceeding 142 and 222 ug/l would typically be required to support macrophytes at high or good ecological status respectively. Such standards would be less suitable as targets (such as for restoration) and the SRP associated with the EQR in the middle of a class might be more appropriate for this purpose. Thus, one could use the predicted SRP associated with the middle of high or good status (EQR of 0.9 or 0.7 respectively), or, as discussed below, a value taking into account prediction error.

The SRP and TON predicted by global and GIG-specific models was determined and compared. In contrast to lakes there was no apparent advantage in the use of GIG-specific models for rivers, in terms of the spread of SRP values between classes (typically four-fold variation between reference and the P/B boundary) or the prediction error within the relationship. Consequently, the standards provided are based on the simpler global relationship, using alkalinity, MEI and GIG type as predictors (Table 7.6). The full set of values for phosphorus and nitrogen for a range of river types, based on their macrophyte response, are presented in Tables 7.7 and 7.8.

Since the final EQR for a site is partly a product of the method used to combine different metrics, it may be instructive to compare the nutrient standards derived using the full EQR with those associated with the RMNI EQR since this metric is intended to reflect the main component of ecological change in relation to nutrient pressures. A comparison of standards derived from the full EQR and RMNI EQR might suggest the need to revisit combination rules in the future. Standards associated with the independent use of RMNI EQR, based on the models in Table 7.6, are given in Tables 7.9 and 7.10. The pure effect of RMNI EQR was slightly greater than that of the overall EQR (5.2 and 2.8 per cent for SRP and TON respectively), and standards based on RMNI alone were consistently more stringent than those derived using the overall EQR, especially in the most productive lowland rivers. The greater weight given to richness metrics in the overall classification of more productive rivers relaxes their nutrient standards. This is because it is possible to find highly diverse macrophyte assemblages in moderately enriched lowland rivers in which vegetation composition is comparatively heavily altered. In this respect, the current weight assigned to richness metrics in the combination rules may slightly overcompensate for the impact of anthropogenic nutrient inputs on composition. However, the initial outcomes of intercalibration in CB-GIG indicate that the overall UK classification of lowland rivers is relatively precautionary compared to some other European countries. Thus, while reducing the weight assigned to richness in the combination rules for lowland rivers might generate more ecologically reasonable nutrient standards, it could also increase the disparity between the UK classification and that of other European countries.

Table 7.6 Models relating SRP and TON to alkalinity, MEI, GIG type and macrophyte EQR to test existence of unique relationships between macrophyte ecological status and anthropogenic nutrient signal

	Phosphorus (SRP)		Nitrogen (TON)	
	Overall EQR	RMNI EQR	Overall EQR	RMNI EQR
Terms	coefficient	coefficient	coefficient	coefficient
Constant	1.955	1.882	2.552	2.585
Log_{10} alkalinity	0.306	0.295	0.522	0.503
MEIr	0.148	0.113	0.040	0.025
CGIG = 1	0.338	0.235	0.234	0.181
EQR	-0.962	-0.892	-0.389	-0.419
r^2 basic subset	53.0	53.0	73.0	73.0
r^2 macrophyte EQR	57.0	58.2	74.5	75.8
SE	0.441	0.435	0.222	0.217
n in model	2040	2040	2270	2270

Table 7.7 Predicted values of SRP (ug/l) in UK rivers to support macrophytes at different ecological status. Types are ranked in order of ascending productivity.

Alkalinity	Slope	GIG	Slope	Alk.	MEIr	Ref	HG	GM	MP	PB
Low	High	NA	15	7	-2.03	9	11	17	34	53
Low	Mod	NA	5	7	-1.55	11	13	20	40	62
Low	Low	NA	1.5	7	-1.03	13	16	24	47	74
Moderate	High	NA	15	25	-1.48	16	20	31	60	94
Moderate	Mod	NA	5	25	-1.00	19	23	36	71	110
Moderate	Low	NA	1.5	25	-0.48	22	28	44	85	132
High	High	NA	15	100	-0.88	30	37	58	113	176
High	Mod	NA	5	100	-0.40	35	44	68	133	207
High	Low	NA	1.5	100	0.12	42	52	82	159	247
High	V. low	NA	0.5	100	0.60	49	62	96	187	291
High	Mod	SC	5	100	-0.40	77	96	149	289	451
High	Low	SC	1.5	100	0.12	92	114	178	346	538
High	V. low	SC	0.5	100	0.60	108	134	209	407	633
Very high	Mod	SC	5	220	-0.06	110	137	213	414	644
Very high	Low	SC	1.5	220	0.47	131	163	254	494	770
Very high	V. low	SC	0.5	220	0.94	154	192	299	582	906

[1]Where slope is the approximate median slope value as m/km of all sites in that type, standardised across types for ease of comparison.
[2]Where alkalinity is the median alkalinity as mg/l $CaCO_3$ of all sites in that type, standardised across types for ease of comparison.
[3]Where MEIr = $\log_{10}$ (alkalinity(as meq/l)/mean depth (m)) and is based on the median alkalinity and slope of rivers in each type.
[4]Where 1.0, 0.8, 0.6, 0.4 and 0.2 represent the EQR for reference and boundaries for high-good, good-moderate, moderate-poor and poor-bad respectively.

Table 7.8 Predicted values of TON (ug/l) in UK rivers to support macrophytes at different ecological status

Alkalinity	Slope	GIG	Ref	HG	GM	MP	PB
Low	High	NA	0.33	0.40	0.48	0.57	0.68
Low	Mod	NA	0.35	0.42	0.50	0.60	0.71
Low	Low	NA	0.37	0.44	0.52	0.63	0.75
Mod	High	NA	0.68	0.82	0.98	1.17	1.40
Mod	Mod	NA	0.71	0.85	1.02	1.22	1.46
Mod	Low	NA	0.75	0.89	1.07	1.28	1.53
High	High	NA	1.49	1.78	2.13	2.54	3.04
High	Mod	NA	1.55	1.86	2.22	2.66	3.18
High	Low	NA	1.63	1.95	2.33	2.79	3.34
High	Very low	NA	1.70	2.04	2.44	2.91	3.49
High	Mod	SC	2.66	3.18	3.81	4.55	5.45
High	Low	SC	2.79	3.34	4.00	4.78	5.72
High	Very low	SC	2.92	3.49	4.18	4.99	5.97
Very high	Mod	SC	4.14	4.96	5.93	7.09	8.49
Very high	Low	SC	4.35	5.20	6.22	7.44	8.91
Very high	Very low	SC	4.54	5.44	6.50	7.78	9.31

[1]Where 1.0, 0.8, 0.6, 0.4 and 0.2 represent the EQR for reference and boundaries for high-good, good-moderate, moderate-poor and poor-bad respectively.

