Environment Agency

Evidence

The ecological classification of UK lakes using aquatic macrophytes

Report – SC010080/R2

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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 standing water bodies 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 lake macrophytes for ecological assessment has been largely restricted to the selection of sites for conservation designations, focussing on species or assemblages, and limited referencing to unimpacted conditions. The wider use of macrophytes in assessment of freshwaters has concentrated largely on their value for diagnosing pressures, such as nutrient enrichment or water abstraction. 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 lake macrophytes followed a number of steps. These included data collation, development of survey methods, construction of a lake 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 classification.

Macrophyte survey data were collated from a wide range of sources including research projects and historical records. Most of the data was provided by UK conservation agencies. Over 4,500 surveys were collated, providing comprehensive coverage of UK lakes in terms of geographical distribution and environmental conditions. Survey data for each water body were subsequently matched to basic environmental and, where possible, pressure data.

A standardised protocol for lake macrophyte surveys is recommended that largely follows the method developed and trialled by the Centre for Ecology and Hydrology to support site condition monitoring of freshwater Special Areas for Conservations (SACs). Since 2004 this method has been used successfully in over 500 lake macrophyte surveys carried out for WFD tool development and monitoring purposes.

The UK lake resource was stratified into 20 types on the basis of environmental variables (alkalinity and depth) that have a strong influence on lake macrophyte community composition and have a controlling effect on lake productivity. The typology largely followed the UK reporting typology but several additional types were required to reflect the influence of a strong gradient of oceanicity and potential fertility of high alkalinity lakes between north-west and south-east Britain.

Metrics reflecting the composition (Lake Macrophyte Nutrient Index), richness (numbers of hydrophyte taxa and hydrophyte functional groups) and abundance (mean cover, relative algal cover and relative invasive species cover) of the vegetation were developed to reflect different aspects of the WFD normative definitions. Relationships between individual metrics and pressures were assessed, mainly for eutrophication. 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 impoverished in terms of cover or richness, and (iii) complementary sensitivity to key pressures such

as eutrophication which exhibit hierarchical effects on vegetation from composition to richness to abundance, according to lake type and degree of enrichment. Although individual metrics appear to be sensitive to pressures such as nutrient enrichment, acidification or hydromorphological alteration they should be used collectively as an indicator of general degradation, rather than individually in any diagnostic sense.

Ecological status is the ultimate currency of the WFD. It is a measure of the degree of deviation of test sites from 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. When this indicated that water bodies within a lake type failed to achieve reference conditions, these conditions were reconstructed using nineteenth century botanical records. 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, 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 lakes, such as lake area, depth, altitude, distance from coast and aspects of catchment geology, such as alkalinity and freshwater sensitivity class, were used as predictors. Trials using the metric LMNI indicated that site-specific predictions using generalised linear models were superior to type-specific predictions in 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.

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 lake 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, and to assess short-term and longer term changes in the status of intensively surveyed water bodies in the Norfolk Broads and West Midland Meres.

Analysis of uncertainty in the overall lake EQR, associated with sampling, temporal and spatial sources of variation, indicated that two macrophyte surveys in separate years within a six-year monitoring cycle will normally be sufficient to classify a lake with greater than 95 per cent confidence when the mean EQR lies in the middle of a class.

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We are grateful to the Joint Nature Conservation Committee (JNCC) for making available the lake macrophyte survey database that provided a large number of samples used in the design of this classification tool. A number of individuals or organisations further assisted by providing datasets derived from research projects, personal records and long-term monitoring. These include Angela Darwell, Bernard Dudley (Centre for Ecology and Hydrology), Jane Fisher (CEH), Iain Gunn (CEH), Andrea Kelly (Broads Authority), Iwan Jones (Queen Mary's College), Gen Madgwick (University of Stirling), Don Monteith (Acid Waters Monitoring Network (AWMN)), Brian Rippey (University of Ulster), Carl Sayer (University College London) and Nick Stewart. Initial cleaning and collation of this data was overseen by Stephen Maberley, CEH.

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1.1.1 Objectives

Implementation of the EU Water Framework Directive (2000/60/EC)(European Union, 2000) by UK environment agencies and other competent authorities in Europe requires a change in approach to monitoring and reporting on the state of our surface waters. Under the terms of the Water Framework Directive (WFD), member states are required to differentiate their surface water bodies into "types" defined by a range of physical and chemical factors (Water Framework Directive Annex II). There is 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 and phytobenthos.

This report describes the development and testing of a tool for classifying the ecological status of standing water bodies in the UK through the use of 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)).

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. At present, diatoms are treated as a proxy for phytobenthos in terms of WFD classification tools in the UK, leaving a relatively clear distinction with macrophytes. 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 our 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 lake functioning

Macrophytes generally have a more integral role in the ecology of lakes than of rivers. For example they influence substrate and water chemistry 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, act as a food source for fish and water fowl, and offer a physical refuge which buffers interactions between fish and zooplankton. Jeppesen *et al*. (1997) provides an extensive bibliography. Macrophytes are of pivotal importance in the functioning of lake ecosystems, and need to be integrated into holistic assessments of the ecological condition of standing waters. Thus there are a number of recent precedents to the WFD in North America and Australia, where macrophytes contribute to an overall assessment of the ecological integrity of lakes using multiple taxonomic indicators (O'Connor *et al*., 2000; Brazner *et al*. 2007).

1.3 Use of macrophytes for lake assessment

1.3.1 Previous use

There is a longstanding interest in the ecology of lake macrophytes in Britain; most County Floras testify to the efforts of nineteenth century botanists to record the flora of their local lakes. The general appreciation of habitat preferences of different species that already existed informally was expanded upon following detailed surveys of lakes in Cumbria and parts of Scotland by Pearsall (1918; 1920) and West (1905; 1910), and more latterly in lakes across Scotland (Spence, 1967) and Wales (Seddon, 1972). Since 1990, with greater availability of direct gradient techniques for analysing species-environment relationships, a range of studies using large datasets (Toivonen and Huttenen, 1995, Vestergaard *et al.*, 2000, Heegaard *et al*., 2001) have substantiated the view, hitherto based mainly on experimentation and site-specific studies, that alkalinity, depth, nutrient supply, colour, area, altitude, substrate and morphometry are major causes of variation in lake macrophyte composition (Sculthorpe, 1967, Hutchinson, 1975).

To date the main use in the UK of information on lake macrophytes has been for conservation assessment and inventory purposes. The approaches used largely have their origins in the 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 1,100 water bodies, a TWINSPAN-based classification was performed to provide a botanical typology of lakes in Great Britain based on their submerged and floating leaved plants. This generated 12 standing water types (Palmer *et al*., 1992). This 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). More latterly standing water types have been used to guide the selection of lakes to fulfil the UK's obligations under the 1992 Habitats Directive to select, designate and protect Special Areas of Conservation. In general, the primary interest in all these analyses has been in differences in lake macrophyte composition over large-scale 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. However, until a reanalysis of the data, and using a much expanded dataset (Duigan *et al*., 2006), there has been little attempt to further

develop an understanding of the environmental basis behind gradients in lake macrophyte composition in British lakes from that which already existed in the 1970s.

The sensitivity of macrophytes to recent or long-term environmental change associated with eutrophication or acidification is well-documented and is supported by a range of contemporary, historical archive and macrofossil evidence from individual sites (Rintanen, 1996; Sayer, *et al*. 1999; Sand Jensen *et al*., 2000). Nevertheless, the conservation-oriented use of lake botanical data has proved fit for purpose, and, in the absence of other drivers, there has been little attempt to progress the use of lake macrophytes for wider environmental assessment that might fit more closely with the objectives of the WFD. To date, the most notable development in the use of lake macrophytes as biological indicators has been the Trophic Ranking Scores (TRS) for lake plants, developed by Palmer *et al.* (1992) from the initial ranking of trophic preferences of aquatic plants pioneered by Newbold and Palmer (1979). At the same time Haslam (1978) developed similar concepts on trophic preference of river plants. The approach used by Newbold and Palmer (1979) was developed into an expert ranking system for river plants by Holmes and Newbold (1984) and subsequently refined into the Mean Trophic Rank (MTR) by Holmes (1995). Through use of the MTR river plants contribute to the assessment requirements of the Urban Waste Water Treatment Directive (UWWTD). An expert ranking system based on macrophyte response to flow is also used in water resource management in rivers (through input to the Catchment Abstraction Management Strategies (CAMS) in England and Wales). In the case of lakes, TRS was used in an interpretive sense but was subsequently revised into the Plant Lake Ecotype Index (PLEX). Duigan *et al.* (2006) give examples of the use of this index for measuring biological changes due to acidification or eutrophication in lakes. However, in contrast to rivers, lake macrophytes have never been used formally in the UK for statutory environmental assessment.

The biological assessment of lakes lags behind that of rivers, where reference-based systems of assessment using macroinvertebrates have existed in many countries for several decades (Wright *et al*., 2000). Palaeoecological approaches to lake assessment based on the record of diatoms, chironomids or crustaceans in lake sediment (see Battarbee and Bennion, 2007) focus on reconstruction of environmental change and could contribute to assessing deviation from reference conditions. Most uses of botanical data for lake ecological assessment, whether in the UK or elsewhere, deal with contemporary values and use expert opinion to define the quality of sites directly from these 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 of 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 a secondary consideration.

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; 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 realism.

The protection of ecological structure and function lies at the heart of the WFD. Success in achieving this is measured in terms of 'ecological status' which represents '*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. 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 are 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 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 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) 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 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 needed 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 will 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 *prima facie* evidence from the data to pinpoint the reasons 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 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, as aspired to by the WFD. Indeed, strong diagnostic potential comes at the price of reduced sensitivity to multivariate pressures. From this perspective, macrophytes are well-suited to the assessment of general degradation, and their value is heightened by their potential to reflect secondary effects on dependent organisms and processes. To achieve this type of assessment, it is necessary to go beyond traditional weighted-average compositional metrics used in invertebrate and diatom-based monitoring in Europe, and to introduce a broader spectrum of metrics reflecting the structural and functional attributes of macrophytes. Recent examples with aquatic macrophytes 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 stream fisheries in the US in the form of an Index of Biotic Integrity (IBI) (see Hughes *et al*., 1998, McCormick *et al*., 2001). This approach has been adapted into multimetric systems to assess the habitat quality or biological integrity of wetlands or lakes in parts of North America using aquatic and emergent plants (see Miller *et al*., 2006; Mack, 2007; Rothrock *et al*. 2008). Elsewhere, examples include the Lake Submerged Plant Index (Lake SPI) developed in New Zealand to assess ecological status based on the proportion of native and alien species (Clayton and Edwards, 2006). In Europe, however, the development of plant-based multimetric systems to assess aquatic habitats is in its infancy. In the UK, Predictive System for Multimetrics (PSYM) for ponds (Pond Action, 2002) provides a contemporary, albeit isolated example; elsewhere in Europe, assessment of the integrity of rivers in Portugal (Fereira *et al*., 2005) and aquatic habitats of the Rhine floodplain (Tremoliere *et al*., 2007) are among the few published studies to successfully apply the IBI template to macrophytes.

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 higher 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 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 and cross-match archived data	• Maximise value of large pre-existing datasets • Review data collection methods • Raw material for establishing metric-pressure	2.2 - 2.4

Step	Objective	Functions	This report
		relationships • Allows assessment of temporal change • Contributes to uncertainty assessment	
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
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	8.0
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	9.0

2 Methods

2.1 Biological data acquisition

2.1.1 Approaches to lake macrophyte surveys

A variety of approaches, equipment and sampling designs have been employed in the assessment of macrophyte cover and composition in UK lakes. These range from basic, informal observational surveys of a whole lake from a boat, or by walking the perimeter and noting whatever is seen, or sampled by a grapnel or rake, through to intensive, quantitative surveys of fixed plots or transects by divers. A detailed review is given by Gunn *et al*. (2004). The majority of data has, however, been obtained by semi-quantitative methods, based on sampling from boats or the shoreline, with data presented on a whole-lake basis, usually on the DAFOR abundance scoring system (Dominant, Abundant, Frequent, Occasional, Rare) or some related scale, to provide a measure of cover, abundance, frequency of occurrence, or volume inhabited. There are numerous regional variations on this approach, as well as pragmatic modifications undertaken in the field depending on local conditions and equipment availability. Unfortunately, such modifications are often unrecorded. In general, macrophyte surveys of lakes have been undertaken by several personnel simultaneously, at least one of whom is experienced in the survey approach and macrophyte identification. Material is usually identified to species level, with the exception of filamentous algae and sometimes charophytes. Voucher specimens of rare or critical taxa are usually retained for confirmation by experts.

Data obtained for this project and for the purpose of comparing the results of different classification tools was collected exclusively using a standardised approach to macrophyte surveying in lakes developed and trialled by the Centre for Ecology and Hydrology (CEH) during 2003-2004 (see Gunn *et al*., 2004). This approach was developed in accordance with JNCC Common Standards Monitoring (CSM) guidance. It was designed for Site Condition Monitoring (SCM) of standing waters in Scotland for Habitat Directive purposes under contract to Scottish Natural Heritage (SNH). In the interests of harmonising data collection, and given that this approach has undergone extensive testing, consultation and revision, and is compliant with current CEN guidance on lake macrophyte surveys (CEN, 2003), this survey method has been adopted by SEPA (Scottish Environment Protection Agency) and the Environment Agency for future use in lake macrophyte surveys for WFD purposes (a different method (Free *et al*., 2006) developed by the Irish EPA (Environmental Protection Agency) is currently used to assess lake macrophytes in Northern Ireland). Full details of the SCM method are reported in JNCC (2005) and are summarised in Section 3.1.2. In essence, this method formalises previous approaches through the use of standard shoreline and boat transects, and the use of a sampling grid, which helps recording in terms of frequency of occurrence. Other standard measurements, often missing from previous approaches, include maximum depth of colonisation. Data is then aggregated and expressed on a whole-lake basis. A useful feature of this approach is that several components of variability (such as between observers at the same transect, between transects on the same date, within transects on different dates) are readily derived, thus assisting the calculation of confidence of classification (CoC). The JNCC (2005) report is flexible on the number and type of transects and sectors to be applied in a

lake, although a standard effort and survey design should always be applied to individual lakes to allow an unbiased assessment of change. For WFD purposes, it is necessary to make comparisons across large numbers of lakes and deviations from the standard method reported below should be minimised.

2.1.2 A standardised method for lake macrophyte survey

The lake to be surveyed is divided into sectors. Different types of recording transects are applied to each of four sectors located within a lake. Sectors are located visually and are designed to capture as far as possible the range of variability in dominant physical habitats (such as sheltered bays, exposed shorelines). As lake size increases above 50 hectares (ha) the number of sectors sampled increases accordingly, with eight sectors used to assess very large lakes (above 500 ha). In small or uniform water bodies fewer than four sectors may suffice, but it is unlikely that this will be adequate for WFD water bodies. Recording transects fall into three types:

- Perimeter transects (red in Figure 2.1) extend for 100-m parallel to the shore covering the area between the water line and the high water mark (HWM). Species are recorded on a DAFOR scale according to their cover over the length and width of this zone and are stratified into aquatic species found in the strandline, and amphibious or emergent species. The method can also accommodate presence-absence data only from perimeter transects if, for example, it was difficult to access the whole length of the shore, or if it was necessary to undertake the survey from a boat.
- Shore transects (green in Figure 2.1) extend for 100-m parallel to the water's edge and are sampled as five equally spaced 'teeth' along a 'comb' with samples taken at 0.25, 0.5, 0.75 and above 0.75 m water depths along each 'tooth' of the comb, giving a total of 20 sample points per transect. Species are recorded in terms of presence/absence and an assessment (scale zero to three) of the volume of aquatic plants and filamentous algae respectively is made at each sample point.
- Boat transects (blue in Figure 2.1) extend from the mid-point of each shore transect to the centre of the lake or the maximum depth of colonisation if shallower, and run perpendicular to the shore. Twenty spot measurements of macrophyte composition and abundance are made along each boat transect at equally spaced intervals. Species are recorded in terms of presence/absence and an assessment (scale zero to three) of the volume of aquatic plants and filamentous algae is made at each sample point.

The emphasis on intensive sampling of different littoral habitats within four discrete areas largely standardises a strategy that was followed informally in previous surveys of large standing water bodies. Thus, it is impossible to intensively survey every part of a lake greater than 50 hectares and most whole-lake surveys are in fact based on a judgement by surveyors as to which parts of that lake look 'most interesting' or representative, with survey effort concentrated at these sites. Hence, when the total number of species recorded in the standard method is compared with a previous whole-lake survey of the same water body the differences tend to be small. The intensive and standardised nature of the current method has often resulted in the detection of species not recorded in previous whole-lake surveys of that site (Gunn *et al*., 2004). Local circumstances may occasionally prevent the completion of all transect types in all sectors. However, due to the type and size of lakes considered for WFD purposes, a boat should always be available.

The final data for each site are reported as a composite of all transects and survey types, being based on frequency of occurrence (maximum of 20 points per species per boat or shore transect) and a cover rating for the perimeter transects. The final aggregated data are based on a weighting of perimeter, shore and boat transects of 0.25:0.5:1 respectively and expressed in terms of percentage frequency of occurrence. These aggregated data form the basis for the calculation of all metrics used in classification. The treatment of data in an aggregated form reflects the fact that:

i. almost all pre-existing data available for developing the classification tool was in the form of whole-lake surveys;

ii. it is difficult to justify treatment of separate sampling sectors as independent samples, except perhaps in very large lakes;

iii. it is not possible to calculate a sector-specific expected value for the various metrics used since these predictions rely upon whole-lake level environmental data (such as altitude, average depth, water body area).

To this extent spatial variability (in the form of within transect, between transect, within sector and between sector) is absorbed into the final aggregate data.

Repeat surveys of a water body should adhere as closely as possible to the original locations of shoreline and boat transects (for example, within the limits of GPS accuracy) unless it is felt that a high frequency of resurvey of the same section of shoreline is directly modifying the vegetation (only likely if surveys are done annually).

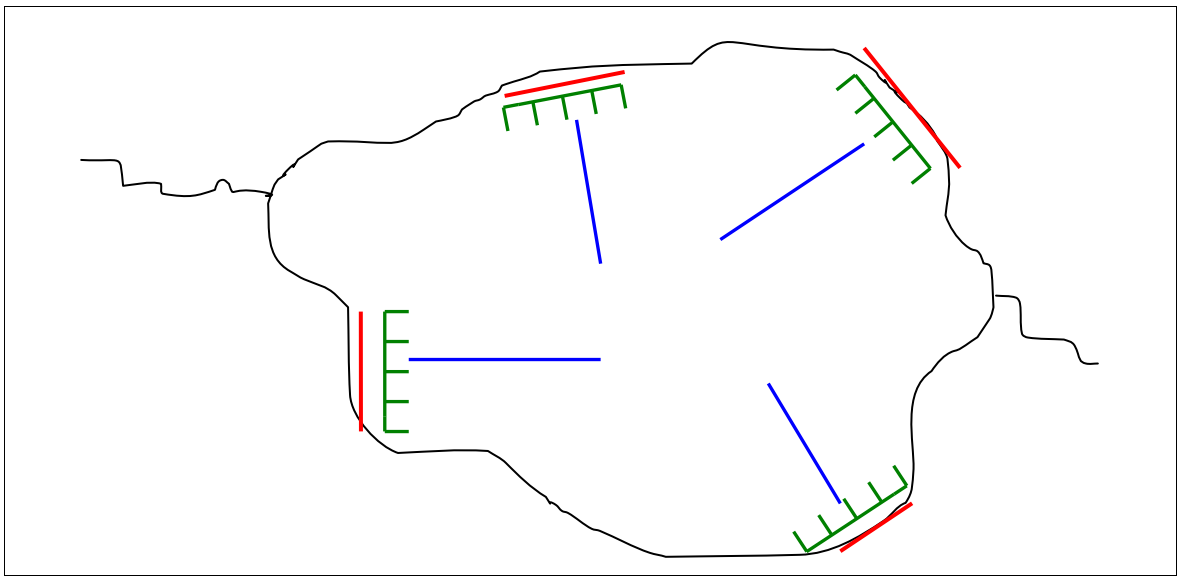


Figure 2.1 Diagram to illustrate final sampling strategy for lake macrophytes developed for CSM.

2.1.3 Extracting data from standardised lake macrophyte surveys

Table 2.1 provides an example of raw data collected in a single shoreline transect. The transect comprises five ‘teeth’ (A-E) that run perpendicular to the shore at 20-metre intervals. On each of these teeth the vegetation is sampled at four different depths, 25, 50 and 75 cm and a grapnel throw into the water body from a depth of 75 cm. At every

point the sample is considered to cover an area of around one-metre squared, and in each of these points all taxa present are assigned a value of one, while an overall rating from one to three is used to indicate the overall volume of aquatic vegetation and filamentous algae respectively. Thus in this example, Point 1 represents a sample collected at 25 cm depth on the first tooth of the transect, and supports six aquatic plant taxa with a volume rating of one, plus filamentous algae with a rating of one.

Table 2.2 illustrates the method of processing the raw survey data. For each point sampled, it is assumed that the volume of vegetation is distributed equally across all the taxa recorded at that point of the transect. Although this assumption may not be true, species that are more abundant locally tend to be more widely distributed so any underestimation in the importance of species locally will be compensated by their greater frequency. To distribute the volume between the species recorded, volume ratings of one, two and three are first assigned a value of 0.2, 0.5 and one, respectively, to reflect the relative volume of material associated with each score. A cover rating of zero is given a nominal value of 0.05 to reflect the fact that some transect points may have taxa associated with them at very low cover, or which are drifting, and which must therefore attract some positive cover value. The numeric cover value is then subdivided between the taxa recorded at each point. Therefore for Point 1, which had a volume rating of one, and a numeric equivalent of 0.2, the volume is split equally between the six non-algal taxa (*Apium inundatum*, *Callitriche hamulata*, *Lemna minor*, *Elatine hexandra*, *Juncus bulbosus* and *Littorella uniflora*). This gives each taxon a value of 0.033 for that point on the transect. This step is repeated for filamentous algae. At Point 1, filamentous algae and non-algal taxa attract the same volume rating. A numeric equivalent of 0.2 is therefore awarded to filamentous algae. This process is repeated (automatically within the spreadsheet calculator) for each point on the transect. Values for each taxon are averaged across the 20 points sampled on the transect. The calculator does not currently allow for deviant surveys that sample fewer than 20 points on a transect. The mean value for each species on a transect represents a cover or volume-weighted frequency for the transect. As an example, a species that occurred in isolation at every point on the transect, with a volume rating at each point of one, would obtain a value for the transect of 0.2, or a value of 0.1 if at each point it shared the cover with one other taxa. If it occurred in isolation at every point with a cover of two it would obtain a value for the transect of 0.5, or a value of 0.25 if at each point it shared the cover with one other taxa. In this example the most abundant taxa, excluding filamentous algae, is Nitella flexilis, which obtains a value for the transect of 0.129.

This step would be repeated for all transects of each transect type (for a minimum of four each of perimeter, shoreline and boat transects per water body), and the data for each transect would be averaged across all transects belonging to that transect type. This would yield three sets of data for the water body, one for each transect type, with each set of values corresponding to the mean of the four or more transects sampled.

Table 2.1 Example of raw lake macrophyte survey data for a standard shoreline transect

RAW DATA	A				B				C				D				E			
SAMPLING POINT:	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Water depth (cm)	**25**	**50**	**75**	**>75**	**25**	**50**	**75**	**>75**	**25**	**50**	**75**	**>75**	**25**	**50**	**75**	**>75**	**25**	**50**	**75**	**>75**
Distance along transect (m)																				
Aquatic plant volume rating (0-3)	**1**	**2**	**2**	**3**	**1**	**1**	**2**	**2**	**2**	**2**	**3**	**2**	**1**	**1**	**1**	**1**	**1**	**2**	**3**	**3**
Filamentous algae rating (0-3)	**1**	**1**	**0**	**0**	**2**	**1**	**0**	**0**	**1**	**2**	**0**	**0**	**2**	**0**	**0**	**0**	**1**	**1**	**0**	**0**
Substrate																				
SPECIES (present = 1, absent = blank)																				
Apium inundatum	1																			
Callitriche hamulata	1																			
Lemna minor	1																			
Najas flexilis															1					
Lobelia dortmanna													1		1		1			
Elatine hexandra	1		1																	
Chara aspera				1																
Elodea canadensis				1																
Myriophyllum alterniflorum								1												1
Nuphar pumila															1	1				1
Potamogeton gramineus							1	1								1				
Juncus bulbosus	1	1			1				1											
Isoetes lacustris				1				1											1	1
Littorella uniflora	1	1	1		1				1	1			1				1	1		
Nitella flexilis agg.							1	1			1	1		1				1		1
Filamentous algae	1	1			1	1			1	1			1				1	1		

Table 2.2 Example of lake macrophyte survey data for a standard shoreline transect after processing

PROCESSED DATA	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
Aquatic plant volume	0.2	0.5	0.5	1	0.2	0.2	0.5	0.5	0.5	0.5	1	0.5	0.2	0.2	0.2	0.2	0.2	0.5	1	1
Filamentous algae volume	0.2	0.2	0.05	0.05	0.5	0.2	0.05	0.05	0.2	0.5	0.05	0.05	0.5	0.05	0.05	0.05	0.2	0.2	0.05	0.05
Distributed volume per taxa																				
Apium inundatum	0.03																			
Callitriche hamulata	0.03																			
Lemna minor	0.03																			
Najas flexilis															0.07					
Lobelia dortmanna													0.1		0.07		0.1			
Elatine hexandra	0.03		0.25																	
Chara aspera				0.33																
Elodea canadensis				0.33																
Myriophyllum alterniflorum								0.13												0.25
Nuphar pumila															0.07	0.1				0.25
Potamogeton gramineus							0.25	0.13								0.1				
Juncus bulbosus	0.03	0.25			0.1				0.25											
Isoetes lacustris				0.33				0.13											1	0.25
Littorella uniflora	0.03	0.25	0.25		0.1				0.25	0.5			0.1				0.1	0.25		
Nitella flexilis agg.							0.25	0.13			1	0.5		0.2				0.25		0.25
Filamentous algae	0.2	0.2			0.5	0.2			0.2	0.5			0.5				0.2	0.2		

Compilation of results on whole-lake basis

The basic approach to combining information from the three methods is to take account of the number of transects completed for each method, reduce each method to a single set of frequency values and then weight each method to derive a set of values for the lake as a whole. There is no difficulty in dealing with different numbers of transects for each type of survey method, or if, for pragmatic reasons, surveys of a particular type have not been possible. Table 2.3 shows the survey data for the example site in its consolidated form. At this stage surveyors are only required to enter the number of surveys of each transect type and indicate if the data on perimeter surveys are of a presence/absence or DAFOR type. The data shown in the column for each survey type are the average proportion of possible survey points and cover at each survey point along a transect attributable to that species. It is based on the sum of cover-weighted frequencies of each species in the perimeter, wader and boat transect sheets, divided by the number of transects of each type that were completed. Thus a value of one for a boat or transect survey would imply that a species occurred at all of the 20 possible points in each of the transects surveyed and accounted for all of the aquatic plant cover that was recorded. A value of 0.5 might imply that a species occurred at all of the possible points surveyed and shared the cover recorded with one other species, or that it occurred at half the points surveyed but was the sole species at each of these points. The values for the different transect types are combined into a figure for the lake as a whole by weighting perimeter, shore and boat transects in the ratio 0.25:0.5:1 respectively. Values for each species are expressed in terms of percentage frequency of occurrence. As an example, *Lobelia* was absent from the perimeter surveys and had a proportional frequency of 0.0338 and 0.0365 in the shoreline and boat transects respectively. Its value for the lake as a whole is therefore obtained by:

$$(0 * 0.25 + 0.0338 * 0.5 + 0.0365 * 1) * 100 = 5.34$$

When data for the lake as a whole are reported surveyors are instructed to check these to ensure they are consistent with the raw data, since a spreadsheet tool cannot correct for data that is not entered or is entered wrongly. In the final collation the figures for species with cover values above five per cent are highlighted to aid data checking.

2.1.4 Quality control of lake macrophyte surveys

Macrophyte surveys (of lakes or rivers) are rather different to surveys involving other quality elements. Data are derived by field observation and recording and it must be assumed that what is recorded is a true reflection of what was actually seen. Similarly, when a species is recorded as absent (by inference) it must be assumed that the correct techniques were employed to find that species and consequently that it was indeed absent or too rare to be detected within the area sampled with the methods used. 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.

Table 2.3 Collation of lake macrophyte survey data based on spreadsheet tool

SURVEY REPORT	Perimeter	Wader	Boat	Water body
Number of sectors sampled	6	4	4	
DAFOR? (Y/N)	y			
Lemna minor	0.00	0.00	0.00	**0.02**
Baldellia ranunculoides	0.00	0.00	0.00	**0.08**
Ludwigia palustris	0.00	0.00	0.00	**0.17**
Pilularia globulifera	0.01	0.00	0.00	**0.29**
Ranunculus aquatilis agg.	0.01	0.00	0.00	**0.29**
Tolypella glomerata	0.00	0.00	0.00	**0.29**
Potamogeton filiformis	0.00	0.01	0.00	**0.30**
Apium inundatum	0.01	0.00	0.00	**0.31**
Utricularia minor	0.00	0.00	0.00	**0.31**
Chara virgata	0.00	0.00	0.00	**0.41**
Utricularia intermedia sens.lat.	0.00	0.01	0.00	**0.42**
Chara aspera	0.00	0.00	0.00	**0.46**
Sparganium angustifolium	0.00	0.00	0.00	**0.68**
Sparganium natans	0.00	0.01	0.00	**0.70**
Nymphaea alba	0.00	0.01	0.00	**0.73**
Nitella translucens	0.00	0.00	0.01	**0.92**
Potamogeton praelongus	0.00	0.00	0.01	**0.92**
Callitriche hamulata	0.04	0.00	0.00	**0.94**
Elatine hexandra	0.03	0.00	0.00	**0.97**
Najas flexilis	0.00	0.00	0.01	**0.99**
Potamogeton gramineus	0.00	0.01	0.01	**1.10**
Potamogeton natans	0.03	0.01	0.00	**1.22**
Potamogeton alpinus	0.00	0.01	0.01	**1.49**
Elodea canadensis	0.00	0.00	0.01	**1.51**
Nuphar pumila	0.00	0.01	0.01	**1.55**
Fontinalis antipyretica	0.07	0.00	0.00	**1.75**
Potamogeton rutilus	0.00	0.01	0.02	**2.44**
Nitella flexilis agg.	0.00	0.05	0.01	**3.27**
Lobelia dortmanna	0.00	0.03	0.04	**5.34**
Filamentous algae	0.00	0.05	0.03	**5.69**
Juncus bulbosus	0.05	0.01	0.05	**6.36**
Littorella uniflora	0.00	0.08	0.06	**10.47**
Isoetes lacustris	0.01	0.06	0.08	**11.43**
Myriophyllum alterniflorum	0.03	0.06	0.09	**12.78**

Intensity of colour coding is applied to aid rapid identification of taxa for a water body as an aid to detecting errors in data entry.

Apart from confirmation of voucher specimens by 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 those leading a survey team should have at least two full seasons of previous experience as a survey assistant.

Working in groups or pairs is an effective way to reduce errors in recording or identification and improve safety. Nevertheless, unforced errors such as in field recording and transcribing data are inevitable, and all biological databases contain a small number of 'unrecoverable' errors. Such errors are likely to be reduced if electronic entry of data is undertaken by those who carried out the survey and is completed shortly after a survey is carried out, rather than data being entered in a large batch weeks or months later.

2.2 Environmental data

Most lake macrophyte surveys have been undertaken for conservation inventory purposes and recording of anything other than routine environmental data (grid references, lake area, altitude and so on) has not been the priority. However, such information is required to help screen reference sites, develop lake typology, and enable site-specific prediction of expected metric values. Therefore, surveys were linked to contextual GIS-derived environmental data via the GB Lakes Inventory (Bennion *et al*., 2005) subsequently updated to the GB Lakes Inventory (www.UKlakes.net). Nutrient pressure data (total phosphorus, TP, and total oxidised nitrogen, TON) was acquired via the 2006 CEH lake chemistry database compiled by Dudley and Carvalho from lake chemistry data collected by or on behalf of UK agencies. Where available, these data were used in the selection of reference sites, validation of pressure-sensitive metrics and development of environmental standards.

2.3 Data sources and database compilation

Data used in this project were derived from a wide range of sources, including archived historical data, and collected for a range of purposes (Table 2.4). Only a small percentage of the data in the database (10 per cent) was collected post-2000 and specifically for WFD tool development purposes, using the standard method described above. The bulk of the biological data derives from lake surveys carried out between 1983-1997 as part of the Scottish Loch Survey organised by the Nature Conservancy Council for Scotland and subsequently SNH. To ensure widespread geographical coverage, these data were supplemented by a number of contemporary data sources, including the Northern Ireland Lake Survey and Broads Authority surveys. Where contemporary surveys were considered unlikely to capture reference conditions for a given lake type, archived historical data were also used. 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. The project obtained permission to use all these data sources, this use being licensed where necessary.

Table 2.4 Summary of data sources used in tool development

Source/provider	Number of surveys	Timing	Type of data
JNCC	2,457	1978-2000	Whole-lake surveys usually including DAFOR assessment of cover, mostly in Scotland. Range of methods.
NILS	614	1991	Whole-lake surveys including PVI based assessment of abundance of lakes in Northern Ireland.
Acid Waters Monitoring Network	51	2000	Whole-lake, DAFOR level surveys of acidified or recovering lakes in Scotland and Wales.
Broads Authority	383	1982-2004	Annual whole-lake boat-based surveys of 22 broads
Environment Agency/ENSIS	173	2003-04	Commissioned whole-lake surveys to SCM standard protocol. Abundance as percentage frequency, partly boat-based.
Irish EPA	111	2001-02	Whole-lake surveys based on boat transects. Data converted to percentage frequency.
Jane Fisher, CEH	21	1998	Whole-lake surveys, abundance as percentage frequency, boat-based.
Michael Jackson, Gen Madgwick, Carl Sayer, UCL	92	1850	Decadal composite surveys for Norfolk Broads assembled from botanical records.
Gen Madgwick, Alex Lockton, BSBI	38	1830-1870	Composite surveys for West Midlands Meres based on early nineteenth century records.
Liverpool University	19	1992	Whole-lake surveys of West Midlands Meres. Abundance as DAFOR. Surveys boat-based.
Nature Conservancy Council	48	1979-87	Whole-lake surveys of West Midlands Meres. Abundance as DAFOR.
ECRC, UCL	31	1980	Whole-lake surveys of acidified lochs in SW Scotland. Three-point cover score. Boat-based.
Carl Sayer, UCL	71	2002	Whole-lake surveys of smaller broads and ornamental lakes in East Anglia. DAFOR assessment.
Seddon (1972)	70	1965	Surveys and records collated for Welsh

Source/provider	Number of surveys	Timing	Type of data
			lakes. Range of methods. Presence/absence data.
Ralph Stokoe, FBA	290	1978-80	Surveys of Cumbrian lakes and tarns. Range of methods. Presence/absence.
West (1905, 1910)	31	1900	Whole-lake, boat-based surveys of lochs in SW Scotland. Presence/absence. Boat-based.
Natural England/Whild Associates	37	1999-2005	Whole-lake surveys and record collation of West Midlands Meres. Presence/absence.

The current database amounts to 4,538 macrophyte surveys of 3,500 water bodies providing wide and comprehensive coverage of the UK (Figure 2.2). Since 1983 a total of 2,720 sites have been surveyed to a minimum standard (DAFOR or better assessment of cover) and for these sites, the full suite of metrics described can be calculated and used in classification.

Two significant recent datasets are not currently incorporated in the project database: the 2006 survey of about 100 lakes in Northern Ireland on behalf of Environment and Heritage Service (EHS) (now Northern Ireland Environment Agency), and the 2004 survey of about 240 lakes in Scotland on behalf of SNH. Both these campaigns were undertaken for SCM purposes to meet the requirements of the Habitats Directive and surveys were done using standard methods supported by this project. Earlier data from most of these sites is already in the database and it is unlikely that inclusion of these new surveys would materially affect the classifications derived. However, the recent data would provide a perspective on decade-scale changes at a number of sites and would offer a more up–to-date perspective on the status of UK lakes. Consequently, it is hoped that data from these recent campaigns can be incorporated at the earliest opportunity.

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 synonyms. The data and analyses reported here are based exclusively on hydrophytes since many surveys excluded emergent species or conceded to having under-recorded members of this group. The exclusive focus on hydrophytes is restrictive since many helophytes have strong aquatic tendencies as well as being potentially useful indicators of shoreline modification or riparian zone quality. Future surveys should consider this group of species. Opportunities to use data on emergent species from CSM surveys (such as mean number of emergent species per perimeter transect) for hydromorphological assessment are currently being investigated.

Several metrics described here require the use of continuous scale numeric data, expressed as percentage abundance. Thus, all data reported at a DAFOR scale were recoded according to R=1%, O=5%, F=10%, A=25% and D=50%. This is a pragmatic solution supported by experience in the field using this scoring system at hundreds of sites. Unfortunately, this assumes that PVI data (Plant Volume Infested or Inhabited) on a one to five scale can be rescaled in the same manner and that abundance data can be considered interchangeably with data expressed as percent frequency of occurrence. The main justification for this assumption lies in the general biological rule that the number of sites a species occupies is positively related to its average abundance across those sites. Thus species with high percentage frequency of occurrence should, on average, have higher local abundance.

Macrophyte surveys are carried out almost exclusively during the period June to September. Unfortunately, in many 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. Where historical records were used to infer reference conditions, unless a comprehensive survey was available for a given year, individual records and herbarium specimens were aggregated, typically on a decadal basis, to provide a composite sample of vegetation at that site in a given decade.

2.4.2 Environmental data

Biological data was linked to contextual and pressure environmental data via the UK Lakes unique water body identifier (WBID). Wherever available, long-term environmental data (such as alkalinity) was used to override spot environmental data collected at the time of biological sampling. However, for many small or remote sites the best available data were spot measurements of pH, alkalinity or conductivity from a single sample, usually collected at the time the biological survey was carried out. Obvious errors in units of measurement were removed but any predictions based on single spot measurements would retain a high degree of uncertainty. Recent pressure data were matched to biological surveys in the last 10 years, predominantly within the previous five years. Where only spot-measured pressure data were available, these were converted to an estimated annual mean based on a regression between summer and annual mean pressure data for 300 sites where monthly or bimonthly sampling had been carried out.

In subsequent modelling (for example, to derive site-specific expected metric values) the input predictive environmental data for a site was considered to be fixed. Thus, the value used for temporally variable data such as alkalinity was 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.

Since a minimum standard for environmental data was required (enough to assign a lake to the appropriate type), sites which could not be linked to a WBID or which had no independently derived supporting environmental data were excluded. This mainly affected data derived from the JNCC database relating to canals, ditches, small ponds or newly created water bodies that were in any case considered to be outside the scope of this project.

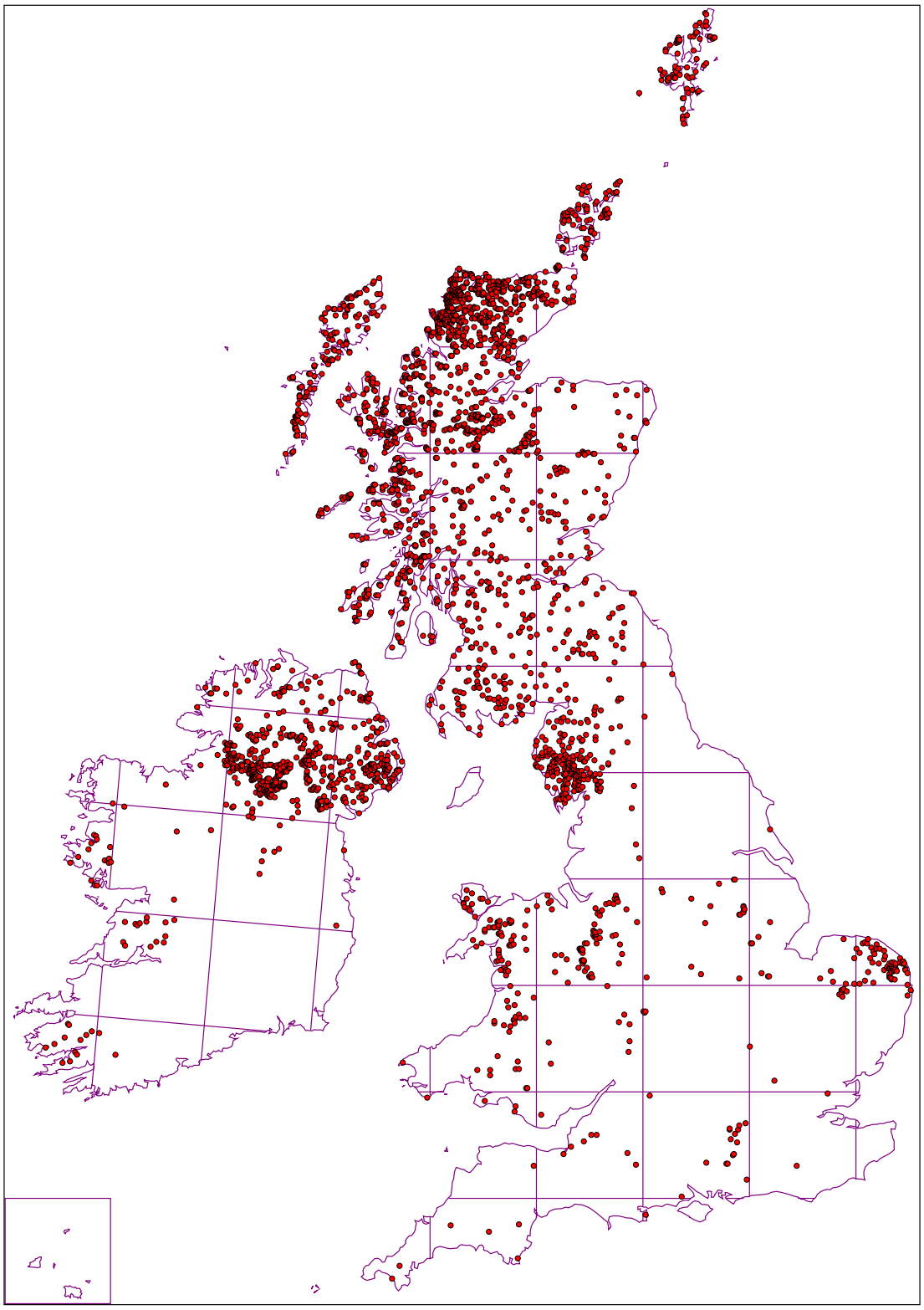


Figure 2.2 Global distribution of lakes with macrophyte surveys used in the development of the tool

3 A lake 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 goal is a site-specific predictive system, 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, area, depth and geology) each of which is stratified into prescribed classes (such as mean lake depth of three to 15 metres).
- System B provides a set of obligatory and optional factors that can be used to construct a typology.

In the case of lakes the GB core lake typology is based on geology and mean lake depth, with subsequent subdivisions on the basis of lake altitude and area (see UKTAG, 2003). In practice, alkalinity and conductivity are used as surrogates for geology, since the assignment of mapped geological types to calcareous and siliceous categories only permits lake types to be assigned with low confidence, when used in isolation. However, typologies are largely for reporting purposes and it is highly unlikely that a single typology would be optimal for expressing the variation in all biological quality elements (BQEs). Consequently, it is necessary to confirm whether the existing typology for lakes is appropriate to stratify the variation in macrophyte assemblages. In making refinements that are specific to a particular BQE, the only requirement is that any site can be mapped against the original typology when reporting classifications.

3.2 Approach

A TWINSPAN analysis using WinTWINS (Hill and Šmilauer, 2005) was undertaken of the entire macrophyte survey database, using presence-absence data for hydrophytes only. The analysis was allowed to run to six levels (potentially generating 64 groups). Since this process was not designed to describe fine-level variation in macrophyte assemblages, a minimum group size of 25 was specified (a division that would result in a group size below 25 is halted at that node). This resulted in 22 botanical types. These types were then related to an environmental dataset composed of temporally invariant, ‘unimpactable’ variables, including geology, lake area, depth, altitude and other contextual variables. An alternative measure of the base status of catchment geology was derived using mapping of soil and geology in five Freshwater Sensitivity Classes (FSC), based on their sensitivity to acidification (Hornung *et al*., 1995). Each FSC was weighted by its percentage cover in the polygon used to define each lake catchment to give a single value (wtd FSC) from one to five for each lake. One-way ANOVA of

normalised data was used to filter the variables that discriminated most strongly between botanical types, and boxplots were used to assess the suitability of the divisions in parameters employed in the GB lake typology. 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 different 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 (MDA), to predict reference biological assemblages (as in the approach adopted for RIVPACS). Detailed comparisons were not made with a recently published revised analysis of the vegetation communities of British lakes (Duigan *et al.*, 2006) since our analysis excluded close to 1,000 surveys included in the published analysis (mostly small sites below 0.1 ha without a WBID, or closely spaced sites that could not be assigned to a unique WBID from the available information), as well as adding a further 2,000 new surveys. Nevertheless, there are obvious parallels between results of these analyses.

3.3 Results

3.3.1 Overall typology

Geology, expressed in terms of alkalinity, was the primary variable in explaining variation in lake macrophyte composition, explaining more variation than all other variables combined. When viewed as box plots (Figures 3.1a-i) with botanical types ranked along the base, there is a largely predictable gradient from base-poor, upland, deep, peaty, acid to circumneutral lakes through to base-rich, lowland, shallow, alkaline lakes, reflecting the major environmental NW-SE axis in Britain.

Mean depth is one of a suite of secondary variables which explain a small yet significant amount of variation in macrophyte assemblages. In some respects, mean depth is not the best variable to create a typology for macrophytes, since macrophyte surveys usually report data for the area inhabited or potentially inhabited (typically under six-m depth) rather than for the lake as whole. The main influence of depth on compositional data acquired in this way may therefore be indirect, via the association between depth and variables such as lake area, altitude or wave fetch. However, coupled with alkalinity, mean depth, has a well-resolved relationship with potential productivity since, for example, very shallow lakes will be too well mixed to thermally stratify. The relationship between productivity and alkalinity and depth is summarised through the Morpho-Edaphic Index (MEI), which was originally designed as a measure of potential fisheries production (Ryder, 1965). Mean depth classes also separate lakes in which vegetation potentially covers the entire bed (very shallow; under three metres), lakes with extensive, but generally not ubiquitous aquatic vegetation (shallow; 3-15 m) and sparsely vegetated lakes (deep; over 15 m). It is also convenient to retain a typology that closely matches the core GB typology. Consequently the typology used in this project is as defined in Table 3.1. The botanical clusters used to set thresholds for environmental variables are detailed in Table 3.2.

Minor refinements or conditions for typing a water body were imposed. In the case of Brackish lakes, which the GB typology nominally defines as having conductivity over 1,000 uS/cm, it was found that UK lakes rarely included a distinct brackish water

element to their flora (based on Ellenberg salinity indicator scores, Hill *et al*., 1999) unless conductivity was much higher. Consequently, a threshold of 10,000 uS/cm was used to define brackish lakes from a macrophyte perspective.

Peat lakes are defined as those in which over 90 per cent of the lake catchment is composed of peat. Since small numbers of sites with alkalinity above 0.2meq/l fell into this category, it was effectively reserved for low alkalinity lakes. The term peat lake should be interpreted with caution and cannot be used as a surrogate for coloured or humic lakes since the small amount of data available from sites with colour and drift geology data indicate that coloured lakes (above 30 mg/l platinum (Pt)) will often occur where peat cover is below 90 per cent, while clear lakes often occur when peat cover exceeds 90 per cent.

The definition of marl lakes was reviewed by Willby (2005). Here, it was suggested that marl lakes should be regarded as being geologically defined (CO_2-rich groundwater upwells through limestone, CO_2 reduces to equilibrium, pH increases, and dissolved calcium (Ca) is precipitated), as distinct from lakes in which marl is sometimes formed due to photosynthetic activity. Many high-alkalinity lakes in GB, often with extensive *Chara*, will have phases of biologically-induced marl formation but this may be a temporary or seasonal phenomenon. Marl lakes should occur exclusively on hard limestone-dominated catchments, and will be relatively rare in Britain but commonplace in Ireland. Given data on alkalinity and major ion chemistry, it is possible to extract marl lakes reasonably reliably from large datasets. Thus, marl lakes should normally have high calcium concentrations (above 30 mg/l) and low concentrations of other base cations such as magnesium (below 6 mg/l), sodium (below 10 mg/l) and potassium (below 3 mg/l). However, regional variations in limestone type indicate that such thresholds cannot be applied universally (for example, Durness limestones in NW Scotland have higher magnesium concentrations).