Table 7.9 Predicted values of SRP in UK rivers to support macrophytes at different ecological status based only on the metric RMNI

Alkalinity	Slope	GIG	Ref	HG	GM	MP	PB
Low	High	NA	10	13	19	35	53
Low	Mod	NA	12	14	21	40	60
Low	Low	NA	13	16	25	46	69
Mod	High	NA	17	21	32	59	89
Mod	Mod	NA	19	24	36	67	101
Mod	Low	NA	22	27	41	77	115
High	High	NA	30	37	56	104	157
High	Mod	NA	34	42	63	118	177
High	Low	NA	39	48	73	135	203
High	Very low	NA	44	55	82	152	230
High	Mod	SC	59	72	109	202	305
High	Low	SC	67	83	125	231	349
High	Very low	SC	76	94	141	262	395
Very high	Mod	SC	81	100	150	279	420
Very high	Low	SC	93	114	172	319	481
Very high	Very low	SC	105	129	195	361	545

Table 7.10 Predicted values of TON (mg/l) in UK rivers to support macrophytes at different ecological status based only on the metric RMNI

Alkalinity	Slope	GIG	Ref	HG	GM	MP	PB
Low	High	NA	0.35	0.42	0.51	0.62	0.75
Low	Mod	NA	0.36	0.43	0.52	0.64	0.77
Low	Low	NA	0.37	0.45	0.54	0.66	0.80
Mod	High	NA	0.68	0.82	1.00	1.21	1.47
Mod	Mod	NA	0.70	0.85	1.03	1.25	1.51
Mod	Low	NA	0.72	0.87	1.06	1.28	1.56
High	High	NA	1.41	1.71	2.08	2.52	3.06
High	Mod	NA	1.45	1.76	2.14	2.59	3.14
High	Low	NA	1.50	1.82	2.20	2.67	3.24
High	Very low	NA	1.54	1.87	2.26	2.74	3.33
High	Mod	SC	2.20	2.67	3.24	3.93	4.77
High	Low	SC	2.27	2.75	3.34	4.05	4.91
High	Very low	SC	2.33	2.83	3.43	4.16	5.05
Very high	Mod	SC	3.34	4.05	4.91	5.96	7.23
Very high	Low	SC	3.44	4.18	5.06	6.14	7.45
Very high	Very low	SC	3.54	4.29	5.21	6.31	7.66

7.8.3 Interpretation of nutrient standards

The purpose of classification tools is not the prediction of environmental conditions with high precision, as is the case, for example, in the application of transfer functions to diatom data from sediment cores to reconstruct changes in water chemistry. Thus, the relationships derived here and used to generate the figures in Tables 7.7 and 7.8 are correlational not causal and have relatively poor precision. The values should be considered indicative rather than prescriptive. Used for guidance purposes, increasing deviation above these values represents a greater risk that a river of a given type will not achieve high or good ecological status as a result of nutrient enrichment. Conversely achieving values lower than those presented should in no way be interpreted as a guarantee that the desired ecological status will be attained. However, the figures provide an insight to the scale of reductions in nutrient concentrations needed on some rivers before ecological benefits are likely. Thus, while small-scale reductions may serve to alleviate some undesirable disturbances, such as excessive filamentous algal growth, substantial reductions will often be needed to support meaningful changes in ecological quality (Hilton *et al*., 2006).

Several points need to be made with reference to the concentrations of nutrients associated with class boundaries. Firstly, for a given amount of change in nutrient concentration, macrophytes in streams may be rather less sensitive than other quality elements in view of the buffering effects of flow-related disturbance. Consequently, if the underlying hydromorphology and associated buffering processes have been compromised, lower nutrient standards would be necessary until the hydromorphology had been restored. The values presented cannot be treated as values to protect the overall ecological status of a site, or for that matter, to protect the status of other interconnected water bodies, such as downstream lakes.

Secondly, although it is possible to find rivers with macrophyte assemblages that indicate high or good ecological status when nutrient concentrations are higher than the true baseline, these almost certainly represent the tail end of a distribution, most of which can no longer be found in the modern environment. Thus, historical records, export coefficient modelling and reconstructions based on palaeo-channel sediment cores indicate that nutrient concentrations in lowland base-rich rivers a century ago were in the order of 50-100 ug/l SRP and 2-3 mg/l TON (Johnes, 1996; Hilt *et al*., 2008). These values are less than half those predicted in our analysis but are still an order of magnitude higher than concentrations that would be encountered in truly pristine forested catchments, even on well-buffered geology, such as can still be found in parts of the US (e.g. Likens *et al*., 1971).

Concentrations of available phosphorus and nitrogen, as low as 5 ug/l and 100 ug/l respectively, are clearly unachievable at present human population densities, anywhere in lowland Europe, even with the most radical reform of land management practices and resource use. Therefore, more pragmatic values need to be sought that align ecological status with reference conditions based on minimally impacted sites, or which are equivalent to sustainable agriculture and resource use. For example, Herlihy *et al*. (2008) used thresholds of 150 ug/l total phosphorus (TP) and 4.5 mg/l total nitrogen (TN) in screening for least disturbed sites within the southern and temperate plains ecoregions of the US to identify candidate reference sites for measuring biotic integrity. However, in the less disturbed Appalachians, upper mid-west and coastal plain ecoregions it was possible to apply thresholds of 20-30 ug/l TP and 0.75-1.0 mg/l TN. In comparing a range of methods for determining reference nutrient concentrations in the US, Dodds & Oakes (2004) concluded that potential reference stream sites with values exceeding 60 ug/l TP or 600 ug/l TN should generally be treated as suspect. Stevenson *et al*. (2008) employed thresholds in algae-P relationships to guide the setting of ecologically-based nutrient standards for streams in the Mid-Atlantic Highlands of the US. Here, it was possible to identify ecological thresholds of 10-20 ug/l TP, thereby supporting land use TP models, and the distribution of TP values in reference sites which suggested a value of 10-12 ug/l TP to protect high quality biological assemblages. Such values are reasonably consistent with the outputs of our analyses for low- and moderate-alkalinity streams in the UK. Evidently a greater degree of pragmatism will be required to identify concentrations of nutrients compatible with good ecological status in lowland base-rich rivers in the UK or Europe as a whole (for example, below100 ug/l SRP), than is perhaps the case in the US.

Thirdly, river sites with matched macrophyte survey data and available nutrient data are likely to represent a highly biased sample of the true population of sites and associated nutrient concentrations for a given river type. This effect is amplified by the fact that most nutrient data that is available is itself derived from routine assessment points, which in many cases have been located to monitor specific discharges, and therefore are probably atypical of rivers of a particular type at a particular status. The predicted values presented above reflect the average nutrient concentration for a given combination of alkalinity, MEIr and EQR. However, these values are already biased towards the upper extremity of the distribution and are therefore misleading if used as standards.

When applied to lakes the approach followed above generated sensible values to protect the ecological status of macrophytes in lakes compared with standards derived from lake chlorophyll. The same cannot be said for rivers, partly for the reasons outlined above. This may also reflect the nature of the lake survey data, which better integrates spatial variation in macrophytes within a lake water body, as well as reflecting the less heavily biased sampling of water chemistry in lakes. A reasonable alternative might be to follow the same approach but instead to use as standards the predicted nutrient concentrations minus their prediction error since this is likely to more closely approximate the true mid-point of the distribution of nutrient concentrations at a given alkalinity, MEI and EQR. The results of such an analysis are given in Table 7.11.

Values obtained by this modified approach provide a more realistic guide to the concentrations of nutrients required across the full range of river types to support macrophytes at high or good ecological status, and compare much more favourably with historical data and existing standards for lakes. On this evidence, for example, concentrations of 100-130 ug/l SRP and 3-4 mg/l TON will typically be required to support macrophytes at good ecological status in the most productive lowland river types in the UK. Nutrient standards derived from this dataset, and based on the predicted value minus the prediction error, are close to the predicted *mid-point* values obtained in a separate analysis of a much smaller independent dataset assembled to compare river classifications based on macrophytes and diatoms. In the latter case the dataset was based exclusively on recent standard surveys undertaken for WFD purposes and was linked to contemporary water chemistry from the same site. This resulted in a much smaller (about 500 sites) but also less noisy dataset, indicating the importance of deriving standards from rigorously screened data if the approach presented here is to be followed. Datasets that cover several biological quality elements linked to contemporary chemistry data also permit better comparisons of the chemistry needed to support each biological element at a given status.

Table 7.11 Concentrations of SRP and TON in UK rivers considered likely to support macrophytes at the indicated ecological status. Types are listed in order of ascending productivity.

			Annual mean SRP (ug/l)					Annual mean TON (mg/l)				
Alkalinity	**Slope**	**GIG**	**Ref[1]**	**HG**	**GM**	**MP**	**PB**	**Ref**	**HG**	**GM**	**MP**	**PB**
Low	High	NA	3	5	8	12	19	0.20	0.24	0.29	0.34	0.41
Low	Mod	NA	4	6	9	14	22	0.21	0.25	0.30	0.36	0.43
Low	Low	NA	5	7	11	17	27	0.22	0.26	0.31	0.38	0.45
Mod	High	NA	6	9	14	22	34	0.41	0.49	0.59	0.70	0.84
Mod	Mod	NA	7	11	16	26	40	0.43	0.51	0.61	0.73	0.87
Mod	Low	NA	8	13	20	31	48	0.45	0.54	0.64	0.77	0.92
High	High	NA	11	17	26	41	64	0.89	1.07	1.28	1.53	1.82
High	Mod	NA	13	20	31	48	75	0.93	1.11	1.33	1.59	1.91
High	Low	NA	15	24	37	58	90	0.98	1.17	1.40	1.67	2.00
High	Very low	NA	18	28	43	68	105	1.02	1.22	1.46	1.75	2.09
High	Mod	SC	28	43	67	105	163	1.60	1.91	2.28	2.73	3.27
High	Low	SC	33	52	80	125	195	1.67	2.00	2.40	2.87	3.43
High	Very low	SC	39	61	95	147	229	1.75	2.09	2.50	3.00	3.58
Very high	Mod	SC	40	62	96	150	233	2.49	2.97	3.56	4.26	5.09
Very high	Low	SC	47	74	115	179	279	2.61	3.12	3.73	4.47	5.34
Very high	Very low	SC	56	87	135	211	328	2.73	3.26	3.90	4.67	5.58

[1]Values are based on the predicted nutrient concentrations from a global relationship based on alkalinity, river MEI, and GIG type minus the prediction error.
[2]Ref, HG, GM, MP and PB refer to EQRs of 1.0, 0.8, 0.6, 0.4 and 0.2 respectively.