Table 3.1 Lake typology employed in this project

Mean depth (m)	Alkalinity (meq/l)				Other characteristics		
	<0.2	0.2-1.0	1.0-2.5	>2.5	>60% hard limestone	>90% peat	>10,000 µS/cm conductivity
<3	LAVSh	MAVSh	HAVSh-C HAVSh	VHAVSh-C VHAVSh	MarIVSh	PeatVSh	BVSh
3 - 15	LASh	MASh	HASh-C HASh	VHAVSh-C VHAVSh	MarISh	PeatSh	BSh
>15	LAD	MAD					

L = Low, M = Moderate, H = High, V = Very, Sh = Shallow, D = Deep, A =Alkalinitiy, B = Brackish, C = Continental type

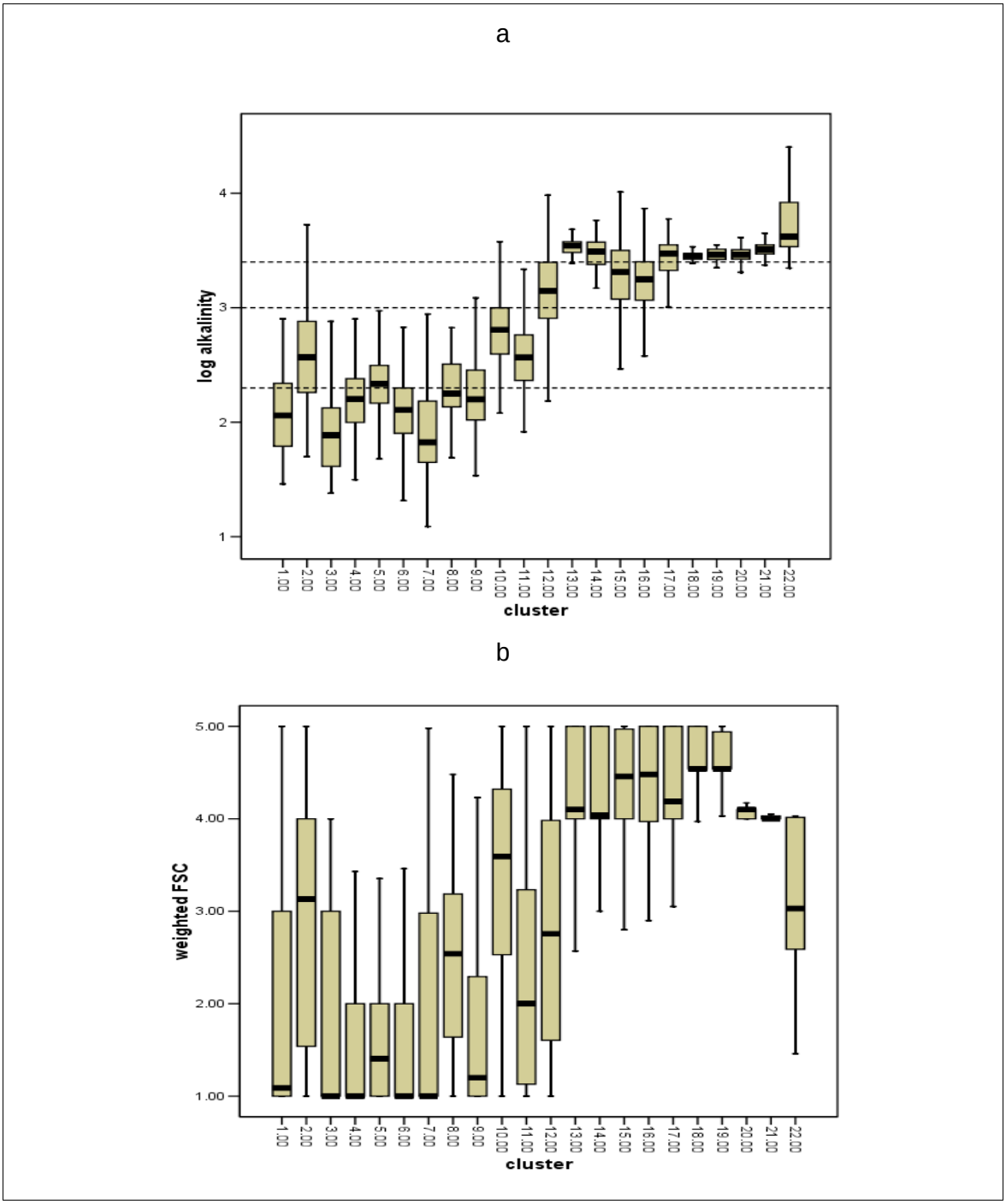


Figure 3.1 Distribution of lake botanical clusters in relation to a) alkalinity as log_{10} µeq/l and b) FSC weighted by coverage in lake catchment. Dashed lines show separation of clusters in relation to alkalinity thresholds of 0.2, 1.0 and 2.5 meq/l. Note restriction of types 13-21 to high alkalinities and geologies with high FSC.

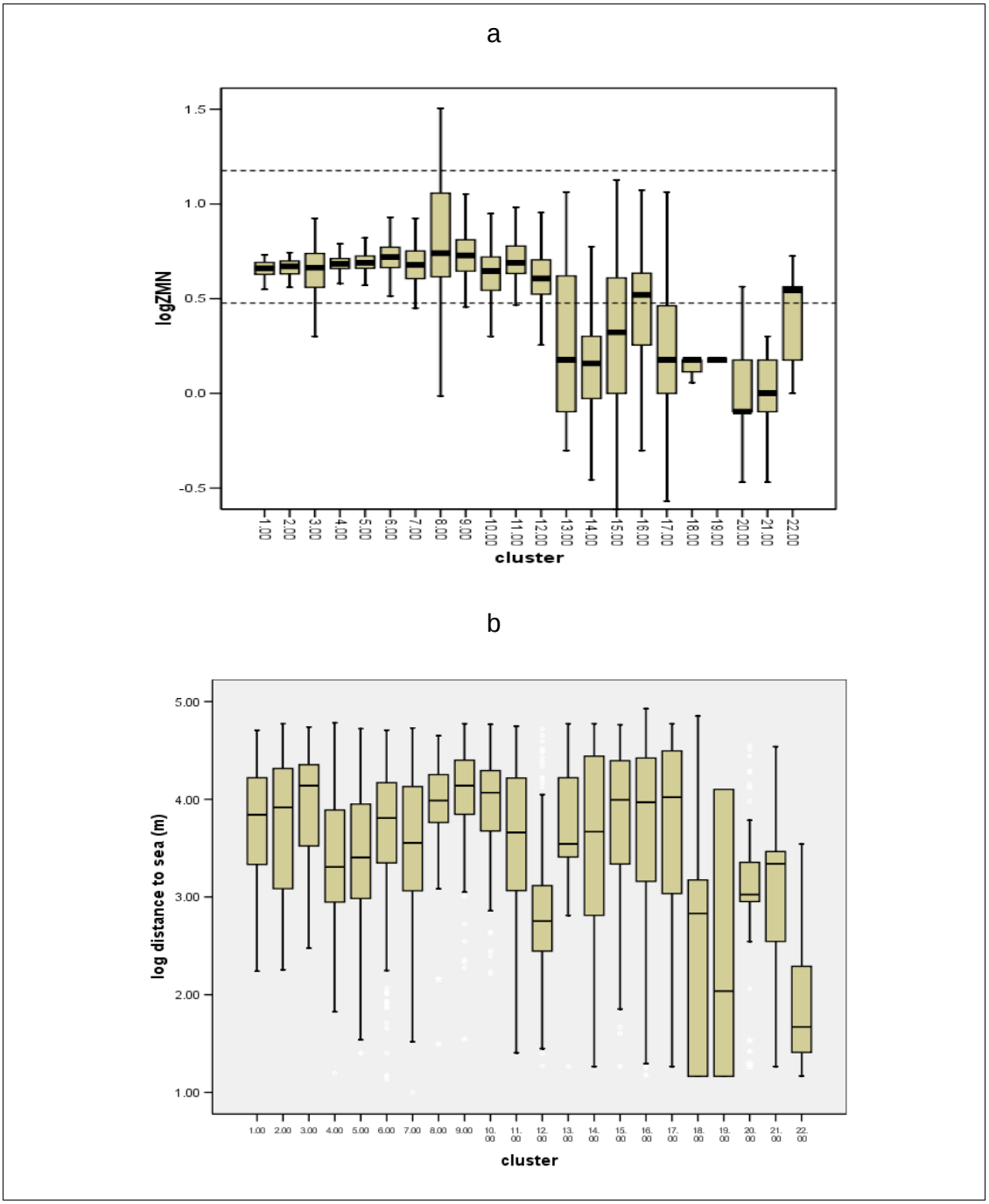


Figure 3.2 Distribution of lake botanical clusters in relation to a) mean lake depth (log_{10} m) and b) nearest distance to sea (log_{10} m). Dashed lines show the mean depth cut off at 3 and 15 m respectively. Note the separation of clusters 1-12 and 13-22 in relation to an average depth of 3 m and the subset of semi-coastal (12,20, 21) and coastal lakes (18,19,22).

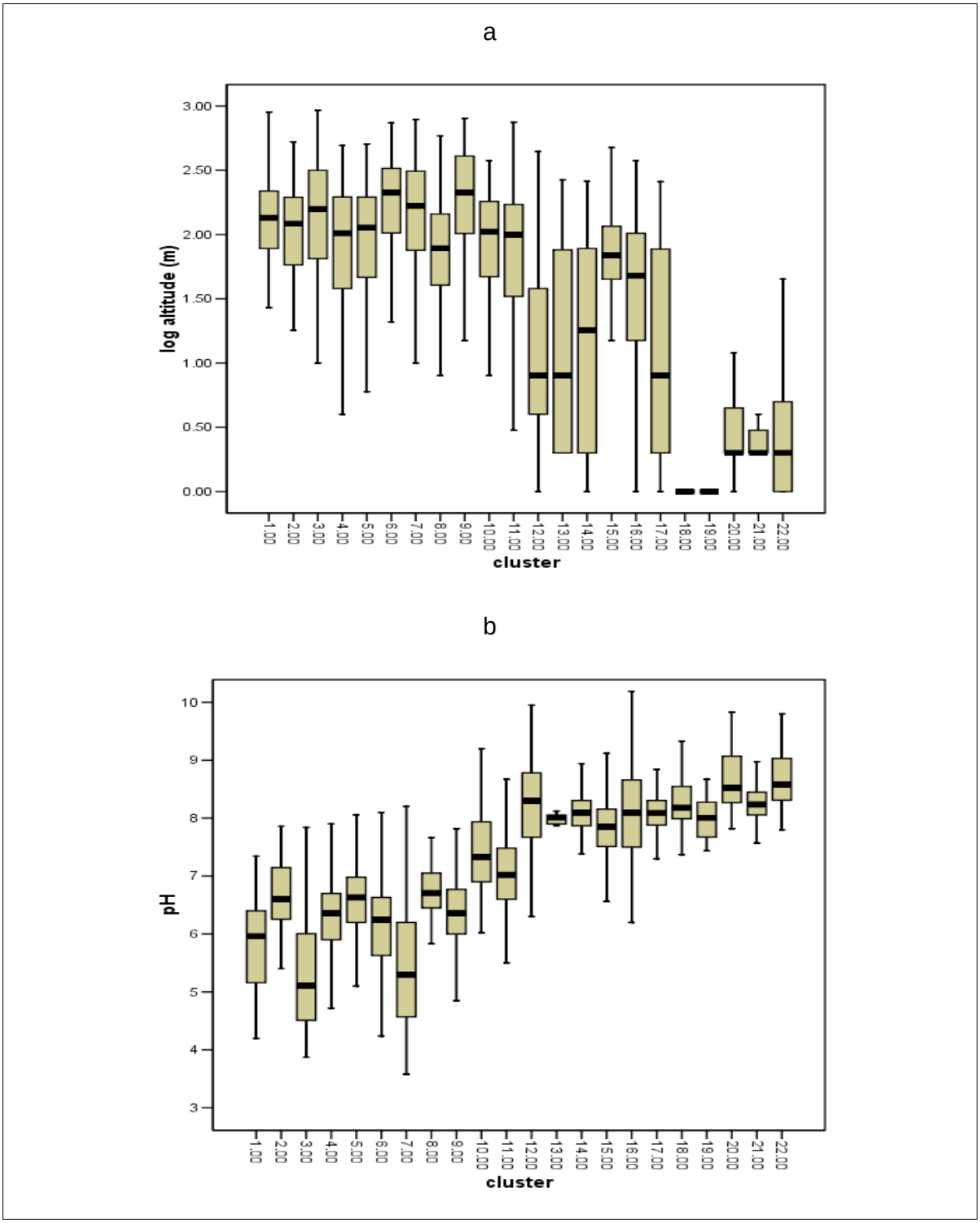
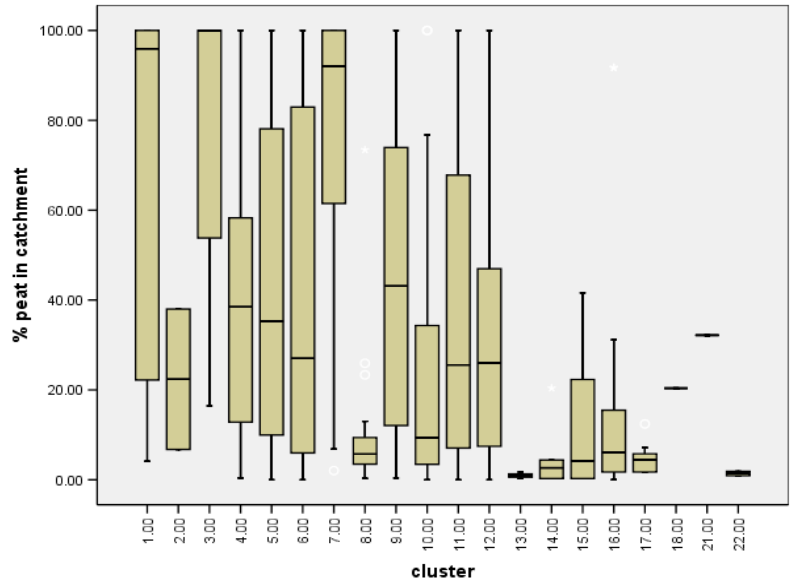


Figure 3.3 Distribution of lake botanical clusters in relation to a) altitude (as log_{10} m) and b) pH. Note the concentration of types 15 and 16 (very high alkalinity and very shallow) at 'higher' altitudes, the subgroup of lakes 12-14 and 16,17 at middle altitude and remaining types restricted to lowlands. Types 12-22 represent alkaline lakes, 1, 3 and 7 acidic lakes and others circumneutral.

a



b

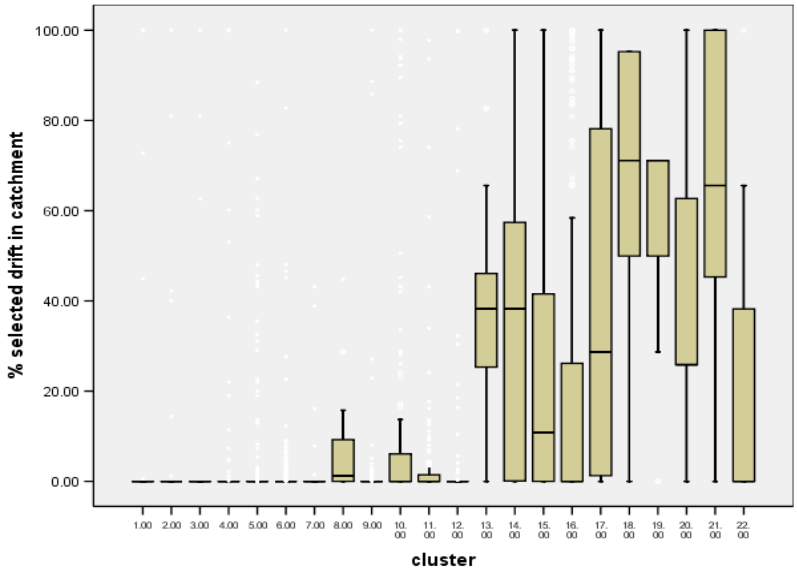


Figure 3.4 Distribution of lake botanical clusters in relation to a) extent of peat and b) extent of principle drift geology types. Note dominance of peat in types 1, 3 and 7, sparse cover or absence of peat in types 8 plus 13-21 and variable extent in other types. Types 1-12 are essentially drift free, 13-16 and 22 have a moderate extent of drift while 17-21 tend to have moderate-high drift cover.

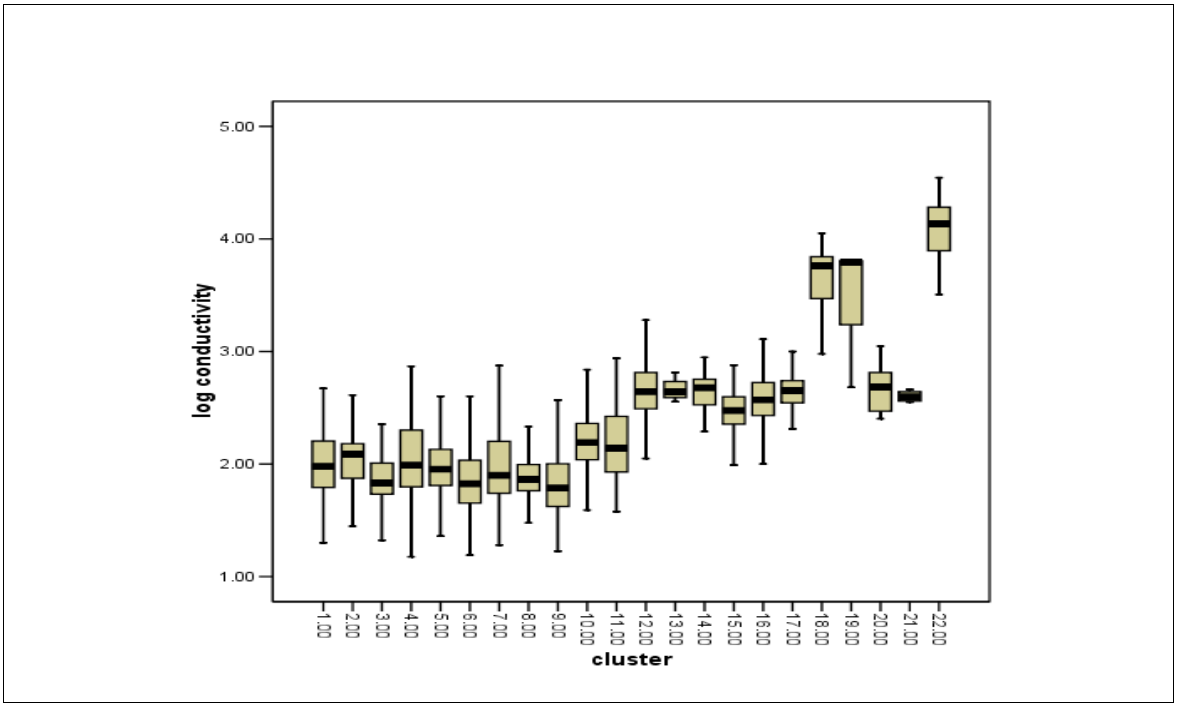


Figure 3.5 Distribution of lake botanical clusters in relation to conductivity (log_{10} uS/cm). Note low-moderate subgroup (types 1-11), moderate-high subgroup (12-17 and 20,21) and very high subgroup (18,19,22) associated mainly with coastal locations.

3.3.2 High-alkalinitylLakes

High-alkalinity lakes (over one meq/l) present a special case for several reasons. Firstly there are marked differences in baseline productivity for a given level of alkalinity. Hard calcareous rocks (situated principally in the north and west of the British Isles) naturally yield low amounts of phosphorus through weathering while, in marl lakes, phosphorus is co-precipitated with calcium resulting in low or undetectable amounts of phosphorus in the water column. Proximity to the sea also influences the productivity of high-alkalinity lakes through the addition of marine-derived nutrients via spray. By contrast, soft calcareous geologies in the south and east of Britain, which are often overlain by fertile drift deposits (alluvium, glacial sands and gravels, or Crag), furnish significantly more phosphorus during weathering and consequently such catchments have higher baseline productivity. Secondly, and partly in response to the above, there are marked biogeographical differences in species composition of these lakes within the UK. Thus, southern and eastern species that have a characteristically temperate-continental type distribution (such as *Potamogeton lucens*, *Potamogeton compressus*, *Myriophyllum verticillatum*, *Najas marina*) show little overlap with northern-Atlantic congeneric species found in the same broad lake type (such as *Potamogeton filiformis*, *P. polygonifolius*, *Myriophyllum alterniflorum*, *Najas flexilis*). Moreover, under naturally very low fertility this latter group of lakes frequently support species more characteristic of base-poor oligotrophic lakes, such as *Juncus bulbosus*, *Sparganium angustifolium* and various isoetids, that would largely never be found in naturally more fertile high-alkalinity lakes.

The separation of high-alkalinity lakes into high alkalinity (1-2.5 meq/l), very high alkalinity (above 2.5 meq/l), and marl lakes (located on hard limestone where calcium

is precipitated naturally) improves resolution but there remains a need to stratify high- and very high-alkalinity lakes into naturally infertile and naturally fertile types. A generic type-specific reference condition would be largely meaningless for such lakes, as what represents a valid reference condition for infertile lakes in northern-Atlantic parts of the UK is inappropriate for naturally more fertile lakes in temperate-continental areas (and vice versa). The need to resolve this dichotomy is heightened by the position of the UK in both the Northern and Central-Baltic Geographical Intercalibration Groups (GIGs), all of which have sought to intercalibrate national classifications of high-alkalinity lakes. This approach is largely consistent with the definition of Special Areas of Conservation (SAC) freshwater habitats in the UK which separates hard oligo-mesotrophic waters with benthic vegetation of *Chara* spp. (type 3140) from natural eutrophic lakes with *Magnopotamion* or *Hydrocharition*-type vegetation (type 3150) of which southern, northern or western and coastal variants are recognised.

We propose subdividing high- and very high-alkalinity lakes on the basis of rainfall distribution (see Figure 3.2) into a northern-Atlantic type and a southern-continental type since this is compatible with the ecoregion approach to defining GIGs, as well as with concepts of hydro-ecoregions and bioclimatic zones. An annual rainfall figure of 750 mm was used to make this separation. Clearly, lakes should be split into these subtypes in the most objective way possible given that a more fertile baseline (and therefore a more relaxed standard) is permitted for one group. This climate-based distinction essentially splits the UK high-alkalinity lake resource into those lakes on hard limestone and/or or with less fertile drift deposits, and those on predominantly soft, calcareous rock with fertile drift deposits. This is illustrated in Figure 3.3. Other criteria could be used to achieve essentially the same subdivision. The resultant geographical distribution of lake types structured by geology is illustrated in Figure 3.4.

Table 3.2 Constancy table summarising TWINSPAN 22 cluster botanical types

	1	*2*	*3*	*4*	*5*	*6*	*7*	*8*	*9*	*10*	*11*	*12*	*13*	*14*	*15*	*16*	*17*	*18*	*19*	*20*	*21*	*22*
Utricularia intermedia sens.lat.	I	I		I	II	I		I			I	I	I									
Eleocharis multicaulis	II	II		III	IV	I	I	I		I	I	I				I			I			I
Eleogiton fluitans	I	II	I	III	III	I		I		I	I	I										
Utricularia minor	II	I		II	II	I		I	I	I	I	I	I									
Utricularia stygia	I	I		I		I	I				I											
Batrachospermum	I		I	I		I	I															
Eriocaulon aquaticum				I	I	I		I														
Lobelia dortmanna	I	I		IV	V	IV	I	III	II	I	III	I										
Utricularia ochroleuca	I	I		I	I	I	I				I											
Potamogeton polygonifolius	III	IV	I	IV	IV	IV	II	I	II	I	III	II	I		I	I			I			
Sphagnum (aquatic indet.)	III	I	V	III	I	II	III	I	I	I	I			I	I	I	I					
Najas flexilis					I						I	I										
Pilularia globulifera				I				I		I	I											
Hypericum elodes		I		I						I	I											
Isoetes (indet)				I	I	I		I			I											
Nitella translucens		I		I	I	I		I	I	I	I	I							I			
Isoetes lacustris			I	I	III	IV	I	IV	III	I	III	I										
Juncus bulbosus	III	III	IV	V	V	V	V	III	IV	II	IV	II			I	I						
Subularia aquatica				I	I	II		II	II	I	I											
Isoetes echinospora				I		I	I	I	I	I	I											
Myriophyllum alterniflorum	I	I		III	V	IV	I	III	III	III	V	IV			I	I						I
Sparganium angustifolium	I	I	I	III	III	III	III	I	II	I	IV	I			I							
Sparganium natans	I	III		I	I	I		I	I	I	I	I	I		I							
Utricularia australis	I	I		I	I	I		I		I	I	I										
Littorella uniflora	I	II	I	IV	V	V	II	IV	IV	IV	V	V		I	I	II	I	I	I			I
Utricularia (indet)	I			I	I	I		I			I		I					I	I	I		
Menyanthes trifoliata	V	V	I	IV	IV	II	II	I	I	II	III	III	II	I	I	I	I		I	I		I
Nitella (indet)	I	I		I	I	I		IV	I	I	I	I			I	I	I	I	I		I	
Nuphar pumila	I				I					I	I						I					
Potamogeton natans	I	IV		IV	V	III	I	II	II	III	IV	III	I	I	III	II	I	I	I			
Luronium natans						I		II	I	I			I									
Callitriche hamulata		I	I	I	I	II	I	III	V	III	III	I		I	I	I	I					
Elatine hexandra				I	I	I		III	I	I	I	I				I	I					
Fontinalis squamosa								I	I	I										I		
Ranunculus omiophyllus		I				I			I	I		I				I	I					
Apium inundatum		II		I	I	I		I	I	II	I	I	I		I	I		I				
Chara virgata var.annulata					I					I	I	I	I									
Nitella opaca		I		I	I	I	I	I	I	I	II	I				I						
Potamogeton gramineus		I		I	I	I	I	I		I	II	III	I		I	I						
Lythrum portula						I		I	I	I	I	I				I	I					
Potamogeton x nitens		I		I	I			I		I	II	I				I				I		
Fontinalis antipyretica	I	I	I	I	I	II	I	V	IV	III	IV	II	I	I	II	I	I	I	II		I	
Nitella flexilis agg.		I		I	I	I		I	I	II	II	I	I		I	I	I	I				
Nuphar x spenneriana						I		I			I						I					
Utricularia vulgaris	I			I	I	I		I	I	I	I	I	II	I	I		I	I	I	I	I	
Potamogeton filiformis					I						I	III			I	I		I		I		I
Baldellia ranunculoides		I		I	I			I		I	I	I	I			I						
Chara virgata	I	I	I	I	I	I		I	I	I	II	I			I	I	I	I	I			
Nymphaea (exotic indet)		I								I				I								
Potamogeton x zizii					I						I	I	I							I		
Potamogeton perfoliatus				I	I	I		II	I	II	IV	IV	I		I	II	I	I	I	I	I	
Potamogeton alpinus		I		I	I	I		I	I	I	III	I	I		I	I						
Potamogeton praelongus					I	I			I	I	I	I	I		I	I		I	I	I		
Callitriche stagnalis	I	I	I	I	I	I	I	I	I	II	II	III	I	I	I	II	I	I	I	I		I
Chara (indet)		I		I	I	I		I	I	I	I	II	I	I	I	I	I	I	I	II	I	I
Potamogeton berchtoldii		I		I	I	I		II	I	II	III	I	I	I	II	II	I	I		I	I	
Ranunculus baudotii											I	I				I		I		I		I
Ranunculus peltatus	I							I	I	I	I	I	I		I	I						
Ranunculus aquatilis agg.		I						I	I	I	I	I	I		I	I	I		I	I	I	
Ranunculus hederaceus	I							I	I	I	I	I				I	I	I				

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
Ranunculus trichophyllus										I	I	I	I			I	I		I			
Fucus (indet)												I										I
Ruppia cirrhosa												I										I
Chara curta											I	I	I					I	I			
Hippuris vulgaris	I	I		I	I	I	I		I	I	I	II	II	I	I	I	I	IV	V	II	I	I
Callitriche agg.	I		I	I		I		I	I	I	I	I	I	I	II	I	I	I	III	I	I	
Callitriche platycarpa								I	I	I		I			I	I	I					
Eleocharis acicularis		I						I		I	I	I			I	I	I					
Elodea nuttallii								II		I	I			I	I	I	I	I	I	I	I	
Potamogeton lucens		I			I			I		I	I	I	II		I	I	I	I	I	I	I	
Potamogeton obtusifolius		I		I	I	I				II	I		I		III	I	I	I	I	I		
Ranunculus aquatilis sens.str.										I		I				I					I	
Sparganium emersum	I			I	I	I		I		II	I	I	I	I	III	I	I					
Callitriche hermaphroditica								I		I	I	II			I	II	I				I	
Persicaria amphibia		I						I	I	II	I	II	I	I	I	III	II	I	I	I		
Elodea canadensis		I			I	I		I	I	III	II	I	I	I	IV	III	III	I	IV	I	I	
Drepanocladus fluitans	I		I											I		I						
Hottonia palustris		I											I			I	I		I	I		
Hydrocharis morsus-ranae													II		I		I	I	I	I	I	
Lemna minor	I	II	I		I			I	I	II	I	I	I	II	IV	III	IV	I	I	I	I	I
Lemna trisulca		I								I			I	I	III	II	II	I	I	I	I	
Nitella mucronata														I			I				I	
Nuphar lutea	I	I	I	I	I	I		II	I	II	I	I	IV	V	IV	I	IV	I	IV	I	I	
Oenanthe aquatica													I	I			I					
Ranunculus lingua										I			I	I	I	I	I					
Riccia fluitans	I		I											I		I	I					
Sagittaria sagittifolia								I					I		I	I	I	I	I			
Spirodela polyrhiza															I	I	I				I	
Stratiotes aloides													II	I	I		I	I	I	I	I	
Butomus umbellatus													I			I	I		I		I	
Elatine hydropiper								I		I						I	I					
Potamogeton crispus	I							I		I	I	I	I	I	I	III	II	II	I	I	II	
Potamogeton trichoides													I			I	I	I				
Ranunculus circinatus													I		I	I	I	I	III	I	I	
Zannichellia palustris	I									I	I	I	I	I	I	II	IV	I	IV	III	I	I
Ceratophyllum demersum								I		I			I	II	I	I	III	I	IV	II	V	
Chara aspera					I					I	I	I	I			I		II	III	I		I
Chara baltica																		II	I			
Chara connivens														I			I	II	I	I	I	
Chara globularis					I					I		II			I	II	I	II	IV	I	I	
Chara hispida												I	I	I	I	I	I	III	IV	I	I	
Chara intermedia																		IV	II	I		
Chara pedunculata													I					I	II			
Chara vulgaris		I										I	I		I	I	I	I	II	II	I	
Chara vulgaris var. papillata												I				I		I	I			I
Chara contraria var.contraria												I	I		I	I	I	I	I	I		
Lemna minuta														I			I	I			I	
Leptodyction riparium																	I	I	I		I	
Myriophyllum spicatum					I			I		I	I	II	I		I	III	I	IV	V	I		I
Myriophyllum verticillatum													I					I	I	I	I	
Najas marina													I	I			I	IV	V	III	II	
Nitellopsis obtusa													I					III	III	I		
Potamogeton coloratus												I	I	I				I				
Potamogeton compressus													I						I	I	I	
Potamogeton friesii												I	I	I	I	I	I	I	II	II	I	
Potamogeton pectinatus	I				I			I		I	I	II	I		I	III	II	V	V	IV	II	III
Potamogeton pusillus		I						I		I	I	II	I		I	II	II	II	III	II	I	
Ruppia maritima												I						I				IV
Ruppia (indet)																		I				II
Nymphaea alba	III	II	I	III	IV	I		I	I	II	II	I	IV	IV	II	I	II	I	II	II	I	

I = 2-20% frequency in that cluster, II = 20-40%, III =40-60%, IV = 60-80%, V => 80%. Thirty taxa occurring at level I only in two or fewer clusters were excluded for clarity.

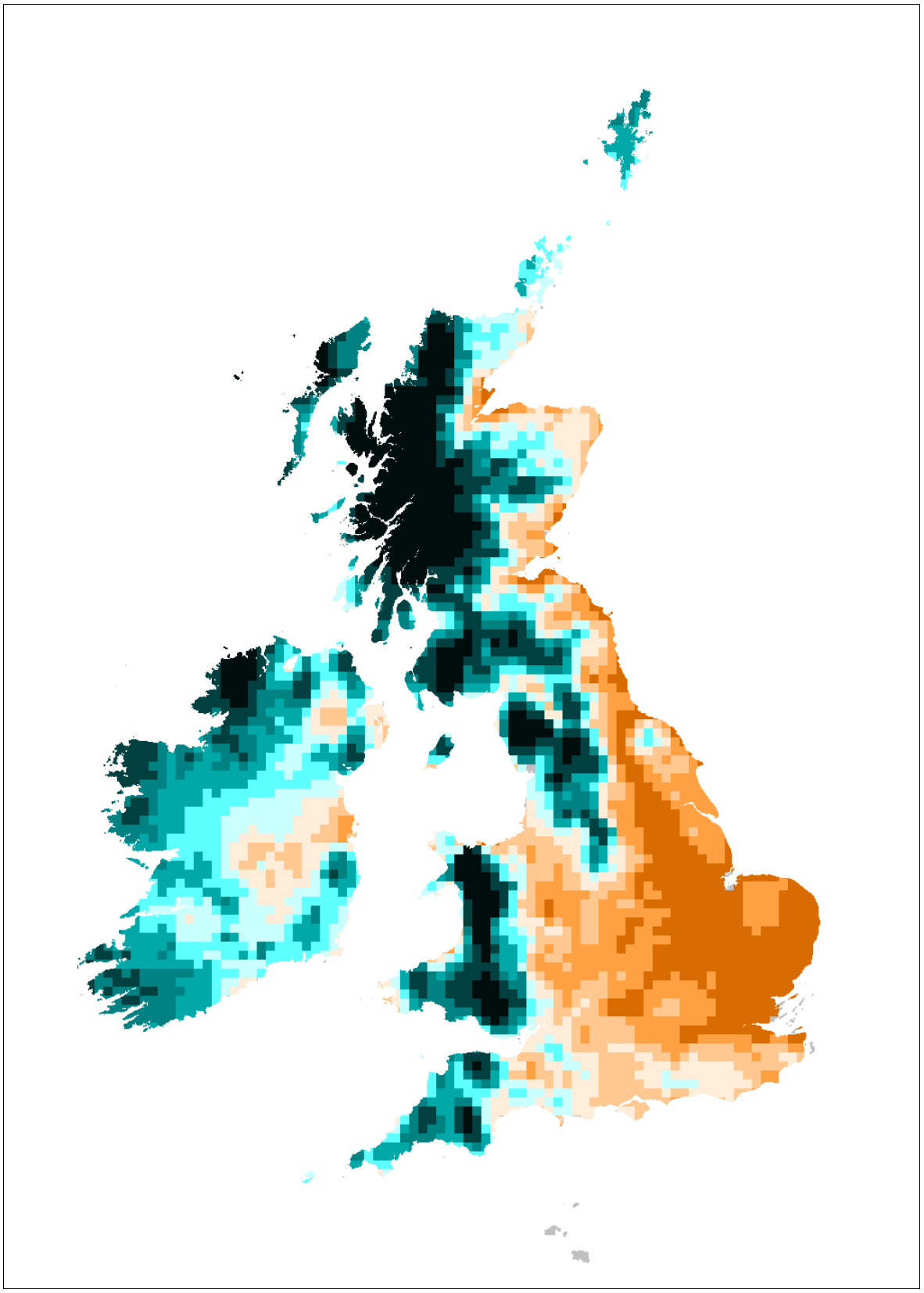


Figure 3.6 Annual rainfall distribution in the British Isles, based on UK Meterological Office annual mean data for the period 1960-1992. The lightest brown cells experienced a mean annual rainfall over this period of 750 mm (reproduced from Preston *et al*., 2002).

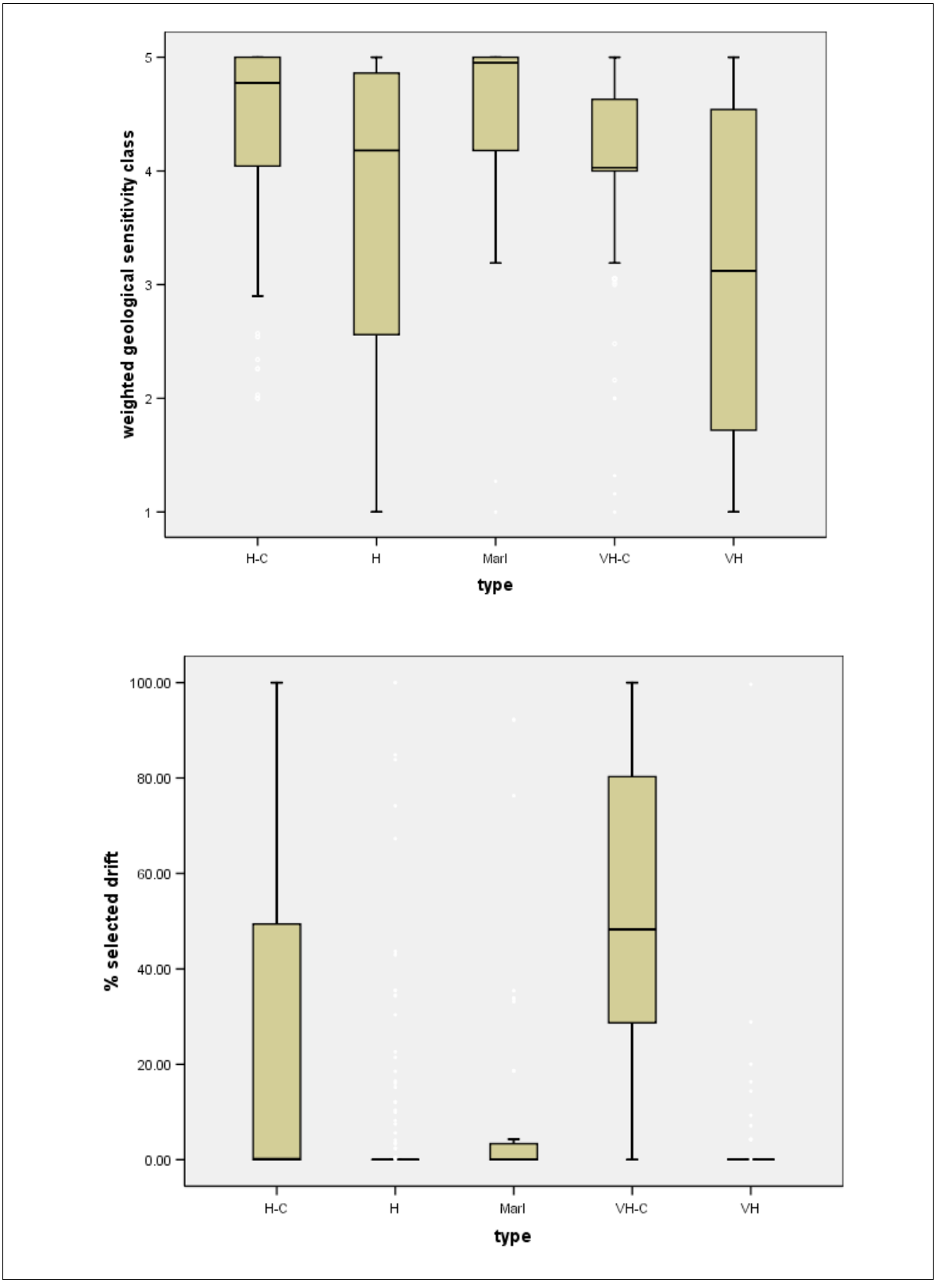


Figure 3.7 Relationship between high alkalinity lake types and base-status of underlying solid geology and extent of selected drift (alluvium, glacial sands and gravels and crag)

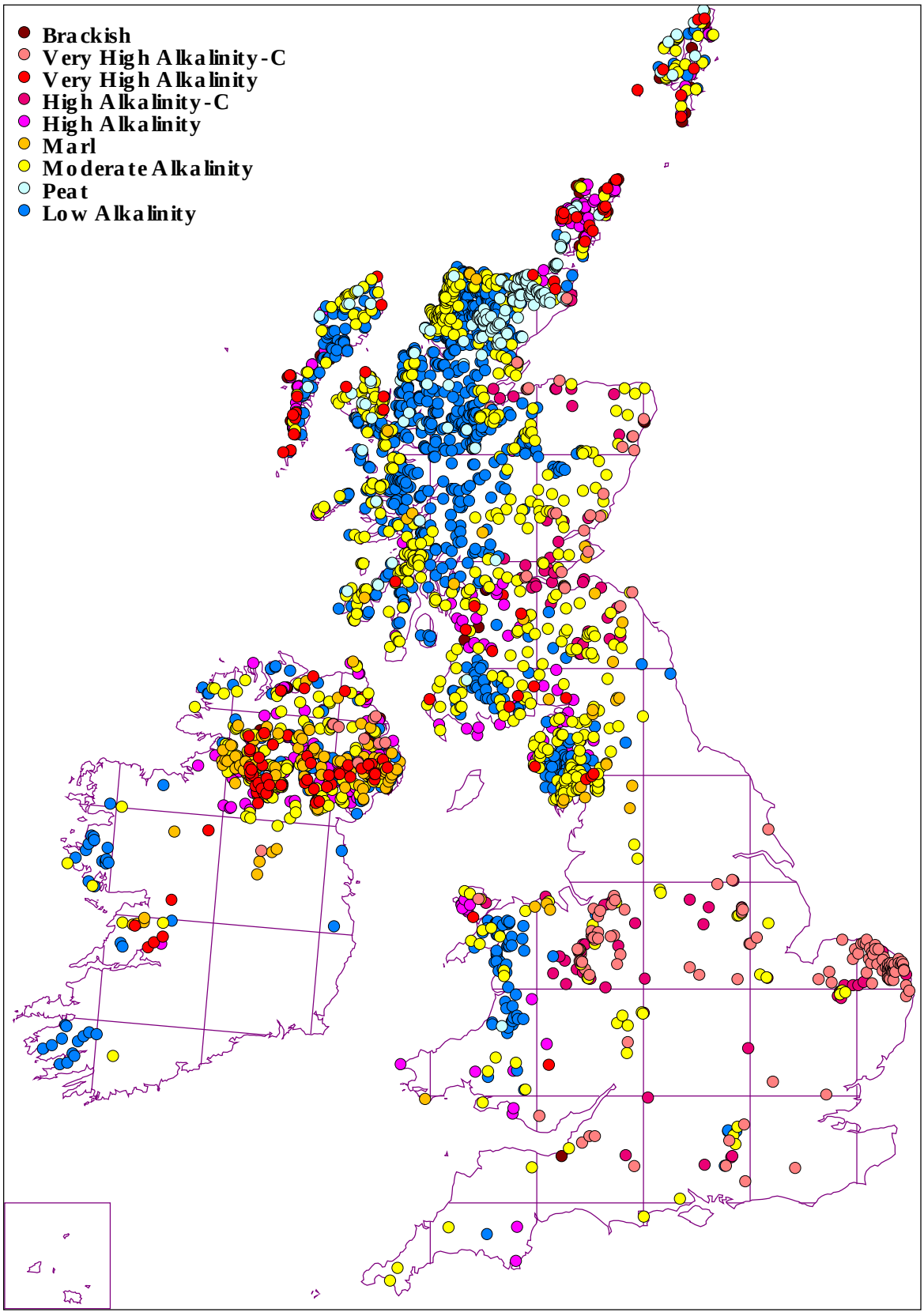


Figure 3.8 Distribution of lakes by major geological type for which there is supporting macrophyte data

3.4 Assessment of typology

3.4.1 Relationship between botanical and environmental types

A cross tabulation of botanical types versus the proposed environmental lake typology (Table 3.3) indicates a far from clear alignment between biology and environment at this level. For example, each lake type contains on average 12 of the 22 botanical clusters, while each botanical cluster occurs, on average, in 11 of the 20 lake types. There are, however, clear patterns. Thus, peat lakes are dominated by botanical types 4, 6 and 7, low-alkalinity lakes are dominated by clusters 4, 5 and especially 6, while moderate-alkalinity lakes show some overlap with this pattern as well as supporting clusters 9, 10 and 11. Moreover, there are only two lake types (peat VSh and HA Sh) in which more than two botanical clusters are common (over 20 per cent of the number of surveys in that type). The core GB lake typology (UKTAG, 2003) cannot, by definition, cover a range of secondary variables that influence botanical composition; natural variation in botanical type within a given lake type is therefore to be expected. Since the observed botanical lake 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 environmentally-defined lake type for non-natural reasons.

3.4.2 Relative importance of other environmental factors

The results of a CCA in which site biology is constrained by data on a wide range of environmental variables confirms the overriding importance of alkalinity in structuring lake macrophyte communities (Figure 3.9). Thus, alkalinity explains as much variation in macrophyte composition as the six next best factors, including lake area, perimeter, depth, conductivity and drift geology, combined. In a forward selection analysis (Table 3.4) the high ranking of both depth and factors related to drift geology and freshwater sensitivity class support a typology based on alkalinity, depth and geographical distribution (via the correlations discussed in Section 3.3.2). However, it is also evident from Figure 3.9 that there is a degree of redundancy in the use of depth since depth tends to decrease strongly with alkalinity for obvious reasons.

Figure 3.10 offers a refined perspective by mapping all surveys in ordination space (Detrended Correspondence Analaysis (DCA) axes 1 and 2) coded by their lake type. The two dominant themes to emerge are:

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

v. Turnover between surveys increases with increasing alkalinity, as reflected by the widening range of scores on axis 2. While this might reflect an increasing range and intensity of impacts on lakes as their alkalinity and fertility increase, the general response to pressures is for homogenisation to a common pool of “tolerant” species. Hence, it is more likely that the observed pattern reflects increasing opportunities for niche diversification and/or for different species to dominate at individual sites. This is likely to

be a response to the relaxation of nutrient limitation at higher fertility coupled with the greater relative importance of both physical disturbance and competition for light. Thus, assemblages in high-alkalinity lakes may vary between the extremes of usually small, wave-sheltered, isolated sites in which free-floating or floating-leaved rooted species are the major contributors, through to larger, more exposed sites with extensive open water and relatively short retention times where submerged, streamlined growth or body flexibility is necessary to withstand greater hydraulic forces. By contrast, at low alkalinity the vegetation of sheltered and exposed sites is more similar due to the overarching importance of nutrient limitation. This interpretation of the distribution of sites on DCA axis 2 is supported by Figure 3.9 in which the environmental vectors most strongly correlated with axis 2 are physical factors such as perimeter and fetch.

The forward selection analysis in CCA highlights a pool of variables and their derivatives (alkalinity, perimeter, drift geology, conductivity, geology base status (as FSC), area, depth and altitude in descending order of importance) that together account for 90 per cent of the explainable variation in lake macrophyte composition. In general, it seems that a typology built on alkalinity, depth and geographical distribution will adequately describe the variation in macrophyte composition in UK lakes, even if these are (sometimes) correlates of, rather than the specific variables driving, differences in plant composition. Hydraulic disturbance linked to physical factors such as lake area, perimeter and shape is probably the closest parallel in lakes to the importance of slope in governing river macrophyte communities. However, data on variables such as perimeter length and fetch is currently not universally available for UK lakes, making it more difficult to stratify the resource in this way. Such variables also may not adequately reflect the true disturbance regime at a site. In large lakes with complex shorelines, vegetation of sheltered and exposed habitats may coexist in the same water body so the structuring role of physical factors is a less striking feature of the analysis than is the case in rivers.

Table 3.3 Cross-tabulation of botanical clusters and lake types. Figures indicate percentage of sites in that lake type (see Table 3.1) that belonged to a given botanical cluster. N-clusters refers to the number of botanical clusters found within a given lake type while n-types indicates the number of lake types over which a cluster was distributed.

Type	c1	c2	c3	c4	c5	c6	c7	c8	c9	c10	c11	c12	c13	c14	c15	c16	c17	c18	c19	c20	c21	c22	n-clusters
PtSh	5.5		6.1	6.1	14.0	33.5	18.9		9.1	0.6	4.9	1.2											**10**
PtVSh	2.9		11.8	20.6		20.6	20.6		8.8	5.9	2.9	5.9											**9**
LAD				2.4	4.8	57.1		10.7	17.9		7.1												**6**
LASh	1.5	0.9	1.9	14.3	17.7	43.5	3.6	2.4	7.2	1.8	4.4	0.3		0.2	0.2	0.2	0.1						**16**
LAVSh	7.2	1.1	8.3	14.9	13.8	32.2	5.4	1.1	6.5	2.2	2.9	0.7		1.1	1.4	0.7	0.4						**16**
MAD					6.9	6.9		6.9	13.8	13.8	48.3	3.4											**7**
MASh	0.7	1.5	0.4	6.3	20.1	12.2	1.0	2.6	3.8	16.0	17.3	4.6	0.3	0.6	4.8	6.6	0.6			0.3	0.1		**19**
MAVSh	1.3	2.6	1.0	5.6	4.6	6.3	1.0	0.3	7.6	25.2	8.3	1.7	0.3	1.7	17.9	12.9	1.0			0.7			**18**
MarlSh					1.0	1.0				11.8	3.9	9.8	3.9	5.9	40.2	19.6	2.0	1.0					**11**
MarlVSh										3.0			4.0	5.0	74.0	13.0	1.0						**6**
HASh		1.5		2.0	3.5	4.0	0.5		2.5	9.1	6.1	20.2		1.5	23.7	22.2	2.0			0.5		0.5	**15**
HAVSh					1.7				1.1	7.8	1.1	7.3		1.1	37.4	36.3	3.9	1.1				1.1	**11**
HASh-C	2.4	1.2			1.2					9.4	1.2	7.1	2.4	5.9	4.7	38.8	25.9						**11**
HAVSh-C	2.0			1.0						4.0	1.0	1.0	3.0	10.9	16.8	36.6	21.8	1.0			1.0		**12**
VHASh				1.4	7.1	1.4				4.3	5.7	25.7		1.4	31.4	14.3	4.3	1.4				1.4	**12**
VHAVSh	1.4	1.4			4.1					1.4	2.7	6.8	4.1	4.1	39.2	27.0	6.8			1.4			**12**
VHASh-C	2.7			1.4		1.4				4.1	1.4	4.1	6.8	6.8	5.4	33.8	25.7		4.1	2.7			**13**
VHAVSh-C	0.2	0.2		0.2		0.2		0.8		0.5		0.2	3.6	12.4	8.2	5.0	24.7	11.7	7.6	9.9	14.0	0.9	**17**
BSh					4.5	2.3					4.5	27.3			6.8	15.9		6.8				31.8	**8**
BVSh												16.0			4.0	20.0		12.0		4.0		44.0	**6**
n-types	**11**	**8**	**6**	**12**	**14**	**14**	**7**	**7**	**10**	**17**	**17**	**18**	**9**	**14**	**16**	**16**	**14**	**7**	**2**	**7**	**3**	**6**	

>10% >20% >40%

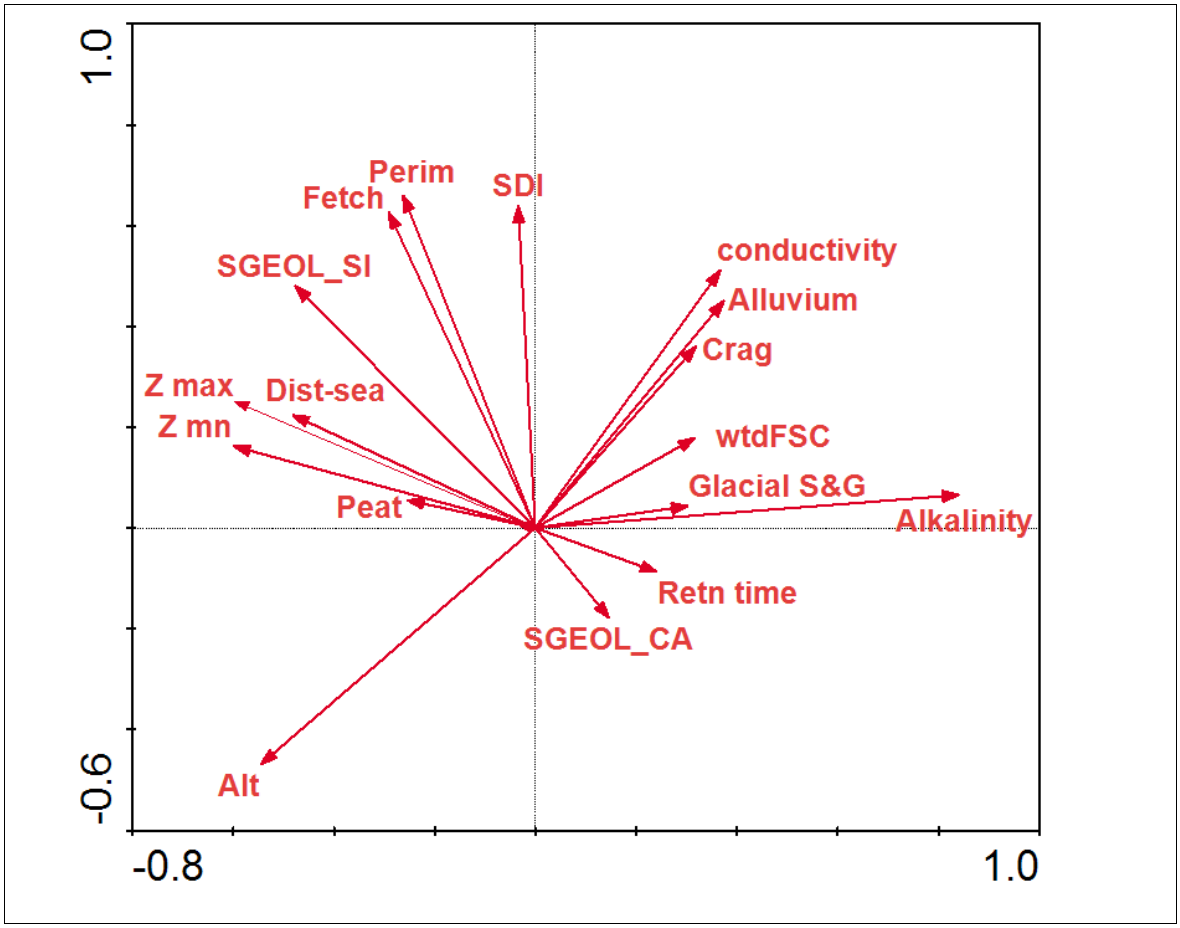


Figure 3.9 Results of a CCA summarising the distribution of surveys (not shown) in relation to environmental variables (as identified by forward selection – Table 3.4). For clarity, some variables positioned close to the origin are not shown.

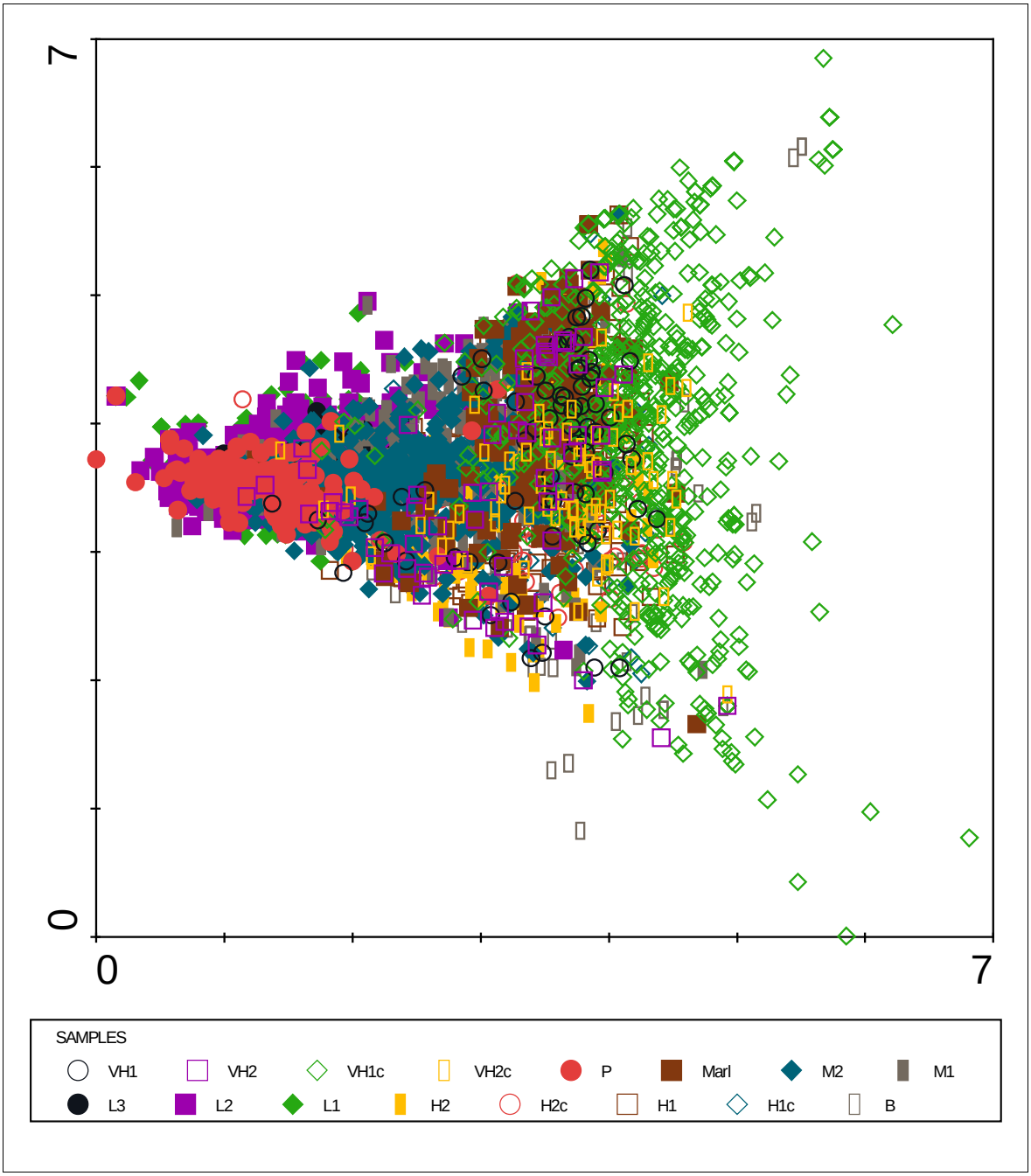


Figure 3.10 Ordination by DCA of the full lake macrophyte survey dataset showing site scores on axes 1 and 2. Note the increasing dispersion of surveys on axis 2 with increasing alkalinity (see VH1c (very high-alkalinity, shallow, continental lakes) versus P (peat) and L (low-alkalinity) lakes).

Table 3.4 Results of forward selection analysis and Monte Carlo permutation testing of the importance of different environmental variables in a CCA in sequentially explaining variation in lake macrophyte composition

Variable	Conditional effects		
	LambdaA	P	F
Alkalinity	0.34	0.003	249.71
Perimeter (log_{10})	0.09	0.003	64.05
Crag	0.07	0.003	55.72
Conductivity	0.06	0.003	44.54
FSC5	0.05	0.003	40.07
Area (log_{10})	0.05	0.003	34.62
Max depth (log_{10})	0.04	0.003	30.74
Altitude	0.03	0.003	27.48
Altitude (log_{10})	0.03	0.003	19.41
FSC4	0.02	0.003	18.8
Alkalinity (log_{10})	0.02	0.003	12.47
SDI	0.02	0.003	12.55
Distance to sea	0.01	0.003	13.29
Alluvium	0.01	0.003	11.66
Distance to sea (log_{10})	0.01	0.003	9.71
Calcareous geology	0.01	0.003	8.39
Glacial sands & gravels	0.02	0.003	8.55
Conductivity (log_{10})	0	0.003	7.48
Siliceous geology	0.01	0.003	6.91
Retention time (log_{10})	0.01	0.003	6.18
FSC3	0.01	0.003	6.93
Fetch	0.01	0.003	5.53
Mean depth (log_{10})	0	0.003	5.64
Peat	0.01	0.003	5.32
Boulder clay	0.01	0.003	5.34
SDI (log_{10})	0	0.003	4.48
FSC1	0.01	0.003	3.83
Area	0	0.007	2.83
Perimeter (log_{10})	0	0.01	2.04
Fetch (log_{10})	0.01	0.003	2.72
Retention time	0	0.023	2.49
Mean depth	0	0.003	2.71
Max depth	0	0.037	1.57
Wtd FSC	0.01	0.03	1.47

4 Developing metrics for ecological assessment 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. Thiebaut *et al*. (2002) review approaches to the assessment of riverine ecosystems using macrophytes, several of which could be used to assess lakes. These include:

i. Classical phytosociological approaches, as used mainly in France and Germany.

ii. Measurements of cover or biomass. In lakes, equivalent measures include PVI or maximum depth of colonisation.

iii. Typing of lakes based on the composition of their vegetation.

iv. Application of expert-based or empirically derived water quality indices to community level data.

v. Description of vegetation in terms of biological traits or functional groups.

vi. Use of diversity indices (for example covering total richness, or the richness of rare or indicator taxa).

The suitability of these approaches depends on the methods used to collect the data, limitations of this data and purposes for which this data is used. Thus, standardised survey approaches 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 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 a taxon is, or was, absent from a site. 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. This problem has preoccupied some critics of river macrophyte surveys approaches (Lansdown, 2007). Phytosociological and typological approaches are especially vulnerable because they use raw lists of species to generate associations or floristic types, and require data of the same format to classify new observations. Approaches heavily reliant on finding specific taxa and identifying them correctly may be especially vulnerable to detection bias. Willby and Casas-Mulet (2009) discuss the issues of detection bias in assessing changes in lake vegetation. Detection bias appears to be an accepted and unavoidable reality of lake

macrophyte surveys; the chance of recording some taxa depends purely on where a rake or grapnel lands, or which side of a boat is sampled from, and sampling issues are analogous to those associated with the use of grabs for collecting littoral or profundal invertebrates, where the sampler is effectively operating blind. Nevertheless, in current WFD classification systems for lake macrophytes, several European countries have based their method on the proportion of characteristic type-specific taxa that can be found in a water body (see Schaumburg *et al*., 2004; Stelzer *et al*., 2005; Coops *et al*, 2007). Dispersal limitation creates an additional complication in the case of water plants since, in contrast to mostly mobile macroinvertebrates, the majority of 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 taxa can have similar scores they are treated as ecologically equivalent, while strongly taxonomic approaches will treat them as discrete entities, irrespective of their ecological similarity. Two sites may thus have few species in common, while still having similar scores for a given biological metric. Some metrics will bypass issues of detection bias (such as biomass or cover) although issues of sampling error will remain, 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.

The main disadvantages of current metric-based approaches are firstly their ‘remoteness’ from biological data, secondly, the weak evidence base from which to validate the use of some metrics and to support interpretation, and thirdly, the typical focus on single pressures (typically chemical water quality) that is incompatible with a more holistic assessment. The following analyses are designed to identify a pool of candidate metrics to assess ecological status in lakes 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 of attaching numerical ranks to taxon names, to derive a site index relating the biota to environmental variation, whether natural or pressure-related, measured or merely inferred. Such ranks may be derived from empirical data (in terms of species optima on a given environmental gradient, as determined by weighted or reciprocal averaging, for example), 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. The basic principles of using water plants for bioindication appear to have originated in rivers, where such approaches were being trialled by European workers such as Kohler and Carbiener in the early 1970s (Kohler, *et al*., 1973; Carbiener *et al*., 1990). Newbold and Palmer (1979) appear to have been among the first to adapt this approach to lakes. In this project, a method was devised for empirical adjustment or recalibration of expert ranks as a hybrid between strict empirical and expert approaches. Site scores derived from this ranking were subsequently validated against measured pressure data.

There are various practical and philosophical strengths and weaknesses in the use of empirical or expert ranking systems. Diekmann (2003) provides a useful review. These are discussed below.

Expert ranking systems reflect a diffuse evidence base incorporating, on the one hand, literature reports or anecdotal observations of 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 relationships between biological traits and environment. Such systems may be closer to the concepts of ecological structure and function advocated by the WFD, but it is difficult to extract precisely what such systems are measuring. Consequently, such indices can be related (or not) to measured environmental data which may reflect the pressure to which the index is designed to respond, but the index itself is effectively unfalsifiable. Expert ranking systems 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 and thus it is clear precisely what they are measuring. However, it is unclear whether a single directly measured environmental variable (such as annual mean water column total phosphorus) 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. 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. A further concern relates to the use of chemistry from routine water sampling programmes to derive metric values, since routine sampling usually 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; the comparability and reliability of such data can be questionable, yet this becomes buried within the resulting species scores. A final problem in the case of macrophytes is that naturally productive sites may exhibit surprisingly low nutrient concentrations because most of the available nutrients are sequestered by the macrophytes themselves.

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 have usually been applied in assessing nutrient enrichment (Palmer *et al*., 1992; Holmes, 1995; Penning *et al*., 2008). Other pressures that could be considered usefully via this approach include hydromorphological alteration in the form of water level regulation (such as through sensitivity to water level fluctuations) or shoreline modification (such as through substrate-associations), or acidification. Such metrics for lake macrophytes would parallel the development of the Lotic Invertebrate Flow Evaluation (LIFE) Index and the Acid Waters Indicator Community Scores (AWICS) currently used for riverine invertebrates.

4.2.2 Approach

In developing a compositional metric to assess lake ecological status, the focus has been on the assessment of nutrient impacts. The grounds for this are quite simple: nutrient enrichment is the most widespread pressure affecting European lakes. Analysis by CCA in which intrinsic and impact variables were included as predictors (Table 4.1) reveals that total phosphorus (TP) and the extent of impacted land cover are the second and third best predictors of variation in lake macrophyte composition after alkalinity. Consequently, there is a strong empirical basis for the assessment of nutrient-related pressures using macrophytes.

In the UK the Trophic Ranking System (TRS) devised by Palmer *et al*. (1992) has been used hitherto to infer information on the nutrient status of lakes based on their macrophyte communities. This has recently been replaced by the Plant Lake Ecotype Index (PLEX) in the light of a new botanical classification of lake types (Duigan *et al*., 2006), although PLEX focuses on change in base status. The TRS is an expert system based exclusively on catchment characteristics such as land use and geology, since little contemporary nutrient data exists for UK lakes, or was collated into a single resource at the time. Many European countries including Sweden, the Netherlands and the Czech Republic operate ranking systems derived and structured in similar ways. It is common within such systems to refer to species as characteristically ‘eutrophic’ and so on, or even at finer levels of description, such as ‘meso-eutrophic’. However, the rhetoric and reality of compositional metrics lie somewhere apart. Until recently, few metrics quoted measured nutrient concentrations and it is unclear how reliably labels describing trophic status have been applied. They do not appear to be referenced in any way to the original Organistion for Economic Cooperation and Development (OECD) definitions of different levels of trophic status, while hybrid terms such as ‘meso-eutrophic’ are often long-hand for the numerical position of a species within a ranking system and consequently are tautological. For some species, the range of tolerance may be wide but this may not be apparent from the rank applied.

Table 4.1 Results of a CCA on global UK lake macrophyte survey data followed by forward selection of intrinsic and impact environmental variables. Possible indicators of impact are asterisked; gs and ann refer to growing season and annual means respectively.

Variable	Conditional effects		
	LambdaA	P	F
Alkalinity	0.34	0.002	249.71
TP ann (log_{10})*	0.1	0.002	73.7
Impact LC*	0.09	0.002	65.97
Crag geology	0.07	0.002	54.39
Conductivity	0.06	0.002	47.61
Perimeter (log_{10})	0.06	0.002	42.41
Altitude	0.03	0.002	25.6
Area (log_{10})	0.03	0.002	24.56
Altitude (log_{10})	0.03	0.002	22.72
pH	0.02	0.002	15.96
Distance to sea (log_{10})	0.02	0.002	15.84
FWSC1	0.02	0.002	11.54
FWSC2	0.01	0.002	12.85
Alluvium	0.02	0.002	12.06
SDI	0.01	0.002	10.59
Alkalinity (log_{10})	0.01	0.002	9.65
Chlorophyll gs (log_{10})*	0.01	0.002	7.93
FWSC5	0.01	0.002	7.85
Conductivity (log_{10})	0.01	0.002	7.63
Distance to sea	0.01	0.002	7.27
NO_3-N gs (log_{10})*	0.01	0.002	6.38
TP gs (log_{10})*	0.01	0.004	6.08
FWSC4	0	0.002	6.18
Max depth (log_{10})	0.01	0.002	7.03
Retention time (log_{10})	0.01	0.002	5.77
Calcareous geology	0.01	0.002	6.14
Mean depth (log_{10})	0	0.002	5.52
Peat	0.01	0.002	5.3
Fetch	0.01	0.004	4.73
Glacial sands and gravels	0	0.002	4.55
Chlorophyll gs*	0.01	0.01	4.37
SDI (log_{10})	0	0.002	4.12
Siliceous geology	0.01	0.002	4.27
FWSC3	0	0.002	5.91
Boulder clay	0.01	0.002	3.81
Fetch (log_{10})	0	0.002	2.96
Area	0.01	0.006	2.5
Retention time	0	0.012	2.69
Mean depth	0	0.008	2.75
NO_3-N gs*	0	0.044	1.68
Perimeter	0.01	0.02	1.93
Max depth	0	0.056	1.44
wtdFSC	0	0.092	1.33

One of the difficulties of expert systems is that they reflect the limited 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. Consequently there have been attempts to improve expert systems, either by ground-truthing them against measured environmental data (see Ertsen *et al*., 1998; Warnelink *et al*., 2005), or by adjusting scores statistically to reflect

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 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.* (2000) to form an expert index for lake hydrophytes. Recalibration of existing expert indices using such approaches is discussed by Diekmann (2003). The adjusted Ellenberg N scores for the UK flora formed the basis of this index since these covered a wider range of species, provided UK wide geographical coverage, and had a more explicit empirical basis than TRS scores. 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. For species without initial expert scores, such as Charophytes and selected bryophytes, scores are obtained at this step based on the regression between axis 1 species scores and the expert scores of known species. A second iteration is then 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 were rescaled to run from one to 10. In deference to the convention adopted by other indices (such as Ellenberg, Trophic Diatom Index, Trophic Ranking Score) high scores are associated with the most nutrient-tolerant species. The LMNI 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.

LMNI scores of all scoring species are listed in Table 4.2. This covers all hydrophyte species and common hybrids found in UK lakes which have at least two occurrences in the database. Minor changes in the species scores first calculated have been implemented, either to bring the taxonomy of Charophytes into line with Bryant *et al*. (2002), or to reflect a better method for deriving scores when taxa are only identified to genus level (see Appendix 1). The effect of these changes on site scores and the EQRs reported is negligible since the weighted overall change in species LMNI score amounts to a reduction of only 0.16 units. Several bryophytes that had been previously included have also been removed since it appears that previous recording of these was too variable for records to be of use. Also, many taxa appear to be confined to intermittently submerged sections of the upper shore that are covered in perimeter surveys, or associated flush zones or small inflows and are therefore not indicative of the vegetation of the water body itself.

A consequence of the statistical approach to deriving species scores is that few species will have extreme scores (because few species occur exclusively with other high- or low-scoring species), while most species will have scores located near the central part of the gradient, due to varying degrees of co-occurrence (Figure 4.1). A metric based on these species scores must therefore produce a narrower range of site scores, in this case typically ranging from around three to eight, rather than values that cover the full range of species scores (one to 10).

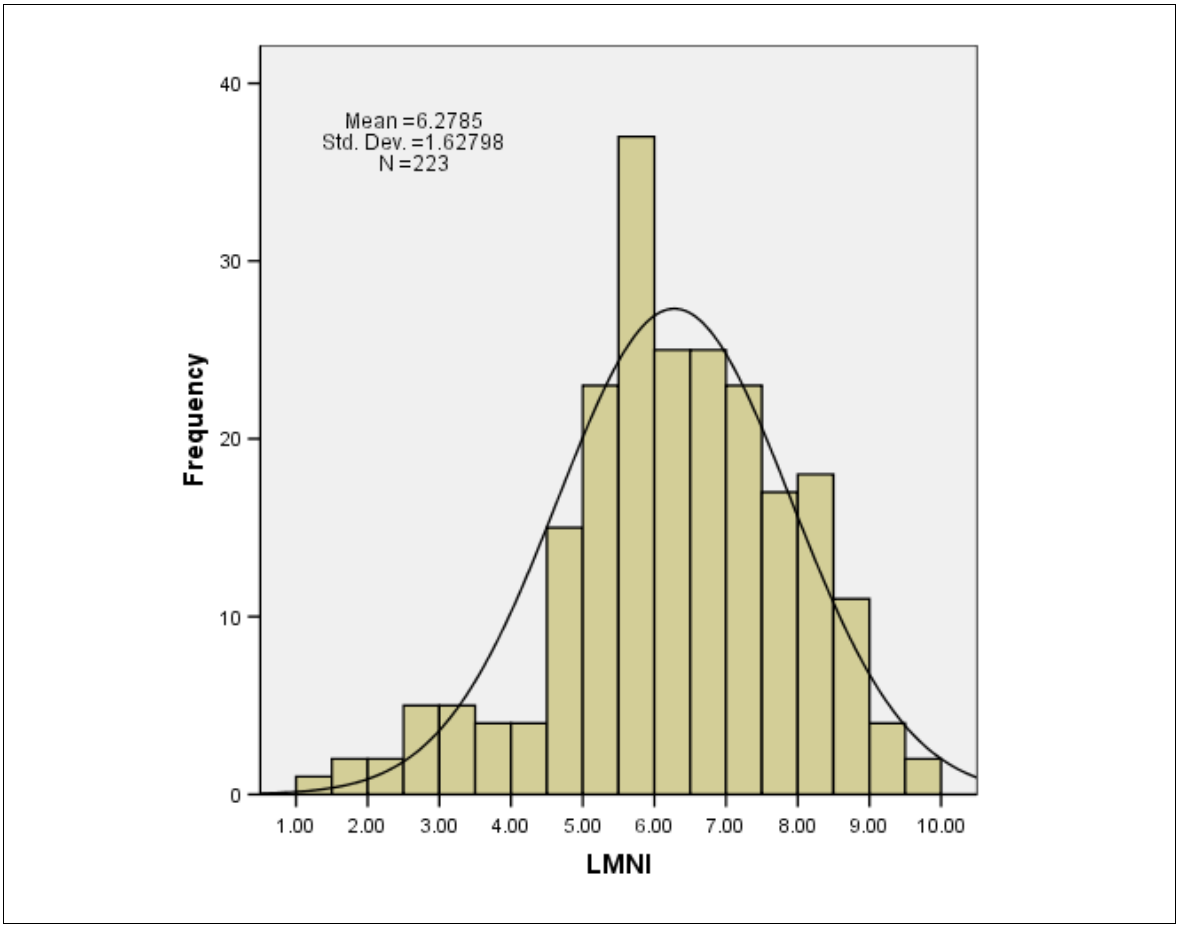


Figure 4.1 Distribution of LMNI species scores following adjustment of Ellenberg N scores

Table 4.2 Lake Macrophyte Nutrient Index (LMNI) scores and Functional Group membership for 186 taxa treated as lake hydrophytes

Taxa	LMNI	FG membership	Taxa	LMNI	FG membership
Alisma gramineum	5.57	13	Najas marina	8.84	14
Apium inundatum	5.69	7	Nitella (indet)	5.47	2
Aponogeton distachyos	7.38	16	Nitella confervacea	4.91	2
Azolla filiculoides	9.28	1	Nitella flexilis agg.	5.6	2
Baldellia ranunculoides	5.58	13	Nitella gracilis	4.38	2
Batrachospermum	3.02		Nitella mucronata	8.42	2
Butomus umbellatus	8.46	13	Nitella opaca	5.27	2
Callitriche sp(p).	5.97	6	Nitella translucens	5.17	2
Callitriche brutia	6.49	6	Nitellopsis obtusa	7.62	2
Callitriche hamulata	5.47	6	Nuphar lutea	6.92	12
Callitriche hermaphroditica	6.71	5	Nuphar lutea x spenneriana	5.61	12
Callitriche obtusangula	7.83	6	Nuphar pumila	5.33	12
Callitriche platycarpa	7.45	6	Nymphaea spp (exotic spp or cultivars)	5.63	12
Callitriche stagnalis	5.98	6	Nymphaea alba	5.54	12
Callitriche truncata	8.35	6	Nymphoides peltata	8.07	10
Ceratophyllum demersum	8.67	5	Oenanthe aquatica	8.31	7
Ceratophyllum submersum	8.82	5	Persicaria amphibia	7.25	10
Chara aculeolata	6.54	2	Pilularia globulifera	5.18	4
Chara aspera	6.39	2	Potamogeton alpinus	5.79	16

Chara baltica	8.6	2	Potamogeton x griffithii	5.24	16
Chara canescens	8.13	2	Potamogeton berchtoldii	6.07	14
Chara connivens	7.92	2	Potamogeton coloratus	6.7	16
Chara contraria var. contraria	7.47	2	Potamogeton compressus	8	14
Chara contraria var. hispidula	7.35	2	Potamogeton crispus	7.64	17
Chara curta	6.52	2	Potamogeton epihydrus	2.78	16
Chara globularis	7.18	2	Potamogeton filiformis	6.16	15
Chara hispida	6.87	2	Potamogeton x suecicus	6.11	15
Chara intermedia	8	2	Potamogeton friesii	7.64	14
Chara rudis	6.94	2	Potamogeton x lintonii	8.35	14
Chara sp	6.31	2	Potamogeton gramineus	5.51	16
Chara virgata	5.55	2	Potamogeton x zizii	5.69	16
Chara virgata var. annulata	5.62	2	Potamogeton x sparganifolius	5.54	16
Chara virgata var. virgata	5.31	2	Potamogeton x nitens	5.6	17
Chara vulgaris	7.2	2	Potamogeton lucens	7.02	17
Chara vulgaris var. longibracteata	8.37	2	Potamogeton x salicifolius	6.89	17
Chara vulgaris var. papillata	7.21	2	Potamogeton natans	5.16	16
Chara vulgaris var.vulgaris	7.35	2	Potamogeton obtusifolius	6.72	14
Crassula helmsii	6.18	5	Potamogeton pectinatus	8.25	15
Damasonium alisma	4.64	13	Potamogeton perfoliatus	5.83	17
Drepanocladus fluitans	6.65	3	Potamogeton polygonifolius	3.5	16
Elatine hexandra	5.41	11	Potamogeton praelongus	5.77	17
Elatine hydropiper	7.39	11	Potamogeton pusillus	7.61	14
Eleocharis acicularis	6.75	4	Potamogeton rutilus	5.62	14
Eleocharis multicaulis	1.93	4	Potamogeton trichoides	8.39	14
Eleogiton fluitans	3.45	15	Potamogeton x cooperi	5.67	17
Elodea callitrichoides	7.92	5	Ranunculus aquatilis agg.	6.5	18
Elodea canadensis	7.14	5	Ranunculus aquatilis sens.str.	6.61	18
Elodea nuttallii	6.92	5	Ranunculus baudotii	6.82	18
Enteromorpha (Ulva)	8.42		Ranunculus circinatus	8.64	5
Eriocaulon aquaticum	1.67	4	Ranunculus fluitans	7.42	18
Filamentous algae	6.39		Ranunculus hederaceus	6.6	11
Fontinalis antipyretica	5.42	3	Ranunculus lingua	7.61	10
Fontinalis squamosa	4.56	3	Ranunculus omiophyllus	5.76	11
Groenlandia densa	6.49	5	Ranunculus peltatus var peltatus	6.48	18
Hippuris vulgaris	6.4	7	Ranunculus peltatus var diffusus	6.68	18
Hottonia palustris	7.33	7	Ranunculus penicillatus	6.49	18
Hydrocharis morsus-ranae	8.26	8	Ranunculus penicillatus var penicillatus	5.78	18
Hydrodictyon reticulatum	9.11		Ranunculus (sub sect. Batrachian) sp.	6.84	18
Hypericum elodes	4.95	11	Riccia fluitans	6.63	1
Isoetes sp.	3.21	4	Riccia sp.	6.53	1
Isoetes echinospora	4.06	4	Ricciocarpus natans	6.3	1
Isoetes lacustris	3.09	4	Ruppia cirrhosa	8.13	15
Juncus bulbosus	3.72	4	Ruppia maritima	10	15
Lagarosiphon major	7.42	5	Ruppia sp.	9.57	15
Lemna gibba	9.24	1	Sagittaria sagittifolia	7.88	12
Lemna minor	7.58	1	Sparganium angustifolium	3.65	13
Lemna minuta	8.64	1	Sparganium emersum	6.59	13
Lemna trisulca	7.82	1	Sparganium natans	4.84	13
Leptodyction riparium	8.44	3	Sphagnum (aquatic indet.)	3.37	3
Limosella aquatica	6.49	11	Spirodela polyrhiza	8.79	1
Littorella uniflora	4.7	4	Stratiotes aloides	8.51	8
Lobelia dortmanna	2.46	4	Subularia aquatica	2.93	4
Ludwigia palustris	5.57	11	Tolypella glomerata	7.18	2
Luronium natans	5.13	13	Utricularia sp(p)	3.58	9
Lycopodiella inundata	3.01		Utricularia australis	4.75	9
Lythrum portula	5.56	11	Utricularia intermedia sens.lat.	2.19	9
Menyanthes trifoliata	4.76	10	Utricularia minor	2.97	9
Myriophyllum alterniflorum	4.54	7	Utricularia ochroleuca	1	9
Myriophyllum aquaticum	4.64	7	Utricularia stygia	2.06	9
Myriophyllum spicatum	7.84	7	Utricularia vulgaris sens.lat.	5.28	9
Myriophyllum verticillatum	8.67	7	Utricularia vulgaris sens.str.	5.13	9
Najas flexilis	5.39	14	Zannichellia palustris	8.49	15

4.2.3 Validating a compositional metric sensitive to nutrient enrichment

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 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 suggest that the metric was independent of the pressure and therefore of little utility, a perfect or near perfect correlation would mean that the biology was reflected closely by a readily measured environmental variable and was thus redundant.

Figure 4.2 describes the relationship between annual mean TP and LMNI, where LMNI is the average of the recalibrated expert-based metric scores described above. Prior to this analysis, annual mean TP was found to be a superior predictor of the vegetation metric than summer TP (r^2 = 0.5 and 0.45 respectively). Where only summer TP was available, annual mean TP was modelled from the correlation between summer and annual mean TP from sites where monthly or bimonthly sampling was carried out (n=834, r^2 = 0.96).

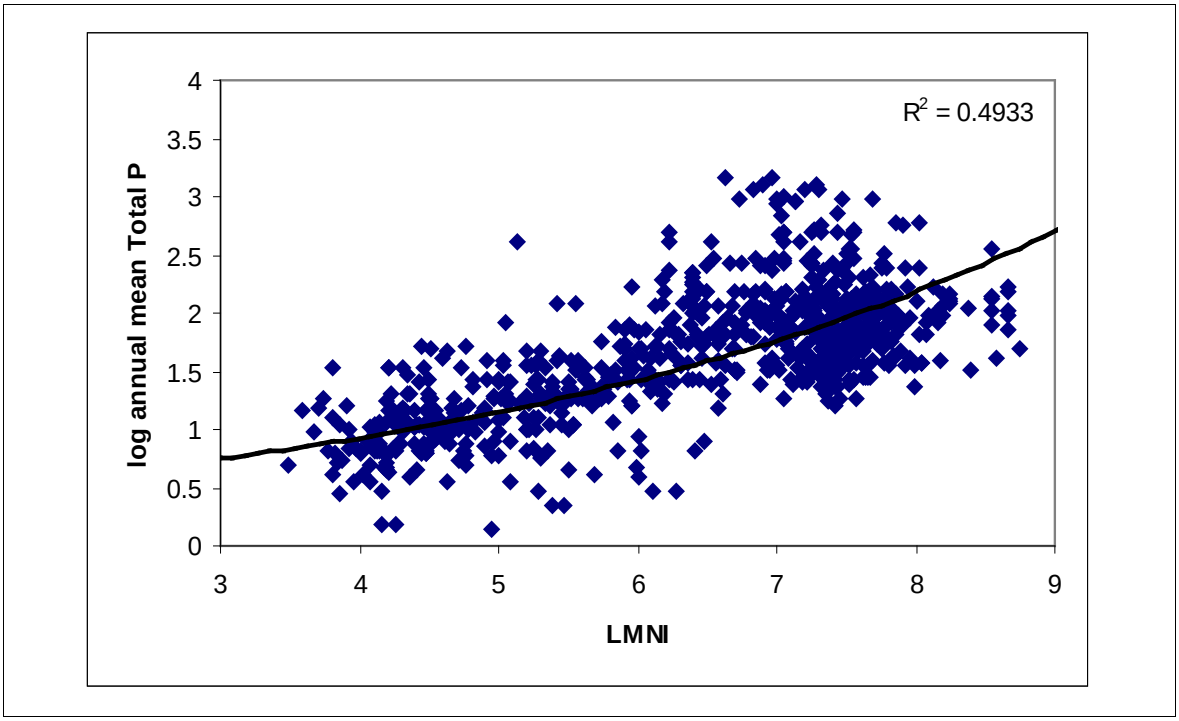


Figure 4.2 Global relationship between LMNI and lake annual mean TP where LMNI is determined by the average rank of the species present

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. Since the LMNI metric is calculated using presence-absence data it is potentially sensitive to the number of contributing taxa. When the number of taxa at a site was used to filter surveys used to derive the LMNI versus TP relationship it was found, by sequentially changing the filter, that the relationship was optimised when sites with five or fewer taxa were excluded (Figure 4.3). When the number of species recorded is small, it is

likely that a site is heavily impacted, or small, or the survey was incomplete or carried out under suboptimal conditions.

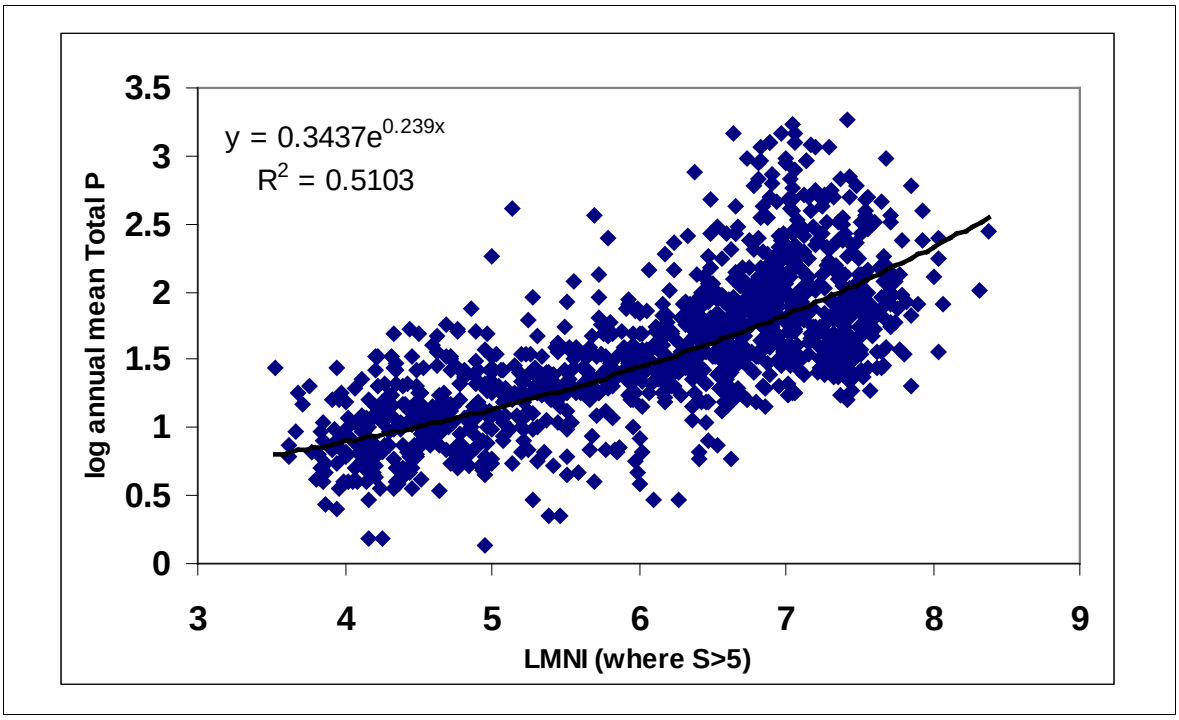


Figure 4.3 LMNI versus TP relationship after removing sites with five or fewer taxa. Subsequent screening of sites by number of taxa produced no further improvement in the relationship.

An analogous approach is to compare the ranks of individual species to their TP optima. Figure 4.4 provides a rigorous test of this relationship, taking the UK species ranks and comparing them with the TP optima of the same species in an independent dataset from five European member states constructed as part of the Northern GIG intercalibration process. Although there are a small number of outliers, it is clear from this analysis that LMNI scores strongly reflect TP optima, with scores over eight typically associated with species in lakes with TP above 50 ug/l compared to species with scores under three associated with lakes where TP is around 10 ug/l.

To give a measure of the effect of recalibrating expert scores, the relationship between TRS and TP was compared with that between LMNI and TP for a subset of the dataset where all surveys were derived from a single source and had supporting nutrient data. In the Northern Ireland Lake Survey, the correlation between TRS and summer TP increased from $r^2 = 0.19$ (n=620) to $r^2 = 0.33$ when LMNI was used instead of TRS.

A second stage in assessing the utility of the new site scores is to include this as a supplementary variable in an ordination of lake macrophyte data that is constrained by available environmental data. An example of this is shown below (Figure 4.5) using a CCA ordination of macrophyte composition for 1,250 European lakes in the lake dataset compiled for the recent EU REBECCA (Relationships Between Ecological and Chemical Status of Surface Waters) project (data supplied and managed by Bernard Dudley, CEH). The environmental variables retained are the minimum necessary to explain variation in the species data. The Lake Macrophyte Nutrient Index (LMNI) has been supplied as a supplementary variable to illustrate the strength of correlation with chlorophyll and phosphorus and its general utility as an aggregative measure of enrichment pressure. Since this is an entirely independent dataset, it represents a particularly good test of the utility of LMNI.

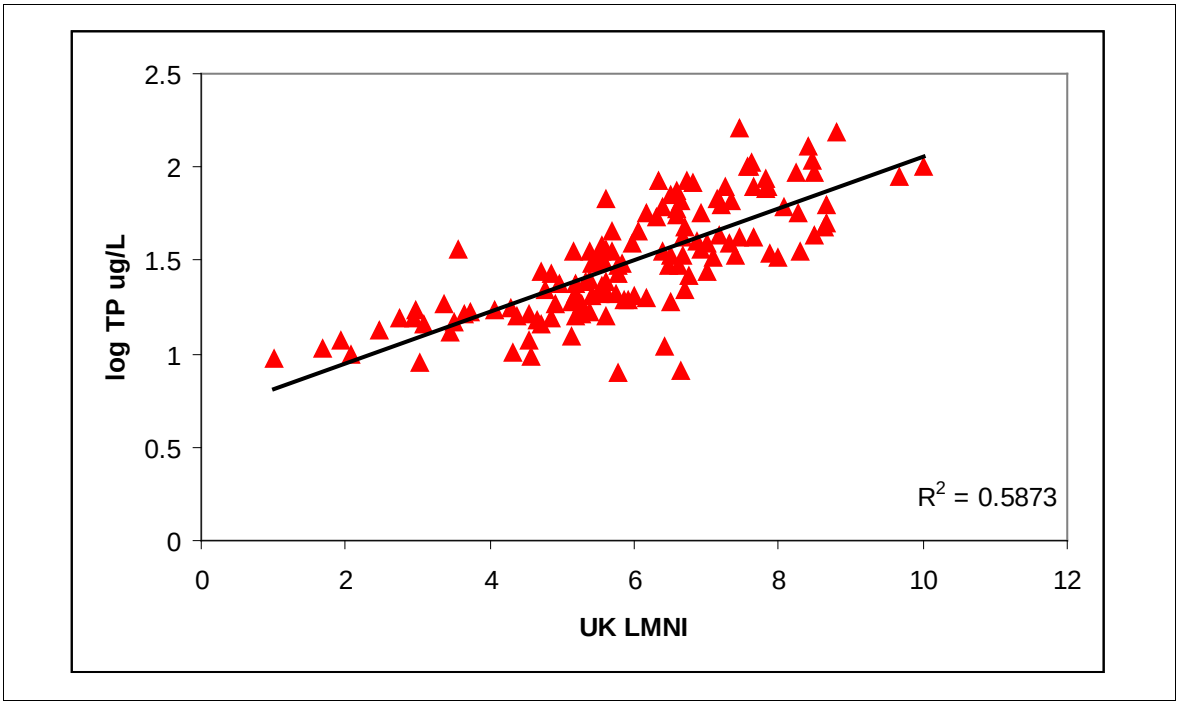


Figure 4.4 Correlation between species LMNI scores and their average TP optima as provided by EU Member States participating in the Northern GIG

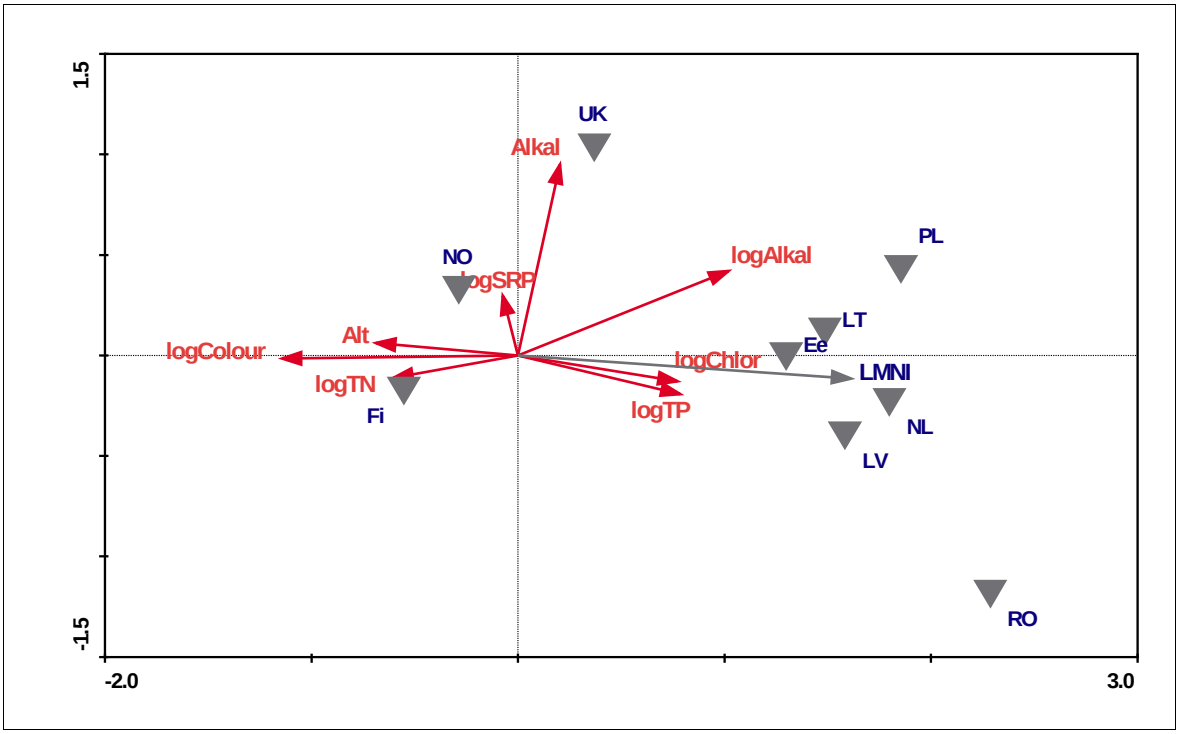


Figure 4.5 CCA analysis of lake macrophyte data (REBECCA European dataset). LMNI is fitted as a supplementary variable. Grey triangles represent centroids of data supplied by different countries.

4.2.4 Calculating a compositional metric sensitive to nutrient enrichment

The simplest way to calculate a compositional metric is to average the ranks or scores of the species present. This is likely to be most suitable when assemblages are species rich or species abundances are relatively evenly distributed, and is the approach employed in the above validation. Weighting scores (i) by the relative abundance of different species and/or (ii) by the indicator strength or tolerance of different species scores introduces increasing degrees of sophistication.

Since surveys undertaken to the standard protocol will provide a measure of abundance, and most archived surveys have been undertaken to at least a DAFOR recording level, it is possible to assess whether weighting by abundance strengthens the pressure-metric relationship. In an assessment of the global dataset it was found that weighting by cover scores (standardised to a range of one to five) had a negligible impact on the strength of the LMNI versus TP relationship, increasing it by under one per cent. Given that the global dataset aggregates surveys from various sources, it was considered preferable to double check the influence of cover weighting on the LNMNI versus TP relationship by using data obtained from a single source. This was done using the Northern Ireland Lake Survey since this offers one of the largest datasets in which biology and chemistry are linked across a wide range of lake types and surveys were undertaken by a small number of individuals. In this instance TP values are based simply on summer spot samples.

Species tolerance was estimated using CCA taking the standard deviation of species scores on the first axis as a measure of tolerance, which is effectively a measure of niche breadth in unimodal methods (ter Braak and Smilauer, 2002). To reflect the fact that species with a low standard deviation have higher indicator potential, the reciprocal of the reported standard deviation was used as a measure of indicator potential. When site scores were calculated as the average of the LMNI score of each species present weighted by the cover and indicator value of that species, the global LMNI versus TP relationship again improved by less than one per cent (Figure 4.6).