7.9 Summary

i. This chapter describes the process used to combine several metric EQRs to achieve an overall classification of the ecological status of a river water body based on its macrophytes.

ii. A conceptual basis for combining metrics that reflects structure, diversity and abundance is presented and various options for combining metrics in the light of this are explored.

iii. On the basis of intercalibration at Northern and Central-Baltic GIG levels, a rule-based approach for combining metrics was developed to ensure harmonisation of UK lake classifications at high-good and good-moderate boundaries. The approach used in the river tool uses the intercalibrated lake method as a template.

iv. The first rule takes compositional information in the form of the two metrics RMNI and RMHI, and the two richness metrics, N_TAXA and N_FG to create an interim EQR. This interim value is based on the lowest two of each of the pair of metrics. Where the lowest richness EQR is higher than the lowest composition EQR, the richness EQR is multiplied by a weighting factor dependent on baseline productivity and added to the composition EQR. This product is then divided by the weighting factor plus unity. If the lowest of the richness EQRs is less than the lowest composition EQR, then the mean of the composition EQR and the richness EQR is calculated where the weight assigned to richness is half that given to composition. The second rule takes information on the extent of green filamentous algae (ALG_COV) in a site as a measure of disturbance to macrophyte abundance. If the interim EQR based on composition and richness is lower than the ALG_COV EQR, then the interim EQR forms the final EQR for the site. If the ALG_COV EQR is lower than the interim EQR the ALG_COV EQR is multiplied by a weighting factor, which gives it greater weight in more fertile rivers, and added to the interim EQR. This product is then divided by the weighting factor plus unity to give the final EQR.

v. The consequences of the rule based approach are that: (i) impoverished sites will have a lower final EQR for the same composition EQR; (ii) with increasing baseline productivity diverse sites with an impacted composition will have a higher EQR than impoverished sites with the same compositional metric, but at low productivity, high richness will have a proportionally weaker effect on the final EQR; (iii) sites on more productive rivers that support extensive filamentous algal growth will have a lower EQR than other comparable sites with the same composition and richness.

vi. Based on the final EQR, high and very high-alkalinity rivers, especially in England and Northern Ireland exhibit a relatively high level of impact, with typically over 70 per cent of sites on such rivers failing to achieve good ecological status. By contrast in the low-and moderate-alkalinity river types, 80-90 per cent of sites achieve good ecological status based on their macrophytes.

vii. The geographical distribution of rivers by class is presented graphically, which highlights the preponderance of high and good status rivers in the upland areas of NW England, Wales and much of Scotland. In contrast, over much of southern and eastern England, Northern Ireland, and to a lesser extent central and eastern Scotland, rivers of moderate or lower status are the norm.

viii. An examination of the mean overall EQR of sites in UK rivers designated as Special Areas of Conservation for their aquatic vegetation shows that whilst many such rivers perform well relative to the global average, some rivers would generally fail to achieve Good Ecological Status (GES) because their vegetation deviates from what that river would be expected to support under reference conditions. In such cases a river may still afford the best example of a particular type of vegetation within a region.

ix. An assessment of longitudinal trends in ecological status in larger, well monitored rivers suggests that, on average, the lower end of a water body is no more impacted than any other part of the water body, although rivers in northern England that rise at high elevation in sparsely populated areas, and pass through densely populated areas in their middle and lower reaches, generally indicate a trend for declining status downstream.

x. The issue of covariation between pressures and background environmental variables is explored using the final EQR as the overall measure of macrophyte ecological status. There is a unique macrophyte signal to nutrient enrichment of rivers, even when this covariation is removed. This signal is stronger for SRP than for TON.

xi. Using alkalinity, MEI, GIG type and EQR as predictors it is possible to establish values for phosphorus and nitrogen above which macrophytes would be unlikely to achieve a given ecological status. Such nutrient concentrations could contribute towards the derivation of environmental standards for rivers. Difficulties in this approach in terms of data quality and bias are discussed and a more precautionary approach is recommended to generate ecologically sensible standards for nutrients in rivers.

8 Uncertainty of ecological classifications of rivers

8.1 Introduction

A template for the analysis of uncertainty in classification is provided in Chapter 8 of the accompanying report on ecological classification of lakes using macrophytes. The nature of the available data places considerable constraints on a similar analysis for classifications on rivers. Thus, different methods were used with sampling by different or unknown observers, at different densities per water body and at temporal frequencies substantially outside those relevant to WFD reporting timescales. The analysis presented below focuses on aspects of spatial variability. The emphasis is on identifying an appropriate level of sampling effort that will, *on average*, produce an acceptably high confidence of classification (over 95 per cent) when a water body EQR (mean EQR from the sites assessed) lies within the middle of a class. The analysis should be thought of as the basis for assessing resources and effort needed for river macrophyte assessments, rather than a statement of the definitive variability associated with any given water body EQR.

A separate project funded by the Environment Agency and conservation agencies has dealt in detail with sources of variability in macrophyte surveys based on a dedicated hierarchical sampling design (Davy & Garrow, 2009). Such designs are the only adequate approach to determining specific components of variability, in particular those associated with inter (or intra)-operator variability. The reader is referred to this report for further details.

8.2 Spatial variability

This section provides a brief analysis of the variation in EQR associated with macrophyte surveys of river water bodies. The analysis is focused on the effect on confidence of classification of varying the number of standard sites per water body. The conclusions should assist in the planning of future surveys and allocating resources.

During 2006 and 2007, SEPA (Scottish Environment Protection Agency) commissioned or undertook surveys of 54 river water bodies in Scotland. Across these water bodies, a total of 219 standard 100-m sites were surveyed following MTR guidance and cover scoring. The working recommendation was to survey five 100-m sites in each water body. In practice the average number of sites per water body was four.

Each survey was classified by SEPA using the necessary metrics and environmental predictors required by LEAFPACS. The mean EQR per water body and the standard deviation associated with this EQR was subsequently determined for each water body.

An initial analysis to determine confidence of class was undertaken based on this dataset and the results reported verbally. However, this is a relatively small dataset and contains no water bodies classified below the middle of moderate status (EQR = 0.5). Consequently it was necessary to extrapolate the fit between EQR SD and mean EQR to estimate variability associated with EQR at the lower end of the range. In this case there is a significant risk that the (unknown) error associated with a given mean EQR will be greatly under- or overestimated, with knock-on effects on the risk of classification. A second analysis was therefore carried out on a much larger dataset (810 unique year x water body combinations in which more than one site was sampled within the water body) which spans almost the full range of EQRs (0.06 to 1.0).