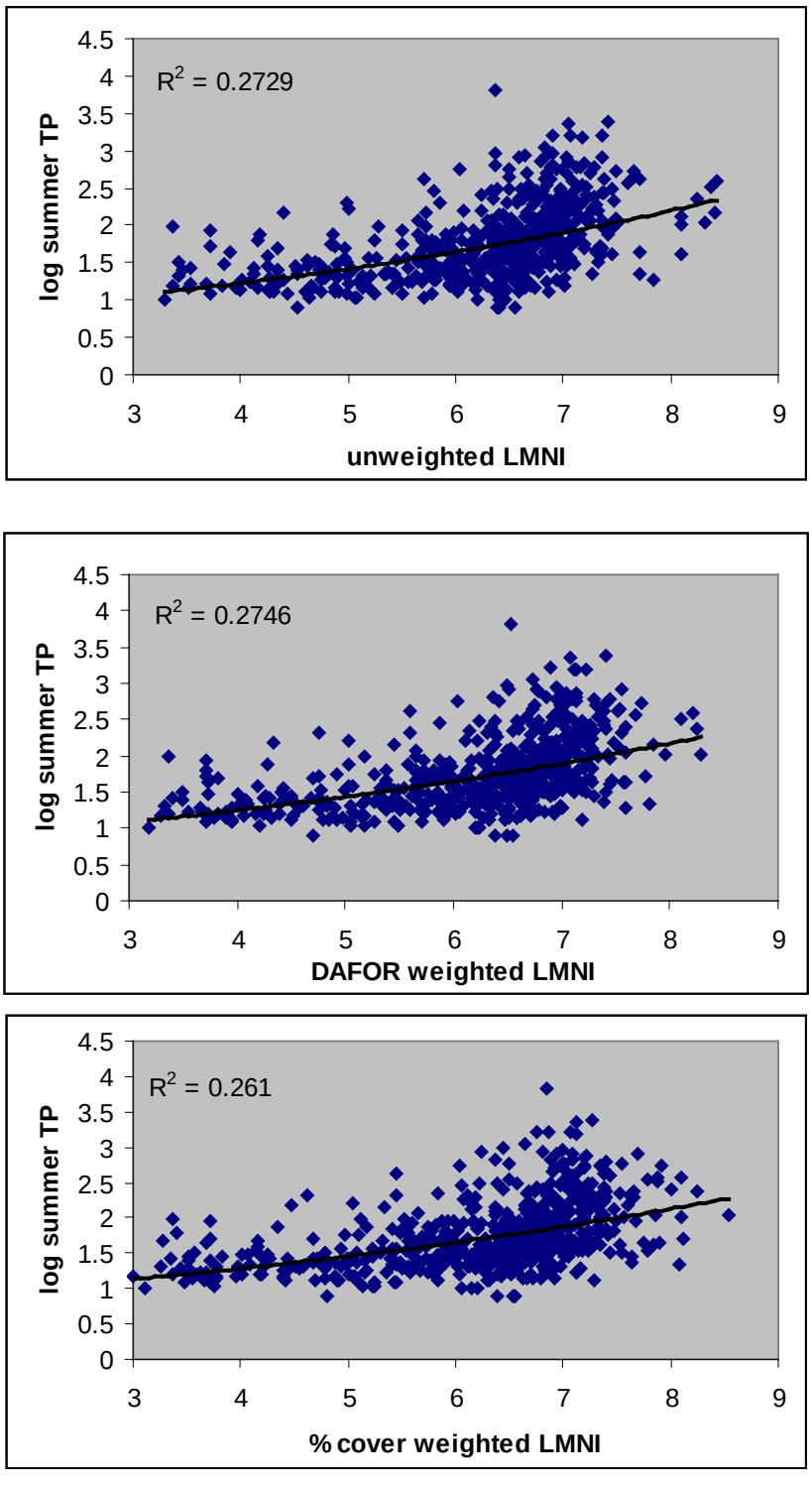


Figure 4.6 Comparison of relationship between LMNI metric and summer TP in a survey of Northern Irish lakes (n=620) when LMNI is calculated by different weighting options

These results suggest that the methods used to calculate LMNI have a conservative effect on the LMNI versus TP relationship. Consequently there is little to be gained by using alternatives to simple presence or absence for calculating the metric. Initially this

appears somewhat surprising, since intuitively, abundant species or species with a high indicator potential would be expected to merit greater weight and to improve the pressure-metric relationship. A number of possible explanations are offered below.

i. LMNI ranks for each species are initially derived from a presence-absence dataset (this is inevitable given the type and coverage of data and inclusion of surveys based on a composite of historical records) so are relatively insensitive when other weightings are applied retrospectively.

ii. Cover is difficult to assess in the field, leading to high variability between observers.

iii. Cover of individual species shows high interannual variability at individual sites and is somewhat uncoupled from nutrient conditions during the year of monitoring.

iv. Distribution of cover among individual species is primarily a function of physical habitat availability at a site rather than a product of nutrient regime. Loss of sensitivity associated with the decline, but not loss, of a sensitive species (which would have no effect on the site score) is offset by the effect of the first appearance of a tolerant species, even at low cover (which will affect the site score).

v. The LMNI site score is a relatively conservative metric insensitive to outliers. For example, if a site has an LMNI score of six and supports 10 species, the addition of another species with a score of three or nine will reduce or increase the site score by under 0.3 units. Given that species with low LMNI scores tend to be less abundant, while species with very high LMNI scores only occur abundantly in association with other high scoring taxa, the influence of weighting by cover will be even less pronounced.

vi. The influence of indicator potential is small because the range of variability in species tolerance is surprisingly low (interquartile range of indicator potential = 1.02). It is also difficult to distinguish species that have a high indicator potential because they have a genuinely narrow niche breadth from species that appear to have a narrow niche breadth simply as a consequence of under sampling. Furthermore, the effect of combining abundance and indicator potential tends to be neutral because species with high indicator potential tend to be narrowly distributed and rare and therefore carry relatively low overall weight, while widely distributed and locally more abundant species associated with lower indicator potential carry more weight. A further feature is that the global average richness of hydrophyte taxa per site of 9.4 will tend to ensure that a mix of values of indicator potential are represented at all sites, diminishing the influence of species with high indicator values.

4.2.5 Correlation and causation

Validation of metrics is based on correlation, not a demonstration of causation. The following points should be borne in mind when interpreting raw metric values. The issue of covariation is revisited in Section 7.9 in deriving supporting standards for nutrients in lakes.

i. Annual mean water column TP is used simply as an indicator of eutrophication stress because it is the most widely measured useful determinand. Macrophytes may respond more directly to other correlates of

water column TP such as sediment nutrient supply, nutrient loading rate, land use, siltation, sediment oxygen supply, characteristics of the nutrient supply regime (other than annual mean), particular nutrient fractions, or sediment as opposed to waterborne concentrations.

ii. Nutrient chemistry is highly covariable with major ion chemistry, especially calcium, since most naturally-derived phosphorus in freshwaters originates from rock weathering. Identifying anthropogenic impacts in spatial datasets therefore depends on identifying an effect of nutrients after covariation with natural factors has been removed. This requires sufficient variation in nutrient concentrations for a given alkalinity, which requires large datasets and matching nutrient data. Distinguishing between the influence of covariable factors is problematic. While it is fair to state that a site with an LMNI of eight is more fertile than a site with an LMNI of four, it is impossible to state categorically that they are different *because* one is more fertile than the other. It is possible to show a unique significant effect of TP on LMNI or TP optima, after removal of the covariation between TP and alkalinity. However, this effect is small (three to six per cent of metric variation explained) compared to the shared effect of TP (about 50 per cent of metric variation explained), due to the scale of intercorrelation between TP and alkalinity. One practical concern is that the use of agricultural fertilisers will also increase mean alkalinity. Separating alkalinity and nutrient effects may therefore be somewhat artificial, while use of alkalinity as a predictive variable may inadvertently allow a higher level of degradation.

iii. It is impossible to take metric values and state on the basis of these that one site is more *impacted* than another since the dependency on intrinsic factors, notably alkalinity, must first be removed.

4.2.6 Comparative value of expert and empirical metrics

Two 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 TP concentration at that site. The first approach uses the adjusted expert-based metric described above (LMNI), calculating the average species rank for each site for which there is matching TP data. The second approach follows an empirical route, using Canonical Correspondence Analysis with annual mean TP as the dependent variable. The axis 1 site scores from this analysis represent the average of the species scores where these are constrained by TP. Figure 4.7 illustrates the correlation between these site scores and the TP data from the same sites. There is an encouragingly strong correlation which is slightly superior to that obtained using the adjusted expert score approach. However, in this case the site scores are derived directly from the species TP optima. Consequently, the relationship shown is only an internal validation since the correlation is with the data on which the model was trained. The expert scores have not been previously trained on directly measured TP data and the relationship shown in Figure 4.3 is therefore an external validation.

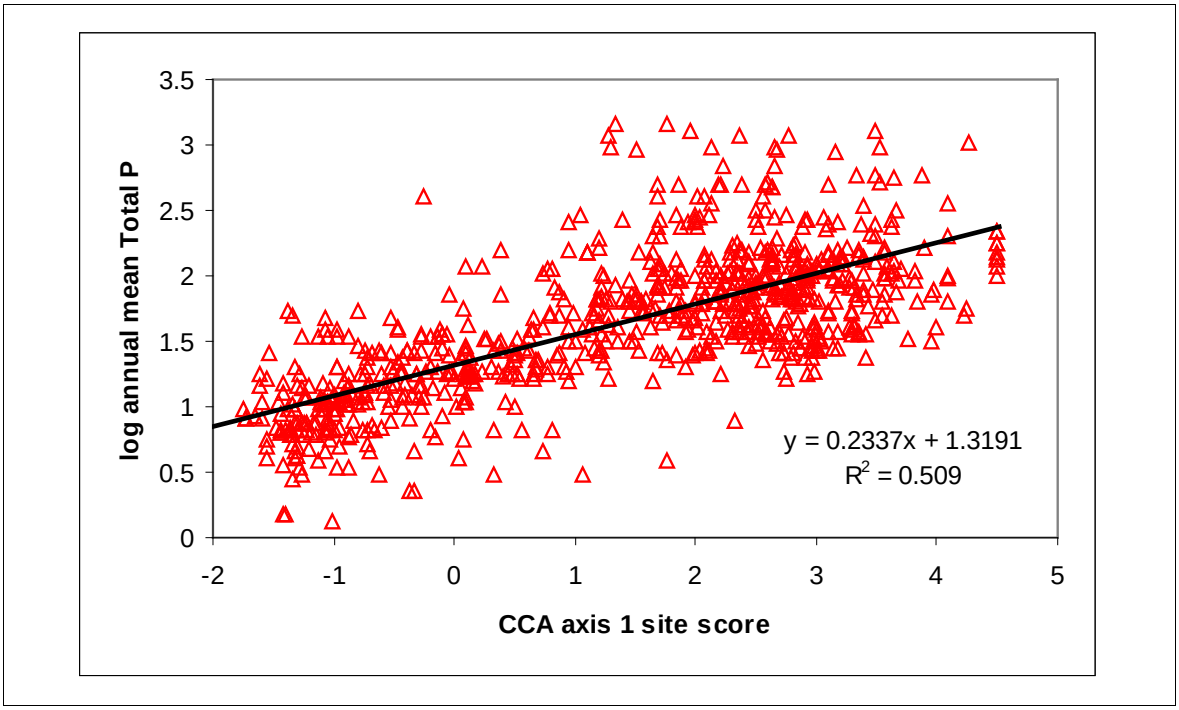


Figure 4.7 Correlation between axis 1 site scores of a CCA constrained by annual mean TP and TP data from the same sites. This represents an internal validation and should be compared with Figure 4.3.

4.2.7 Difficulties in the use of compositional metrics

Compositional metrics focused on the assessment of saprobity have existed since the early 1900s and have been used as the basis for biomonitoring using macroinvertebrates and diatoms for many decades (Chandler, 1970; Round, 1991). The idea of defining different aspects of the niche of individual species by discrete values and combining this information to represent the biology at a site through a single number is appealing in its logic and simplicity. However, there are a number of less widely discussed practical and statistical problems associated with compositional metrics. These problems are generic rather than being unique to a nutrient enrichment metric and are discussed below.

1. Sensitivity of site metric values to the number of contributing species

In dividing the sum of metric scores by the number of scoring species the dependency of the metric on richness is reduced but not removed. This is because the more species that are present, the more species from higher ranks must be incorporated to achieve that richness. Thus, if species are arranged in ascending order of LMNI scores the minimum possible LMNI score for any site with a single species is one but with three species is (1+1.67+1.93)/3 = 1.53. This effect is accentuated when a continuous scaling of species ranks is employed and when site ranking scores are not evenly distributed because some increases in richness will result in disproportionately large increases in LMNI. Addition of species to highly eutrophic sites (LMNI above eight) will on average reduce scores, whereas the addition of species to oligotrophic sites (LMNI below four) will on average increase scores. In both cases, with a sufficiently large species richness, scores would converge at the global mean rank of 6.3. Figure 4.8 illustrates the dependency of LMNI in a set of putative high status sites on the number of taxa and how the theoretical minimum LMNI increases with the number of taxa. In

general, owing to the distribution of species scores, LMNI is not heavily dependent on the number of taxa. This is in contrast to metrics for other quality elements such as macroinvertebrate ASPT which increases strongly with increasing taxa due to the numerical dominance of high BMWP scores. Factors such as lake area or altitude, that are likely to influence species richness, will therefore indirectly affect metric scores and should preferably be incorporated within models to predict compositional metric scores at reference condition. Similarly, caution should be used in screening sites by the metric value itself, unless factors such as lake area are taken into account.

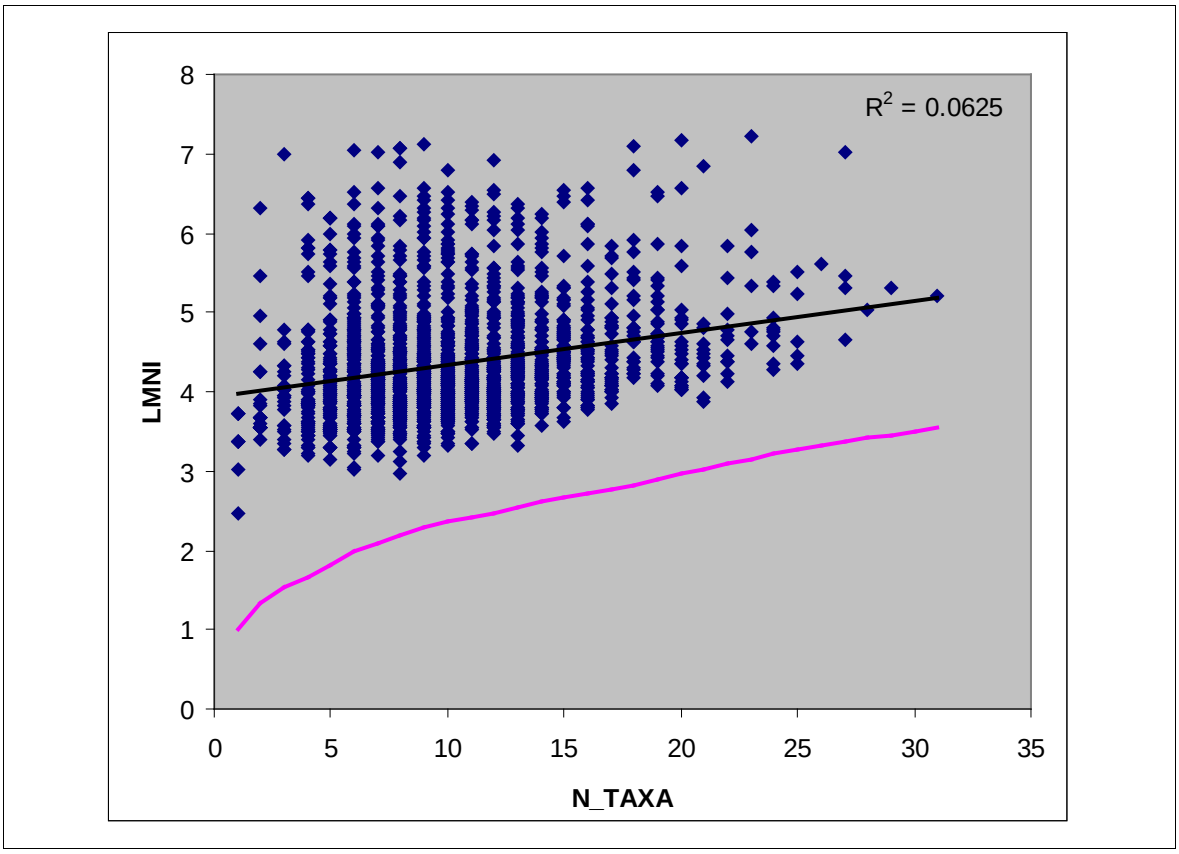


Figure 4.8 Relationship between LMNI and N_TAXA in a set of putative high status lakes. The pink line illustrates the theoretical minimum possible LMNI score for a given number of taxa.

2. Ability to derive site metric values from poor quality biological data

It is possible to derive a metric score for a site based on trivial quantities of a single species. Although this is an extreme scenario, it shows that the quality of compositional data from which a metric is derived may be poor or constrained by other factors, yet these constraints are hidden by the metric's face value. If composition alone was used as the basis for assessment, a sparsely vegetated, species-impoverished lake might be classified as good or even high status when the richness and amount of vegetation would be incompatible with the maintenance of macrophyte-dependent functions, as well as being at odds with the normative definitions.

There is also a statistical risk associated with the use of metric values based on a very small number of species. Hence, when the number of species is three or fewer there is a greater than five per cent probability that a score as low as that observed could have arisen purely by randomly sampling the global species pool. Once the number of species exceeds three, this risk becomes negligible (under 0.1 per cent).

3. Issues over transferability of specific metric scores

Developing pan-European metric systems has often been debated in terms of assisting intercalibration of national methods. Among the objections raised is the difficulty of applying scores from one country to another, where some species may have different ecological affinities due to the composition of the national species pool or position of that country geographically with respect to the overall range of a species. This is a relatively minor problem when the number of taxa in a sample is large, as is likely to be the case for invertebrates or diatoms. However, when the number of taxa is quite small (five to 10), as may be the case for macrophytes, one species with an unrepresentative score may have a relatively large influence.

The same problem arises when widely distributed species are expected to score differently in different parts of their range within a country. The approach used to develop index values for each species takes data from individual sites across the entire UK lake resource. Consequently, it is inevitable that, for example, the associations of species in low-alkalinity lakes in north and west Britain will have the strongest influence on the scores of widely distributed taxa since these lakes will provide the greatest number of records for those species. This means that the scores for individual species should be applicable to the great majority of lakes but may apply less well in a lake type with fewer members if the widely distributed species occur across several different associations. Consequently, a number of widely distributed species such as *Nymphaea alba*, *Nuphar lutea* and *Potamogeton perfoliatus* have relatively low LMNI scores compared to other species found in high-alkalinity enriched lakes, because these species occur alongside a large number of low-moderate scoring species in low- and moderate-alkalinity lakes in north and west Britain, that are themselves absent from more enriched high-alkalinity lakes in the south and east. A simple solution to this problem would be to stratify the UK lake resource by lake type and weight each type equally when deriving species scores, but this may reduce the applicability of compositional metrics to the majority of lakes.

4. Differential sensitivity of metrics in relation to lake type

The anatomy of pressure-metric relationships is somewhat complex. The global pressure-metric relationship is a composite of data from different lake types arranged predictably, as illustrated in Figure 4.9. However, it is not obvious from this the extent to which the pressure-metric relationship is type-specific. Figure 4.10a illustrates the type-specific relationships between LMNI and TP. There are two points of note:

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

ii. Sensitivity to higher concentrations of phosphorus is weak in the most base-rich lake types. Thus, above 20-60 ug/l TP, LMNI is virtually independent of TP in high- and very high-alkalinity lakes, yet continues to increase with increasing TP in other lake types.

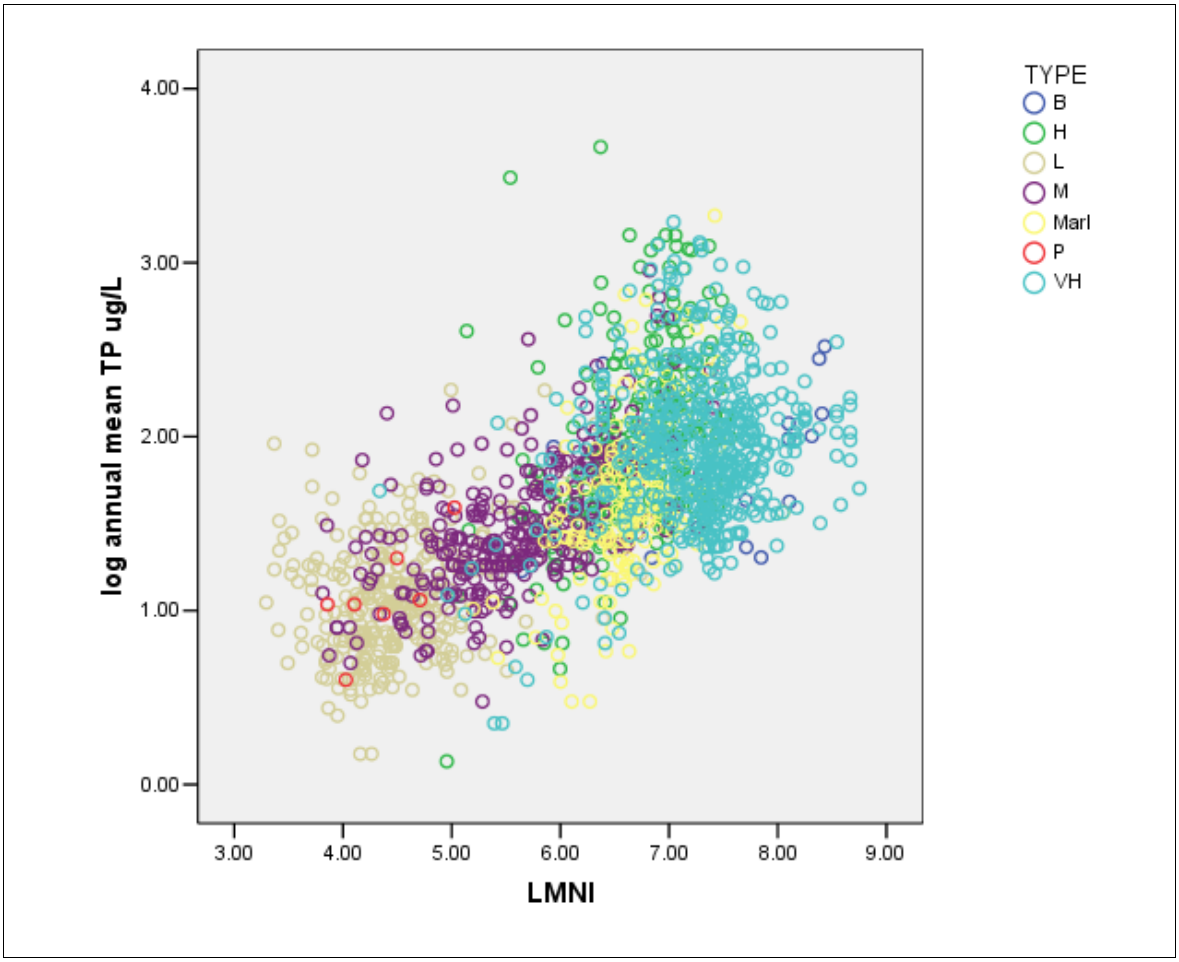


Figure 4.9 Change in LMNI versus TP relationship when stratified by lake type. Simplified lake typology (ignoring depth and continentality) based only on alkalinity, peat cover and conductivity, as defined in Table 3.1.

Figure 4.10b illustrates the type-specific relationships between the site scores based on species TP optima derived from CCA and TP. It is clear that the two properties described above are not simply an artefact of the expert score approach since they are reproduced when the TP 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 tend to characterise greater productivity. Consequently, for a given concentration of phosphorus the productivity of vegetation will increase with increasing alkalinity which will be reflected in higher LMNI scores. In this respect LMNI is more akin to a productivity index than a fertility index.

The second point is more difficult to explain and could have several contributory factors. The scores for all lake types tend to converge at around seven. It is probably an artefact of the statistical approach to deriving species scores that site scores will rarely exceed eight, since the majority of species scores lie in the range 4.5-7.5. 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 occur at extremes of fertility tend to be relatively rare when present. Given the underlying tendency for LMNI to increase strongly with increasing alkalinity at low P,

this means that there is little remaining ‘room for manoeuvre’ in the highest alkalinity types with further increases in P. The relationship between LMNI and TP would thus be expected to saturate at progressively lower TP with increasing alkalinity. The relationship for very high-alkalinity lakes may be slightly distorted by the lack of contemporary sites with TP below 20 ug/l. However, species turnover still occurs in higher alkalinity lakes at higher fertility, as captured by changing LMNI (LMNI 6.5-8) yet this turnover is apparently independent of water column TP. This may reflect, *inter alia*, greater influence of nitrate on composition when P is no longer limiting, sequestering of nutrients within plant tissue at high productivity, a shift towards competition for light as the dominant control on vegetation, or an increasing influence of top-down controls on macrophytes in the naturally most productive lakes. Even the most enriched lakes do not converge to a single community type; dominance by either *Lemna* species or *Ceratophyllum demersum*, or co-dominance by either *Nuphar lutea* and *Nymphaea alba*, or *Potamogeton pusillus*, *Zannichellia palustris* and *Potamogeton pectinatus* all appear from the survey data to be alternative endpoints. Such communities would be indistinguishable from LMNI scores alone, and perhaps reflect a combination of historical factors and hydraulic constraints on plant morphology.

4.2.8 Calculating the metrics

The LMNI metric is calculated as the sum of the LMNI scores of hydrophyte taxa present at a site, divided by the number of hydrophyte taxa recorded. The LMNI scores to be used are listed in Table 4.2. In the unlikely event of recording a taxon without an LMNI score, the site score is based only on the average LMNI value of the scoring taxa.

4.2.9 General use of compositional metrics

While there is substantial empirical support for the use of compositional metrics the above problems suggest 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 some 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.

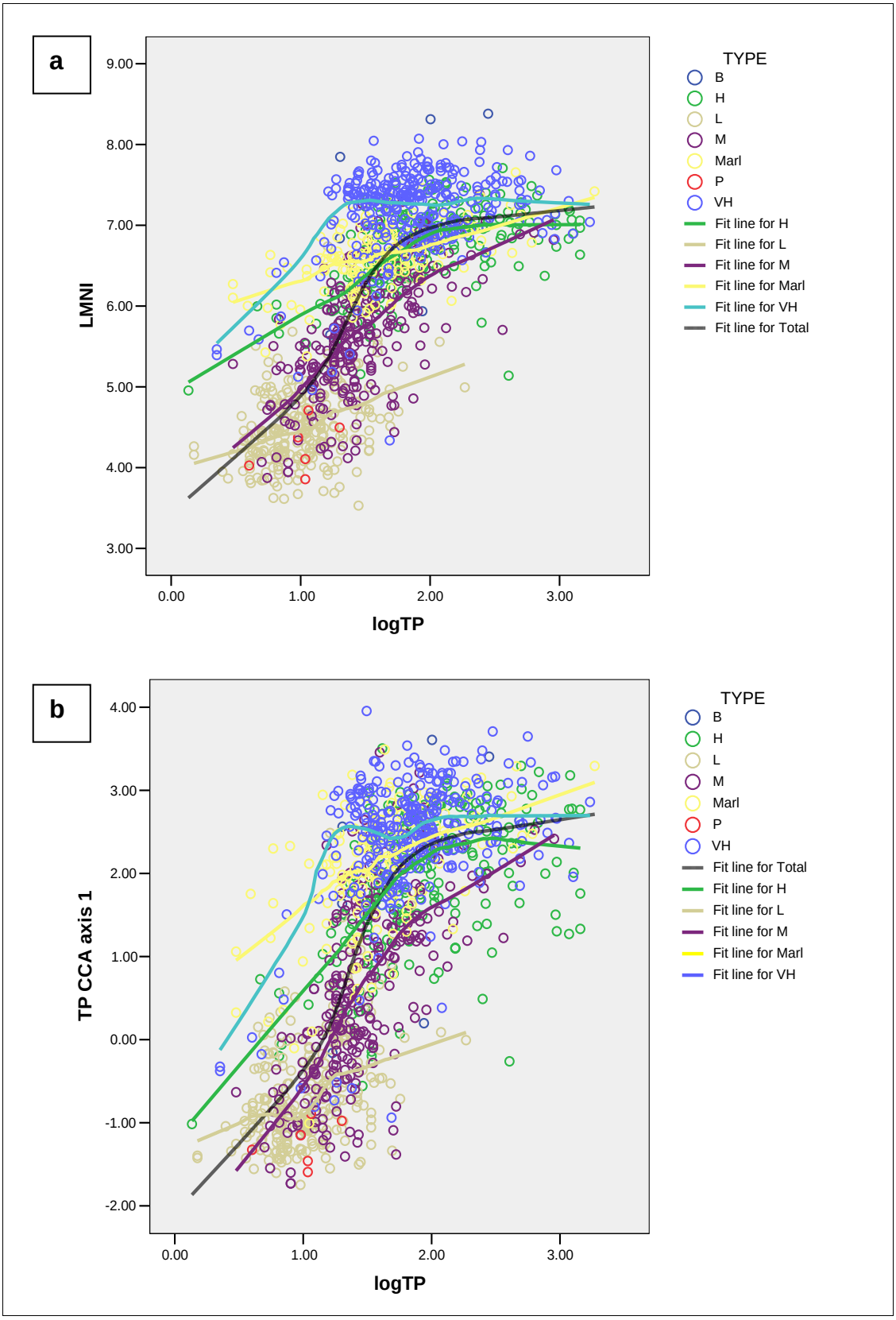


Figure 4.10 Comparisons of geology type-specific metric versus lake TP relationships using LMNI (a) or CCA derived phosphorus optima (b). Note increasing metric values with increasing alkalinity when TP is low, saturation of response at lower TP in higher alkalinity lakes and convergence to a common value at the highest TP in most lake types.

4.3 Richness metrics

4.3.1 Background

Richness is often viewed as an indicator of biological quality (Ricklefs and 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 require the use of diversity metrics to assess deviation from reference condition for macrophytes, indeed diversity is only mentioned explicitly in the case of benthic invertebrates (European Union, 2000). However, given that diversity will be 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 significantly altering its composition, while there is widespread evidence of the link between biodiversity and ecosystem functioning (Hooper *et al*., 2005). For example, interannual fluctuations in total cover in shallow high-alkalinity lakes are lower in sites that support larger number of species (Figure 4.11). Macrophyte cover is an important ecosystem attribute because it buffers interactions between higher trophic levels (Diehl and Kornijow, 1998). Consequentl,y high macrophyte richness is likely to contribute to ecosystem stability. Given the inevitable influence of richness on values of some other metrics, as discussed above (Section 4.2.7), there is also an underlying statistical justification for its inclusion. The inclusion of richness can also mitigate against over reliance on compositional metrics from species-poor sites.

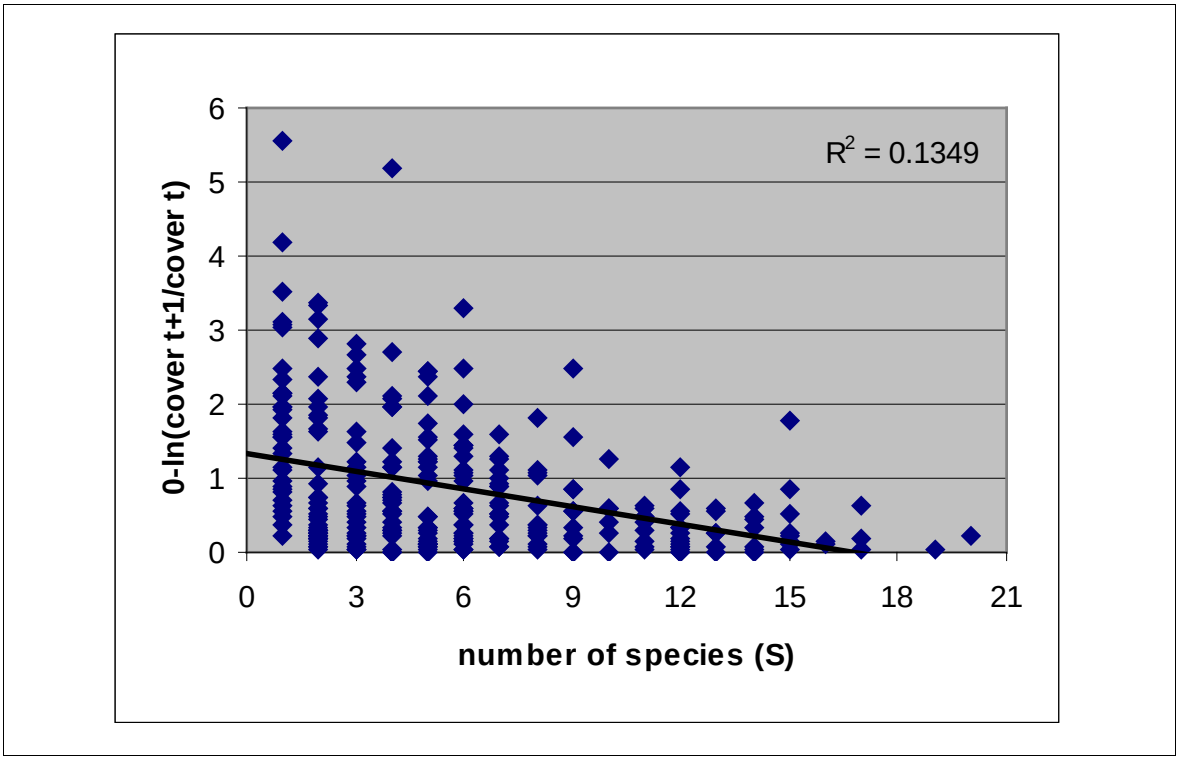


Figure 4.11 Interannual changes in total aquatic plant cover in 20 lakes in the Norfolk Broads in relation to plant species richness. The y axis measures the relative change in cover between successive years with large values indicating low stability of cover. All data collected by Broads Authority (Willby *et al*. in preparation).

The use of richness as a metric is less straightforward compared to other metrics. If richness is assumed to (partly) reflect the diversity of resources available, it can be argued that low richness, relative to that expected, could be indicative of low resource diversity. Thus, hydromorphological pressures such as some forms of water level regulation or shoreline modification that are liable to reduce physical habitat heterogeneity, are likely to be associated with lower than expected richness. On the other hand, in fertile sites physical disturbance associated with water level fluctuations may prove sufficient to arrest the effects of competitive exclusion by larger canopy-forming species sensitive to physical disturbance, which would otherwise dominate. In the absence of a suitable database to cover such pressures, these relationships are assumed rather than proven. Links to the Lake Habitat Survey database at a later date is therefore imperative to validate richness metrics as measures of hydromorphological impact.

4.3.2 Approach

There are well-recognised macroecological relationships between area or altitude and species richness (Gaston and Blackburn, 2000). These have been described for lake plants by Rorslett (1991) and Jones *et al*. (2003) respectively. Moderate-alkalinity or naturally mesotrophic lakes are also commonly regarded as species-rich relative to other lake types (see. Murphy, 2002, JNCC, 2005) and the database compiled in this project supports this assertion (Figure 4.12).

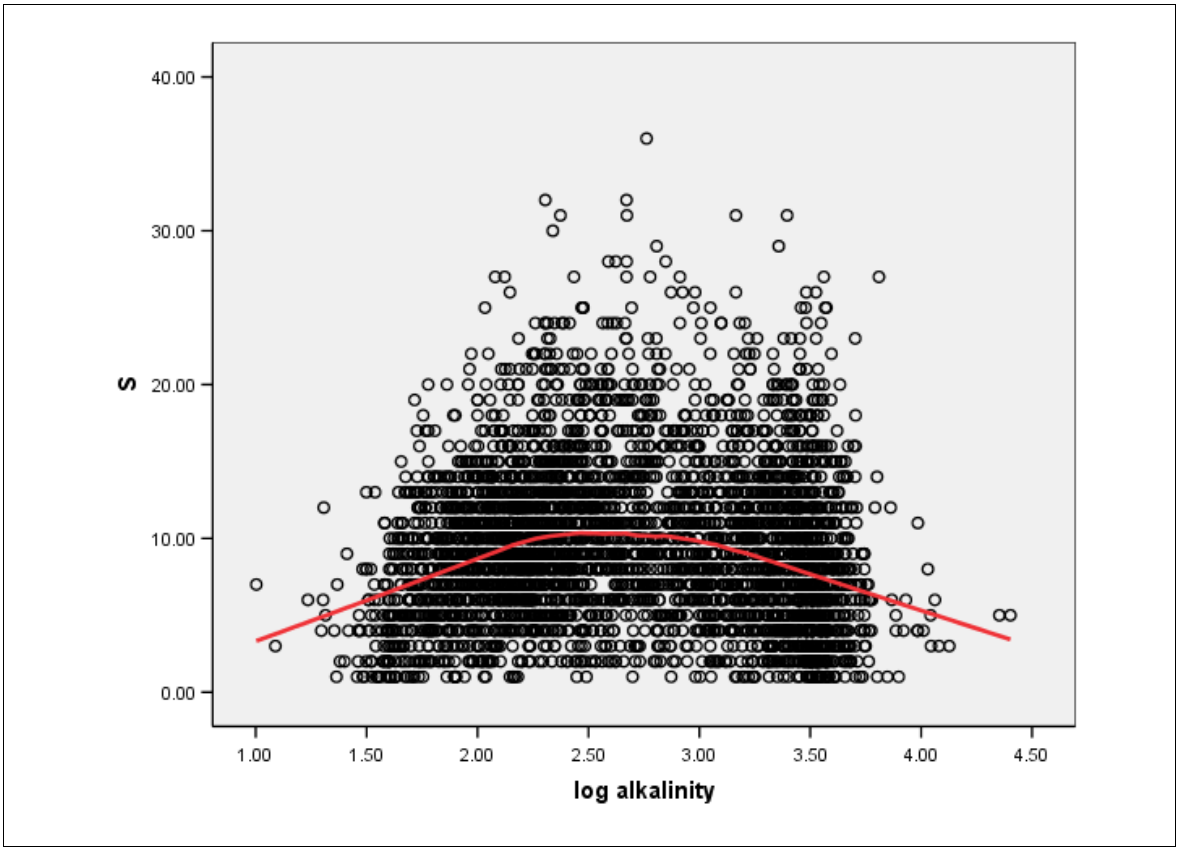


Figure 4.12 Global relationship between alkalinity and lake species richness fitted using logit regression with Gaussian link function. Sites with moderate alkalinity (0.5-1 meq/l) support on average twice as many as species as sites at the tails of the alkalinity range.

As well as being a function of variables such as area, altitude and alkalinity, taxa richness will vary at reference sites in relation to their degree of isolation, the richness of the regional species pool, survey effort, surveyor expertise and level of taxonomic resolution. Thus, exhaustively surveyed sites covered by highly experienced botanists with critical taxa all identified to species level will appear more taxa-rich than if the same sites were surveyed strategically or superficially by less experienced personnel, or with difficult groups identified only to genus level.

A metric based on functional group diversity (N_FG) was also developed. Functional groups were defined as described in Willby *et al*. (2000), based on a matrix of morphological and regenerative traits. Eighteen groups 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'). Macroalgae other than charophytes were excluded (Table 4.3). 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 and Segal, 1964; Hutchinson, 1975; Wiegleb, 1991). Our analysis does not explicitly include ecophysiological traits (such as bicarbonate usage) due to lack of adequate information. Consequently, the approach 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 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; for example, all lakes, more or less, 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). N_FG 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 and Li, 1997; Yachi and Loreau, 1999) This situation may be especially relevant to sites in which FGs such as charophytes or isoetids dominate, since the number of FGs present at a site may then be relatively small (three to five), while the numbers of taxa per FG may be large (four to six, for example). See Figure 4.13.

Table 4.3 Lake macrophyte functional groups

Code	Label[1]	Growth form and morphology	Stand structure and habit
1	Lemnids and ricielids	Very small, free-floating plants.	Monospecific, surface canopy, usually restricted to marginal growth except in small sheltered water bodies.
2	Charophytes	Small-medium, predominantly submerged perennials or annuals with simple branched structure of capillary leaves, high reproductive output.	Associate in deep to moderately deep water. Forms large low-growing pure stands in shallower water.
3.1	Mosses	Small-medium sized, submerged or amphibious perennials with trailing multiple leafy branches or short rosettes of small stiff evergreen laminar leaves.	Associate, usually in very shallow water or on hard intermittently inundated marginal substrates. Occasionally loosely anchored pure benthic mat in moderately deep water.
3.2	Leafy liverworts	Very small to small spreading attached plants with tiny, rounded, evergreen soft laminar leaves	Very small, dense submerged or emergent patches on hard substrate in shallow water.
4	Isoetids	Small-medium sized, submerged or amphibious, rosette-forming plants with stiff, tubular evergreen leaves.	Associate in moderate to deep water or as pure lawns of low-growing plants in very shallow water,
5	Elodeids and ceratophyllids	Medium-large, submerged, canopy-forming, multi-branched perennials with densely arranged small laminar or rigid dissected leaves. Mainly vegetative.	Moderate to large, dense single-species beds in moderately deep to shallow water, or as associates in deeper water.
6	Peplids	Small-medium, multiple-branched, submerged or amphibious plants with small linear leaves and floating rosettes.	Associates, or extensive low-growing open beds in shallow to very shallow water or on fine sediment exposed by fluctuating water levels.
7	Myriophyllids and herbids	Medium-large, submerged or partially emergent, rhizomatous perennials with flexible, dissected-leaved submerged foliage.	Mixed or pure submerged beds with open stand structure over a range of water depths. Herbids potentially emergent in very shallow water.
8	Hydrocharids and stratiotids	Medium-large, free-floating rosettes of mainly floating or aerial leaves linked by stolons.	Large, mainly pure floating stands in marginal or sheltered areas.
9	Utricularids	Small-medium, submerged, loosely or unanchored perennial with multiple branches and small, dense, flaccid, capillary-leaves. Conspicuous aerial flowers.	Small-medium, pure or mixed beds in undisturbed, shallow or very shallow water.

Code	Label[1]	Growth form and morphology	Stand structure and habit
10	Magno and parvonymphaeids	Large, mostly emergent or floating leaved, stand-forming, stoloniferous or rhizomatous perennials with large, insect-pollinated aerial flowers.	Medium to large, semi-floating beds with crowded canopy structure in shallow water marginal habitats.
11	Herbids and elodeids	Small-leaved, amphibious or submerged, annual or perennial, prostrate plants.	Small extensive low-growing open beds in very shallow water or on fine sediment exposed by fluctuating water levels.
12	Magnonymphaeids and sagittarids	Large to very large, unbranched, rhizomatous perennials with large expanded, submerged and floating leaves.	Medium to very large, pure or co-dominant beds with very open structure, usually in fairly sheltered, shallow to moderately deep water.
13	Vallisnerids and sagittarids	Medium-sized, perennial, basal rosette of submerged elongate leaves with expanded or strap-shaped floating and/or emergent foliage.	Associate or as small, open-structured pure stands with extensive surface cover in shallow water.
14	Parvopotamids	Small-medium, submerged, fine, linear leaved pondweeds with multiple branched foliage.	Associate or forming medium-large dense stands in sheltered shallow water.
15	Magno- and parvopotamids	Submerged, medium-sized, rhizomatous perennials with fine or tubular leaves.	Small-medium, dense stands in shallow water, sometimes with vigorous subsurface canopy.
16	Parvonymphaeids and magnopotamids	Medium-large, branched, submerged rhizomatous perennials with expanded medium submerged and floating leaves.	Medium-large, open stands in shallow to moderately deep water with dense apical growth in shallow water.
17	Magnopotamids	Medium-large, branched, submerged rhizomatoius perennials with expanded medium-large submerged laminar leaves.	Medium-large, open stands in moderate to deep water, with dense apical growth in shallow water.
18	Batrachids	Small-medium sized, branched, submerged plants with small capillary leaves and, potentially, small laminar floating leaves. Small insect pollinated flowers.	Open stands with apical canopy, usually in shallow water or on exposed fine sediment.

[1]After Hutchinson (1975) and Wiegleb (1991)

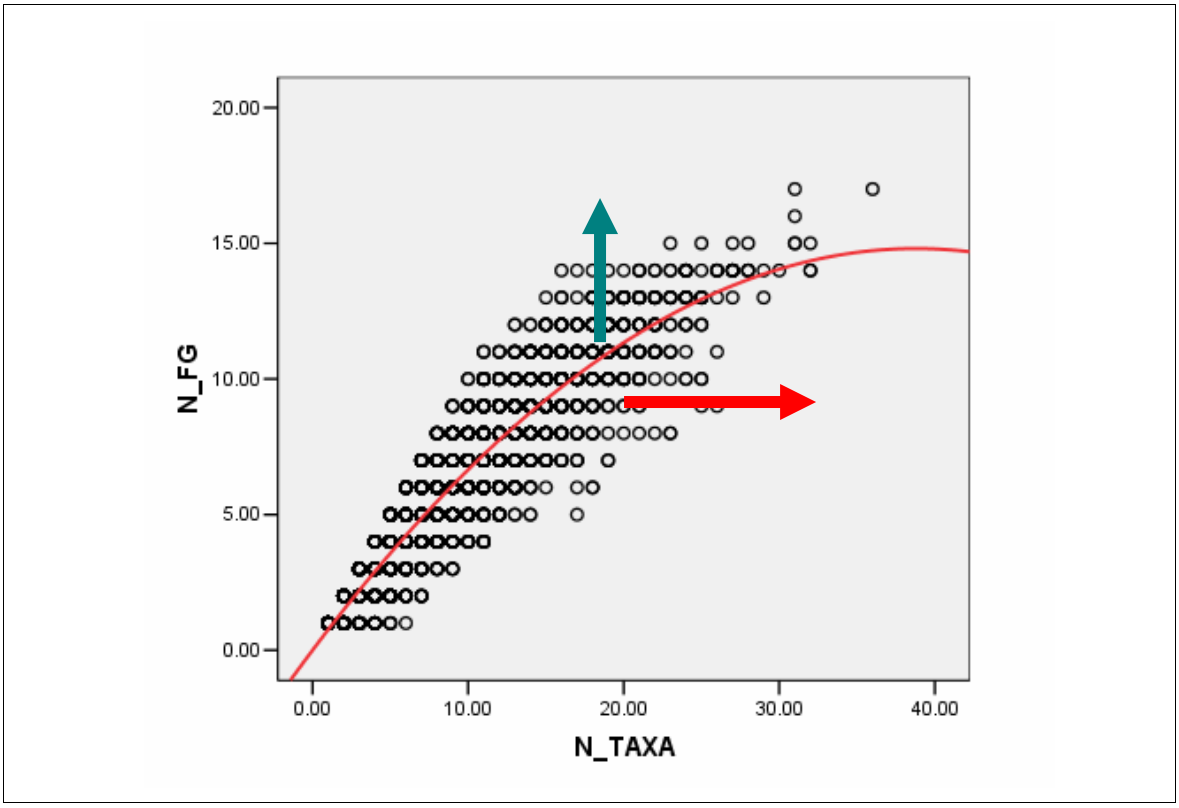


Figure 4.13 Relationship between observed N_TAXA and N_FG in UK lake dataset. Red arrow symbolises increasing functional redundancy (increasing number of taxa for given number of FGs), typical of sites where isoetids or charophytes dominate. High functional redundancy may contribute to ecosystem stability by insuring against functional group loss. Blue arrow symbolises increasing functional diversity (increasing number of functional groups for given number of taxa), reflecting greater physical habitat diversity or finer partitioning of resources in more fertile sites.

4.3.3 Validation

The global relationship between fertility and richness is generally one of negative response, although relationships are noisy (Figure 4.14). Thus, numbers of taxa decline with increasing TP up to values of around 100 µg/l (annual mean) and are thereafter stable. There is a weak negative relationship with nitrate (winter concentrations reveal the same pattern but substantially less data was available) although high taxa richness is clearly confined to sites where NO_3-N is below 2 mg/l. The relationship with summer chlorophyll is most striking, with numbers of macrophyte taxa relatively stable up to concentrations of 25 µg/l chlorophyll a and declining sharply thereafter. Collectively, these findings emphasise the importance of P rather than N as a controlling factor of lake macrophyte species richness. This contrasts with James *et al*. (2005) who found that nitrate (especially winter nitrate) was the major determinant of hydrophyte species richness in a set of mainly high-alkalinity shallow lakes in the UK and Poland, with richness declining significantly above NO_3-N of 1-2 mg/l.

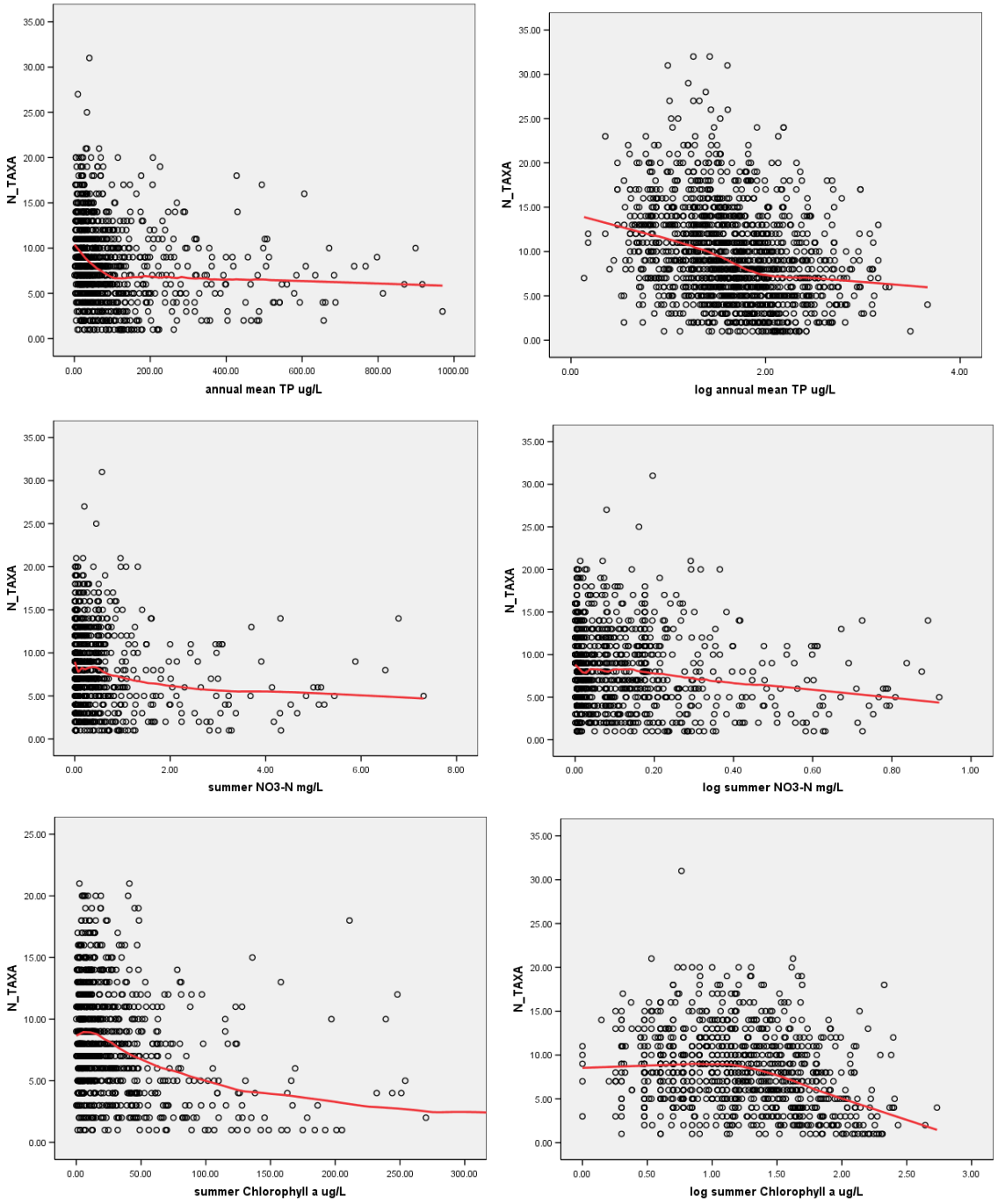


Figure 4.14 Global relationships between lake macrophyte species richness and several indicators of eutrophication. Lines fitted by LOWESS with Gaussian link function. Relationships based on untransformed data are shown on the left to illustrate the overall shape of the relationship.