8.2.1 Effect of number of standard length surveys

The relationship between mean EQR and EQR SD in the global dataset is illustrated In Figure 8.1 with the SEPA surveys overlain. Although variability between some surveys is high, in most cases the standard deviation is below 0.15. The two datasets overlap at the high end of the EQR gradient. However, the considerable uncertainty associated with extrapolating the model based on the SEPA data means that variability is substantially overestimated (by about 50 per cent) at the lower end of the range compared to the 'true' value derived from the global dataset. Consequently the number of surveys required for a classification with a given high level of confidence will be overestimated using the SEPA data.

A supporting perspective on the risk of classification can be obtained by focussing on the G/M boundary and considering the risk of misclassifying a site in the middle of moderate status as being good or better or a site in the middle of good status as being moderate or worse. This is shown in Figure 8.3. When the focus is on passing or failing GES, two surveys per water body will suffice in the middle of the class to provide a classification with 95 per cent confidence (see Table 8.1).

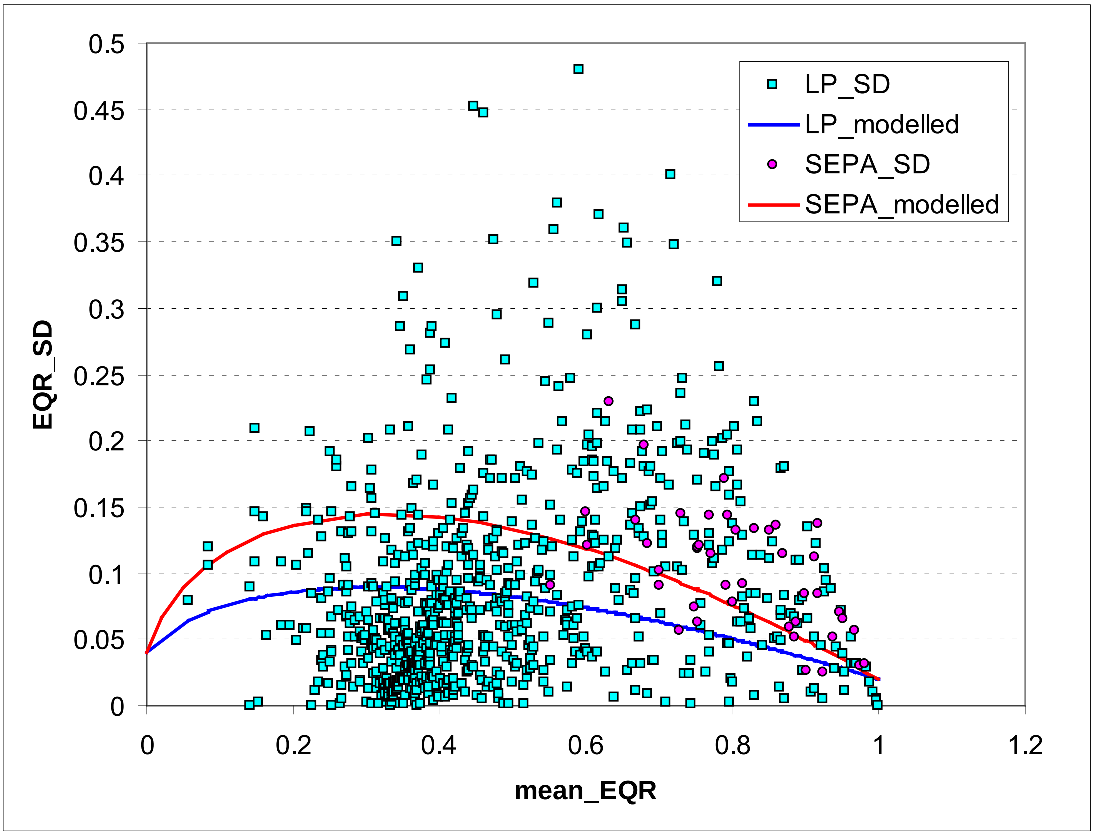


Figure 8.1 Relationship between observed and modelled EQR standard deviation in two datasets. Lines fitted by a power function anchored at 0.04 and 0.02 EQR SD at mean EQR zero and one respectively.

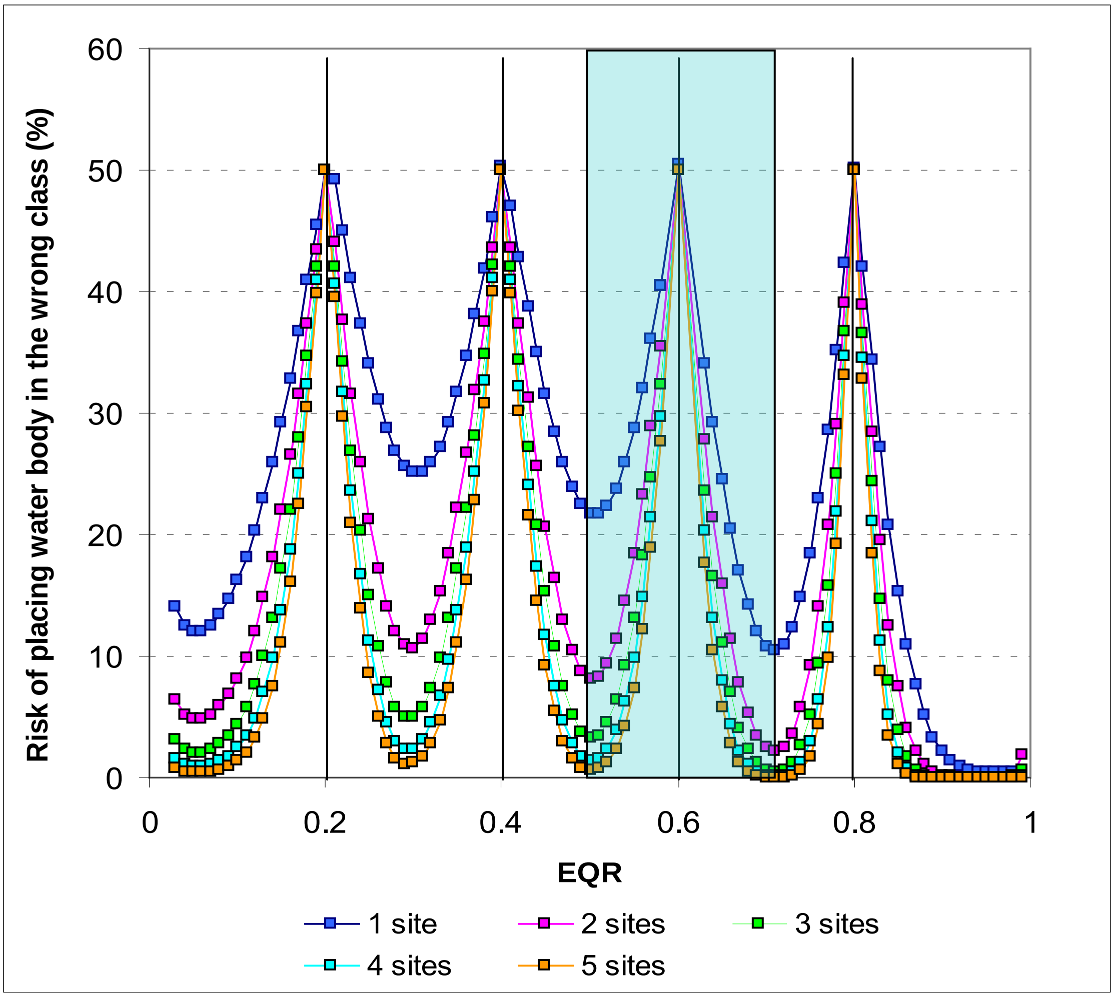


Figure 8.2 Risk of misclassification of a water body for different levels of survey effort, in terms of numbers of macrophyte survey sites per water body. Three surveys ensure the risk of misclassification of the water body is below five per cent in the middle of all classes.

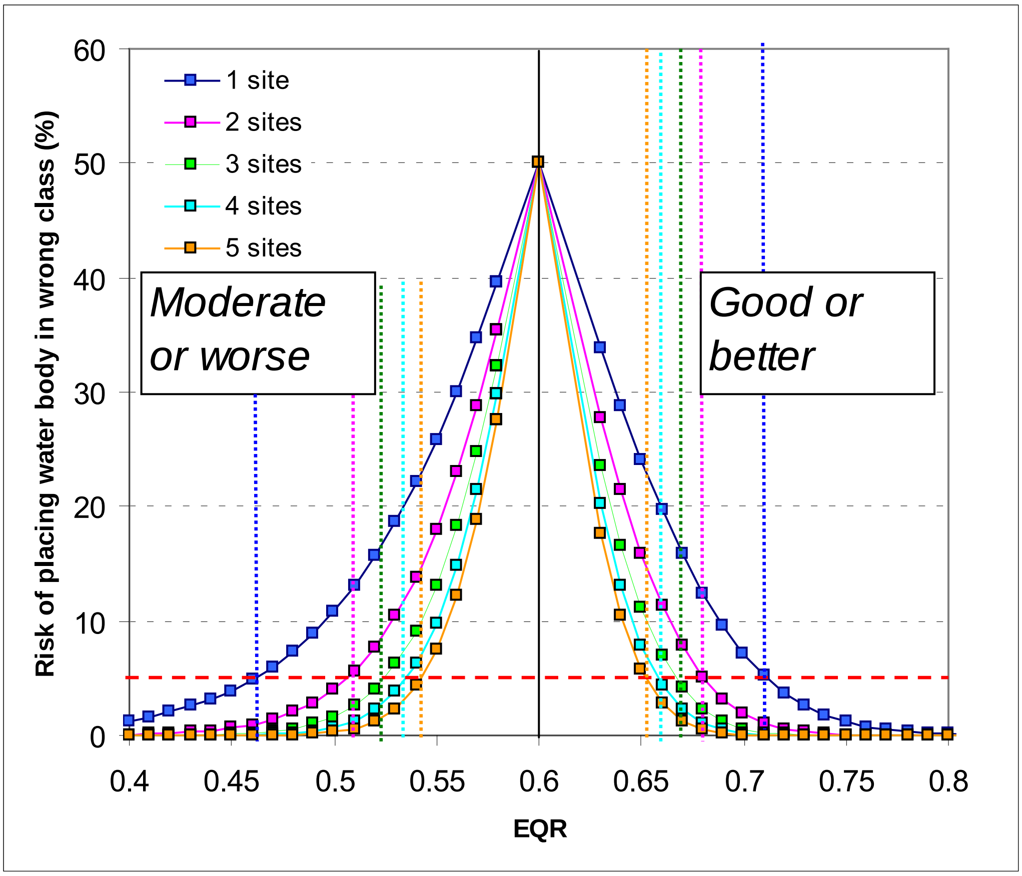


Figure 8.3 Risk of misclassification of a water body for different levels of survey effort, when the focus is on the risk of wrongly classifying a water body as good or better or moderate or worse. Two surveys ensure the risk of misclassification of the water body is below five per cent in the middle of good (EQR = 0.7) or the middle of moderate (EQR = 0.5).

Table 8.1 Confidence of classifications in relation to good or better or moderate or worse status with different levels of survey effort

	Number of sites per water body				
	One	***Two***	***Three***	***Four***	***Five***
Confidence at middle of moderate that WB is less than good	89	96	98	99	100
Confidence at middle of good that water body is at least good	93	98	99	100	100

8.2.2 Effect of type of survey

As well as containing surveys conducted to the standard 100-m survey length recommended for MTR and adopted for WFD purposes, the LEAFPACS database also contains surveys undertaken for conservation purposes. The JNCC survey method comprises paired 500-m reach surveys either side of a central point (one-km of river). It is therefore possible to determine the variability associated with surveys conducted with this method and to compare this to the variability associated with the shorter length of survey traditionally used by UK environment agencies. In this case there are 436 unique year x water body combinations in which more than one paired 500-m reach survey was undertaken within the water body.