In general, for a given level of impact on taxa composition from a specific pressure, a site which supports twice the expected number of taxa may be more stable and retain more elements of the original ecosystem function than a site which contains half the expected number of taxa. However, a more detailed exploration of the relationship between richness and fertility reveals a significant element of type-specificity. Thus, richness, whether as N_TAXA or N_FG, declines in all lake types with increasing TP, with this decline stabilising around 100 µg/l TP in the moderate and higher alkalinity lakes, depending on the lake type (Figure 4.15). However, the very high-alkalinity lakes display a much more acute response than other lake types to increases in TP in the range 10 to 100 µg/l. Relative to the global pattern, the decline in N_FG is particularly

marked in this lake type. This is an important finding because it extends the window of sensitivity of macrophytes in this lake type from that revealed by LMNI. In terms of chlorophyll (Figure 4.16) there is a weak response by both richness metrics up to values of 25 µg/l. Thereafter, a pronounced decline occurs in the richness of very high- and moderate-alkalinity lake types that does not flatten off at high levels of chlorophyll, as is the case with TP. In low-alkalinity lakes this decline starts at around 5 µg/l chlorophyll a. Curiously, the richness of high-alkalinity and marl lakes shows little signal in relation to chlorophyll. There was no evidence of type-specificity in the relationship between richness and nitrate.

The evidence from the above relationships is of a negative relationship between richness and fertility. However, some pressures such as low nutrient enrichment of a naturally oligotrophic lake may increase resource diversity and therefore increase richness relative to the baseline state. If this was indeed the case, careful consideration would be required for methods of scaling and integrating metrics in different lake types into an overall classification to avoid 'rewarding' a potential impact. This component of the richness-fertility relationship appears to be missing from the UK dataset, possibly due to the scarcity of sites with very low measured phosphorus concentrations. Use of an alternative dataset assembled to intercalibrate methods of member states in Northern GIG confirms that enhancement of richness (N_TAXA and N_FG) from very low to moderate fertility is a real phenomenon. Thus, Figure 4.17 indicates that for low- and moderate-alkalinity lakes richness increases to a peak at 10-25 µg/l TP. In this dataset the response of high-alkalinity lakes to TP may be diminished due to the shortage of higher fertility observations. Suitable approaches for integrating richness metrics into the final classification in the light of these analyses are considered in detail in Section 7.

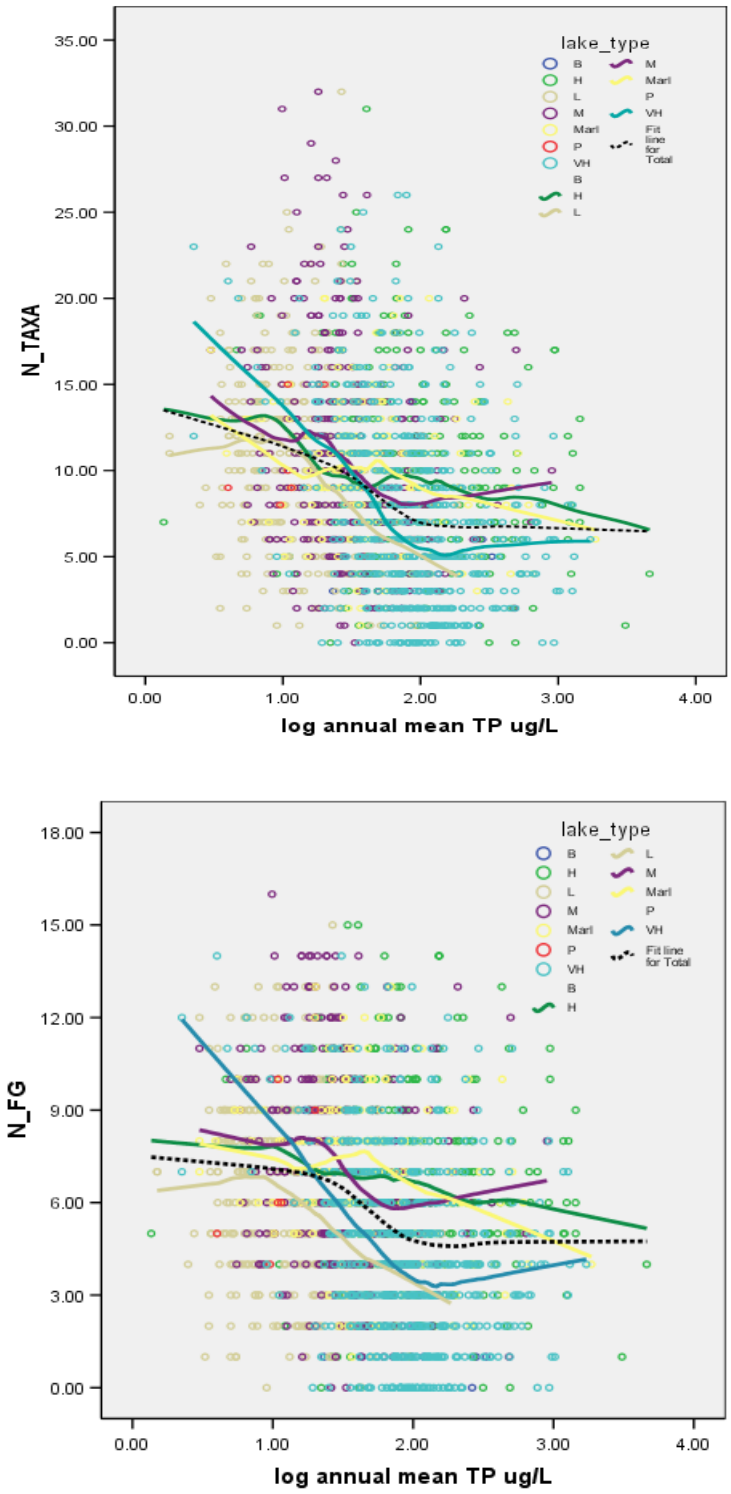


Figure 4.15 Type-specific relationships between richness metrics and fertility (as TP) in UK lakes. Note the relatively pronounced sensitivity of very high-alkalinity lakes and the saturation of responses at around 100 µg/l TP.

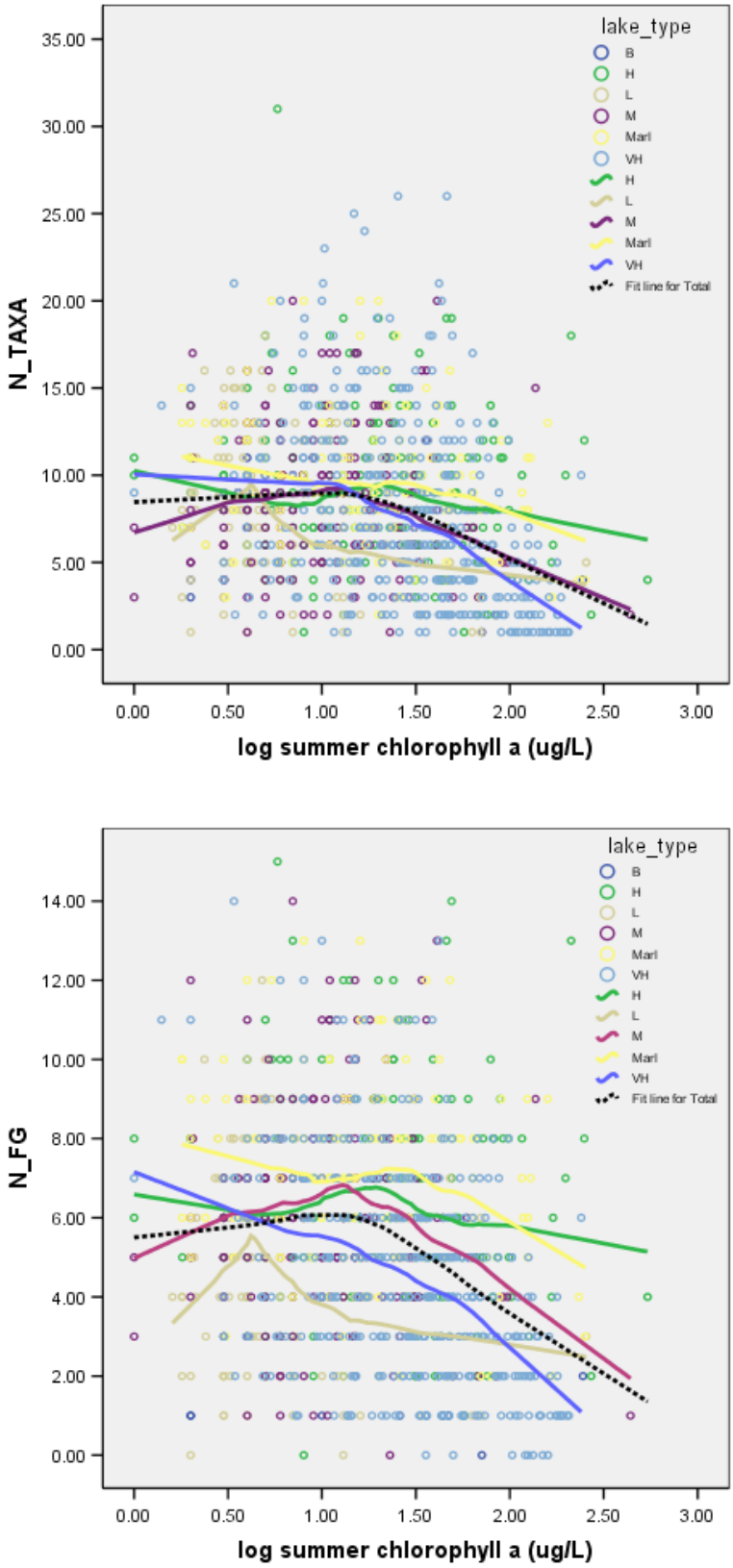


Figure 4.16 Type-specific relationships between richness metrics and fertility (as chlorophyll a) in UK lakes. Note the relatively pronounced sensitivity of very high-alkalinity and moderate lakes over 25 µg/l TP.

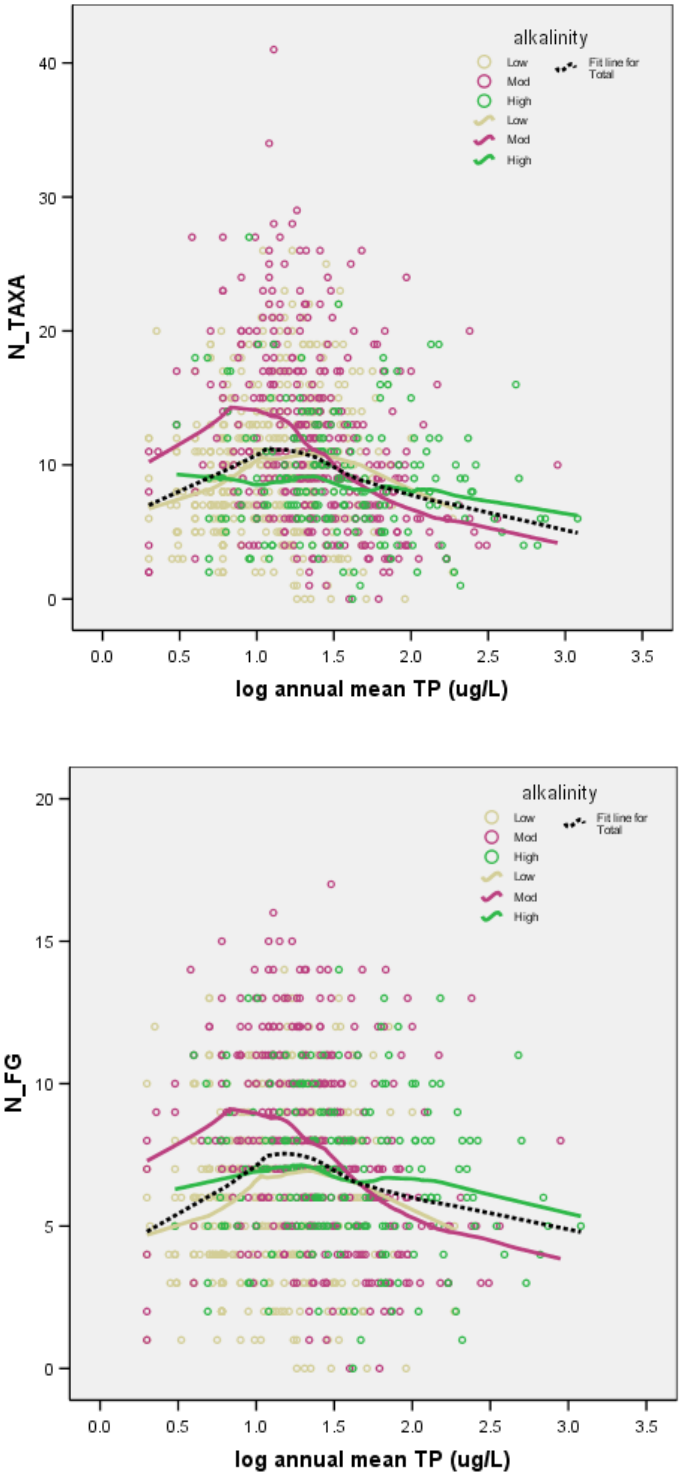


Figure 4.17 Type-specific relationships between richness metrics and fertility (as chlorophyll a) based on Northern GIG lake dataset. Note unimodal form of relationship in low- and moderate-alkalinity lake types that is not exhibited by UK data.

Richness indices may also prove useful in inferring other pressures, such as acidification of low-alkalinity lakes, which are not being specifically assessed by a dedicated metric. Acidification is likely to reduce macrophyte species diversity (Figure 4.18) and should be considered, even if indirectly, since it is most likely to affect sites where perceived pressures from nutrient enrichment are low. Figure 4.18 confirms the type-specific nature of pH versus N_TAXA relationship with peaty and low-alkalinity

lakes exhibiting marked reductions in number of taxa when pH drops from about six to four, while richness in higher alkalinity lakes seem largely insensitive to change in pH over the upper range.

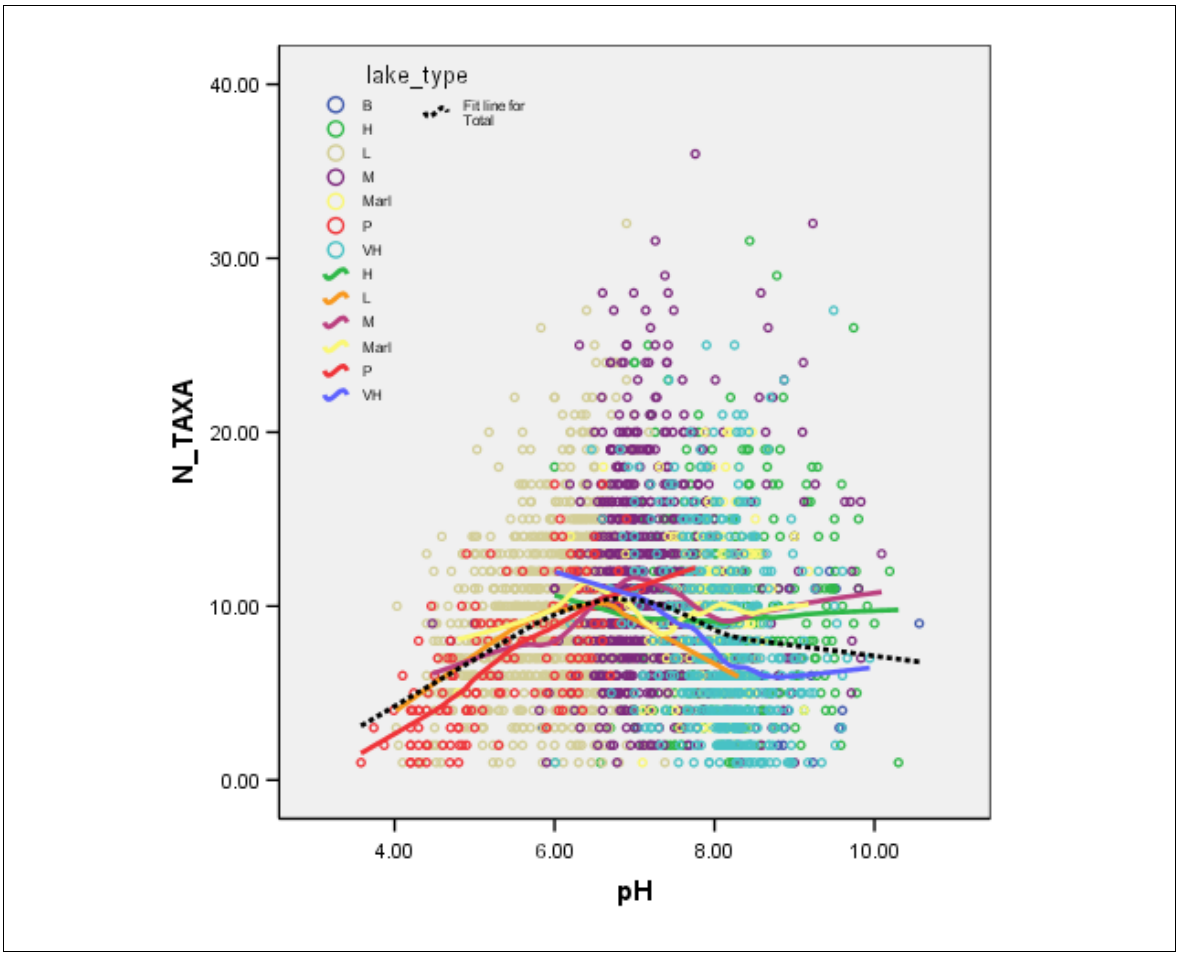


Figure 4.18 Global relationship between laboratory pH of summer samples and N_TAXA. Note the strong decline in observed N_TAXA at pH values below six in low-alkalinity and peaty lakes.

4.3.4 Comparative value of richness metrics

This analysis shows that richness is a valuable indicator of macrophyte response to enrichment in base-rich lakes and that it continues to respond at the higher levels of fertility where the LMNI metric response is largely saturated. The weight given to richness metrics in classification should be accordingly greater in the more productive lake types. To avoid giving undue reward to sites where increased richness is a direct response to an impact, careful consideration must be given to the scaling of richness metrics and how they are integrated into an overall site classification. The present evidence indicates that the response to nutrient enrichment in low and moderate alkalinity lakes is well served by the compositional metric, LMNI. Enhanced diversity of such lakes should attract proportionally less weight than in high-alkalinity lakes since there is a risk that it reflects modest levels of enrichment.

Richness will be an important metric to consider alongside other metrics and should be used to condition information on composition. Thus, for a given level of impact on taxa composition, a site which supports more than the expected number of taxa may be more stable, and retain more elements of the original ecosystem function, than a site

which contains half the expected number of taxa. The integration of richness metrics into the final classification is considered in detail in Section 8.

4.3.5 Calculating the metric

The metric N_TAXA is calculated as the number of hydrophyte taxa present at a site where the scoring species are as listed in Table 4.2. The metric N_FG is calculated as the number of different functional groups represented by the hydrophyte taxa recorded at a site where functional group membership is as listed in Table 4.2.

4.4 Abundance metrics

4.4.1 Background

The WFD explicitly encourages the consideration of abundance as part of the assessment of deviation from reference condition. Thus, at high status there are '*no detectable changes in the average macrophytic…abundance*' while at moderate status '*moderate changes in the average macrophytic…abundance are evident*'. More specifically at good status there are slight changes in the abundance of macrophytic taxa but '*such changes do not indicate any accelerated growth of…higher plant life resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water'*. The tone of these comments suggest that the primary concern is with nuisance growths that choke water bodies, and lead, *inter alia*, to deoxygenation and fish kills. Increased abundance is therefore considered as one of the undesirable disturbances that accompany eutrophication (ECOSTAT, 2005). However, this change in abundance is usually preceded or accompanied by a shift in composition to tolerant, dominant canopy-forming species, such as *Elodea* or *Ceratophyllum*, invasion by alien species and/or loss of species or functional group richness. Other metrics have been developed to cover these scenarios. Regardless of the statutory requirement to consider abundance in ecological classification, there are practical arguments to incorporate some measure of abundance to protect against classifications being based only on relative composition or richness metrics when absolute cover is unnaturally very low.

The abundance metric developed here was specifically designed to recognise *low* abundance. At very low abundance macrophytes in lakes will fail to maintain key dependent functions such as habitat support for higher trophic levels, bed stabilisation or nutrient cycling (Jeppsen *et al*., 1998). Low aquatic plant cover may occur naturally in dystrophic lakes, or where there is high wave exposure or a shortage of substrate suitable for rooting. It may also be a response to high levels of grazing by waterfowl, most notably swans or coots. Loss of shallow water areas to rapid expansion of reedswamp will also reduce the area available for hydrophytes, although there may be an underlying anthropogenic basis for such change, such as nutrient enrichment, water level fluctuations, or high rates of siltation. Generally it is assumed that abnormally low aquatic plant abundance in lakes is likely to have some anthropogenic basis and that the most likely cause of this will be nutrient enrichment leading to increased chlorophyll concentrations and a corresponding reduction in transparency. Shoreline modification or alterations to the water level regime are the other most likely contributors.

Probably the single most useful measure of total macrophyte abundance in a lake is provided by the maximum depth of colonisation. This measure is unambiguous, unbiased by surveyor visual assessment and has a clear relationship to light regime

(see Canfield *et al*., 1985). Unfortunately, insufficient data on depth of colonisation are available to develop a metric based on this value, although this situation is likely to improve in the next few years as more standardised surveys that include this measure are carried out. Macrophyte abundance in its broadest sense has been assessed by a variety of approaches ranging from visual estimates of cover, or the proportion of the water column occupied (PVI), to point frequency counts, often supported by physical recovery of samples by grapnel or snorkelling. Although the data from these different approaches cannot be regarded as interchangeable, there are strong positive relationships between cover, frequency and PVI rooted in basic biological laws linking occupancy with local abundance (the more sites a species occupies, on average the greater its abundance at those sites).

The basis for the abundance metric is the mean cover per taxa. The use of this value has a firm theoretical basis in the form of species abundance-frequency distributions which have an underlying log-normal form. The rationale is that abnormally low cover per taxa represents a significant distortion of the abundance-frequency distribution reflecting a loss of more abundant taxa and a preponderance of rare species. This would be expected to occur where stress or disturbance levels have increased above a baseline. Lake macrophyte surveys generally report cover for the zone surveyed or the euphotic zone, whichever is smaller, rather than for the lake as a whole. Thus, the mean cover per species should be seen as reflection of the distribution of cover between taxa within the habitable zone, rather than an expression of the overall abundance of macrophytes in lakes or the extent of the euphotic zone. Mean rather than total cover (sum of cover scores) is used since total cover is highly dependent on the number of taxa present and therefore replicates part of the information conveyed by richness metrics.

4.4.2 Approach

All data were converted to percentage values. Raw measures of PVI or frequency expressed in percentage terms were unchanged. Values expressed on a DAFOR scale were converted to percentages based on 1 = 1%, 2 = 5%, 3 = 10%, 4 = 25% and 5 = 50%. Clearly this metric cannot be applied to presence-absence data. The following histogram (Figure 4.19) illustrates the distribution of records between percentage ‘cover’ classes in the global dataset (35,652 records).

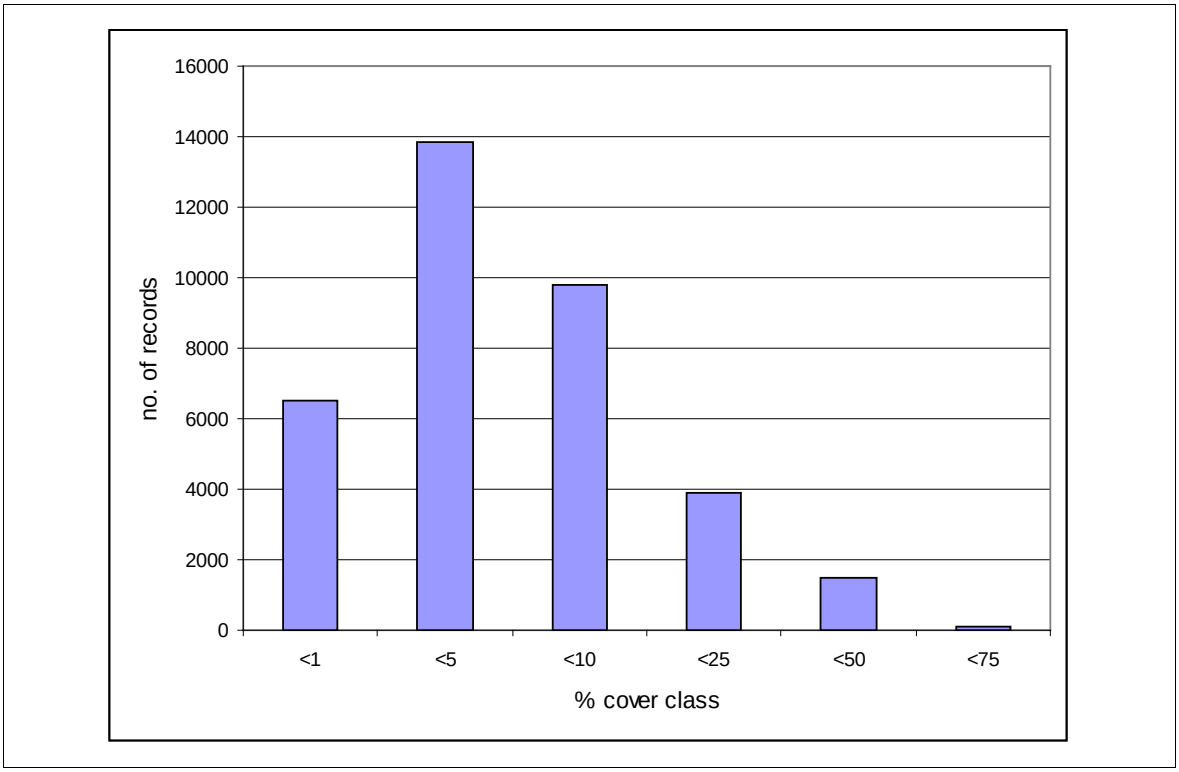


Figure 4.19 Distribution of cover values in global dataset (35,652 records)

The weighted mean percentage cover across all surveys is 9.9 per cent. The under-representation of the lowest cover scores is surprising. This may reflect surveyor bias, for example, in awarding too few low cover scores or in underdetecting genuinely rare species. Alternatively, it may reveal a trend for suitable habitat patches in lakes to occur at a sufficiently large size or wide distribution such that, when coupled with high within-lake dispersal potential, relatively few species have very low cover.

The rationale of the mean cover approach can be understood by reference to Figure 4.20 below which shows the distribution of cover values in three hypothetical sites, A, B and C, each of which support the same number of species. The weighted mean percent cover value in each of these sites is 4.9, 10.6 and 22.7 per cent respectively.

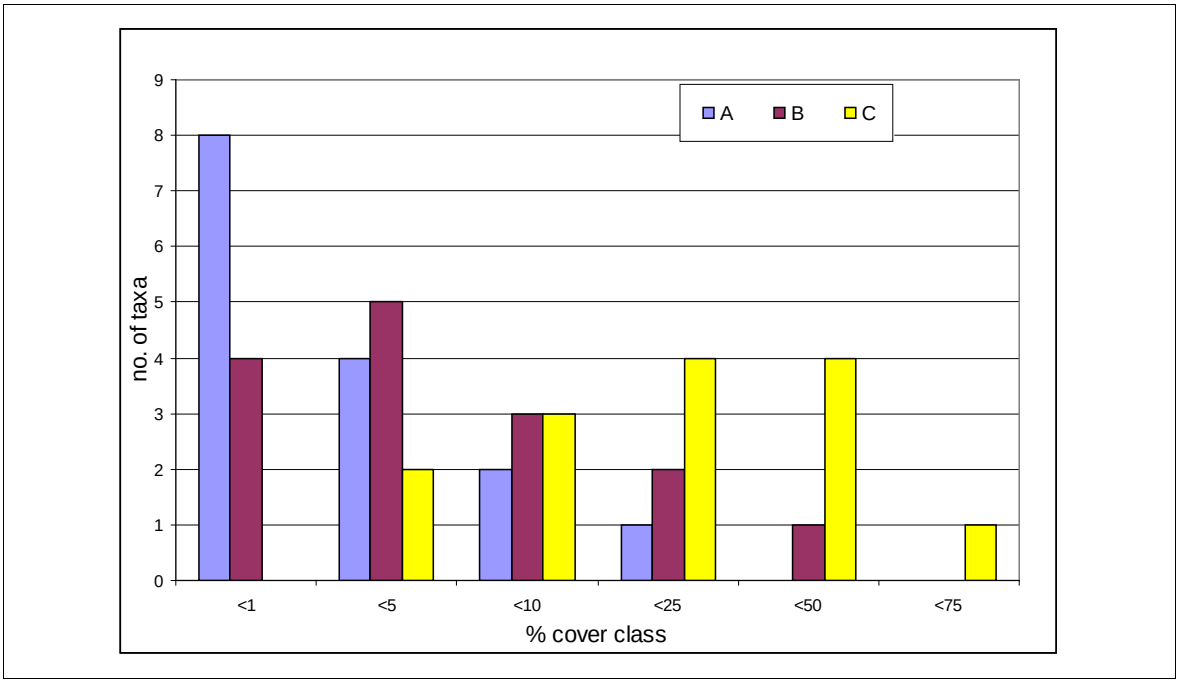


Figure 4.20 Distribution of cover values in three hypothetical sites, each supporting 15 species. The distribution in site A and C is significantly distorted from that found in Site B, leading to a much lower or higher mean percent cover. Higher cover, as reflected in Site C, is generally detected via compositional and richness metrics. The abundance metric developed here is designed to detect low abundance.

Site B represents a situation close to the norm while cover at A is shifted towards a preponderance of rare species (individually low cover), suggesting increased stress or disturbance, while at C mean cover is very high suggesting increasing dominance due to low stress or disturbance. The abundance metric was originally conceived as a bimodal metric for which there was an acceptable mean cover range (five to 25 per cent based on the interquartile range of observed mean cover at reference sites) and deviation above or below this range was assessed. However, the component of this metric measuring deviation above 25 per cent was found to be redundant as a result of inclusion of other metrics, while deviation below five per cent identified numerous sparsely vegetated sites which other metrics had failed to highlight.

4.4.3 Validation and comparative value

The use of the abundance metric was assessed by relating mean cover per species with supporting environmental data (Figure 4.21). This suggests that the abundance metric is poorly related to nutrient data but, perhaps predictably, is more sensitive to chlorophyll concentrations. There is some evidence of type-specificity in this relationship, with the average cover per species decreasing more steeply with increasing chlorophyll in the most base-rich lakes, although this may reflect the greater availability of data for this lake type. In general, the greatest value of this metric will be to reflect significant declines in cover of macrophytes at moderate to high fertility in base-rich lakes. Such declines are liable to have important secondary effects and may occur independently of a change in vegetation composition or richness. The abundance metric may prove redundant in many cases since the compositional metric LMNI should already have shown marked deterioration at the point that cover is severely compromised. However, the abundance metric will discriminate between sites where the expected assemblage is persisting (or re-establishing) but individual populations are small and potentially unstable, and sites where most species have

healthy stable populations and extent of macrophyte cover is likely to be beneficial to the ecosystem.

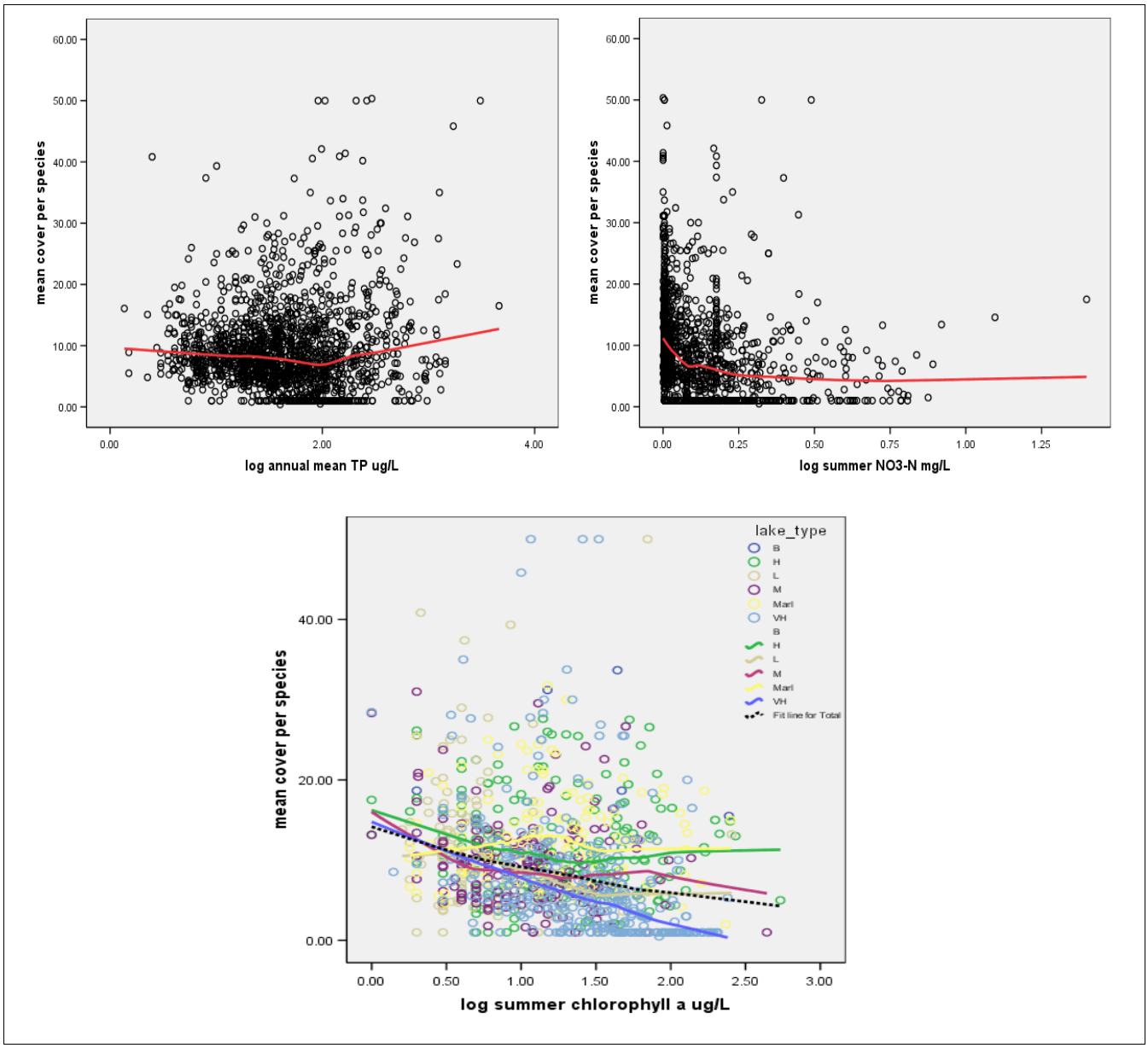


Figure 4.21 Relationships between mean cover per species and environmental parameters. Note the relative sensitivity of this metric to chlorophyll compared to nutrients.

4.4.4 Calculating the metric

This metric is calculated as the sum of the abundance values of all hydrophyte species present (as percentage cover or percentage frequency of occurrence) divided by the number of hydrophyte taxa recorded.

4.5 Other metrics

Two further metrics were developed to cover rare circumstances where a lake might be considered to be at high or good status when assessed by compositional, richness and

abundance metrics, yet inspection of the biological data would suggest otherwise. Thus, it would be possible for a site to achieve good or better status on the basis of LMNI, cover and richness yet to contain significant cover of one or more invasive alien or translocated native species, or of filamentous algae. In reality this scenario is likely to be rare since significant cover of invasive species or filamentous algae tends to be associated with nutrient enrichment and/or loss of diversity and will therefore be communicated by the metrics already described. Both of the following metrics were included in the initial sift to identify reference sites (see Section 6.5). The value of and rationale for these metrics is discussed below in more detail.

4.5.1 Invasive species (INV)

Invasive alien macrophyte species represent a threat to biodiversity and lake ecosystem integrity. At high cover they may reflect anthropogenic pressures (such as spikes in nutrient loading, shoreline disturbance) but invasive species may also be independent of such pressures, either colonising naturally from existing populations or being introduced intentionally. In this respect they may constitute a pressure in their own right. The term invasive species can also be applied to translocated native species, such as *Stratiotes aloides* or *Nymphoides peltata* that can behave as invasives when dispersed outside their native range. Several invasive alien species (notably *Elodea canadensis* and *E. nuttallii*) are now so pervasive in the UK that they could be considered naturalised. While their colonisation of new sites is clearly not desirable, it is frequently unpreventable and it may be preferable to consider them in the same way as translocated native species. Equally, there are cases of high nutrient loading where the persistence of an invasive taxa at high cover in the absence of more sensitive native taxa might be considered beneficial if it stabilises some of the core elements of lake ecosystem function (for example, by providing refugia for zooplankton).

A list of non-native macrophyte taxa that are known to occur in standing waters in the UK is provided below (Table 4.4). About half of these taxa currently have no records in the lake macrophyte database and it seems likely that they will remain rare in standing waters in the UK for the foreseeable future. It is almost impossible to provide a definitive list of non-native species as the potential for new introductions constantly exists. No stratification of non-native species according to threat level or invasiveness was carried out. Some form of ranking of species in relation to threat level, as advocated by the UKTAG Alien Species Group, may prove necessary in the future. This applies most notably to *Crassula helmsii* which appears to have especially strongly negative impacts. Once established in large water bodies, which are accessible to a range of dispersal vectors, it is likely that these will function as a source of colonists for other uninfected sites (Willby 2008). In the case of translocated species, all records of *Nymphoides peltata*, plus all records of *Stratiotes* outside East Anglia were treated as introductions. Recent evidence (Forbes, 2000) suggests that *Stratiotes* could, in fact, be native to Northern Ireland, whereas in this project it was presumed to be an introduction. The difficulty of determining, with certainty, the status of native species that appear to be distributed outside their native range, suggests that future consideration of invasive species might best be restricted to known alien taxa.

There is little evidence that increases in the cover of non-native macrophyte taxa have a negative impact on the diversity of native species, at least at the water body scale (Figure 4.22). This is almost certainly a consequence of scale dependency, especially in larger water bodies, since invasive species are likely to induce local reductions in species richness as a consequence of dominance effects, yet are unlikely to completely displace native species. Figure 4.22 offers some evidence that a high richness of native taxa is confined to situations where the extent of non-native taxa is low. In general, however, the overall extent of native and non-native taxa increases in

parallel (not illustrated). There is some empirical support for the argument that non-native species are an indicator of wider disturbance, since their cover tends to increase with increasing fertility (Figure 4.22). There was no evidence of type-specificity to the form of these relationships.

Table 4.4 Non-native hydrophyte species established in the wild in the UK

Species	Common name	Notes
Aponogeton distachyos	Cape-pondweed	
Azolla spp.	Water fern	
Cabomba caroliniana	Fanwort	No records in database
Crassula helmsii	Australian swamp stonecrop	
Egeria densa	Large-flowered waterweed	
Elodea callitrichoides	South American waterweed	
Elodea canadensis	Canadian waterweed	
Elodea nuttallii	Nuttall's waterweed	
Hydrocotyle ranunculoides	Floating pennywort	Currently absent from lakes
Lagarosiphon major	Curly water-thyme	
Lemna turionifera	Red duckweed	Recent arrival
Lemna minuta	Least duckweed	
Ludwigia grandiflora	Water primrose	Recent arrival
Myriophyllum aquaticum	Brazilian water-milfoil	
Nuphar advena	Spatter-dock	
Nymphaea spp.	Exotic lily species and cultivars	
Nymphoides peltata		Rare native in south-east England in river valleys. All pond/lake populations treated as introduced
Pontederia cordata	Pickerel weed	
Sagittaria latifolia	Duck potato	No records in database
Sagittaria subulata	Narrow-leaved arrowhead	No records in database
Sagittaria rigida	Canadian arrowhead	No records in database
Stratiotes aloides		Treated as native in East Anglia. All other extant lake populations treated as introduced. May need to review status in Northern Ireland
Vallisneria spiralis	Tapegrass	No records in database

Calculating the metric

This metric is expressed simply as the sum of the relative cover of all non-native hydrophyte taxa (as percentage cover or frequency) as a proportion of the summed cover of all hydrophytes. The non-native species considered are as listed in Table 4.4.

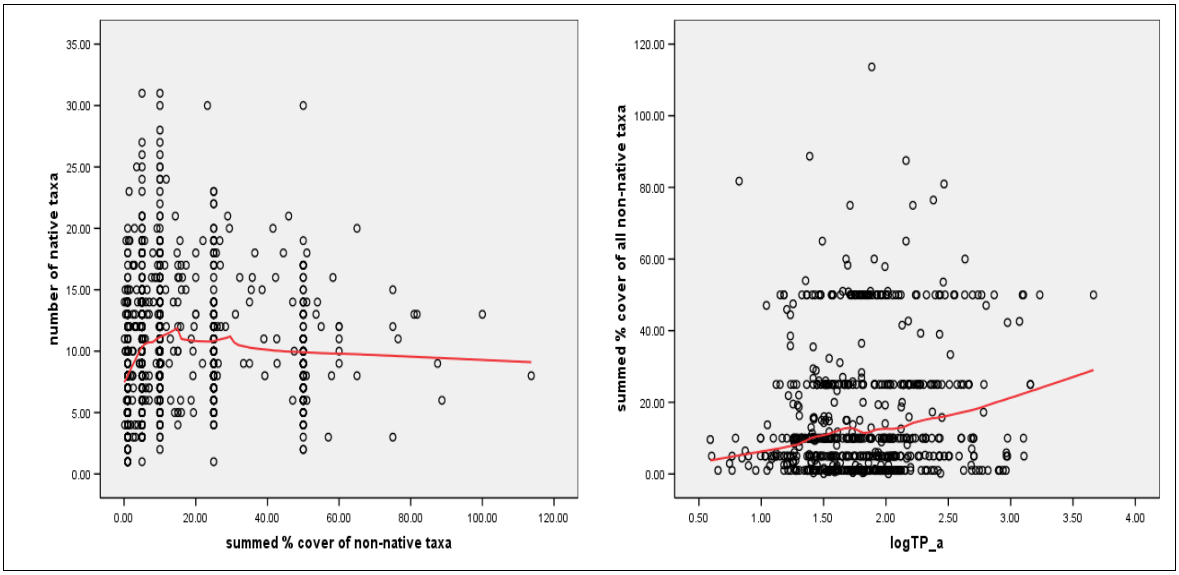


Figure 4.22 Relationship between number of native macrophyte taxa and the cover of non-native taxa (left) and between the cover of non-native taxa and annual mean water body TP (right)

4.5.2 Filamentous algae (ALG)

Filamentous algae exhibit fast growth rates and are therefore potentially highly responsive to nutrient enrichment. Increasing epiphytic algae may also be indicative of reduced grazing by macroinvertebrates and may suggest some form of disruption to the trophic cascade. Extensive epiphytic algal growth may be a precursor to the decline and ultimate loss of rooted macrophytes. Although the relationships are noisy there is support from Figure 4.23 for the use of ALG as a metric to support the detection of nutrient enrichment. However, the relationship is strongly type-specific and is largely driven by the highest alkalinity lakes.

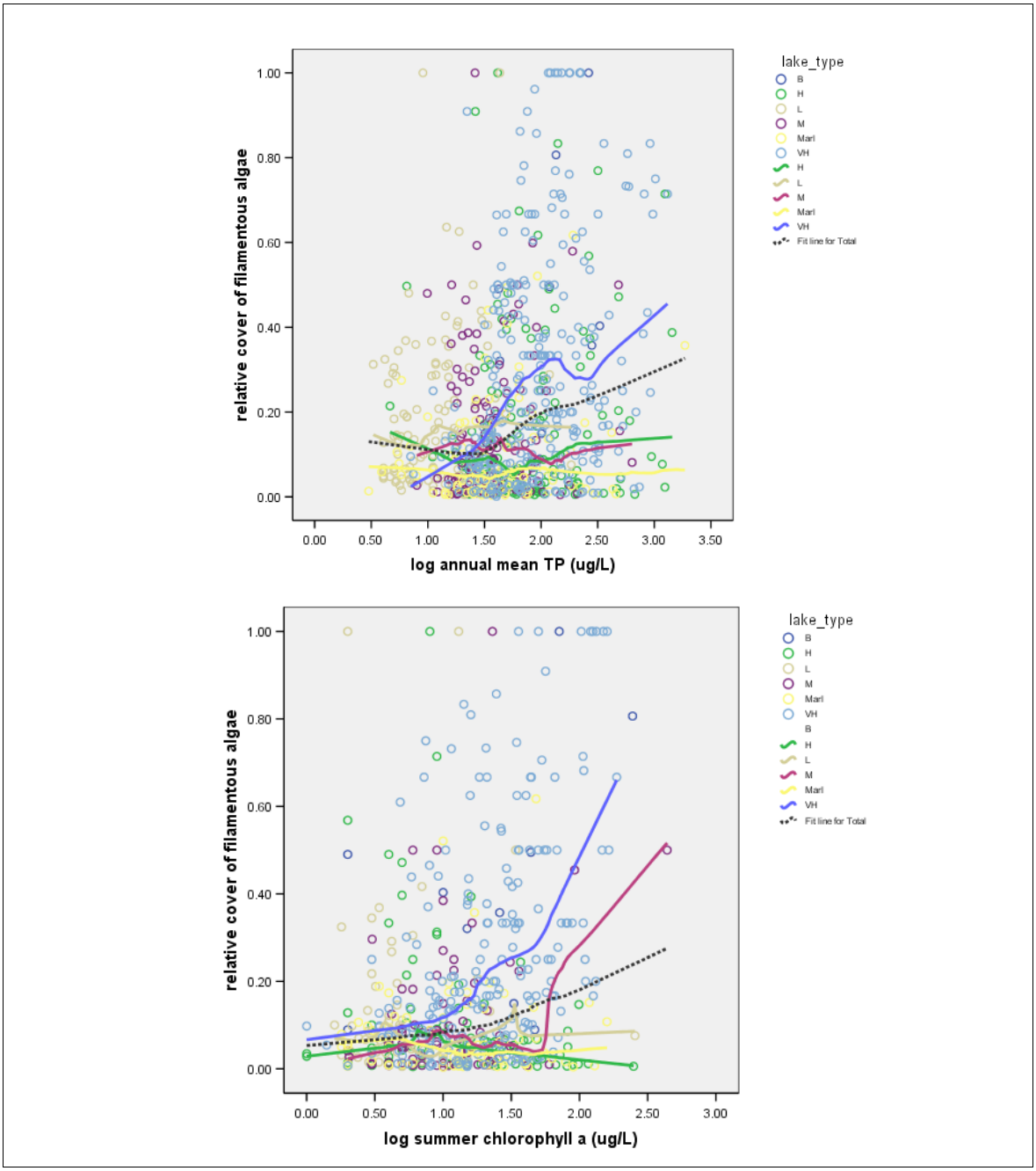


Figure 4.23 Change in relative cover of filamentous algae in lake macrophyte surveys in relation to chlorophyll and phosphorus concentrations. Note the strongly type-specific nature of the relationship and generally low relative cover of filamentous algae at low fertility.

This metric is easily calculated and does not require dedicated data collection. Few data are available for filamentous algae in lake surveys where algal samples have been processed to a higher taxonomic level. Only nuisance taxa (such as *Cladophora*, *Hydrodyction*, *Enteromorpha*) are identfieid with any regularity. Consequently there is currently no scope to create separate ranking scores for most taxa. This is regrettable since there will almost certainly be a gradient of increasing impact from small unbranched filamentous greens (such as *Ulothrix*) through to large, branched filamentous taxa such as *Cladophora*. Even using a simple subdivision between epiphytic and epipelic algae when recording would potentially improve the quality of the information that could be extracted from survey data. Since filamentous algae as a

group are ubiquitous across the full gradient of aquatic plant assemblages, it is not surprising that this taxon carries an LMNI score of 6.39, which is almost equivalent to the global average rank. In this instance information associated with filamentous algae is best considered as a separate metric.

A degree of caution is needed in the use and interpretation of this metric when data is collected using the standard survey protocol. Filamentous algae may occur at high frequency in the marginal areas of lakes, especially on hard substrates, often early in the season when they pre-empt rooted macrophyte growth, or after prolonged calm weather. The use of frequency as opposed to strict abundance data means that widely distributed (frequently occurring) taxa with low cover scores will score high relative to more locally distributed macrophytes that are abundant where they occur. Although frequency is weighted by an abundance ranking at each recording point, there is a risk that in some otherwise high status lakes, filamentous algae will appear to represent a disproportionately large component of the vegetation. This is particularly likely where the number and cover of other taxa is naturally low.

In practice it will be possible to find some filamentous algae in any lake. The absence of records of filamentous algae from older survey data (mainly low and moderate alkalinity lakes in Scotland) needs to be interpreted with caution and may simply reflect a lack of recording (even where filamentous algae were included on standard recording sheets). The practice of carrying out detailed surveys in small areas, as required by the standard method, as opposed to surveying the entire lake may also be liable to inflate the importance of filamentous algae. In future a small downweighting of filamentous algal frequencies as reported by the standard method may be necessary during the data compilation step to deal with this scenario. This is likely to require a comparison of original and standard approaches at a small number of sites where surveys are carried out contemporaneously, or if recently collected, comparable archived survey data is available. An alternative and simpler solution is suggested by Figure 4.23; a metric based on the extent of filamentous algae might be best confined to the highest alkalinity lakes where there is clear evidence of responsiveness to high nutrient levels.

A future iteration should consider the use of absolute as opposed to relative cover of filamentous algae (total absolute cover rather than the extent of algae as a proportion of all macrophyte cover). In a naturally sparsely vegetated lake, a low-modest cover of filamentous algae will currently attract a high relative cover score whereas in a densely vegetated lake, even extensive growth of filamentous algae might be diluted in relative terms by the volume of other plant species.

Calculating the metric

Determine the percentage cover of filamentous algae. In most cases surveys have not attempted to discriminate between different types of filamentous algae. Where separate taxa have been recorded (usually *Cladophora*, *Ulva* (=*Enteromorpha*), *Hydrodyction*, *Spirogyra*, *Vaucheria*) a total for filamentous algae should be obtained. The value is the sum of individual percentage cover scores of each taxon recorded. This value is then expressed as a proportion of the total cover of all hydrophyte species (including macroalgae) at a site (obtained by summing individual percentage cover scores).

4.6 Summary

This chapter describes the process to develop macrophyte-based metrics calculated from standard survey data. These metrics are designed to be complementary and to be

sensitive to a range of pressures to which water bodies are exposed. For each metric, a proof of concept and optimum method of calculation is discussed, based on empirical relationships with measured pressure data. Metrics are summarised as follows (Table 4.5).

Table 4.5 Summary of macrophyte metrics used for water body classification, basis for their use and pressures to which they are sensitive

Attribute	Metric	Basis for use	Sensitivity
Composition	LMNI	Average rank of taxa present, unweighted by cover	Nutrients
Structural diversity	N_FG	Number of functional groups (max 18)	Hydromorphology (Nutrients)
Taxonomic diversity	N_TAXA	Total number of hydrophyte spp present based on fixed list of 188 taxa	Nutrients Hydromorphology Acidification
Abundance	COV	Average percentage cover of hydrophyte taxa present	Nutrients Hydromorphology
	ALG	Relative cover of filamentous algae	Nutrients
	INV	Relative cover of non-native species	Nutrients Hydromorphology

The analyses presented support the use of metrics described for the purposes of assessing ecological status of lakes using aquatic macrophytes and the integration of these metrics into a classification tool. In particular, the analyses show that most metrics have a 'niche' in terms of lake type and position on a pressure gradient, where the information communicated by that metric will be of most use in classification. The following points can be made with regard to the niche of each metric:

LMNI: Sensitive to nutrient enrichment in low- and moderate-alkalinity lakes. Insensitive to moderate to high levels of enrichment in highest alkalinity lakes.

N_FG and N_TAXA: Sensitive to low pH or moderate nutrient loading or high chlorophyll concentrations in naturally productive lakes. Variable signal at low to moderate productivity but suggests stimulation of richness. Potentially sensitive to some forms of hydromorphic alteration.

COV: Weak general decline with increasing eutrophication but only via phytoplankton chlorophyll influence on light regime. Useful additional metric in naturally most productive lake types, but otherwise of limited value.

INV: No type-specificity and weakly correlated with increasing enrichment. Best regarded as pressure in own right.

ALG: Clear increase at moderate to high levels of enrichment in naturally most productive lake types. Otherwise of limited value.

The ultimate currency of the WFD is ecological status, which itself is a reflection of the structural and functional attributes of a suite of quality elements. The metrics discussed are suitable to assess a range of pressures which lakes face, as well as being aligned

with a range of ecosystem functions within lakes that are strongly dependent upon macrophytes. A conventional approach to assessment based only on compositional, weighted-average metrics, such as LMNI, will not suffice for macrophytes. A broader spectrum of metrics is required to compensate for the weaknesses inherent in such an approach, and to support the holistic assessment of ecological status under the WFD.

5 Establishing reference conditions

5.1 Introduction

The Water Framework Directive (WFD) 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.

The UK database of macrophyte surveys in lakes and 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 define and establish 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 a 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 the subdivision of 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 the 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 (HT) taxa (which persist in the most degraded sites) are either absent or rare (or account for a small proportion of species or cover), while the most sensitive (HS) taxa (absent from the most degraded sites) are common (or dominate the cover). This framework envisages that the crossover between tolerant (all T) and sensitive (all S) 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 largely unaltered and there is a subtle shift in taxonomic composition from high to good. In the second state, macrophyte-mediated functions are severely compromised or fundamentally altered from those occurring 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 (sensitive or tolerant taxa) and the metric to which they are related. Thus, if dominance by the most tolerant species is associated with, or contributes directly to, undesirable disturbances (such as loss of habitat support functions, more fish kills, less stability in macrophyte cover) these disturbances will have a high probability of occurrence at the M/P boundary but a low probability of occurrence at the G/M boundary. The general concept of functional response groups, defined in terms of sensitivity to disturbance, does not feature in the normative definitions for macrophytes but the principle 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 the high/good and poor/bad boundaries are effectively 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.

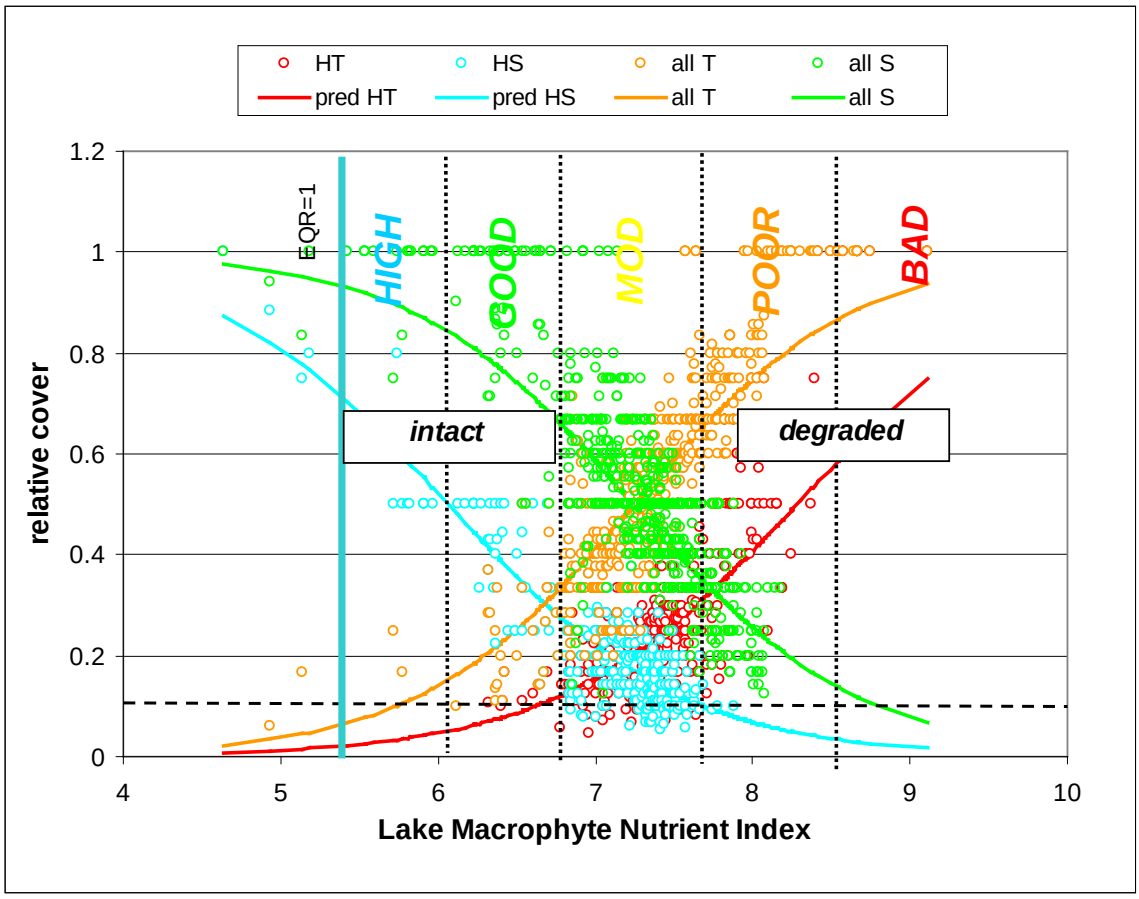


Figure 5.1 The conceptual framework relating structural changes in macrophyte assemblages to normative definitions which was used to provide an ecological interpretation of class boundaries. The Lake Macrophyte Nutrient Index (LMNI) is shown as an example of an inferred pressure gradient that can be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

Table 5.1 Interpretation of normative definitions for lake macrophytes in the context of 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…..no more than very 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 the ecosystem or physicochemical environment	S taxa dominate but HS taxa are scarcer and account for about half the contribution of S taxa. T taxa are present but remain subordinate. HT taxa, if present, are rare.	Functions delivered at H all intact. Undesirable disturbances very 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 and reduced stability of macrophyte populations. Undesirable disturbances 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 increased incidence and persistence of undesirable disturbances. Macrophyte cover often unstable and dominated by small number of taxa.
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 (e.g. fish kills, algal blooms, biological invasions) frequent and dominating. Macrophyte cover highly unstable or absent.

HS = highly sensitive; S = sensitive; HT = highly tolerant; T = tolerant.

5.3 Identifying reference sites – theory and practice

The WFD 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

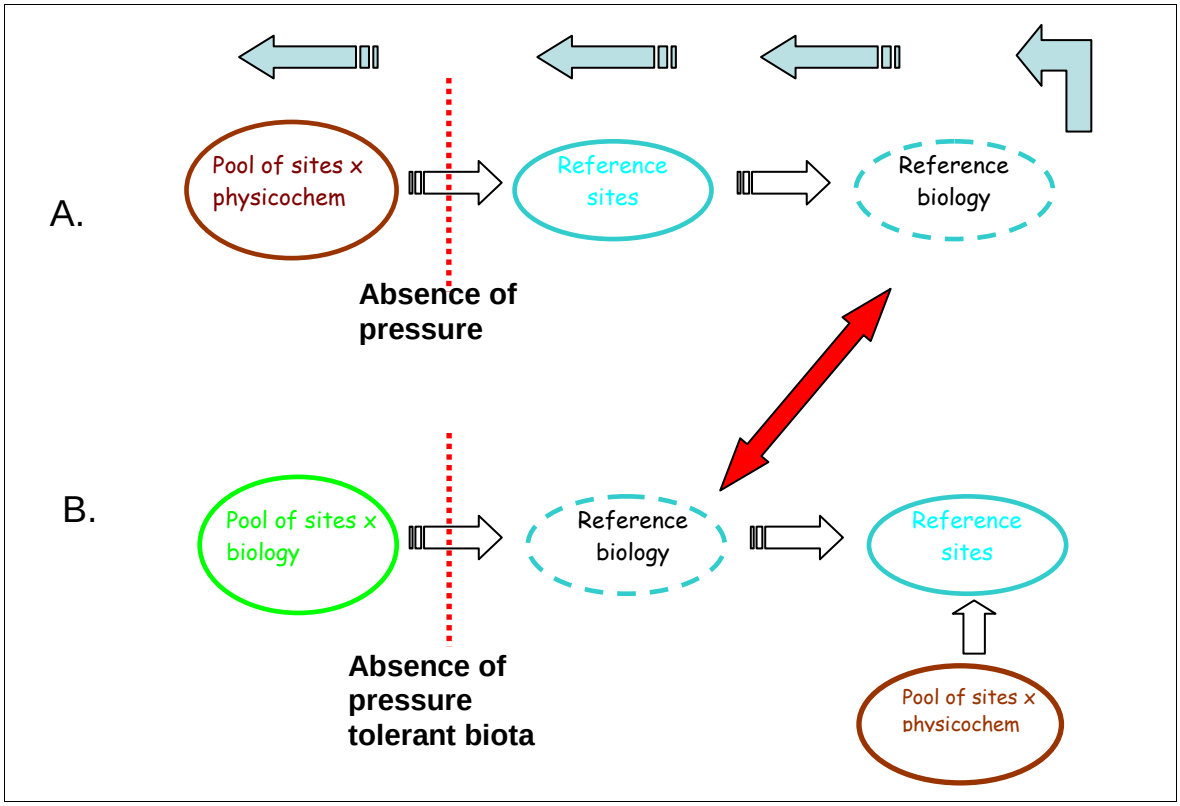


Figure 5.2 Alternative protocols for the identification of biological reference condition sites

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 the 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 temporally matched with the biological dataset and (ii) an ability to define, *a priori*, baseline conditions for pressure indicators; this is likely to involve expert opinion or cross-referencing to biological information to identify response thresholds, or access to palaeo-ecological data. There is also 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 is unavoidably 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 µg/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 requiring confirmation through expert consideration of the biology 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 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, LMNI 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 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 condition. 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. Threshold values for different biological metrics were an important element in the establishment of a reference site network in RIVPACS (Wright, 2000). 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. Supporting chemical pressure data is available for only 10 per cent of putative reference sites selected on biological criteria, since monitoring networks have tended to focus on large and/or high profile sites in more densely populated catchments 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 required 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 inherent weaknesses in this approach it has passed the test of intercalibration across several 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 in other European countries promote the notion of sensitive and tolerant species (see Schaumburg *et al*., 2004). Although this distinction is appealingly simple, sensitivity is a relative concept, referable to pressure gradients, and forms a continuum. Its use in classification therefore has the risk of introducing another tier of discontinuity. Moreover, 'sensitive' and 'tolerant' species have often been defined purely from expert opinion, into which value judgements associated with conservation and rarity are commonly set. 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 LMNI to delimit different response groups uses a type-specific ordination of survey data constrained by the site LMNI 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 5.1. 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 LMNI 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 lake types. For example, if macrophyte species ranks lie on a continuous scale from one to 10, 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 lakes, but would be 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 lake 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.

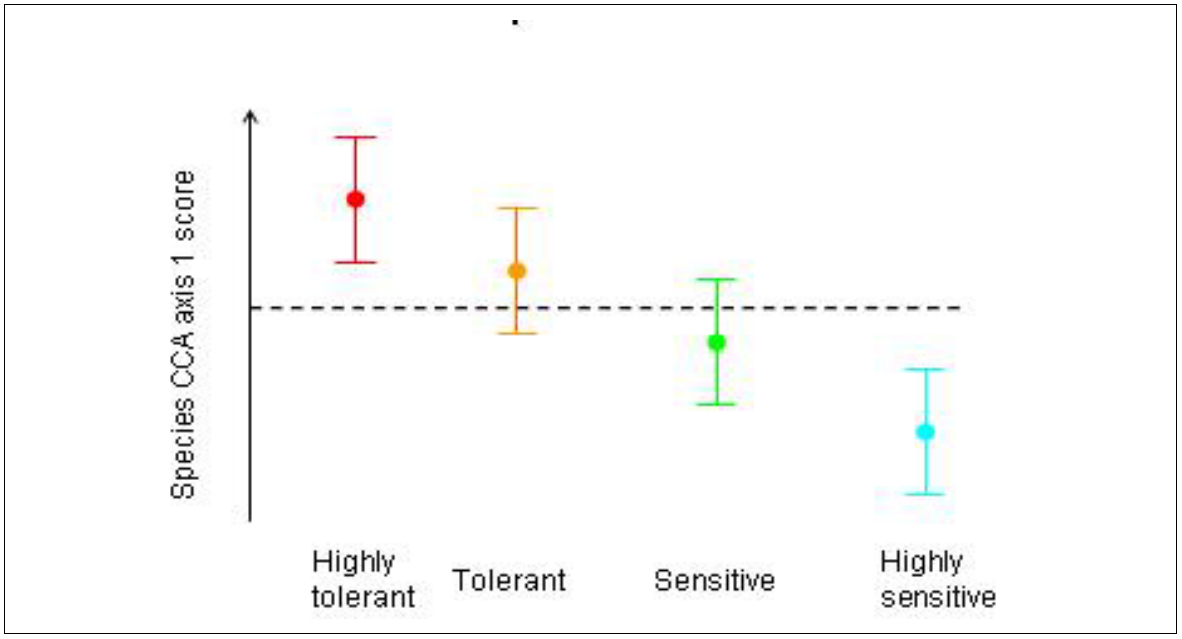


Figure 5.3 Basis for assignment of taxa to response groups. Left axis represents first axis scores of a Canonical Correspondence Analysis in which presence-absence species data for that water body type represents dependent variable and site pressure index is the explanatory variable. Solid dots represent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

5.4.2 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 currently available in low-alkalinity shallow lakes 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 lakes of this type. Conversely, due to a long history of degradation, it is unlikely that the best available high-alkalinity shallow lakes mark the true unimpacted end of a gradient, while it is likely that the worst available sites are correctly anchored.

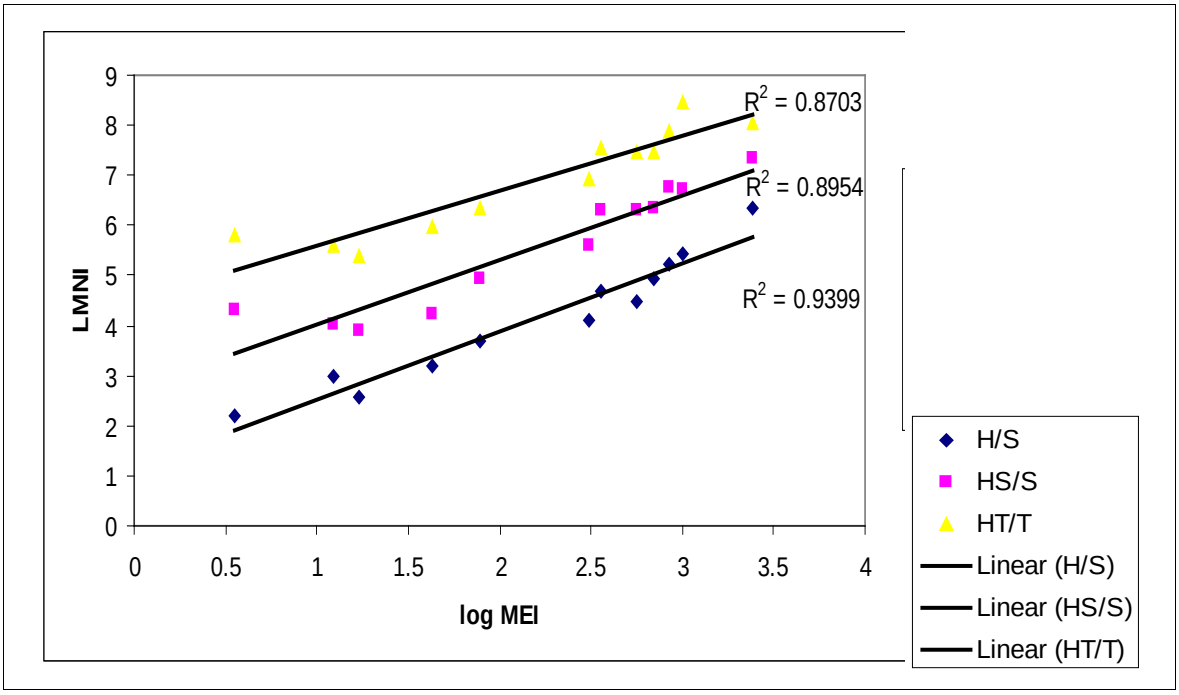


Figure 5.4 Models used to standardise threshold values of LMNI delimiting different response groups in the different lake types. The observed values of LMNI are the thresholds obtained from the CCA approach described.

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

Type	log_{10} MEI	Unstandardised – from CCA			Post standardisation – from linear model		
		HS	S	T	HS	S	T
HA,VSh-N	2.93	5.23	6.75	7.89	5.15	6.52	7.71
HA, Sh-N	2.55	4.69	6.31	7.55	4.64	6.03	7.30
LA, VSh	1.63	3.21	4.22	5.96	3.39	4.84	6.29
LA, Sh	1.23	2.56	3.9	5.4	2.85	4.32	5.85
LA, D	0.55	2.21	4.32	5.8	1.91	3.44	5.10
MA, VSh	2.49	4.1	5.61	6.91	4.56	5.96	7.23
MA, Sh	1.90	3.71	4.95	6.35	3.75	5.18	6.58
Marl, VSh	3.00	5.42	6.72	8.47	5.25	6.62	7.79
Marl, Sh	2.75	4.5	6.32	7.45	4.91	6.29	7.52
Pt	1.09	2.97	4.04	5.6	2.66	4.14	5.70
VHA, VSh-N	3.38	6.33	7.35	8.04	5.77	7.11	8.20
VHA, Sh-N	2.85	4.94	6.36	7.47	5.04	6.41	7.62
HA, VSh-C	2.93	5.4	6.81	8	5.50	6.93	8.14
HA, Sh-C	2.55	4.95	6.31	7.6	5.13	6.62	7.91
VHA, VSh-C	3.38	6.1	7.33	8.51	5.93	7.30	8.42
VHA, Sh-C	2.85	5.35	6.78	8.13	5.41	6.86	8.09

To standardise LMNI scores which delimit the response groups from highly sensitive through to highly tolerant, the scores obtained in Section 5.4.1 for each lake type were regressed against the Morpho-Edaphic Index (MEI) value for the range of lake types (Figure 5.4). MEI was calculated as the log_{10} of (alkalinity (ueq/l)/mean depth (m)) in which values for alkalinity and depth are the median of values of surveys in each lake type. This step also has the benefit of downweighting the influence of any surveys that may have been inadvertently assigned to the wrong lake type due to erroneous environmental data. Standardised scores are given in Table 5.2. Values are the upper thresholds of LMNI for each response group in each lake type. Thus, HT taxa are represented by LMNI values exceeding the upper threshold shown for T taxa.

The full membership of the different response groups is given in Table 5.3 based on LMNI thresholds presented in Table 5.2. Some qualification is needed of the terms high 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 the pool of species 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 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 LMNI 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, but are indicators of an impact in their own right. Other metrics have been devised to cover these scenarios and are described in Chapter 4. Moreover, additional criteria (presented in Table 5.4) are considered in the biological screening of reference sites.

4. Certain species (with an LMNI score above 8.09 such as *Ceratophyllum demersum*) are considered highly tolerant in all lake types. There is good evidence that these species increase with nutrient enrichment and that they 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. Comments in County Floras reveal that many taxa that are common and widely distributed today, and are widely regarded as tolerant of nutrient enrichment, were also widely distributed in lowland areas centuries ago. It is also possible that nutrient enrichment of lakes supports extensive emergent plant growth which creates more sheltered conditions, thus favouring the spread of highly tolerant species from habitats such as ditches and ponds (highly tolerant species may not always have been a feature of the vegetation of lakes in reference condition, but existed at the time in other types of aquatic habitat).

5. Classification of lakes based on macrophytes is achieved through information provided by a range of metrics, not just LMNI. This table 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 lake type. Hence a site with only highly sensitive species will not be classed as high status if the number or cover of such taxa is very low. Conversely, a site with only highly tolerant taxa will not be classed as bad if the diversity of taxa is high, or if other metrics such as extent of invasive species or filamentous algae mitigate the level of impact. The biological characteristics of lakes of different ecological status belonging to different lake types are discussed in Section 8.

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

Species	VHA, VSh-C	VHA, Sh-C	VHA, VSh-N	VHA, Sh-N	HA, VSh-C	HA, Sh-C	HA, VSh-N	HA, Sh-N	Marl, VSh	Marl, Sh	MA, VSh	MA, Sh	LA, VSh	LA, Sh	LA, D	Pt
Utricularia ochroleuca											1	1	1	1	1	1
Eriocaulon aquaticum						1		1				1	1	1		1
Eleocharis multicaulis	1	1	1	1	1	1	1	1		1	1	1	1	1	1	1
Utricularia stygia								1			1	1	1	1	1	
Lobelia dortmanna	1		1	1		1	1	1		1	1	1	1	1	2	1
Utricularia intermedia sens.lat.	1		1	1				1			1	1	1	1	2	1
Potamogeton epihydrus														1		
Subularia aquatica								1		1	1	1	1	2	2	2
Utricularia minor	1		1	1		1		1			1	1	1	2	2	2
Lycopodiella inundata														2	2	2
Batrachospermum spp												1	1	2	2	2
Isoetes lacustris			1	1		1	1	1		1	1	1	1	2	2	2
Sphagnum (aquatic indet.)	1	1	1	1	1	1	1	1	1	1	1	1	1	2	2	2
Eleogiton fluitans	1	1	1	1	1	1	1	1	1	1	1	1	2	2	3	2
Potamogeton polygonifolius	1	1	1	1		1	1	1	1	1	1	1	2	2	3	2
Sparganium angustifolium	1		1	1			1	1	1	1	1	1	2	2	3	2
Juncus bulbosus	1	1	1	1	1	1	1	1		1	1	1	2	2	3	2
Isoetes echinospora			1			1		1	1		1	2	2	2	3	
Utricularia spp	1		1					1		1	1	2	2	2	3	
Nitella gracilis	1		1											3	3	
Myriophyllum alterniflorum	1	1	1	1	1	1	1	1	1	1	1	2	2	3	3	3
Fontinalis squamosa	1		1				1			1	1	2	2	3	3	
Scorpidium scorpioides				1				1		1	1	2	2	3		
Damasonium alisma		1		1												
Myriophyllum aquaticum								1								
Utricularia australis	1		1					1		1	2	2	2	3		3
Littorella uniflora	1	1	1	1	1	1	1	2	1	1	2	2	2	3	3	3
Isoetes sp				1							2	2		3	3	3
Menyanthes trifoliata	1	1	1	1	1	1	1	2	1	1	2	2	2	3	3	3
Utricularia cf. australis			1				1	2		1	2	2	3	3		3
Sparganium natans	1	1	1	1		1	1	2	1	1	2	2	3	3	3	3

Species	VHA, VSh-C	VHA, Sh-C	VHA, VSh-N	VHA, Sh-N	HA, VSh-C	HA, Sh-C	HA, VSh-N	HA, Sh-N	Marl, VSh	Marl, Sh	MA, VSh	MA, Sh	LA, VSh	LA, Sh	LA, D	Pt
Nitella confervacea											2	2		3		3
Hypericum elodes	1	1	1	1	1	1	1	2			2	2	3	3	3	
Utricularia vulgaris sens.str.	1		1									2	3	3		3
Luronium natans	1	1	1	2	1	1	1	2			2	2	3	3	4	
Potamogeton natans	1	1	1	2	1	2	2	2	1	2	2	2	3	3	4	3
Nitella translucens		1		2				2	1		2	2	3	3	4	
Cinclidotus fontinaloides							2	2		2	2	2		3		
Pilularia globulifera		1		2		2		2			2	2	3	3	4	
Potamogeton x griffithii														3		
Nitella sp	1		1	2	1	2	2	2	2	2	2	3	3	3	4	3
Nitella opaca			1	2		2	2	2	2	2	2	3	3	3	4	3
Chara virgata var.virgata						2		2				3		3		
Nuphar pumila					1	2	2	2			2	3	3	3	4	3
Utricularia vulgaris sens.lat.	1	1	1	2	1	2	2	2	2	2	2	3	3	3	4	3
Najas flexilis			1	2			2	2			2	3		3		
Elatine hexandra		1		2	1	2	2	2		2	2	3	3	3	4	
Fontinalis antipyretica	1	2	1	2	1	2	2	2	2	2	2	3	3	3	4	3
Callitriche hamulata	1	2	1	2	1	2	2	2	2	2	2	3	3	3	4	3
Potamogeton gramineus	1	2	1	2	1	2	2	2	2	2	2	3	3	3	4	3
Nymphaea alba	1	2	1	2	2	2	2	2	2	2	2	3	3	3	4	3
Potamogeton x sparganiifolius				2				2					3	3		
Chara virgata	1	2	1	2	2	2	2	2	2	2	2	3	3	3	4	3
Lythrum portula	1	2	1	2	2	2	2	2			2	3	3	3	4	3
Alisma gramineum											2					
Ludwigia palustris											2		3			
Baldellia ranunculoides	1	2	1	2	2	2	2	2		2	2	3	3	3		
Potamogeton x nitens			1	2	2	2	2	2		2	2	3	3	3	4	3
Nitella flexilis agg.	1	2	1	2	2	2	2	2	2	2	2	3	3	3	4	3
Nuphar x spenneriana	1	2	1			2	2	2				3	3	3		
Potamogeton rutilus							2	2			2	3	3			
Chara virgata var.annulata		2		2		2	2	2	2	2	2	3	3	3		
Nymphaea (exotics)		2		2						2	2	3	3	3		
Potamogeton x cooperi		2		2		2		2								

Species	VHA, VSh-C	VHA, Sh-C	VHA, VSh-N	VHA, Sh-N	HA, VSh-C	HA, Sh-C	HA, VSh-N	HA, Sh-N	Marl, VSh	Marl, Sh	MA, VSh	MA, Sh	LA, VSh	LA, Sh	LA, D	Pt
Apium inundatum	1	2	1	2	2	2	2	2	2	2	2	3	3	3	4	4
Potamogeton x zizii	1	2	1	2			2	2			2	3	3	3		
Ranunculus omiophyllus	1	2	1	2	2	2	2	2			2	3	3	3	4	
Potamogeton praelongus	1	2	1	2	2	2	2	2	2	2	2	3	3	3		
Ranunculus p. penicillatus								2		2		3		3		
Potamogeton alpinus	1	2	1	2	2	2	2	2	2	2	2	3	3	3	4	4
Potamogeton perfoliatus	1	2	2	2	2	2	2	2	2	2	2	3	3	3	4	4
Hildenbrandia sp								2			2					
Chara sp	1	2	2	2	2	2	2	2		2	2	3	3	4	4	4
Ranunculus sp.		2		2				2				3		4	4	
Callitriche stagnalis	1	2	2	2	2	2	2	2	2	2	3	3	3	4	4	4
Jungermannia sp						2		2						4		
Chara aspera	2		2	2		2	2	2	2	2		3	3	4		
Riccia sp.												3		4		
Racomitrium sp.	2		2			2		3		2						
Potamogeton berchtoldii	2	2	2	2	2	2	2	3	2	2	3	3	3	4	4	4
Potamogeton x suecicus				2				3								
Potamogeton filiformis			2	2		2	2	3	2	2	3	3		4		
Crassula helmsii						2		3	2		3	3		4		
Ricciocarpus natans	2		2									3				
Callitriche agg.	2	2	2	2	2	2	2	3	2	3	3	3	4	4	4	
Brachythecium sp.		2		2							3					
Filamentous algae	2	2	2	2	2	2	2	3	2	3	3	3	4	4	4	
Hippuris vulgaris	2	2	2	2	2	2	2	3	2	3	3	3	4	4	4	4
Ranunculus peltatus	2	2	2	3	2	2	2	3	2	3	3	3	4	4		
Groenlandia densa	2		2							3		3				
Limosella aquatica				3	2		2	3				3		4	4	
Callitriche brutia														4		
Ranunculus penicillatus												3				
Ranunculus aquatilis agg.	2	2	2	3	2	2	2	3	2	3	3	3	4	4	4	
Chara curta	2		2	3		2	2	3	2	3		3				
Chara pedunculata	2	2	2	3					2	3						
Sparganium emersum	2	2	2	3	2	2	3	3	2	3	3	4	4	4	4	4

Species	VHA, VSh-C	VHA, Sh-C	VHA, VSh-N	VHA, Sh-N	HA, VSh-C	HA, Sh-C	HA, VSh-N	HA, Sh-N	Marl, VSh	Marl, Sh	MA, VSh	MA, Sh	LA, VSh	LA, Sh	LA, D	Pt
Ranunculus hederaceus	2	2	2	3	2	2	3	3		3	3	4	4	4		4
Ranunculus aquatilis sens.str.	2		2	3	2		3	3	2		3	4		4	4	
Riccia fluitans	2	2	2	3	2	3	3	3			3	4		4		
Drepanocladus fluitans	2		2		2	3	3	3			3	4	4	4		
Ranunculus trichophyllus	2	2	2	3	2		3			3	3	4		4		
Scapania sp.						3		3						4		
Potamogeton coloratus	2	2	2	3			3	3	3	3		4				
Callitriche hermaphroditica	2	2	2	3	2	3	3	3	3	3	3	4	4	4		
Potamogeton obtusifolius	2	2	2	3	2	3	3	3	3	3	3	4	4	4	4	4
Eleocharis acicularis	2	2	2	3	2	3	3	3	3	3	3	4	4	4	4	
Ranunculus baudotii			2	3			3	3			3	4				4
Chara hispida	2		2	3	2	3	3	3	3	3	3	4		4		
Potamogeton x salicifolius	2		2					3		3						
Chara contraria var.hispidula		3	2	3				3	3	3						
Elodea nuttallii	2	3	2	3	2	3	3	3		3	3	4		4	4	
Nuphar lutea	2	3	2	3	2	3	3	3	3	3	3	4	4	4	4	
Chara rudis		3		3				3		3		4				
Potamogeton lucens	2	3	2	3	3	3	3	3	3	3	3	4	4	4		
Elodea canadensis	2	3	3	3	3	3	3	3	3	3	3	4	4	4	4	
Tolypella glomerata					3	3	3	3	3			4				
Chara vulgaris	2	3	3	3	3		3	3	3	3	3	4				
Chara vulgaris var. papillata	2		3	3	3		3	3	3	3		4				
Persicaria amphibia	2	3	3	3	3	3	3	3	3	3	4	4	4	4		
Fissidens sp.										3						
Hottonia palustris	2	3	3	3	3	3	3	4			4	4				
Chara globularis sens.lat	2	3	3	3	3	3	3	4	3	3	4	4	4	4		
Elatine hydropiper	2		3	3	3		3	4			4	4		4	4	
Lagarosiphon major								4			4					
Ranunculus fluitans							3					4				
Callitriche platycarpa	3		3	3	3	3	3	4			4	4	4	4		
Chara contraria var.contraria	3	3	3	3	3	3	3	4	3	3	4	4				
Lemna minor	3	3	3	3	3	3	3	4	3	4	4	4	4	4		4
Potamogeton pusillus	3	3	3	3	3	3	3	4	3	4	4	4	4	4	4	

Species	VHA, VSh-C	VHA, Sh-C	VHA, VSh-N	VHA, Sh-N	HA, VSh-C	HA, Sh-C	HA, VSh-N	HA, Sh-N	Marl, VSh	Marl, Sh	MA, VSh	MA, Sh	LA, VSh	LA, Sh	LA, D	Pt
Ranunculus lingua	3	3	3	3	3	3	3	4	3	4	4	4				
Nitellopsis obtusa	3	3	3	4												
Potamogeton friesii	3		3	4		3	3	4	3	4	4	4				
Potamogeton crispus	3	3	3	4	3	3	3	4	3	4	4	4	4	4	4	
Calliergon sp.		3		4		3		4								
Lemna trisulca	3	3	3	4	3	3	4	4	4	4	4	4	4	4		
Callitriche obtusangula	3	3	3	4		3	4	4		4	4	4	4			
Myriophyllum spicatum	3	3	3	4	3	3	4	4	4	4	4	4		4	4	
Fucus												4				
Sagittaria sagittifolia	3	3	3	4			4	4	4	4	4		4	4		
Elodea callitrichoides							4									
Chara connivens	3		3													
Chara intermedia	3		3													
Potamogeton compressus	3		3								4					
Nymphoides peltata	3	3	3	4	3		4		4		4	4				
Chara canescens	3		3													
Chara contraria var. hispidula	3		4		4		4									
Potamogeton pectinatus	3	4	4	4	4	4	4	4	4	4	4	4		4		4
Hydrocharis morsus-ranae	3	4	4	4			4		4	4		4				
Oenanthe aquatica	3	4	4	4	4	4	4	4			4	4	4			
Potamogeton x lintonii	3		4				4									
Callitriche truncata		4		4	4	4	4	4				4				
Chara vulgaris var. longibracteata	3		4					4				4				
Potamogeton trichoides	3		4		4		4				4	4				
Ulva (Enteromorpha)	4	4	4	4	4	4	4	4			4	4				4
Nitella mucronata	4		4		4		4		4							
Leptodyctium riparium	4		4				4					4				
Butomus umbellatus	4	4	4	4	4	4	4	4		4	4	4				
Zannichellia palustris	4	4	4	4	4	4	4	4	4	4	4	4				4
Stratiotes aloides	4		4	4					4	4	4					
Chara baltica	4		4													
Lemna minuta	4	4	4	4	4		4				4	4				
Ranunculus circinatus	4	4	4	4	4	4	4	4	4	4	4					

Species	VHA, VSh-C	VHA, Sh-C	VHA, VSh-N	VHA, Sh-N	HA, VSh-C	HA, Sh-C	HA, VSh-N	HA, Sh-N	Marl, VSh	Marl, Sh	MA, VSh	MA, Sh	LA, VSh	LA, Sh	LA, D	Pt
Ceratophyllum demersum	4	4	4	4	4	4	4	4	4	4	4	4	4	4		
Myriophyllum verticillatum	4	4	4	4					4	4						
Spirodela polyrhiza	4	4	4	4	4	4	4	4	4	4	4	4				
Ceratophyllum submersum	4		4				4	4		4	4					
Najas marina	4		4													
Hydrodictyon reticulatum	4	4	4	4	4	4	4	4				4				
Lemna gibba	4	4	4	4		4		4								
Azolla filiculoides	4		4									4				
Ruppia maritima							4	4				4				4

5.5 Screening survey databases for reference sites

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

Biological indicators
• Less than 15 per cent of species 'pressure tolerant'. • Highly pressure-sensitive species present. • Highly tolerant specied usually absent (max occurrance 1 in 20). • Richness and cover above 25th percentile of residual of type-specific richness versus area relationship. • Mean cover score per species within global mean interquartile range (5-25%). • No established invasive alien or translocated native species (under five per cent of total cover). • Documented acidophiles (*Juncus bulbosus* and aquatic sphagna) under 30 per cent relative cover (based on 75th percentile of cover of these species in sites where acid deposition is below the critical load). • Relative cover of filamentous algae below 10 per cent (median of five per cent).
Pressure indicators
• Total phosphorus below G/M boundary range, set by the MEI approach (UKTAG, 2006). • No evidence of hydromorphological modification. • Impacted land cover (tilled land, permanent pasture, verges, amenity grassland, urban and suburban) less than 20 per cent of catchment area.

In terms of the biological criteria, the significance of 15 per cent of species being pressure-tolerant is explained in Section 5.2. In effect, this value represents the upper prediction error when tolerant species are completely absent. Screening based on the proportion of species rather than proportion of cover is predicated by the inclusion of historical survey data or records. Theoretically, there is a risk that a site meeting other biological criteria could be flagged as a reference site while having a small relative number, yet a high relative cover of tolerant species. In practice, this scenario is unlikely since the proportion of a species in the species pool is strongly related to the proportion of total cover attributable to that species. In other words, it would be improbable to find one highly tolerant species dominating the cover in a lake with a large number of sensitive species, or one highly sensitive species dominating the cover in a lake with a large number of tolerant species. This is consistent with the discussion on the best approach to derive an LMNI score (Section 4.2.4). In the few cases where

cover versus incidence of tolerant species could be distorted to the extremes suggested, such lakes would be likely to fail on direct pressure screening.

The use of land cover data and population data has been trialled as surrogate pressure measurements when no directly measured values 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, screening of a putative set of reference sites was undertaken based on all other criteria. This showed (Figure 5.5) that 90 per cent of reference sites identified by the other criteria 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. Notably, half the non-reference sites also had impacted land cover less than 20 per cent of catchment area, indicating that screening by land cover alone is not a reliable approach to choose reference sites. This probably reflects partly the spatial resolution of the available data and its relation to the lake riparian zone. For example, Hilli *et al*. (2007) found that the extent of change in agricultural land use within 100-m of a lakeshore had a significant influence on aquatic plant species turnover in Finnish lakes but that changes in a zone 100-400 m from the lake shore had a much weaker effect.

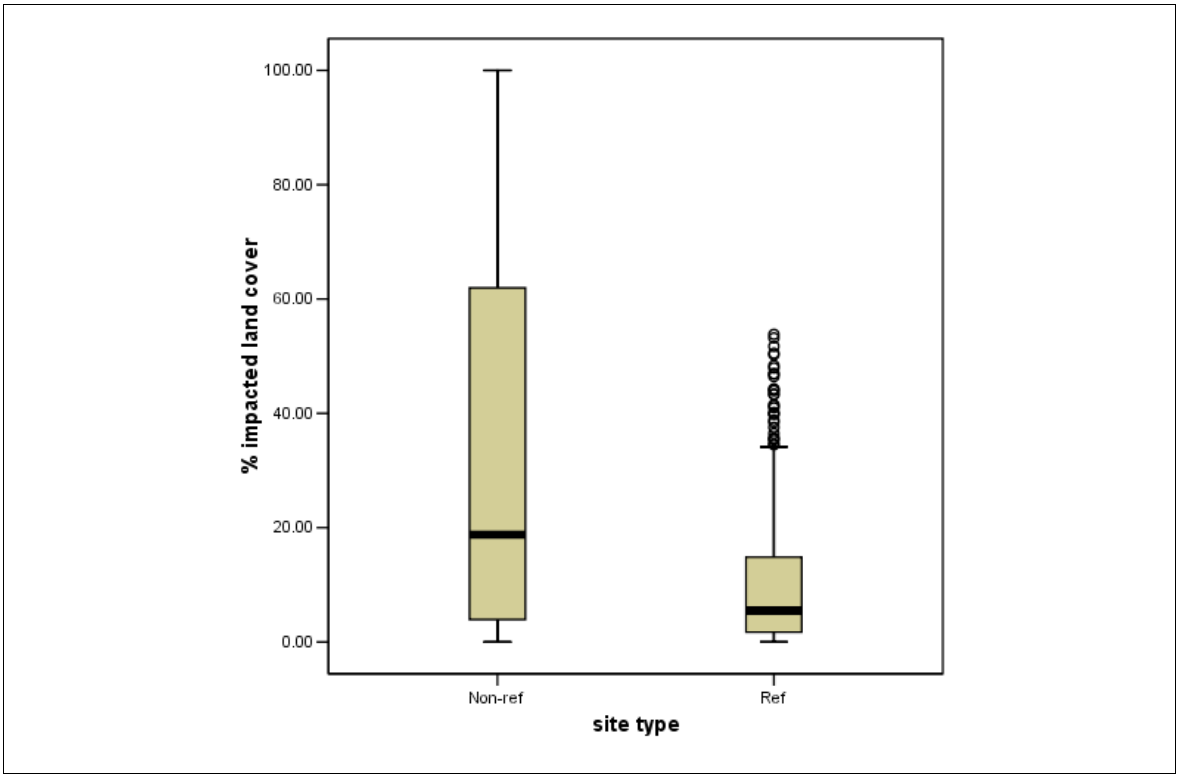


Figure 5.5 Relationship between impacted land cover types in the catchment and reference and non-reference lakes screened on the basis of their biology. Note that biological reference sites generally (90 per cent) have impacted land cover under 30 per cent, but occasionally much higher, whereas half of non-reference sites have impacted land cover under 20 per cent.

6 Predicting the expected flora of lakes at reference condition

6.1 Background

The derivation of ecological status relies on the comparison between the observed value of a metric or set of metrics with the values that would be 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 (Wright, 2000), 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, involves 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 (addition or removal of a site from the reference network requires re-clustering of community structure and calculation of new algorithms to predict reference metric values), and is constrained by the availability of sites which offer reference condition biological assemblages. A second option would be to develop species-specific sub-models (see Barendregt and Bio, 2003) based only on reference site biology, and to use these models to predict the probability of occurrence of each species, 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 provides 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 metric values directly from a set of linked environmental data. Walley and Fontama (1997) affected this step for river macroinvertebrates using a back-propagation neural network and found 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, Maberley *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, following the approach used in the early versions of RIVPACS. Although the option to generate this type of information remains, it was 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 lake types which means that predictions of the flora in certain lakes 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 potentially suitable sites by macrophytes. The majority of species thus have a comparatively low probability of occurrence. Conversely, there is a high risk of failing to find a taxon whose occurrence is expected. Under such circumstances the utility of direct predictions of taxonomic composition seem questionable. As examples, Willby and Eaton (2001) used the MDA approach to predict changes in the vegetation of the Montgomery Canal with increases in boat traffic, while Willby and Birk (in preparation) explored high status plant assemblages of different inter calibration river types *en route* to 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 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 more than 50 per cent probability in comparable lowland river types. Hawkins *et al*. (2000) found that predictions of invertebrate species models improved significantly when they were restricted to species with a probability of capture greater than 0.5. Given that this threshold would exclude all but the commonest and most widely distributed species of macrophytes, comparisons of observed and expected assemblages are of little use.

iii. A further problem is that macrophyte species predicted to occur with the highest probability are invariably common and widespread species distributed over much of the quality gradient with consequently 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 that are 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 to occur 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 unfamiliar with.

6.2 Type-specific classification

6.2.1 Approach

The simplest approach to deriving type-specific reference metric values is to calculate the median value of a metric for the population of reference sites in each lake type. EQRs for all non-reference sites of that type can then be expressed relative to this median value. The difficulty with this approach is that each lake type is treated in isolation when, in reality, even discrete types could be mapped onto a gradient of productivity. Moreover, the populations of some reference sites in a given type are small (less than five), relative to the overall number of sites falling into that type, and are therefore dubiously representative of the true reference condition for that lake type (they are probably situated closer to the H/G boundary than to the middle of reference condition). 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 lake type becomes influenced to some degree by values established by adjacent lake types on a productivity gradient, rather than being defined in isolation. This is consistent with the type-specific screening approach in which LMNI 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 perhaps be seen as a hybrid approach.

The standardisation procedure is accomplished through several steps:

i. Using logistic regression, estimate for each type the site LMNI value that would be associated with seven per cent relative cover of tolerant species. This value is used since it is the median relative cover of tolerant species when a screening threshold of 15 per cent is applied. By using a fixed percentage as the mid-point of 'high' it is possible to model the reference values even if no (or few) reference sites are present.

ii. Regress the modelled reference value against the unstandardised reference LMNI (median LMNI value of the observed population of reference sites after applying the screening criteria from Section 6.5). This step is mainly to correct for any bias in the values generated by the logistic regression models that could have arisen through the inclusion of unusually species-poor, low-cover surveys in the pool of data for each lake type.

iii. Model the values obtained against the morpho-edaphic index (MEI) for the range of lake types (Figure 6.1). This evens out 'observed' reference values from lake types in which the total pool of survey data is small and allows reference LMNI to be predicted for alternative MEI values. The MEI value used here is the MEI derived from the 25th percentile of the distribution of alkalinity (lower) and depth (upper) values within that lake type. The 25th percentile is used because the type-specific screening process will inherently favour sites nearer the end (least productive part) of the range of values that define that lake type and the 25th percentile is thus a more appropriate value to use than the median.

iv. The models derived in Step 3 enable site-specific prediction of reference LMNI given alkalinity and depth data. To allow for circumstances where such data are not jointly available or of sufficient quality (as in some intercalibration exercises), the LMNI values at the middle of 'high' status and the various class boundaries can be calculated for the most typical conditions in each lake type (the MEI that derives from the median of the alkalinity and depth values for all members of that lake type).

Table 6.1 gives the reference, H/G and G/M boundary LMNI values for the various lake types. H/G (0.91) and G/M boundary (0.79) EQR values are based on the modelled LMNI value when all tolerant species account for 15 and 35 per cent respectively of the species present, averaged across all lake types (see Section 7.4.4 for details on boundary setting). The median MEI value for each lake type is also shown. Note that this is the median MEI for surveys of that lake type on which the model has been constructed (lakes with macrophyte surveys and for which depth and alkalinity are available) and not the median MEI of all possible members of that lake type in the UK.

To assess the benefits of standardisation in deriving type-specific reference conditions the distribution of EQRs obtained from unstandardised and standardised approaches were compared. The EQR was calculated for all sites in that type relative to the reference value for the type. Thus, for the LMNI metric the EQR was calculated as:

(Obs LMNI - max LMNI)/(Exp LMNI - max LMNI)

The maximum theoretical LMNI (10) is inserted in this instance simply to reverse the scale so that high LMNI values relative to the maximum achieve a low EQR. This exercise shows that there is a markedly less skewed distribution to EQRs derived by the standardised approach (Figure 6.2b).