Figure 8.4 compares the variability associated with multiple surveys of a water body when the survey unit consists of one km of river as opposed to 100 metres. Most one-km surveys were done for conservation purposes and unsurprisingly almost no water bodies with multiple surveys of this type would be classified below moderate status. Consequently the same problems of extrapolating the fitted line apply. However within the area of the EQR gradient sampled, the fitted line suggests that the variability associated with multiple one-km surveys is on average, as much as 50 per cent higher than for multiple 100-m surveys. Therefore, for a given level of survey effort (or a lower effort if the time taken to complete each survey is considered), multiple 100-m surveys will permit a more confident classification of a water body for WFD purposes than will multiple one-km surveys.

In some respects this result is surprising. Logically, a longer survey reach should better integrate the variation in physical habitat found within a water body and consequently there should be less variation between survey reaches in the biota present. On the other hand, the longer the survey reach the more likely it is to capture rare or localised habitats (or impacts) and associated biota, and therefore the greater the potential for reaches to differ. Shorter reaches will arguably be most likely to characterise the commonest and most widely distributed physical habitats found in a reach and will on average be more similar. However, if the purpose is to characterise all the habitats found in a river, short survey lengths are unlikely to achieve this unless a larger number of such lengths are employed. A secondary factor that cannot be isolated here is that observer error (within or between surveyors) exaggerates the between-reach variability when the survey reach is longer, because the repeatability of surveys of long lengths of channel is inherently lower than that of 100-m reaches within the same river.

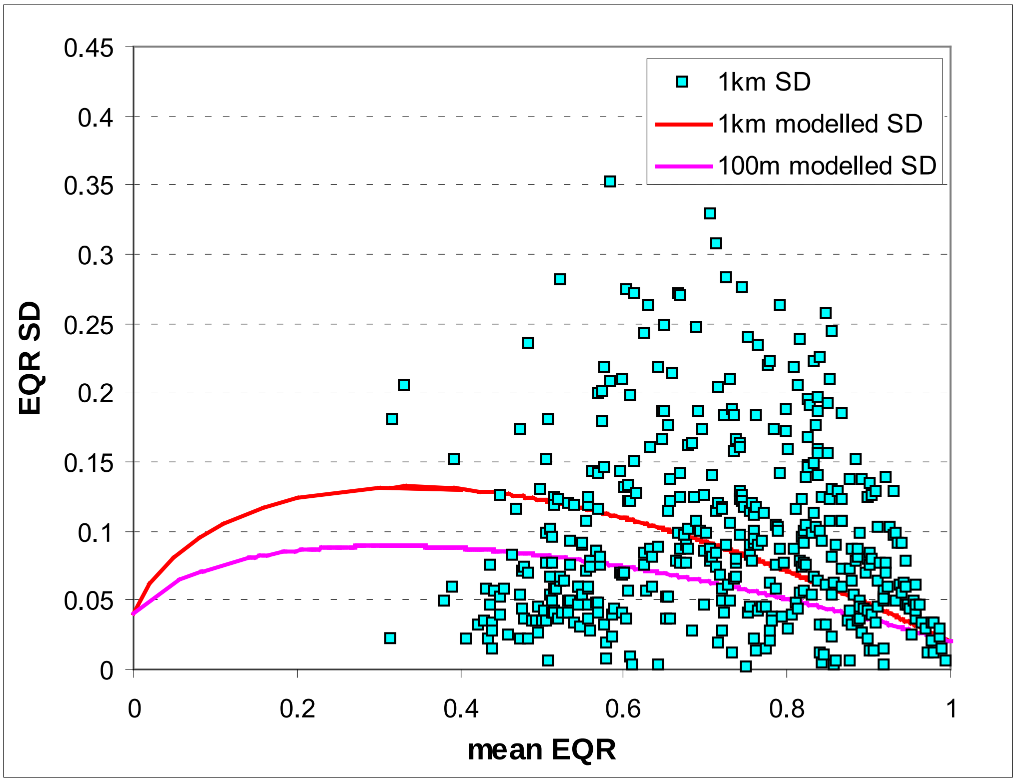


Figure 8.4 Relationship between observed and modelled EQR standard deviation when a water body is characterised by multiple one-km surveys. Also shown is the fitted line for 100-m surveys. Lines fitted by a power function anchored at 0.04 and 0.02 EQR SD at mean EQR zero and one respectively.

8.3 Conclusions

i. Three standard 100-m surveys per river water body will, on average, be sufficient for classification of a water body with at least 95 per cent confidence and therefore should be sufficient for reporting purposes.

ii. This level of survey effort should be realistic in all river water bodies and should be achievable within a single day of fieldwork by any accredited pair of surveyors.

iii. Two standard 100-m surveys per water body will be sufficient to state with 95 per cent confidence in the middle of moderate status or the middle of good status whether the water body fails or meets good ecological status.

iv. An empirical measure of variability based on the standard deviation in the final EQR of multiple surveys in a water body will always be a more reliable means of deriving CoC than will a measure of variability for a mean EQR that is simply modelled from previous data.

v. In some water bodies, empirical variability may greatly exceed modelled variability. In general, this reflects major changes in ecological status over the length of a water body (such as when a river passes from high quality upland areas through degraded reaches in its lower course). Depending on the length of water course at different status, increased survey effort will not necessarily reduce sample variability, it will merely ensure that the sample EQR mean for the water body is a closer representation of the true population mean for the water body. In such cases it will be preferable to report the results of a water body classification with low confidence, or to subdivide the water body.

vi. The variation within a water body between one-km length surveys is greater than the variation between 100m surveys for the corresponding mean EQR.