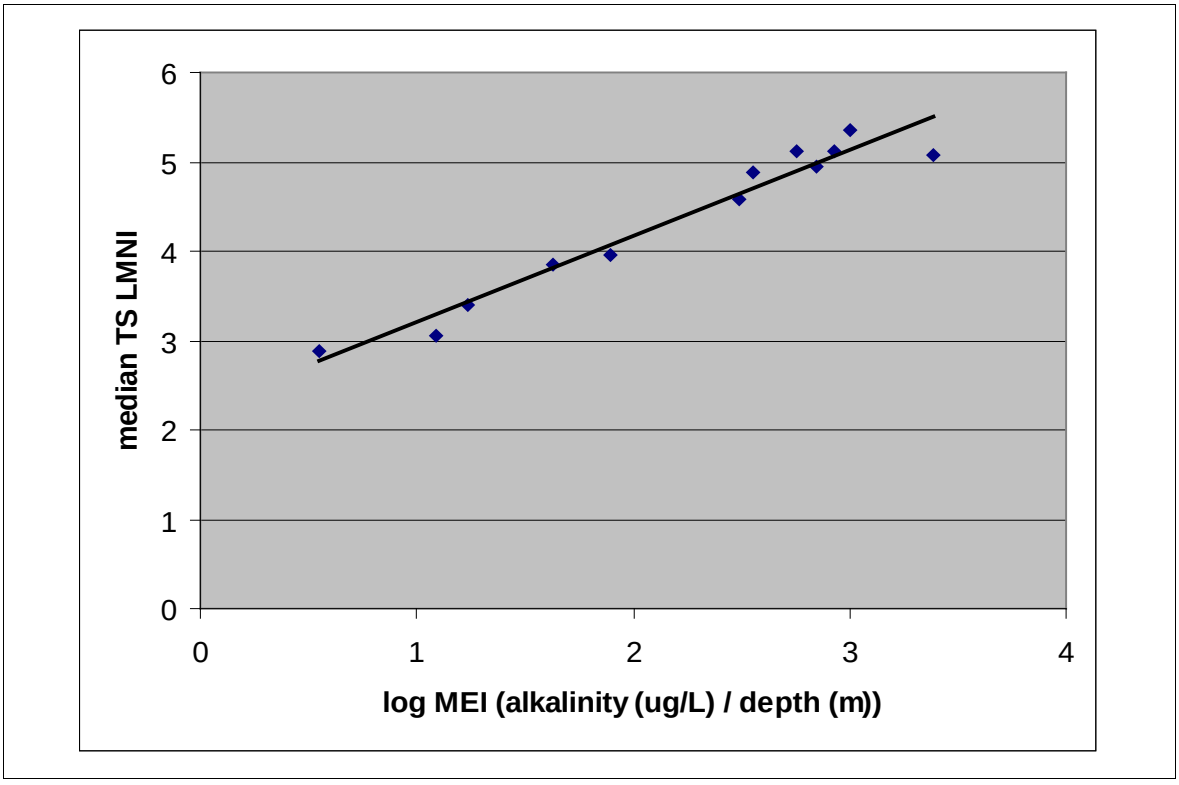


Figure 6.1 Standardisation of type-specific median LMNI values by regression against MEI calculated from lower 25th percentile of alkalinity and upper 25th percentile of depth values for surveys in each lake type

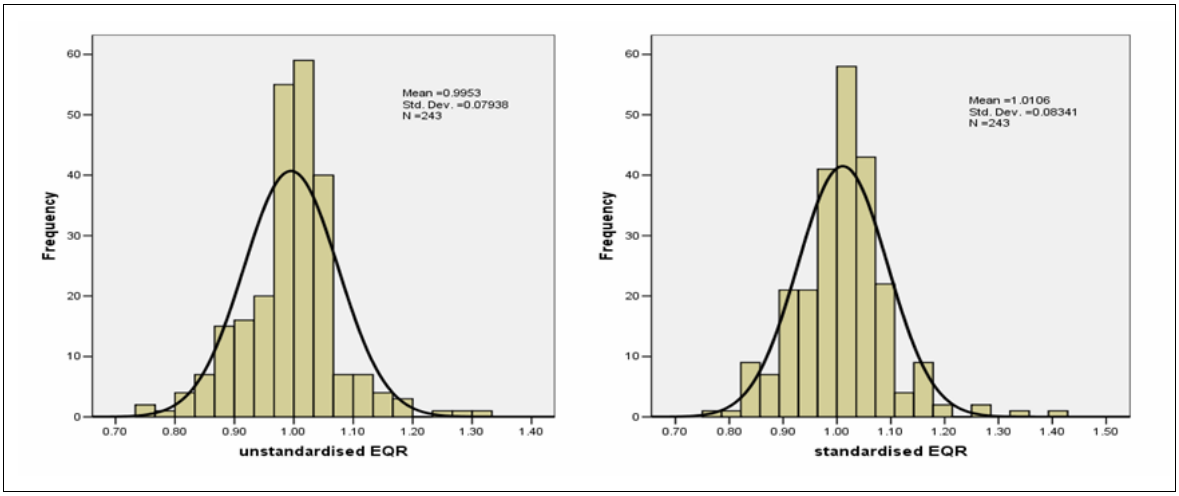


Figure 6.2 Distribution of EQRs from type-specific reference sites pre- (left) and post- (right) standardisation using the steps described above

Table 6.1 Type-specific reference and class boundary values for LMNI metric, as used in intercalibration

Alkalinity	Depth	MEI_{50}	GIG*	Ref LMNI EQR=1	H/G LMNI EQR=0.91	G/M LMNI EQR=0.79
H	VSh	0.896	A/N	5.30	5.72	6.29
H	Sh	0.347	A/N	4.93	5.39	6.00
L	VSh	0.040	A/N	4.10	4.63	5.34
L	Sh	0.017	A/N	3.77	4.33	5.08
L	Deep	0.004	A/N	3.24	3.84	4.66
M	VSh	0.229	A/N	4.77	5.24	5.87
M	Sh	0.079	A/N	4.36	4.87	5.54
Marl	VSh	1.001	A/N	5.34	5.76	6.32
Marl	Sh	0.509	A/N	5.08	5.52	6.11
P	All	0.063	A/N	4.27	4.79	5.47
VH	VSh	1.370	A/N	5.47	5.87	6.42
VH	Sh	0.781	A/N	5.25	5.68	6.25
H	VSh	0.968	C	5.48	5.89	6.43
H	Sh	0.390	C	5.11	5.55	6.14
VH	VSh	2.532	C	5.87	6.24	6.74
VH	Sh	0.662	C	5.33	5.75	6.31

*Geographical Intercalibration Group: A/N = Atlantic + Northern, C = Central

6.3 Site-specific predictions – rationale

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 6.3). 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 are 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.

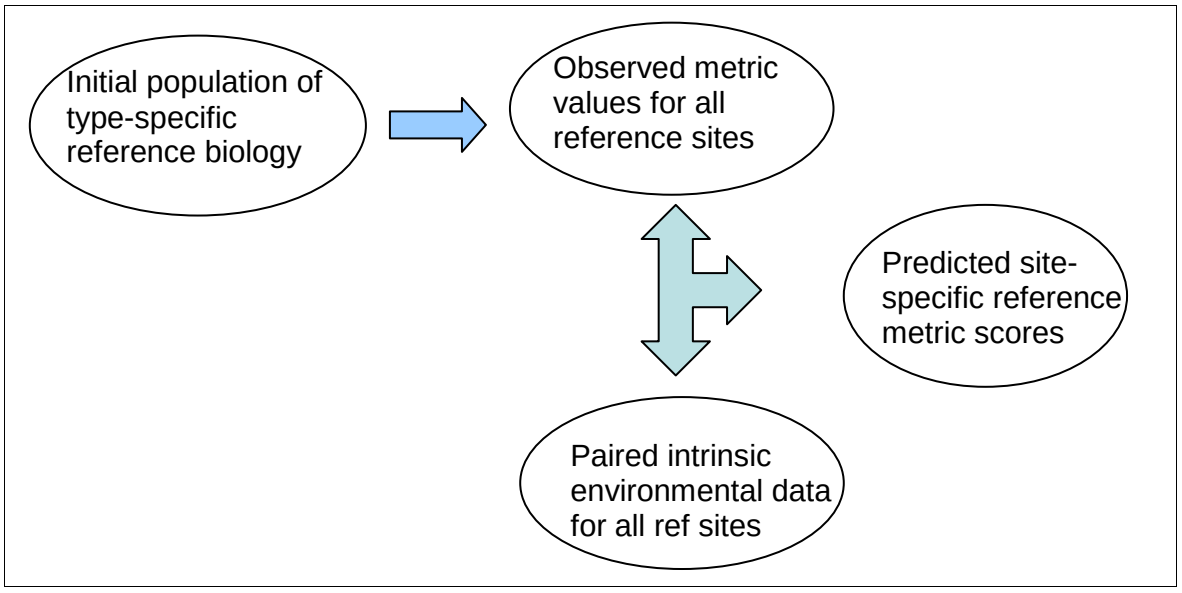


Figure 6.3 Underlying process behind site-specific prediction of reference metric values

The following section explains the development of models of site-specific reference conditions for each of the metrics discussed in Section 4, followed by the calculation of 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 the value of 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 conductivity and alkalinity) rather than face value measurements taken at the time of biological data collection. In reality, the majority of survey data from Scotland and Northern Ireland is linked only to single measurements of key parameters such as alkalinity and conductivity. It is assumed that geographically-based parameters (such as lake area, altitude, distance from coast) 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 will be necessary to revisit these models to adjust the coefficients.

6.4 Compositional metrics

6.4.1 Prediction

LMNI values associated with the population of 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. LMNI was significantly correlated with a range of variables (Table 6.2). Since the scatter in the relationship between LMNI and these variables was generally quite small (Figure 6.4), a stepwise multiple regression procedure was used to identify the most parsimonious models for prediction of LMNI. To accommodate various scenarios of environmental data availability, a number of models were developed. Model performance was assessed based on the percentage variance in LMNI explained by the combination of environmental variables (squared correlation between observed and predicted values), standard error of the prediction, and slope and intercept of the relationship between expected and observed values.

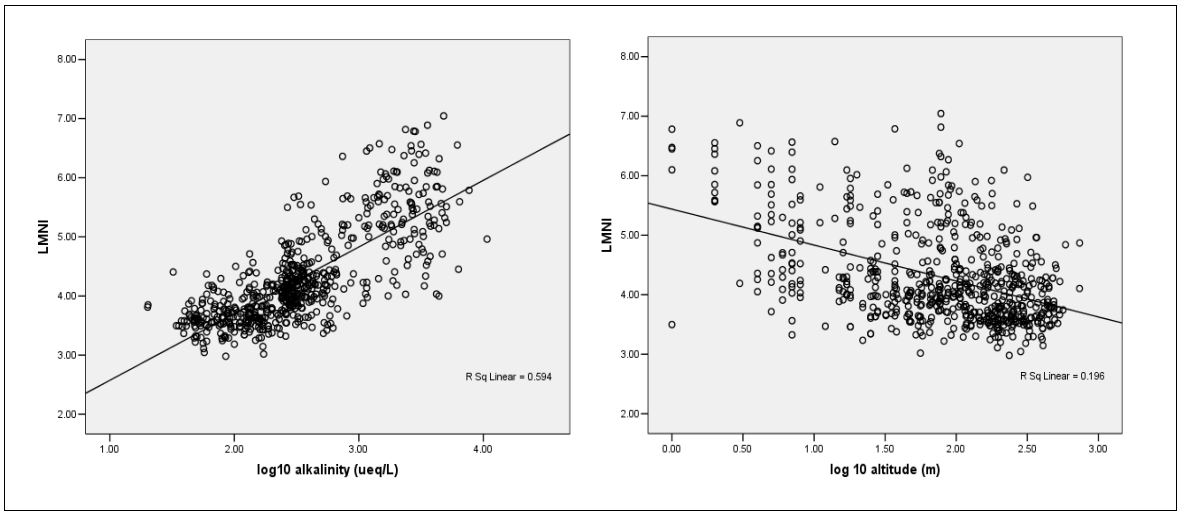


Figure 6.4 Scatter plots of relationship between reference site LMNI and two major intrinsic environmental variables: alkalinity (left) and altitude (right)

The models developed are summarised in Table 6.3. Several redundant models that only operated under circumstances where a superior model would already function were deleted. The first four models (highlighted in grey box in Table 6.4) dealt with the scenarios of environmental data availability pertaining to 94 per cent of reference sites and required between six and nine predictive variables. In terms of the global dataset (reference and non-reference sites) these four models could be used to predict LMNI at 80 per cent of sites, with the optimal model (Model 2; Figure 6.5a) being applicable in 50 per cent of all cases. These models performed strongly, explaining some 72 per cent of the variation in reference LMNI and predicting LMNI to ± 0.38. In all cases alkalinity was the predictor first selected, accounting single-handedly for 60 per cent of the variation in LMNI. A variety of other contextual variables, such as geology (mostly FSC), depth, area, altitude and distance to nearest coast, accounted for the remainder of the explained variation. In the second best model, a combination of geology and geographical location (as northing and easting) compensate for lack of information on conductivity and depth whose terms characterise the optimal model. The value of these secondary variables from a classification perspective is to reduce the uncertainty in reference values from ± 0.51 (using alkalinity as the only predictor) to ± 0.36 (using up

to a further seven predictors). Models which lacked data for both depth and geology (last four models in Table 6.4) tended to perform relatively poorly.

Table 6.2 Pearson correlations between LMNI and transformed intrinsic environmental variables within the 680 sample reference site network

Variable	units	Pearson Correlation	Sig. (2-tailed)	N
Alkalinity(lg_{10})	ueq/L	0.771	<0.001	634
pH		0.647	<0.001	589
Conductivity (lg_{10})	µS/cm	0.643	<0.001	589
weighted Freshwater Sensitivity Class		0.572	<0.001	474
Freshwater Sensitivity Class (FSC) 5	%	0.513	<0.001	474
Altitude(lg_{10})	m	-0.443	<0.001	658
Maximum depth (lg_{10})	m	-0.417	<0.001	533
Mean depth (lg_{10})	m	-0.407	<0.001	553
FWSC1	%	-0.332	<0.001	474
Northing (lg_{10})		-0.310	<0.001	658
Solid calcareous geology	%	0.263	<0.001	526
FWSC4		0.259	<0.001	474
Solid silicieous geology	%	-0.240	<0.001	527
Easting (lg_{10})		0.218	<0.001	658
Glacial Sand & Gravel	%	0.211	<0.001	532
FWSC3	%	0.168	<0.001	474
Area (lg_{10})	ha	0.166	<0.001	658
Alluvium	%	0.165	<0.001	532
FWSC2	%	-0.162	<0.001	474
Mean wave fetch (lg_{10})	m	0.157	<0.001	626
Boulder Clay & Morainic drift	%	0.152	<0.001	532
Crag	%	0.151	<0.001	532
Lake perimeter (log_{10})	m	0.146	<0.001	626
Distance to nearest coast (lg_{10})	m	-0.101	0.012	625
Retention time (lg_{10})	yrs	-0.042	0.333	531
Shoreline Development Index (lg_{10})		0.033	0.407	626
Peat (%)	%	-0.018	0.670	532

Combined use of these models to the reference set (Figure 6.5b) explained an overall average of 71per cent of the variation in LMNI and predicted LMNI to ± 0.43. A model based purely on the typing variables (alkalinity, mean depth plus the geographical descriptor ‘CGIG’ supplied as a dummy variable) was applicable to 528 of the 680 reference sites. This model extracted 70 per cent of the variation in LMNI (with a slightly poorer prediction error of ± 0.47). This shows that the variables used to construct the typology successfully extracted the majority of explainable variation in LMNI.

Table 6.3 Comparison of correlations, slope, intercept and prediction error of regressions relating predicted values of LMNI at reference sites to observed values

Model	Correlation	Slope	Intercept	Error
Model 2	0.868	0.972	0.113	0.364
Model 6	0.860	0.936	0.251	0.378
Model 5	0.857	0.962	0.160	0.411
Model 1	0.853	1.024	0.076	0.422

Table 6.4 indicates the slope and intercepts of the regressions relating predicted values of LMNI to their observed values. Ideally, to minimise bias, these should be one and zero respectively. Consequently there is a small bias in all models. Typically this amounts to an underprediction of expected LMNI under the naturally most productive conditions of about 0.2 (when the observed LMNI is seven, it is predicted as 6.8). Consequently, there is a small bias against very high alkalinity lakes, whereby the EQR of a reference site will be predicted (EQR = 0.94-0.96) to lie towards the H/G boundary (EQR = 0.91), rather than at one. While this is not desirable (and could if necessary be rectified by weighted least-squares regression) it can perhaps help reflect the fact that most of the population of reference lakes in more productive catchments are positioned towards the lower end of high status rather than in the middle of the band.

LMNI is strongly dependent on alkalinity. Strong model performance is therefore effectively guaranteed when alkalinity is used as a predictor of LMNI 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.

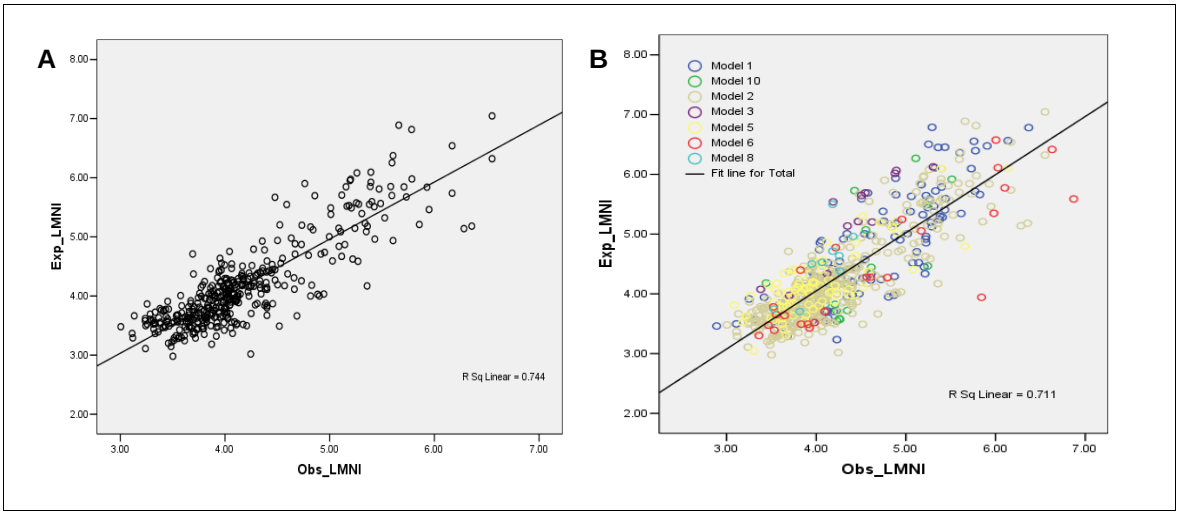


Figure 6.5 a) Application of optimal model (Model 2) to predict LMNI in reference dataset; b) Application of models to cover a range of scenarios of environmental data availability. The aggregate relationship predicts reference LMNI to ± 0.43.

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

Variable / Model name		2	6	5	1	8	3	7	9
Constant		1.605095	8.758946	-12.3919	0.979379	2.582044	1.187018	2.122879	17.09445
Alkalinity (log_{10})	ueq/L	0.635498	0.74562	0.662752	0.698962	0.945879	0.811123	1.008576	
Mean depth (log_{10})	m	-0.53082			-0.70975				
Conductivity (log_{10})	uS/cm	0.440744		0.549979	0.756869		0.552126		
Area (log_{10})	ha	0.17256	0.154732		0.230351				0.219083
Altitude	m			0.000945	0.000948	0.000736	0.001212	0.000948	
Weighted FSC		0.113708	0.183715						0.35113
Distance to coast	m	9.41E-06				1.53E-02			
FSC2	%	-0.00193							
Area	ha			0.001819		0.002436	0.002062	0.001657	
Altitude (log_{10})	m		-0.16426	-0.31997		-0.39754	-0.18306	-0.29756	-0.37414
Northing (log_{10})				2.737386					
Easting			3.81E-06	2.54E-06					6.18E-06
Northing			-9.9E-07	-3E-06					-4.7E-07
FSC1	%		0.002702						0.003865
Easting (log_{10})			-1.26892						-2.68287
Shoreline Devt Index					-0.13164				
Calcareous solid geology	%	0.00196							0.00245
R		0.858	0.847	0.850	0.845	0.795	0.793	0.772	0.760
adj R^2		0.732	0.711	0.718	0.711	0.627	0.625	0.593	0.570
SE Estimate		0.364	0.378	0.411	0.422	0.465	0.474	0.493	0.500
n reference sites in model		415	450	578	474	593	581	626	474
% of surveys applicable		49.7	8.6	3.2	16.6	1.1	14.0	0.2	6.5

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 LMNI explained by LMNI predicted from environmental data. Models are ranked from left to right in decreasing order of desirability (decreasing model strength and increasing prediction error). The percentage of surveys applicable refers to the total number of surveys in the global dataset (n=4642) which contained adequate environmental data for the best available model to be applied. Reference LMNI at 80 per cent of surveyed sites could be predicted using the first four models.

6.4.2 Type- or site-specific predictions?

Once site-specific models are derived it is highly unlikely that one would return to a type-specific approach, unless insufficient environmental data were available for site-specific models to be used, or if a type-specific approach was necessary for intercalibration purposes. This section assesses the advantages of site- over type-specific predictions of LMNI.

The ability to predict observed LMNI values in the population of reference sites using adjusted type-specific median LMNI values, or the site-specific models described above, is compared in Figure 6.6. The initial impression from these analyses is that site-specific models offer few advantages over type-specific models in this dataset. Thus, the observed LMNI values in the reference set can be predicted slightly better via the type-specific route. To some extent, this probably reflects the fine grained nature of the typology with which we are working, and the fact that it is closely aligned to the key variables that structure lake macrophyte communities; the finer the resolution of the typology, the more closely it resembles a site-specific approach and the greater the ratio of between- to within-class variation. Secondly, the extent of post-screening harmonisation of the type-specific reference values is quite sophisticated and already delivers some of the benefits of a site-specific approach through the use of a single model. A slight weakness, however, is apparent in the negatively skewed distribution of observed/expected (O/E) values derived by a type-specific approach.

A major advantage of site-specific predictions is their ability to bridge discontinuities associated with a type-based screening of reference sites. However, this advantage is unlikely to be apparent while the reference site set is composed only of sites that were themselves derived initially from a type-specific screening. Thus, if the analysis is extended to cover all high status sites (those derived from type-specific screening plus those revealed by subsequent analysis) the advantages of a site-specific approach become readily apparent. It is clear from Figure 6.7 that a type-specific approach grossly underpredicts the expected values for high status sites compared with a site-specific approach (slight underprediction is to be expected since the population includes both reference and non-reference high status sites). 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.

It is unlikely that one would attempt type-specific predictions of other metrics, such as richness, especially given their expected dependence on spatial attributes such as lake area or altitude that do not feature in the core typology. Consequently predictions of other metrics at a type-specific level are not considered further.

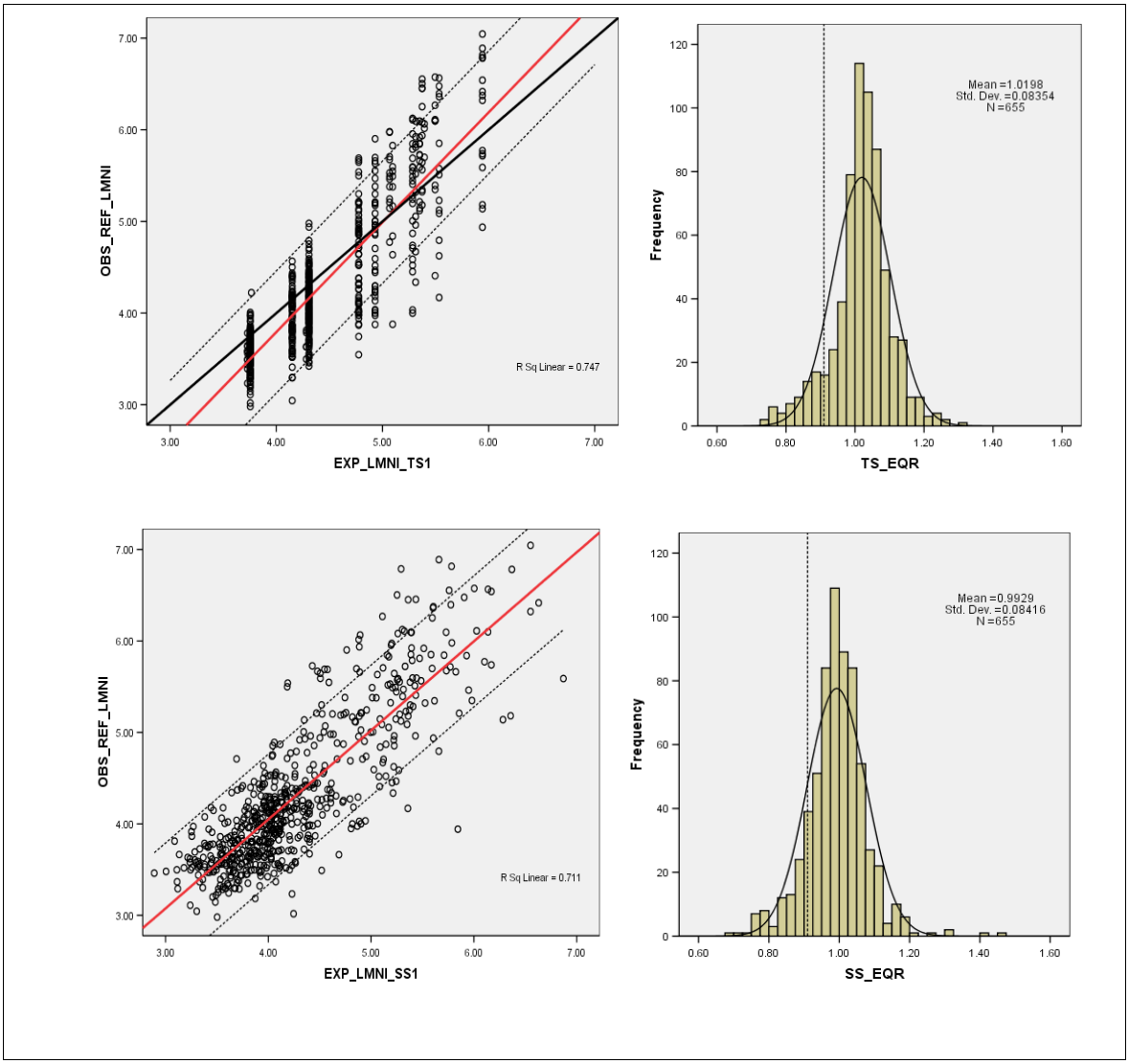


Figure 6.6 Comparison of type-specific (TS) (upper panels) and site-specific (SS) (lower panels) approaches to predict LMNI values in a population of 655 reference sites. Although the type-specific approach slightly outperforms the site-specific approach at this level, it also slightly overpredicts reference LMNI over much of the range of reference values and therefore creates a small negative skew in the distribution of reference EQRs. Red = fitted line, black = one:one relationship. Lines are overlain in bottom left panel.

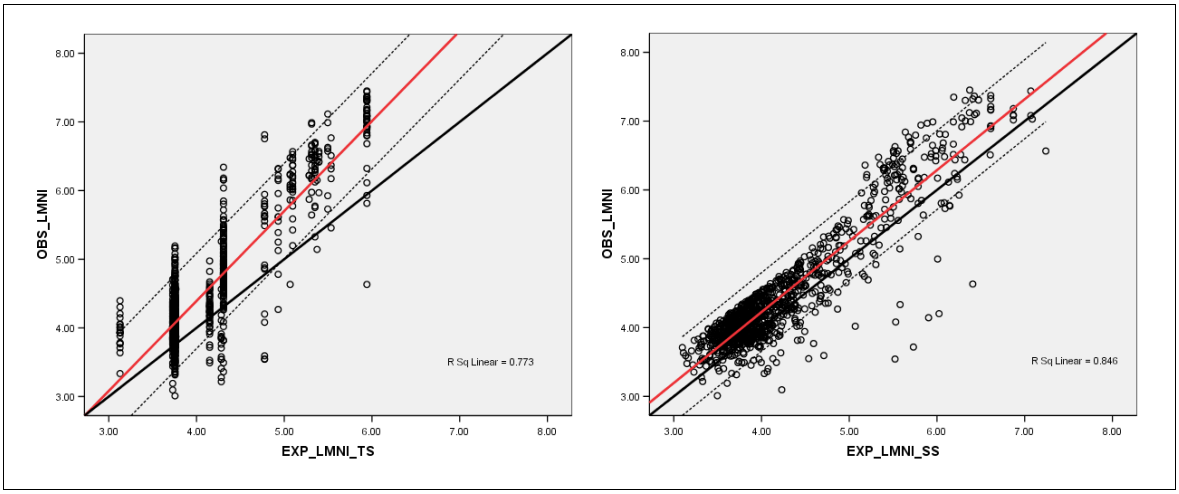


Figure 6.7 Comparison of type-specific (TS) (left) and site-specific (SS) (right) approaches to predict LMNI values in a population of high status sites. Slight underprediction (more precautionarity) is to be expected in both models (one:one black line below fitted red line) because sites with EQR down to 0.91 (lower limit of high status) are included. The type-specific approach significantly underpredicts 'reference' LMNI, with this difference increasing across the LMNI gradient compared to the site-specific approach. A type-specific approach would therefore be especially precautionary if applied to naturally higher productivity sites.

6.4.3 EQR calculation

The LMNI EQR is calculated as

$EQR = (O_1 - E_0)/(E_1 - E_0)$

O_1 = Observed site score

E_0 = Maximum (most degraded) score on scale

E_1 = Expected score under reference conditions (median type-specific reference site score)

In this case $E_0 = 10$ 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. Subtracting the theoretical maximum (worst) LMNI site score of 10 ensures that low LMNI scores achieve a high EQR. Therefore if the observed LMNI value for a site is 6.0 and the value expected at reference conditions, as predicted by a site-specific model is 5.0, the EQR is

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

6.4.4 Placement of class boundaries for LMNI

The process followed to establish class boundaries for LMNI is simply an extension of the conceptual framework used to define biological reference conditions in Section 5. For each lake, and using logistic regression, the LMNI 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 tolerant and sensitive responders, a position that equates to the middle of 'moderate' status. The prediction error between the relative proportions of

tolerant species and LMNI 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 G/M and M/P boundaries. Essentially, at these thresholds there is a low or high probability that the cover of tolerant species will exceed that of sensitive species. To translate values into an EQR the LMNI value associated with each of these thresholds was compared to the modelled LMNI 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 H/G boundary lies at 15 per cent. In several cases the reference LMNI value was derived by modelling because no sites could be found that met the required standard for cover of tolerant taxa in reference sites.

Table 6.5 summarises the results of the logistic regressions as EQRs where the boundaries for each type correspond to the threshold proportions of tolerant species defined above. The EQR was defined as in Section 6.4.3. The highlighted figures represent the average EQR across all types at a given boundary, weighted by the number of sites in each type. When comparing EQR values for different types, because the metric operates over a constrained scale and increases with increasing alkalinity the window of potential metric scores is inevitably much lower in naturally more productive lake types. This results in a tendency to lower EQR values relative to the same point in more fertile lakes. The consequence of a having fixed EQR at the P/B boundary, for example, is that metric scores in base-poor lakes will therefore rarely, if ever, get high enough for sites to be classified as bad on the basis of species composition. This will apply to any metric that operates on a constrained scale and where the metric is intrinsically linked to the typing factors or predictive environmental variables. It is doubtful if, for example, a low-alkalinity deep lake could ever achieve an LMNI of 10 even under the most heavily degraded conditions (without being transformed into another lake type) and one solution may be to scale the maximum (worst) LMNI value used in the EQR calculation relative to the expected value. However, few if any low-alkalinity lakes in Europe could be assigned to genuinely bad status on the basis of nutrient impacts.

Table 6.5 EQR values at class boundaries associated with different lake types based on logistic regression models of the proportion of tolerant species

Alkalinity	Depth	Region	n	H/G	G/M	M/P	P/B
H	VSh		185	0.91	0.79	0.65	0.49
H	Sh		198	0.92	0.81	0.69	0.54
L	VSh		278	0.93	0.83	0.72	0.59
L	Sh		1238	0.93	0.84	0.74	0.62
L	Deep		84	0.94	0.85	0.76	0.64
M	VSh		306	0.91	0.78	0.65	0.49
M	Sh		687	0.92	0.81	0.69	0.55
Marl	VSh		100	0.92	0.81	0.68	0.53
Marl	Sh		103	0.92	0.82	0.71	0.57
P	All		201	0.91	0.79	0.66	0.51
VH	VSh		74	0.87	0.69	0.5	0.27
VH	Sh		72	0.89	0.75	0.6	0.41
H	VSh	C	101	0.91	0.78	0.65	0.48
H	Sh	C	85	0.9	0.78	0.64	0.47
VH	VSh	C	677	0.9	0.78	0.64	0.47
VH	Sh	C	76	0.9	0.77	0.63	0.45
				0.91	0.79	0.67	0.55

On the basis of the above analysis, class boundaries for the LMNI metric are taken as 0.91, 0.79, 0.67 and 0.55. 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). By examining the distribution of site-specific EQR values for LMNI (Figure 6.8) one can put our approach into the context of EQR percentiles. The fifth and tenth percentiles of this distribution correspond to 0.85 and 0.9 respectively. An EQR of 0.91 corresponds to the 12th percentile and thus represents a relatively precautionary position.

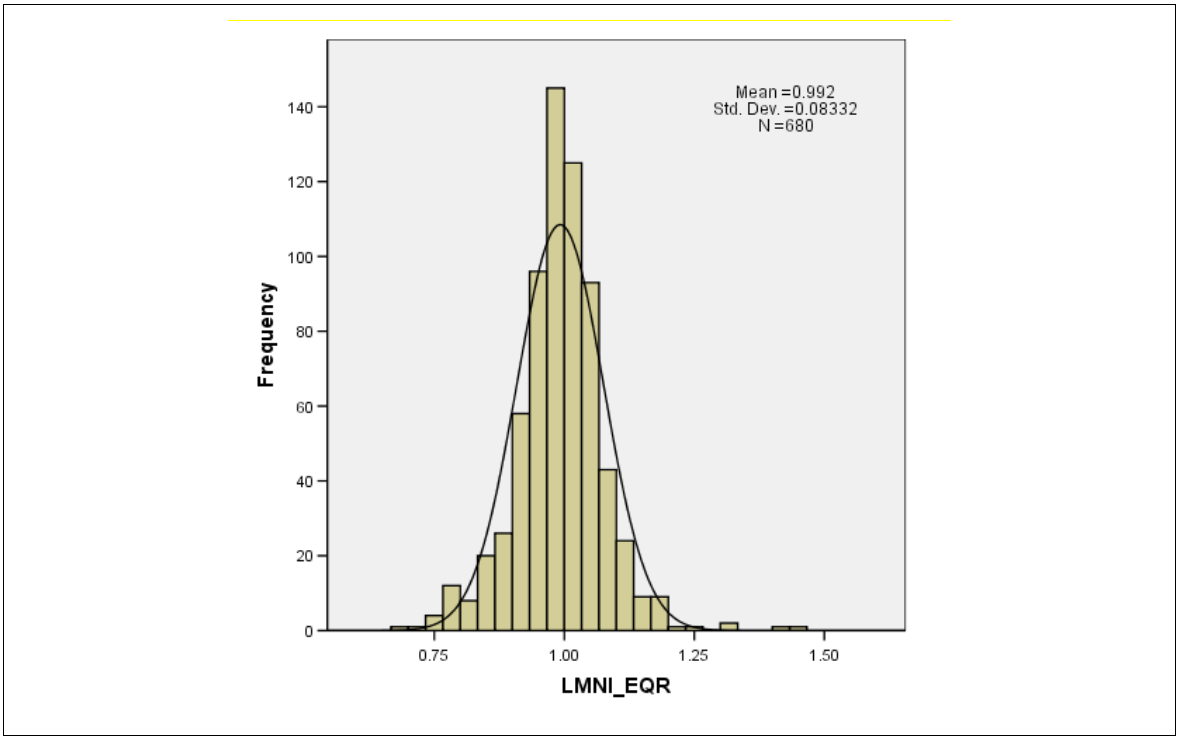


Figure 6.8 Distribution of site-specific reference EQR values for the metric LMNI. The fifth and tenth percentiles lie at 0.85 and 0.9 respectively.

6.5 Richness metrics

6.5.1 Prediction

The reference site database used in the modelling of richness metrics is an expanded version (n=782) of that used for LMNI due to the incorporation of an additional pool of relatively taxa- or FG-poor sites (minimum of two taxa or FG) for which there was no evidence of impact. It is reasonable to suppose that richness at these sites does not vary systematically as a result of anthropogenic pressures, although it is possible that some low level hydro-morphological impacts remain.

Both richness metrics were modelled by stepwise linear regression from the population of reference condition surveys. Lake area (ha), altitude (m), and alkalinity (ueq/l) were incorporated as initial model terms due to their links with richness. Square and cube root terms for these variables were also incorporated to facilitate curvilinear models if appropriate. Initial tests indicated that lake perimeter was slightly superior to lake area as a predictor but this was rejected to maximise the application of the model to datasets for which perimeter was not available. Virola *et al*. (1999) also concluded that lake perimeter rather than area was a more pertinent measure of habitat availability for

aquatic plants. In tests covering a wide variety of other environmental predictors, no other variable could explain a statistically significant amount of variation in N_TAXA. The models developed were tested to ensure that predictions of negative N_TAXA or N_FG are not possible when model terms with very large values (such as derivatives of area) and negative coefficients occur. However, these models are unlikely to be applicable to highly brackish sites where surveys are associated with spuriously high alkalinity data (above 20 meq/l).

Because N_TAXA and N_FG are unconstrained or vary over a much wider range than metrics such as LMNI, a small number of reference sites may 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, since the distribution of 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 (Figure 6.9), while only marginally improving model performance (r^2 = 26 per cent in both cases), 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 therefore as detailed in Table 6.6. Although a rationale exists for the use of richness metrics, in common with other biota there are difficulties in predicting richness which suggest fundamental problems with prediction methods or supporting datasets. For example, within RIVPACS the prediction of numbers of families is substantially poorer than for ASPT, leading to correspondingly wider class boundaries for this metric (Moss *et al*., 1999). Walley and 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. In our analysis there is little evidence that using a substitute metric, such as N_FG, which should be less sensitive to variation in survey effort, actually reduces prediction error though it does reduce model bias. Model bias can be assessed easily by comparing the intercept and slope of the relationship between observed and predicted values which should be close to zero and one in the absence of bias. Figure 6.9 and Table 6.7 confirm that the slope is close to one in both models, but it is clear that the positive intercept in the case of N_TAXA indicates that the best model will tend to underestimate richness. In this respect N_FG is therefore a slightly superior metric.

Table 6.6 Models for predicting N_TAXA and N_FG at reference sites

	Log_{10} (N_TAXA+1)		**Log_{10} (N_FG+1)**	
Term	**coefficient**	**order**	**coefficient**	**order**
Constant	0.554544		0.256635	
Alkalinity	-0.00028	6	-0.0002	3
Log_{10} (Alkalinity)	0.270098	1	0.265515	2
Alkalinity ^2	4.21E-08	8	2.95E-08	4
Alkalinity ^3	-2.1E-12	4	-1.4E-12	5
Altitude			-0.00029	1
Log_{10} (Altitude)	-0.09232	5		
Altitude ^2	7.46E-07	7		
Altitude ^3	-1.4E-09	3		
Log_{10} (Area)	0.060514	2	0.032559	6

All model terms are significant at p = 0.001. Alkalinity as µeq/l and area as ha. Forty (40) is added to all alkalinities to eliminate negative values prior to transformation. Order refers to the order of entry to a stepwise model.

Future prospects to improve the prediction of richness metrics include incorporating additional contextual information (such as lake colour) or information on survey method (boat versus shoreline, whole-lake versus subsample, number of personnel and so on) and survey timing, all of which are likely to contribute to unexplained variation in richness in reference sites. Based on the Northern GIG common dataset, it was possible to model both richness metrics almost twice as effectively (r^2 = 45-50%) in a diverse population of 450 reference lakes using the same explanatory variables (Figure 6.10). This is even more remarkable given that this database was assembled from data from six different countries whose survey methods vary more between countries than within the UK.

Table 6.7 Comparison of slope, intercept and prediction error of regressions relating predicted values of richness at reference sites to observed values

metric	slope	intercept	prediction error
N_TAXA	1.0018	0.16122992	0.164
N_FG	1.0000	-1.4877E-14	0.148

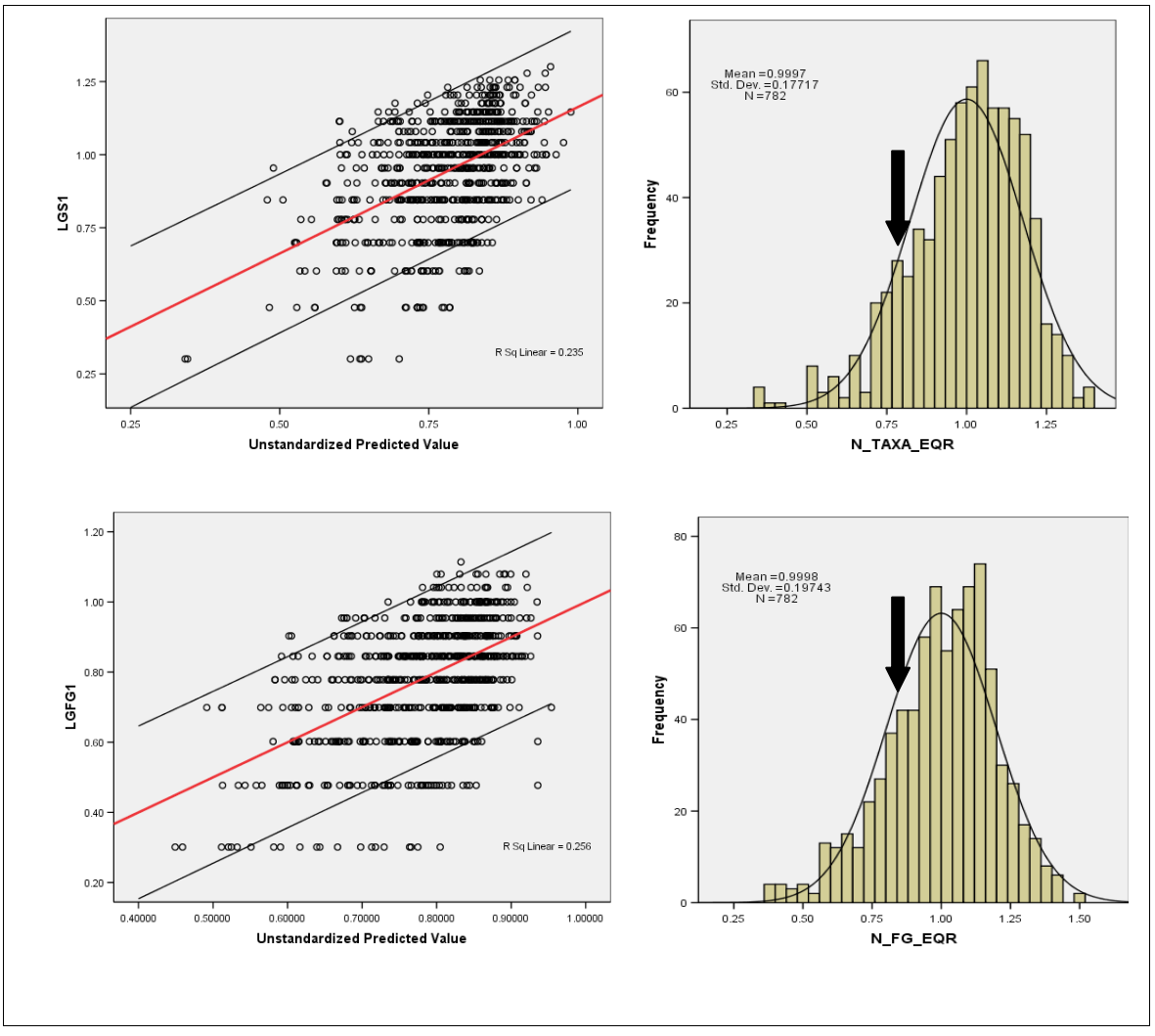


Figure 6.9 a) Predicted versus observed values for log transformed N_TAXA; b) Frequency distribution of EQRs for N_TAXA; c) Predicted versus observed values for log transformed N_FG; d) Frequency distribution of EQRs for N_FG. Arrows in c) and d) indicate 25th percentile of distribution of EQRs used to set H/G boundary. Note lower skewness in distribution of EQRs for N_FG

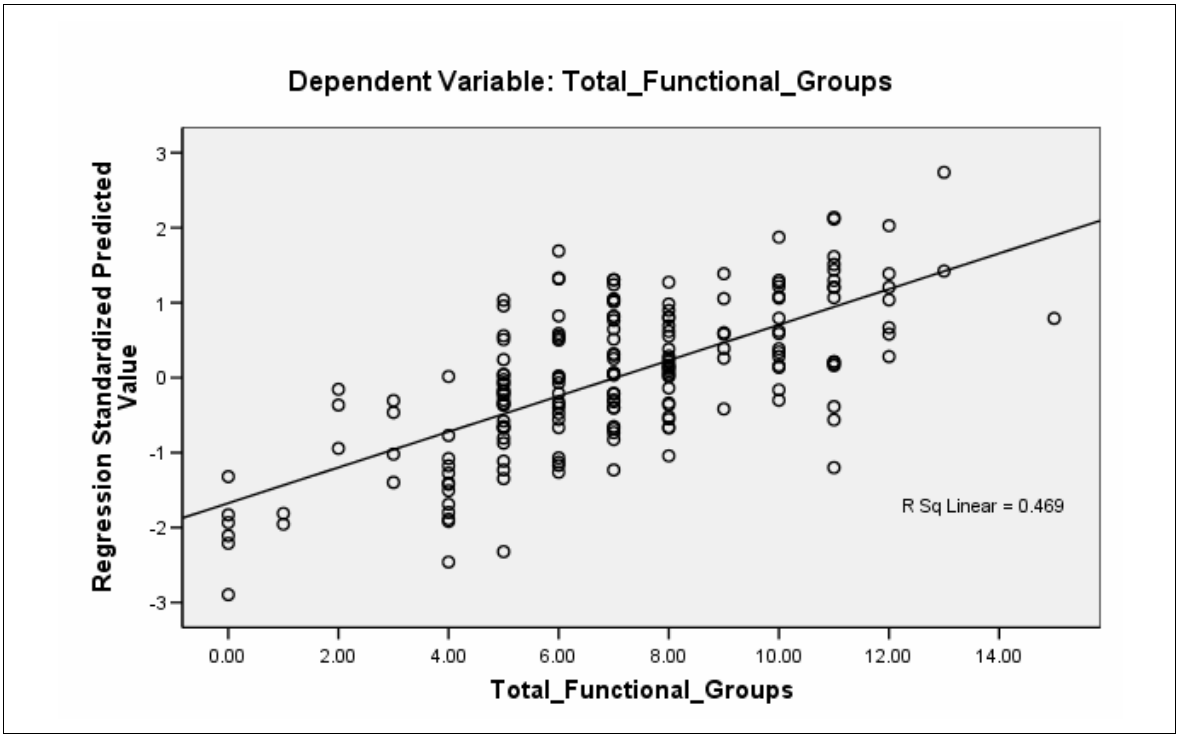


Figure 6.10 Model for predicting N_FG in reference lakes in the Northern GIG common dataset based on alkalinity, lake area and altitude

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 shown 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. On the other hand, when survey quality is high and N_FG is low relative to N_TAXA the higher metric EQR should be used. 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 would potentially act to stabilise against the loss of functional groups.

6.5.3 EQR calculation

Models for both richness metrics were generated from $\log_{10}$ x+1 transformed values and EQR is 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 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 terms of $\log_{10}$ (x + 1)) 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 by calculating the exponent of the value predicted from environmental variables in Table 6.6 and then subtracting one.

6.5.4 Placement of class boundaries for richness metrics

The high-good boundary for both metrics based on the 25th percentile of the EQR frequency distribution was 0.88 (Figure 6.9). The rationale for the placement of class boundaries for this metric is statistical rather than ecological. Thus, boundaries are imposed using the percentile distribution of EQRs in reference sites (0.88 for H/G with subsequent boundaries at 0.12 intervals). Class boundaries or the weight given to this metric must reflect the underlying variability in the reference site model and there are many factors unaccounted for (such as lake isolation, size of catchment species pool) that contribute to variation in the metrics. Indeed, there is no clear evidence that the practical arguments in favour of N_FG as a metric lead to superior prediction. Translating the EQR thresholds proposed, sites at the H/G boundary will support on average 75 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 the stability of macrophyte populations or macrophyte-dependent functions will be impaired if N_FG or N_TAXA falls below these thresholds. 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 are present) it is possible that a site has been undersampled and the quality of the survey or surveyors should be scrutinised.

6.6 Cover

6.6.1 Prediction

Average percentage cover per species was calculated for all putative reference sites which had been subject to the described screening process. Despite the use of a wide range of predictive environmental variables and modelling approaches, it proved impossible to predict expected mean cover at reference sites. The strongest relationships between percentage cover and environmental variables explained only two to three per cent of the variability in percentage cover and were non-significant or only narrowly significant (p = 0.01-0.05). Data transformation failed to improve these relationships. Consequently, expected abundance is based on the global median average percentage cover value across all reference sites (8.5 per cent). This value can be translated for a site supporting 15 taxa as being equivalent to five rare taxa, six occasional taxa, two frequent taxa, one abundant taxon and one dominant taxon.

6.6.2 EQR calculation

When the frequency distribution of EQRs for abundance was examined, a distribution based on log values ($\log_{10}$ (Obs mean % cover + 1)/$\log_{10}$ (8.5 + 1)) was found to

produce a significantly less skewed distribution and for this reason an EQR based on log cover was used. This also has the advantage of reducing the influence of large cover values.