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Appendixes

Appendix 1

Calculating species ranks for a perceived pressure from a global dataset using a draft expert ranking system

i. Take a site x species dataset. The sites should be widely distributed geographically, should represent a randomized sample of the resource, and not be biased towards particular areas or lake or river types (unless these are naturally water body rich). Multiple surveys of single water bodies should be averaged if necessary. Data can be cover values on a standard scale or can be expressed as presence-absence. If large amounts of data from different sources and obtained by different methods are being pooled, analysing presence-absence data is likely to be more robust. If historical data is used the analysis can only be conducted at a presence-absence level.

ii. For each site calculate a site index score using an available expert ranking system, by calculating the average or cover-weighted average rank of the species present. The Ellenberg system, which includes a fertility rank, provides the most comprehensive set of rankings for European plant species and has been adapted for the British flora as part of the ECOFACT project (Hill *et al*., 1999). The MTR or TRS systems for rivers and lakes respectively could also be used, or some hybrid of these systems. If species are present that do not have a rank, ensure that the calculation of the site index is based only on the cover of ranked species.

iii. Take the species x sites dataset and the site index scores and perform a DCCA with the site index scores as the 'environmental' variable.

iv. Take the original expert ranks and regress these against the DCCA axis 1 species scores. For species with no rank in the original system, apply the regression equation to the DCCA1 axis score of that species to obtain a fitted rank scaled according to the original ranking system (this step can be employed during intercalibration if the inclusion of data from other countries within a GIG introduces species absent from the British site database).

v. Repeat steps 2 and 3 until all species have a rank and a site index score is available for all sites. Usually only two iterations are required. Then carry out a DCCA with full set of site index scores as the independent variable. Once rescaled to match the direction and scale of the original system, the axis 1 scores and associated tolerance values produced by this analysis form the new "adjusted expert scores".

vi. Where a genus includes records for several species as well as records identified only to genus level, it is likely that the genus score from the above approach is a poor guide to the general occurrence of the taxa because such data is often specific to particular geographical regions, datasets or surveyors. Therefore, calculate a genus score based on the average of the ranks of all the members of that genus (including records only identified to genus level) weighted by the number of records of each taxa in the dataset, or degrade all identifications at species level to genus level for the specific purpose of deriving a score appropriate to genus level identifications.

vii. Assess the relationship between the original and adjusted scores to identify species that have shown the largest change in rank and see if this can be readily explained. Under such circumstances there is a risk that the adjusted expert score for rare species (under 10 occurrences) is wrong because that species is undersampled and the sites (or vegetation) where it was recorded are misrepresentative of its ecological niche rather than because the expert view is incorrect (indeed, many rare species have been the focus of detailed autecological studies). When species are rare in the dataset and show a marked departure from their original expert score, it is suggested that a global regression of original versus adjusted expert ranks using only those species with more than 10 occurrences is used to produce a new score. Hill *et al* (1999) remark that the adjusted Ellenberg indicator values for the UK flora are in fact 'a mixture of objective results based on calculation and subjectively derived values based on field experience and published sources'. The process described in step 7 effectively brings deviant species more closely into line with the expert view.

viii. Calculate a site index score using the adjusted values, possibly incorporating the (final DCCA derived) tolerance as a measure of the indicator potential of different species.

ix. This approach is described in detail in Hill *et al*. (2000). The details given above differ slightly and do not require the use of specific software but the mechanics of the approach and results are effectively the same (Mark Hill, personal communication).

List of abbreviations

ALG_COV	Metric describing cover of green filamentous algae
ANOVA	Analysis of Variance
ASC	Acid Sensitivity Class
ASPT	Macoinvertebrate metric: Average Score per Taxon
AWIC	Macoinvertebrate metric: Acid Waters Indicator Community
BGS	British Geological Survey
BMWP	Biological Monitoring Working Party
CB-GIG	Central Baltic Geographical Intercalibration Group
CCA	Canonical Correspondence Analysis
CEH	Centre for Ecology and Hydrology
CEN	Comité Européen de Normalisation (European Committee for Standardisation)
CoC	Confidence of Class
CSM	Common Standards Monitoring
DCA	Detrended Correspondence Analysis
ESG	Ecological State Group
EQI	Ecological Quality Index
EQR	Ecological Quality Ratio
EU	European Union
GES	Good Ecological Status
GIG	Geographical Intercalibration Group
GIS	Geographical Information System
GLM	Generalised Linear Model
HMC	Habitat Modification Class (within RHS)
HMS	Habitat Modification Score (within RHS)
HQA	Habitat Quality Assessment (within RHS)
IBI	Index of Biotic Integrity
JNCC	Joint Nature Conservation Committee
LIFE	Lotic Invertebrate Index for Flow Evaluation
MEI	Morpho Edaphic Index
MEIr	Morpho Edaphic Index for rivers
MTR	Mean Trophic Rank

NCC	Nature Conservancy Council
NIEA	Northern Ireland Environment Agency
N_FG	Number of plant functional groups
N-GIG	Northern Geographical Intercalibration Group
N_TAXA	Number of hydrophyte taxa
PoM	Programme of Measures
PSYM	Predictive System for Multimetrics
Q50	Flow which is exceeded 50 percent of the time (median flow)
RHS	River Habitat Survey
RIVPACS	River Invertebrate Prediction and Classification System
RMHI	River Macrophyte Hydraulic Index
RMNI	River Macrophyte Nutrient Index
SAC	Special Area of Conservation
SCM	Site Condition Monitoring
SEPA	Scottish Environment Protection Agency
SNH	Scottish Natural Heritage
SRP	Soluble Reactive Phosphorus (orthophosphate)
SSSI	Site of Special Scientific Interest
STW	Sewage Treatment Works
TDI	Trophic Diatom Index
TON	Total Oxidised Nitrogen
TP	Total Phosphorus
TRS	Trophic Ranking Score
TWINSPAN	Two-Way Indicator Species Analysis
UKTAG	UK Technical Advisory Group on the WFD
UWWTD	Urban Waste Water Treatment Directive
WFD	Water Framework Directive

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