Therefore , if the mean cover per species is seven per cent, based on transforming DAFOR cover values as indicated in Section 4.4.2, cover EQR would be calculated as

$$EQR = \log_{10} (7+1)/\log_{10} (8.5+1) = 0.92$$

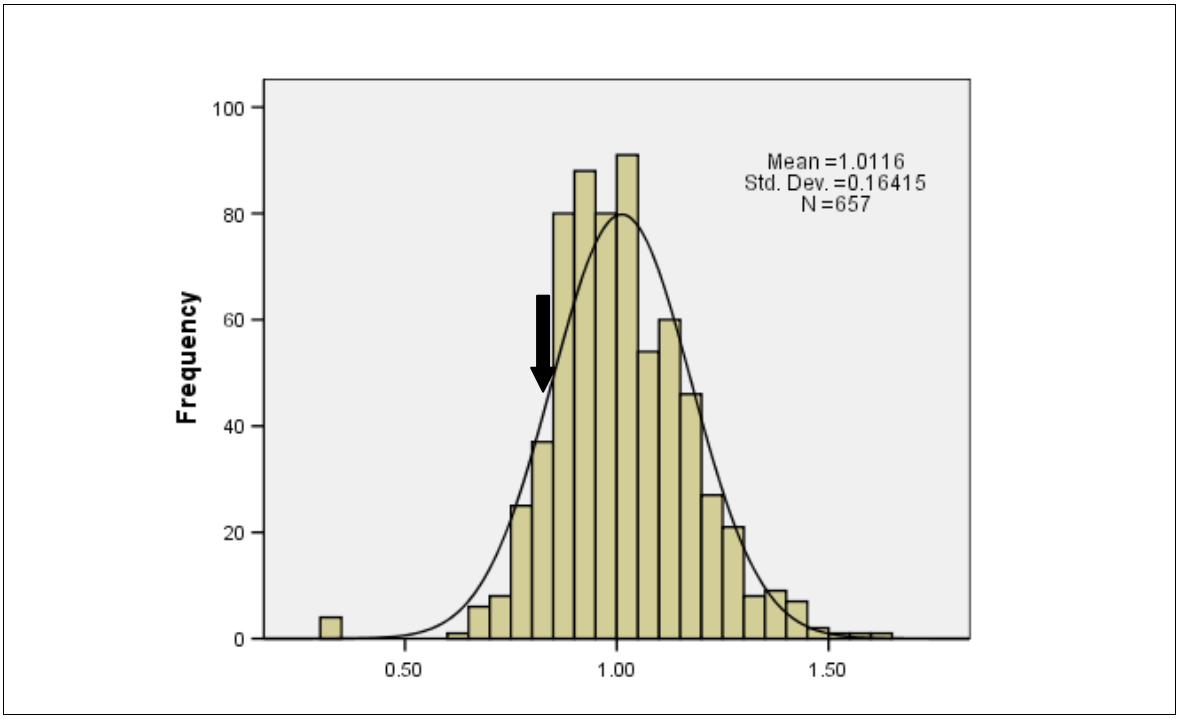


Figure 6.11 Frequency distribution of EQRs for COV. Arrow indicates 10th percentile of distribution of EQRs used to set H/G boundary (0.85).

6.6.3 Placement of class boundaries

The 10th percentile of the distribution of cover values (0.85, Figure 6.11) was employed to set the H/G boundary with subsequent class boundaries placed at 0.15 intervals. At these values the H/G boundary is equivalent to a site supporting on average 68 per cent of its expected mean cover and 46 per cent at the G/M boundary. In the example in Section 6.6.1 above, if a site supports 15 species and has a mean cover per species that is half that expected under reference conditions, the implication is that cover is strongly distorted towards the lower cover classes. To achieve a mean cover per species this low would require a distribution of cover scores such as nine rare taxa, five occasional taxa, one frequent taxon and one abundant taxon. A logical argument is that when a site supports less than half the expected mean macrophyte cover per taxa, a higher proportion of the bed must then be unvegetated (because there can be no dominant taxa) and the site can therefore no longer be said to be delivering the normal suite of macrophyte-dependent functions with the same efficiency.

EQR values in excess of one are disregarded for this metric in the final phase when metrics are combined. It may be of local value to record that the aquatic vegetation at a site is unexpectedly abundant relative to the norm (more species than would be expected have very high cover values). However, it is assumed that when this scenario arises the number of taxa present will have already been reduced by dominance

effects, or composition will have shifted to species with traits (such as dense canopy growth) that permit dominance of large areas of bed and water column.

6.7 Non-native species

6.7.1 Prediction

The median cover of these species in reference sites was zero. By definition zero is the expected value for this metric (non-native species are absent).

6.7.2 EQR calculation

The EQR for this metric can be calculated simply as one minus the proportion of non-native species. Therefore if the metric value is 0.75 (non-native species constitute 75 per cent of the overall vegetation) the EQR is 0.25.

6.7.3 Placement of class boundaries

A threshold of five per cent relative cover of invasive alien or translocated native species was arbitrarily set as the upper limit for these species in reference sites (H/G boundary). Below this level, populations of invasive species cannot be considered to be established and their ecological influence is probably marginal. The latest generation of invasive taxa (*Crassula helmsii*, *Myriophyllum aquaticum* and *Hydrocotyle ranunculoides*) were absent from all reference and high status sites, although no screening was undertaken to ensure absence of these high risk species. This may partly reflect the age of most survey data and distribution of survey sites, plus a tendency for the above species to occur predominantly in shallow ponds, ditches and canals, rather than in lakes.

A threshold of 25 per cent relative cover of invasive species was set as the G/M boundary. The reasoning was that at this level native species are still overwhelmingly dominant, and that if invasive species are leading to community distortion this will be reflected in low N_FG or N_TAXA. If invasive species are a response to nutrient enrichment, this should be reflected in distorted LMNI EQR values.

After inclusion of this metric, it was found that the metric INV had the lowest EQR of six metrics in just 121 out of 4,538 surveys (2.7 per cent). Following an initial approach to combining metrics implemented prior to intercalibration, it was found that of these 121 surveys INV resulted in a final class change from high or good to moderate or worse in only 19 cases (16 per cent). These were all due to high relative cover of *Elodea* spp and/or *Crassula helmsii*.

It is clearly desirable to ensure that reference sites have negligible, and preferably zero cover of invasive alien species. In this sense invasive species are given a similar status to hydromorphology in the screening process (reference sites must show no more than minimal distortion in the attributes of that quality element). Beyond this the information communicated by this metric as far as site classification is concerned appears to be limited. Nevertheless INV as a metric is easily calculated and does not require dedicated data collection. It may, in fact, be preferable to see invasive alien species not in terms of their precise identity, but as indicators of fluctuating resource supply or hydromorphological disturbance, both of which favour or provide recruitment

opportunities for invasive alien taxa (Willby, 2007). Evidence of ecological effect appears to be adequately transmitted via the other main metrics.

The major exception here is provided by the latest generation of invasive macrophytes (*Crassula helmsii*, *Myriophyllum aquaticum* and *Hydrocotyle ranunculoides*) which present a heightened threat to lake ecosystem function due to their morphology and growth habits, which contrast markedly with native and naturalised invasive species (Willby, 2007). The threat level of these taxa is such that their *presence* alone should be sufficient to relegate a site to moderate status, regardless of other metric values; one would clearly not wish to allow populations of such taxa to increase to the level required to downgrade a site from good to moderate status before taking any action. Even if populations of such species fail to reach the threshold needed for downgrading, their establishment in large water bodies potentially transforms such sites into propagule sources and thereby represents a risk to other water bodies within the catchment that should be avoided wherever possible. A simple manual override rule could be implemented to prevent a lake achieving good ecological status if certain high risk taxa were present. This would prompt investigative monitoring to establish the true extent of invasion.

Alien species of all types have been considered by UKTAG, independently of the biological quality elements which they represent, and rules have been established for the inclusion of alien species as a separate tier of the classification process (UKTAG, 2007). Depending on the thresholds set, such an approach might make the incorporation of a dedicated alien species metric-redundant in macrophyte-based classification (or vice versa). In this context, the macrophyte survey protocol should be adequate to determine if invasive species are *established* in a water body but, since such species are often locally abundant in the early stages of colonisation, this method will not be optimal to determine their *presence* alone. In this case additional classification rules such as those proposed by UKTAG (2007) may be useful to allow the inclusion of anecdotal information on high risk alien species that derives from sources other than formal macrophyte surveys.

6.8 Filamentous algae

6.8.1 Prediction

A relative cover threshold of filamentous algae in reference sites of 10 per cent was based on the 10th percentile of relative cover of this taxa in the global dataset. Putative reference sites that exceeded this threshold were excluded from the reference site pool. The median relative cover of this taxa in the remaining reference sites was five per cent and this is treated as the expected value.

6.8.2 Calculation of EQR

Where the proportion of filamentous algae is below 0.05 the EQR for this metric is fixed at one. Otherwise, the EQR for this metric can be calculated from (the proportion of filamentous algae – 1)/(-0.95). Therefore if the metric value is 0.25 (filamentous algae constitute 25 per cent of the overall vegetation) the EQR is 0.79.

6.8.3 Class boundaries

Following the rationale in Section 6.8.1 a relative cover of 0.05 is used as the H/G boundary. A relative cover of 0.25 is used as the G/M boundary, the rationale being that if high cover of filamentous algae is a response to sustained anthropogenic pressure, this will be communicated through impacts on other metrics. This also allows for the fact that in favourable years a cover of filamentous algae may be a natural phenomenon, especially when surveys are undertaken early or late in the season. The value of filamentous algae may be to highlight fluctuations in resource supply to which other macrophyte metrics are not sufficiently responsive. However, it is likely that other quality elements such as diatoms or phytoplankton will adequately cover this possibility

After inclusion of the metric ALG this metric was found to have the lowest EQR of six metrics in 158 out of 4,538 surveys (3.5 per cent). Following an initial combining of metrics implemented prior to intercalibration, it was found that of these 158 surveys ALG would have resulted in a final class change from high or good to moderate or worse in only 61 cases (38 per cent). While this would have been undetected without the use of this metric, its general utility is evidently limited. There is no discrimination between different filamentous algae under this general label and it is probable that at some sites a visible cover of some filamentous algal taxa on coarse substrates is a natural phenomena. Filamentous algae are also highly responsive to growing conditions and survey timing and it is likely that late season surveys of shallow sheltered water bodies, especially in years with hot dry summers, will record higher than normal cover of filamentous algae.

6.9 Harmonising metric EQRs

EQRs for each of five metrics (LMNI, N_FG, COV, ALG and INV) are determined for each survey based on the approaches described above. A full worked example is given in Section 6.10 below. Although each metric is expressed on a common numeric scale, and the EQR value represents the level of distortion for that metric from reference condition, the different EQRs are not 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 LMNI EQR scale (approximate middle of good status),cannot be compared directly with a value of 0.85 on the N_TAXA EQR (approx middle of high status). Consequently, a translation scheme is required to map all EQRs onto a common class boundary system (Table 6.8) before it is valid to combine or select metrics based on their EQR. This becomes redundant once metrics are expressed in terms of per cent confidence of class (see Section 9). The only simple alternative to this approach for standardising EQRs would be to assign each metric to a status band, recode these to a simple numeric scale (1=Bad….5=High) and take the average or minima of these numbers. However, this approach is crude and results in a significant loss of information (such as whether an EQR is just above or below a class boundary).

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

Table 6.8 Equations for harmonising other metrics to class boundaries used for LMNI. After use of this rescaling it is legitimate to average across metric EQR values.

Metric	Rescaling function
N_FG or N_TAXA	=0.95*metric EQR + 0.062
COV	=0.76*metric EQR + 0.252
ALG or INV	=1.3053*metric EQR ^3 - 2.1239*metric EQR ^2 + 1.5245*metric EQR + 0.2802

6.10 Worked example

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

Table 6.9 shows the macrophyte data collected during a standard survey of a high-alkalinity, very shallow lake. Table 6.10 details the environmental data for the site required to predicti reference metric values.

Table 6.9 Summary of macrophyte survey data used in worked example

Species	% cover	LMNI
Chara aspera	10	6.39
Nitellopsis obtusa	2	7.62
Potamogeton obtusifolius	5	6.72
Nymphaea alba	10	5.54
Hippuris vulgaris	5	6.40
Elodea canadensis	1	7.14

Table 6.10 Environmental data for the site

Variable	Units	Value
Alkalinity	µeq/l	1,700
Mean depth	m	2.7
Weighted Freshwater Sensitivity Class (FSC)	Value ranging from 1-5	4.1
Conductivity	µS/cm	350
Distance from coast	m	25,000
Lake area	ha	3.1
FSC2	%	0
Solid Calcareous Geology	%	70
Altitude	m	15

STEP 1

Determine species composition EQR using LMNI index

The **observed LMNI** score for the lake is the average of ranks for individual species:

$$\text{Observed LMNI} = \frac{(6.39 + 7.62 + 6.72 + 5.54 + 6.4 + 7.14)}{6} = 6.64$$

The **expected LMNI** score for this site under reference conditions is calculated using an equation derived from multiple regression with a set of variables from UK reference lakes including: alkalinity, conductivity, lake area, geology, freshwater sensitivity class and distance to nearest coast. Information on most of these variables at test sites should be available through the GB Lakes Inventory (Bennion *et al*., 2002). Applying the equation in Table 6.4, column 1 (Model 2) to the values for the environmental variables in Table 6.9 above results in an expected LMNI score of 5.48. Thus:

Expected LMNI = 1.605095 + 0.635498* ($\log_{10}$(alkalinity +40)+ -0.53082*($\log_{10}$ (Zmn)) + 0.113708*(wtd FSC) + 0.440744*($\log_{10}$ (conductivity) + 9.41E-06* Dist_ coast + 0.17256 * $\log_{10}$ (area) + -0.00193 * FSC2 + 0.00196 * SGEOL_CA = 5.48

Therefore, the EQR for this lake using the LMNI metric alone is:

$$\text{LMNI EQR} = \frac{(6.64 - 10)}{(5.48 - 10)} = 0.74$$

STEP 2

Determine taxonomic diversity as number of taxa (N_TAXA) EQR

The **observed** number of taxa (N_TAXA) is six. The **expected** number of taxa is produced by the equation in Table 6.6 using altitude, alkalinity and lake area as predictors. The equation predicts $\log_{10}$ (N_TAXA+1). In this case the expected number of taxa would be 8.64 ($(10^{0.98})$-1). Thus:

Expected $\log_{10}$(N_TAXA+1) = 0.554544 + -0.000278 * (alkalinity+40) + 0.270098 * ($\log_{10}$ alk+40) + 0.0000000421 * ($(alk+40)^2$) + -0.00000000000205 * ($(alk+40)^3$) + -0.092317 * ($\log_{10}$ altitude) + 7.45721828406972E-07 * ($altitude^2$) + -1.37193346236172E-09 * ($altitude^3$) + 0.060514 * ($\log_{10}$ area) = 0.98

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

This gives an N_TAXA EQR of 0.86 and an adjusted EQR (Table 6.8) of 0.88 (an EQR harmonised to a common scale).

STEP 3

Determine functional diversity as number of functional groups (N_FG) EQR

The **observed** number of functional groups (N_FG) for this lake is five (*Chara aspera* and *Nitellopsis obtusa* are in group 2, *Potamogeton obtusifolius* group 14, *Nymphaea alba* group 12, *Hippuris vulgaris* group 7 and *Elodea canadensis* group 5).

The **expected** number of functional groups is derived from the equation in Table 6.6 and uses the same predictors as N_TAXA. The equation predicts $\log_{10}$ (NFG+1). The expected number of functional groups would be 6.14 ($(10^{0.85})$-1). Thus:

Expected $\log_{10}$(N_FG+1) = 0.2566347 + -0.00020472 * (alkalinity+40) + 0.26551458 * ($\log_{10}$(alkalinity+40)) + 2.94538463516568E-08 * ((alkalinity+40)^2) + -1.40898925977951E-12 * ((alkalinity+40)^3) + -0.00028913 * altitude + 0.03255904 * $\log_{10}$(lake area) = 0.85

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

This gives a N_FG EQR of 0.91, and an adjusted EQR (Table 6.8) of 0.93.

STEP 4

Determine hydrophyte mean percentage cover (COV) EQR

The **observed** percentage cover in this example is *Chara aspera* 10 per cent, *Nitellopsis obtusa* two per cent, *Potamogeton obtusifolius* five per cent, *Nymphaea alba* 10 per cent, *Hippuris vulgaris* five per cent and *Elodea canadensis* one per cent.

Based on summing the cover values and dividing by the number of taxa this gives an observed mean cover per taxa of 5.5 per cent (all species in this example are treated as hydrophytes by LEAFPACS).

For the cover metric no model could be developed as the amount of cover in reference sites was unrelated to the available environmental data. Therefore, the median of the reference set was used (8.5 per cent mean cover) as a generic **expected** value. The percentage cover is per cent cover of the colonised zone and not per cent cover of the whole lake. Hence the cover EQR is calculated as:

$$\text{COV EQR} = \frac{Log.(Observed\,mean\%\,cover+1)}{Log.(8.5+1)}$$

Which is an EQR of 0.83 for this example, and an adjusted EQR (Table 6.8) of 0.88.

STEP 5

Determine relative cover of macro algae (ALG) EQR

Where data are available, the EQR for relative macro algal cover is determined from the following equation. As for COV, no model of reference cover could be determined so a fixed reference condition of 0.05 is used based on the median of the reference population.

If macro-algal cover is above 0.05 then ALG EQR is given by: $\frac{(MacroAlCover - 1)}{(0.05 - 1)}$

If macro-algal cover is below 0.05 then ALG EQR is one.

Hence in the absence of any filamentous algae, the EQR is one.

STEP 6

Determine relative cover of non-native species (INV) EQR

Where cover data are available, or where cover can be estimated from ordinal scale cover scores (e.g. DAFOR), it is possible to estimate the proportion of total cover that is attributable to non-native invasive species. In this example *Elodea canadensis* is the only non-native invasive species having a cover of one per cent out of a combined total cover of 33 per cent. Therefore the **observed** relative cover of non-native invasive species at this site is 0.03.

No model of reference cover is required for this metric since it is assumed that established populations (over 0.05 relative cover) of invasive species should be absent under reference conditions with an EQR of one, approximately zero cover of non-natives. Therefore the **expected** value for this metric is always zero. Hence the INV EQR is calculated as:

INV EQR = *1-relative cover of non-native species*

This produces an EQR of 0.97 and an adjusted EQR (Table 6.8) of 0.95.

7 Achieving an overall lake classification based on macrophyte metrics

7.1 Introduction

Any classification can be secured from the EQR based on a single metric. However, independently of the uncertainty associated with that individual 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 the opposite. 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 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 information from different metrics is combined will influence the final classification of a site.

It is implicit in the WFD 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 different metrics must be employed simultaneously to diagnose a range of pressures. Generally the more metrics 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 lower confidence associated with them when vegetation is sparse or species-poor. By contrast, when the compositional metrics use scores expressed on a constrained scale (1-10 in this case), there is an inevitable tendency in high or good status sites for the metric value (LMNI in this case) to increase with increasing number of species. In this project five metrics were developed, LMNI, N_FG, COV, ALG and INV which, it is believed, collectively address the major anthropogenic and biological pressures to which lakes are exposed (nutrient enrichment, hydromorphological modification, acidification, invasion and grazing). This however, does not preclude the addition of other compositional metrics at a subsequent stage. 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 ASPT, LIFE and 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 multiple pressures on water courses in Northern Ireland using species optima for 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 (Biggs *et al*., 1998) the 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 using 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 an integrated sense, but introduce other metrics, such as richness or cover, as 'bolt-ons' to override assessments based purely on traditional compositional metrics whenever expert opinion indicates this to be necessary.

The LEAFPACS project adopted a multimetric approach, partly to reflect the impacts of different types of pressure, but primarily to explore the use of individual metrics. Only a single compositional metric (LMNI) is used in LEAFPACS, partly because nutrient enrichment is the dominant pressure on European lakes, and partly because it would be difficult to disentangle other related pressures, such as sedimentation, from a general response to nutrient enrichment. Opportunities remain to incorporate other compositional metrics that are sensitive, for example, to changes in water level regime. The additional metrics used in LEAFPACS have a dual basis:

(i) Reliance on a single compositional metric is unwise when metric values may derive from abnormally species-poor or sparsely vegetated sites. This may be a particular 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.

(ii) Different types of metrics are required to provide complementary sensitivity across a full pressure gradient, as well as across a full range of lake types. Thus richness and cover-based metrics appear to be increasingly important at higher levels of enrichment and in more naturally fertile lakes, where the 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.

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 (note that there is an additional caveat here, since the metric that 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 for using the metric with the lowest associated class since the 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 LMNI EQR cannot be raised by the presence of high COV and N_FG EQRs. Taking the minimum EQR is undoubtedly a conservative approach. An important disadvantage is that it does not adequately discriminate between sites where there is 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 LMNI and a high EQR for COV and N_FG would not be discriminated from a site with a similarly low EQR for LMNI but also low EQRs for COV and N_FG. This is significant because, *inter alia*, resources to support Programmes of Measures (POMs) may be allocated differently between these cases. Moreover, conservation of water body ecosystem function lies at the heart of the WFD. Consequently it is arguable that a vegetation which approaches the natural richness and abundance for that water body, yet is altered in its composition (for example through a direct anthropogenic pressure or biological invasion), will still retain and support more of the ecosystem functions to which macrophytes contribute than a similar waterbody with a vegetation that is largely unaltered compositionally, yet is species-impoverished or of sparse cover relative to the expected state. These concepts are summarised in Table 7.1. This 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 ultimately translate to an effect on the class boundaries for that metric), or (iv) because some metrics behave differently across a pressure gradient.

In our classification the three supporting metrics (COV, INV and ALG) are associated with a higher measurement error and a weaker correlation with the available pressure data than the core metrics. Consequently, these supporting metrics receive only half

the weight of the core metrics where they contribute to the final classification. In the case of the richness metrics (N_TAXA and N_FG) consideration of the relationship between richness and pressure (section 4.3.3) 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. Conversely, at high baseline productivity, when conditions might be expected to lead to loss of species through competitive exclusion by dominant, canopy-forming, tolerant taxa, 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 lake type.

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. Rules for combining metrics have therefore been developed iteratively through the process of intercalibration of the UK method at both Northern/Atlantic and Central-Baltic GIGs.

7.3 Application to macrophyte-based classification of lakes

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 metrics;

iii. average of a subset of metrics (LMNI, 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.

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 intervention through a Programme of Measures to achieve recovery. Where ecological status has been destroyed, restoration to a set of alternative objectives other than Good Ecological Status (GES) may be appropriate.

7.3.2 Results

Methods ii and iii above resulted in the clustering of large numbers of surveys in high or good classes and were not sufficiently sensitive. The evidence suggests that even when several metrics exhibit significant distortion, their low EQR is 'rescued' through averaging by metrics which show little impact. Generally, few sites exist where all metrics are degraded to a similarly high degree. The only way that Methods ii or iii 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 10th percentile of the EQR distribution is 0.96 for both these methods, which would suggest a need to adjust the class boundaries accordingly (upwards). However, even after making this adjustment the averaging approaches do not compare favourably with Methods i or iv. Method iv 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 specifically 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 i), moderate by the averaging over all metrics approach (Method ii) and moderate by the averaging across the worst two metrics approach (Method iv). Method ii may give too optimistic a view while Method i 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 here to define the final class. An additional advantage of this approach is that it is amenable to the incorporation 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 7.2, whereby Metric iii carries twice the weight of Metrics i and ii.

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, and are the result of the successful intercalibration of the UK classification method at N-GIG/A-GIG and CB-GIG levels using two alternative approaches to intercalibration. In this sense these rules are fixed (or modifiable only to the extent that the UK remains inside the harmonisation bands agreed by the two GIGs) and additional metrics to reflect these or other pressures would probably need to be developed and integrated, rather than changing the rules for combining the existing metrics. The following rules are used for combining the metrics in this project and yield a classification of the same or similar lakes that is compatible with the view of other GIG MS. Note that these rules can only be applied after the harmonisation of all metrics to a common class boundary system.

STEP 1

Determine maximum indicator of diversity and adjust LMNI EQR to give EQR_p

- If the values of both the adj N_FG EQR and the adj N_TAXA EQR are less than the LMNI EQR, the mean of the LMNI EQR and the greater of adj NFG EQR or adj NTAXA EQR is calculated.
- If the value of the adj N_FG EQR or the adj N_TAXA EQR is greater than the LMNI EQR, then the greater of adj N_FG EQR or adj N_TAXA EQR is multiplied by a weighting factor and added to the LMNI EQR . This product is then divided by the weighting factor plus unity.

$$\frac{\left(LMNI\_EQR + weighting * Max(N\_FG\_EQR, N\_TAXA\_EQR\right)}{1 + weighting}$$

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

Weighting factor = (1/(EXP(LN(2624653085.79034) +expected LMNI* LN(0.0165738290871162))+1/0.5001))

The weighting factor is designed to compensate for an expected increase in productivity of high-alkalinity lakes and the associated decrease in richness with increasing nutrient pressure in such lakes. The weighting factor is defined by a simple logistic regression based on the expected LMNI value, which is used as a surrogate for natural productivity. This gives the richness metrics a maximum of 0.5 of the weight of the LMNI EQR when productivity is naturally high (expected LMNI above 5.5). This increases the EQR of diverse relative to impoverished high-alkalinity sites with the same LMNI EQR. In moderate- and low-alkalinity lakes (expected LMNI below 4.5) the weighting factor gives the diversity metric EQRs a neutral influence (effective weighting of zero). This is appropriate since the evidence from the analysis in Section 4.2.3 implies that giving a positive weighting to relatively high richness at low alkalinity might be equivalent to rewarding an impact. Thus, richness reaches a maximum between 10-25 µg/l TP, which represents an elevated level of fertility in low- and moderate-alkalinity lakes. Note, however, that a *negative* weighting is not used, since it becomes increasingly difficult for compositional metrics to achieve the required standard when richness is high. Hence, any penalty that should be associated with high richness in low-and moderate-alkalinity lakes is imposed via its influence on observed LMNI.

STEP 2

Determine the lower of the EQRs for COV, ALG and INV metrics, and adjust the diversity modified LMNI EQR (EQR_p)

- If the diversity modified LMNI EQR (EQR_p) is *greater* than the minimum of the adj COV EQR, adj ALG EQR or adj INV EQR, the final EQR is determined from the following weighted average. This lowers the final EQR of sites that are sparsely vegetated or support relatively extensive growths of filamentous algae or invasive macrophytes.

$$\frac{\left(EQR_p + 0.5 * Min(COV\_EQR, ALG\_EQR, INV\_EQR\right)}{1.5}$$

  Note that the minimum of adj COV EQR, adj ALG EQR or adj INV EQR can only influence the final EQR when these metrics are below one. In other words if the diversity modified LMNI EQR (EQR_p) is above one and the minima of supporting metrics equals one, there is no effect on final EQR.

- If the diversity modified LMNI EQR (EQR_p) is *less* than COV EQR, adj ALG EQR, or INV EQR the modified LMNI EQR (EQR_p) is used as the final EQR.

7.5 Examples of classification

7.5.1 Reference to worked example

The following examples are 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 6.10 is used as a starting point.

Consider the worked example. EQRs and their adjusted equivalent for the various metrics (based on the equations in Table 6.8) are summarized in Table 7.3 below. In this example, the maximum of the N_FG EQR and N_TAXA EQR is 0.91 (equivalent to an adj value of 0.93). This is greater than the LMNI EQR (0.74) so the diversity adjusted LMNI EQR (EQR_p) is given by

$$0.80 = \frac{\left(0.74 + 0.41 * 0.93\right)}{1 + 0.41}$$

Where 0.41 is the weighting factor given by:

(1/(EXP(LN(2624653085.79034) +expected LMNI*LN (0.0165738290871162))+1/0.5001))

In which the expected LMNI = 5.48

This value of EQR_p is less than the COV EQR (0.88) and adjusted INV EQR (0.95) and is thus the final EQR. This would place the example given on the G/M boundary. Thus the combined effect of relatively high functional diversity (N_FG), combined with largely unimpacted EQRs for other metrics, is to raise the EQR from 0.74 (moderate status) based on LMNI alone to 0.80 (G/M). Elevation of this site to mid good status depends on the acquisition of additional lower scoring taxa which would increase both LMNI and N_TAXA EQR. In this instance regular resurvey would be advisable to check for evidence of improvement or deterioration.

Table 7.3 Summary of metric EQRs and adjusted EQRs in worked example

Metric	Observed	Expected	EQR	adj EQR
a. LMNI	6.64	5.48	0.74	0.80
b. N_TAXA	6.00	8.64	0.86	0.88
c. N_FG	5.00	6.14	0.91	0.93
d. COV	5.50	8.50	0.83	0.88
e. ALG	0.00	0.05	1.00	1.00
f. INV	0.03	0.00	0.97	0.95
FINAL EQR				0.80

7.5.2 Further examples

Airthrey Loch, University of Stirling (Table 7.4)

Small, lowland, high-alkalinity, very shallow lake

Table 7.4 Summary of metric EQRs and adjusted EQRs in Airthrey Loch

Metric	Observed	Expected	EQR	adj EQR
a. LMNI	7.41	4.78	0.50	0.50
b. N_TAXA	11.00	10.20	1.03	1.04
c. N_FG	7.00	7.20	0.99	1.00
d. COV	11.60	8.50	1.13	1.11
e. ALG	0.08	0.05	0.97	0.95
f. INV	0.35	0.00	0.65	0.73
FINAL EQR				0.54

Final EQR places this lake on P/B boundary.

The site has a highly impacted composition but is well-vegetated and relatively diverse.

Final EQR is derived from:

(LMNI + weighting factor* adj N_TAXA EQR)/(1+weighting factor).

This value lies below the EQR of the remaining metrics, which are therefore redundant.

In its current position, improving the status of this site will depend firstly on increasing the LMNI EQR by creating an environment in which more nutrient-sensitive taxa can establish. This will also benefit the diversity metric EQRs. Improvements beyond moderate status will probably depend on increasing the INV EQR through management of non-native invasive/translocated species (in this case *Nymphoides peltata* and *Elodea canadensis*).

Lake of Menteith, Stirlingshire (Table 7.5)

Large, lowland, moderate-alkalinity, shallow lake

The final EQR places this lake on G/M boundary.

The site has a moderately impacted composition but is well-vegetated, diverse and algal and invasive metrics show little impact.

Table 7.5 Summary of metric EQRs and adjusted EQRs in Lake of Menteith

Metric	Observed	Expected	EQR	adj EQR
a. LMNI	5.59	4.27	0.77	0.77
b. N_TAXA	32.00	14.29	1.28	1.28
c. N_FG	14.00	7.88	1.24	1.24
d. COV	6.90	8.50	0.92	0.95
e. ALG	0.03	0.05	1.00	1.00
f. INV	0.03	0.00	0.97	0.96
FINAL EQR				0.78

Final EQR is derived from:

(LMNI + weighting factor* adj N_TAXA EQR)/(1+weighting factor).

This value lies below the EQR of the remaining metrics which are therefore redundant. Despite the high diversity of this site, the weighting factor gives this feature little influence on the final EQR since elevated richness at moderate to low alkalinity may be attributable to enrichment (as has occurred at this site). In its current position recovery of composition is the only way in which the ecological status of this site could be improved. This implies a reduction in the occurrence of more nutrient-tolerant taxa. A drop in the LMNI EQR or a drop in COV, ALG or INV EQR below the present LMNI EQR would move the site further into moderate status.

This site is an SAC for mesotrophic standing water vegetation and supports populations of a number of rare or scarce aquatic taxa including *Najas flexilis*, *Nuphar pumila*, *Pilularia globulifera*, *Elatine hexandra* and *E. hydropiper*. The current status would suggest that this site should not be considered in favourable conservation status.

West Loch Ollay, South Uist (Table 7.6)

Medium sized, lowland, high-alkalinity, very shallow lake

Table 7.6 Summary of metric EQRs and adjusted EQRs in West Loch Ollay

Metric	Observed	Expected	EQR	adj EQR
a. LMNI	5.23	5.34	1.02	1.02
b. N_TAXA	25.00	9.21	1.40	1.39
c. N_FG	12.00	5.87	1.33	1.33
d. COV	10.80	8.50	1.10	1.09
e. ALG	0.01	0.05	1.00	1.00
f. INV	0.00	0.00	1.00	1.00
FINAL EQR				1.07

Final EQR places this lake solidly in high status, affirming the position of this site in the reference network. The site has an unimpacted composition, is well-vegetated, diverse and algal and invasive metrics show little impact.

Final EQR is derived from:

(LMNI + weighting factor* adj N_TAXA EQR)/(1+weighting factor).

This value lies above the EQR of the ALG and INV metrics but these are redundant in this case since they are one or above.

This site is an SAC and supports an outstanding assemblage of mesotrophic vegetation typical of machair lochs, including nine species of *Potamogeton*. The current assessment would support this site achieving favourable conservation status.

7.6 Overall implications for classification

7.6.1 Comparison on a type by type basis

Figure 7.1 shows, at a type-specific level, the average final site-specific EQR of all UK sites surveyed since 1983 to the minimum standard required for calculation of all metrics (cover-based assessment of macrophyte composition). Sites are ranked from left to right in approximate order of increasing baseline productivity and the data include all surveys in each lake type, across all dates. This confirms the generally high status of most low-alkalinity lakes in the UK. Given the overwhelming numerical dominance of this lake type the global average EQR is also high (0.86). However, it is also readily evident that a major proportion of higher alkalinity lakes and especially those in the very high-alkalinity bracket (above 2.5 meq/l) fail to meet good ecological status for macrophytes. For a number of lake types, perhaps most notably continental type very high-alkalinity, very shallow lakes, this represents the majority of surveys. Note also the greater impacts on deep compared to shallower lakes at low and moderate alkalinity.

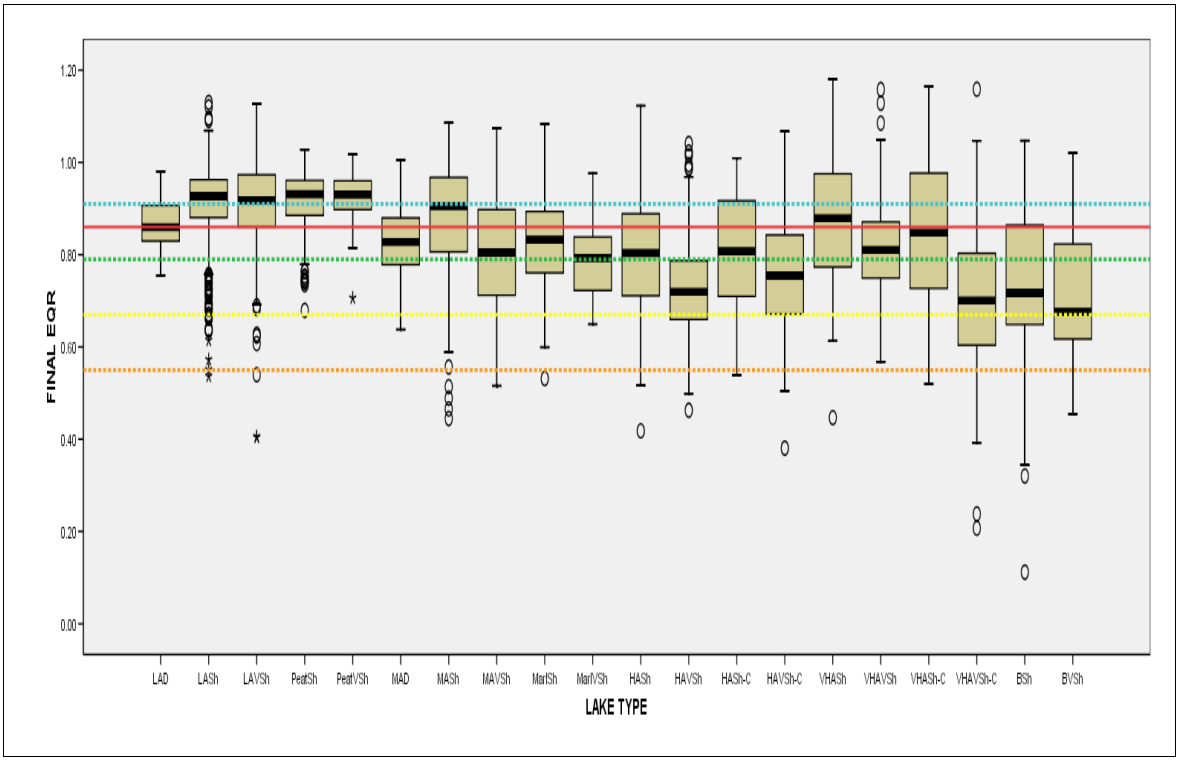


Figure 7.1 Distribution of mean final EQRs for individual lake types based on surveys since 1983. Class boundaries are indicated by dashed lines. The global average EQR across all surveys and all types (0.86) is shown as a solid red line.

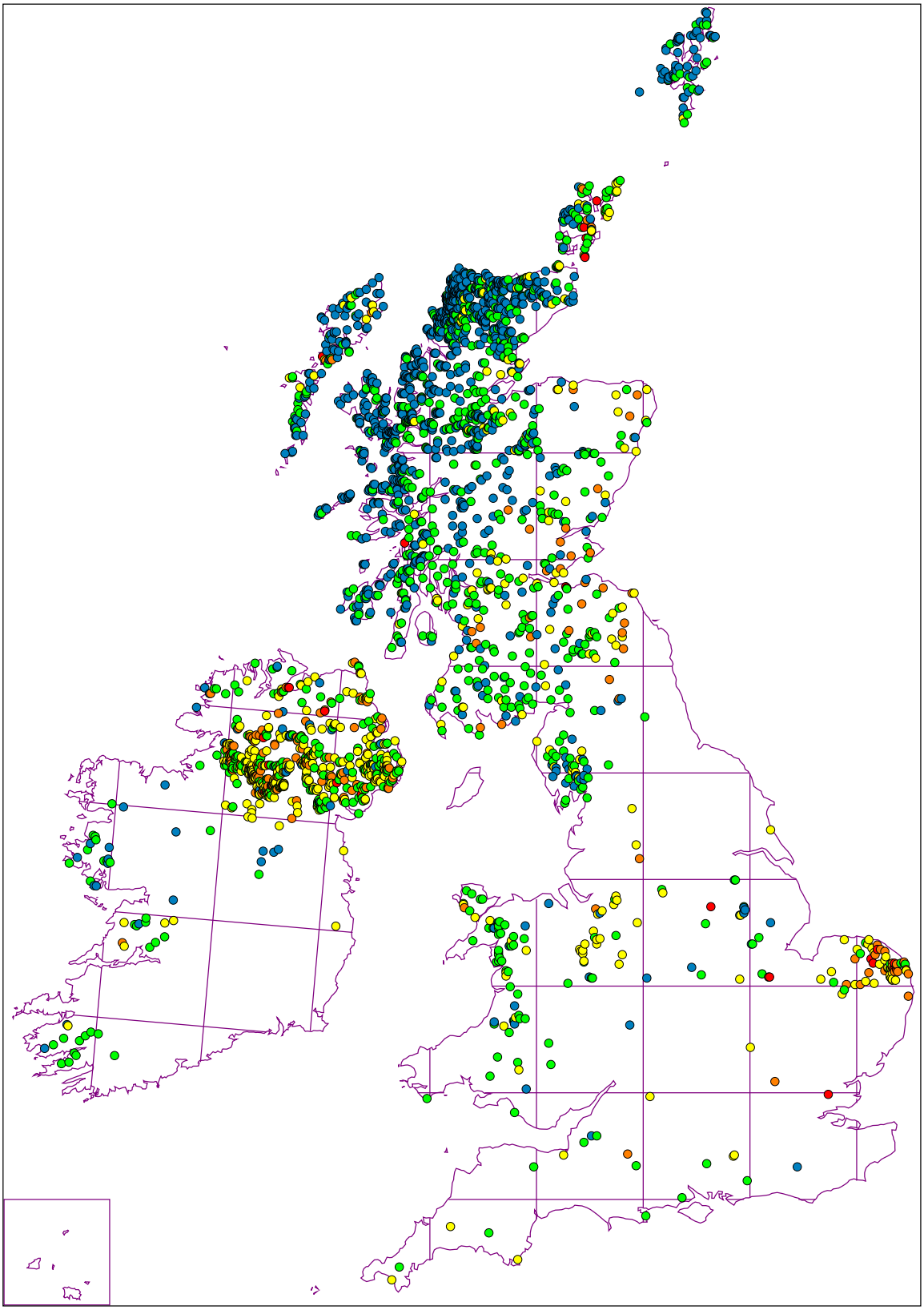


Figure 7.2 Final classification of UK lakes based on the average EQR from post-1983 data for sites surveyed to a minimum standard. Irish sites are shown for comparative purposes only. Colour coding follows WFD convention.

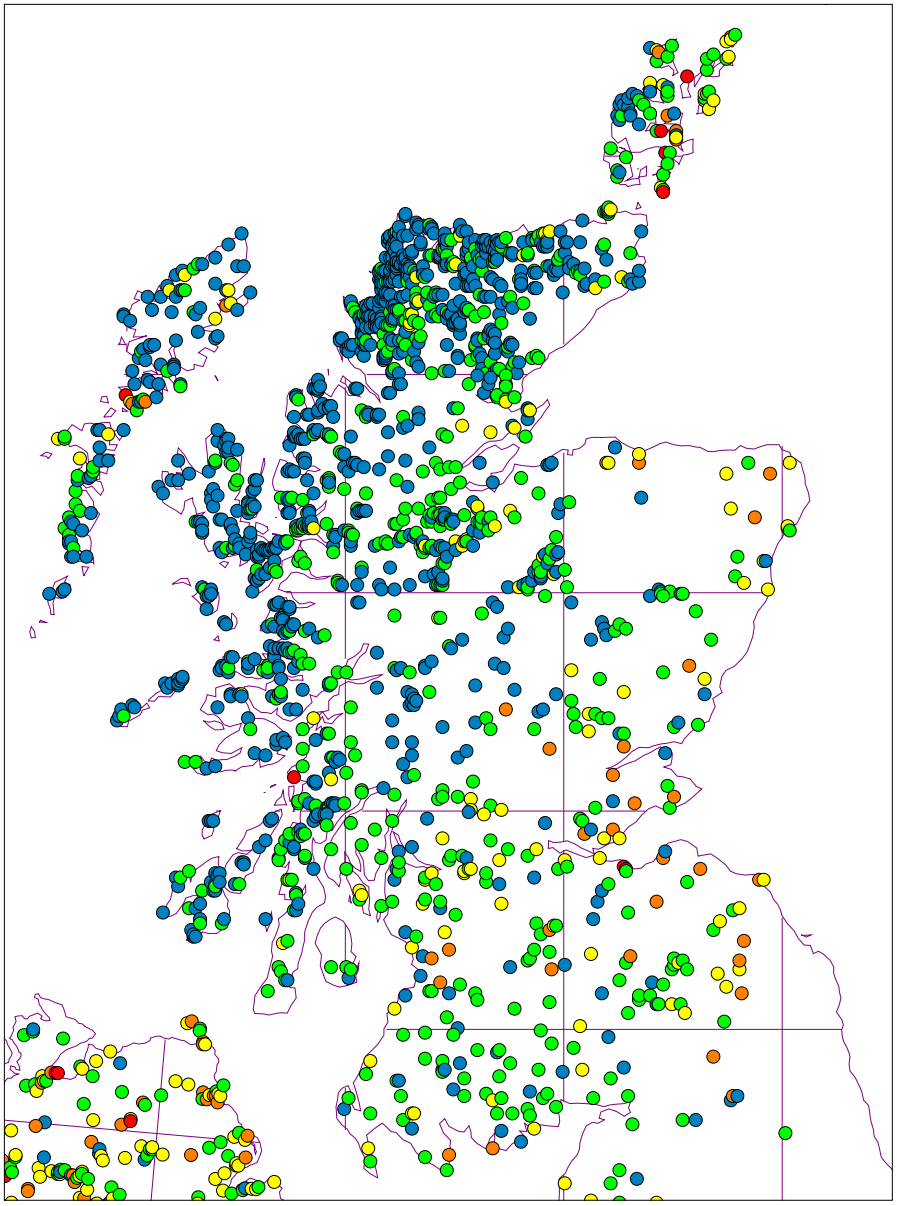


Figure 7.3 Distribution of post-1983 survey lakes by mean class in Scotland

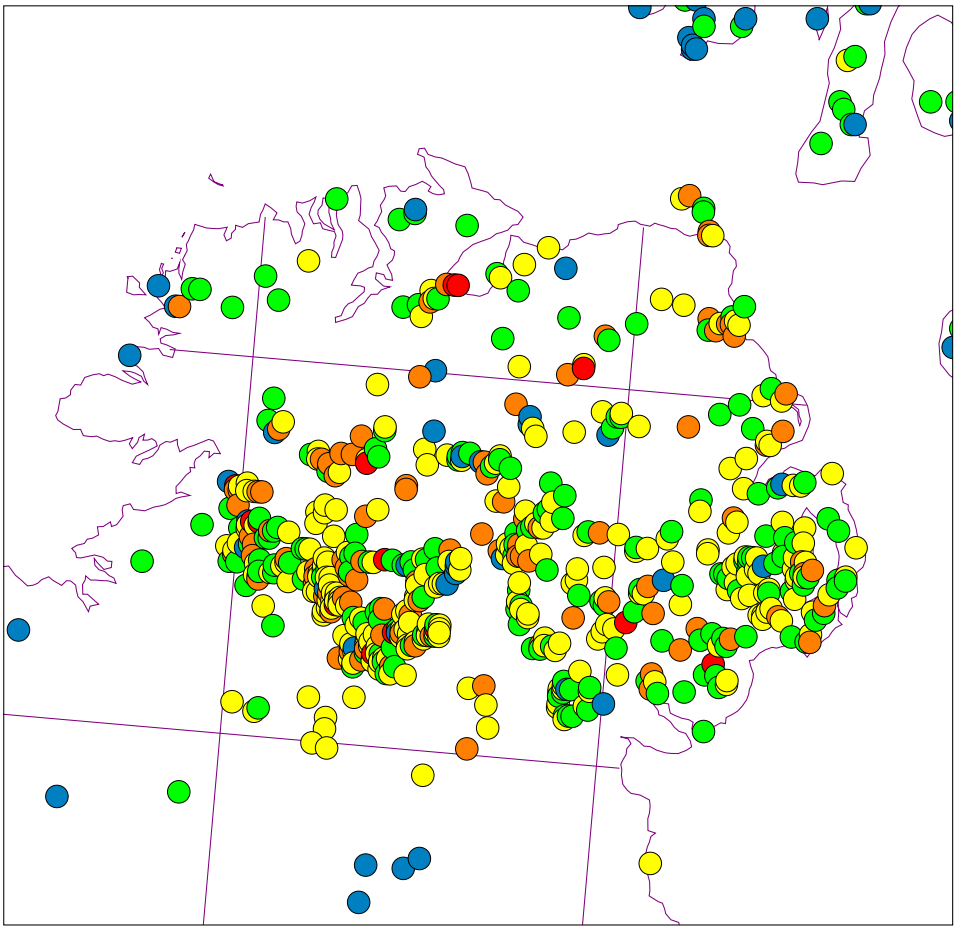


Figure 7.4 Distribution of post-1983 survey lakes by mean class in Northern Ireland

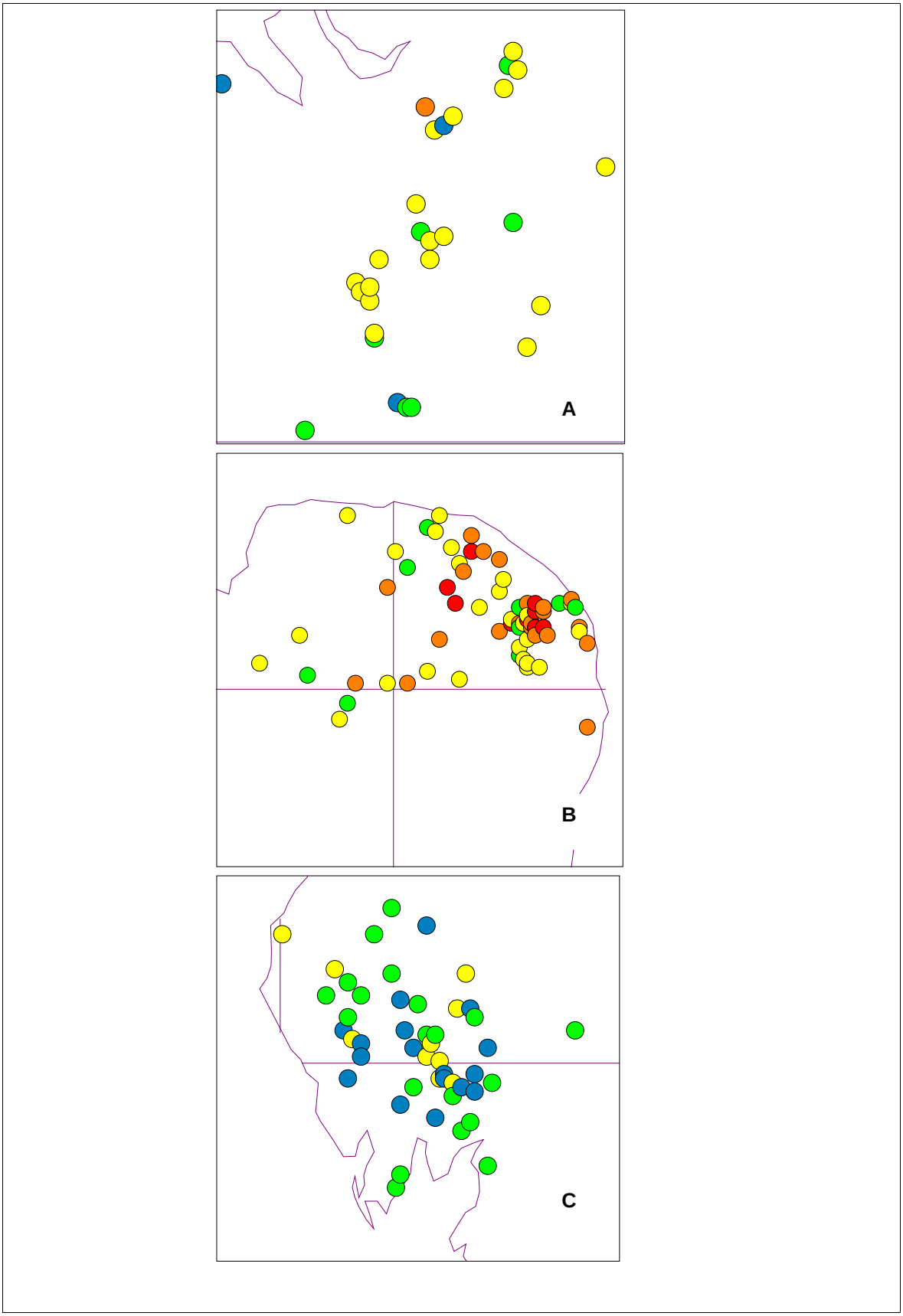


Figure 7.5 Distribution of post-1983 survey lakes by mean class in three major English lakes districts: A) West Midland Meres, B) Norfolk Broads, C) Cumbria.

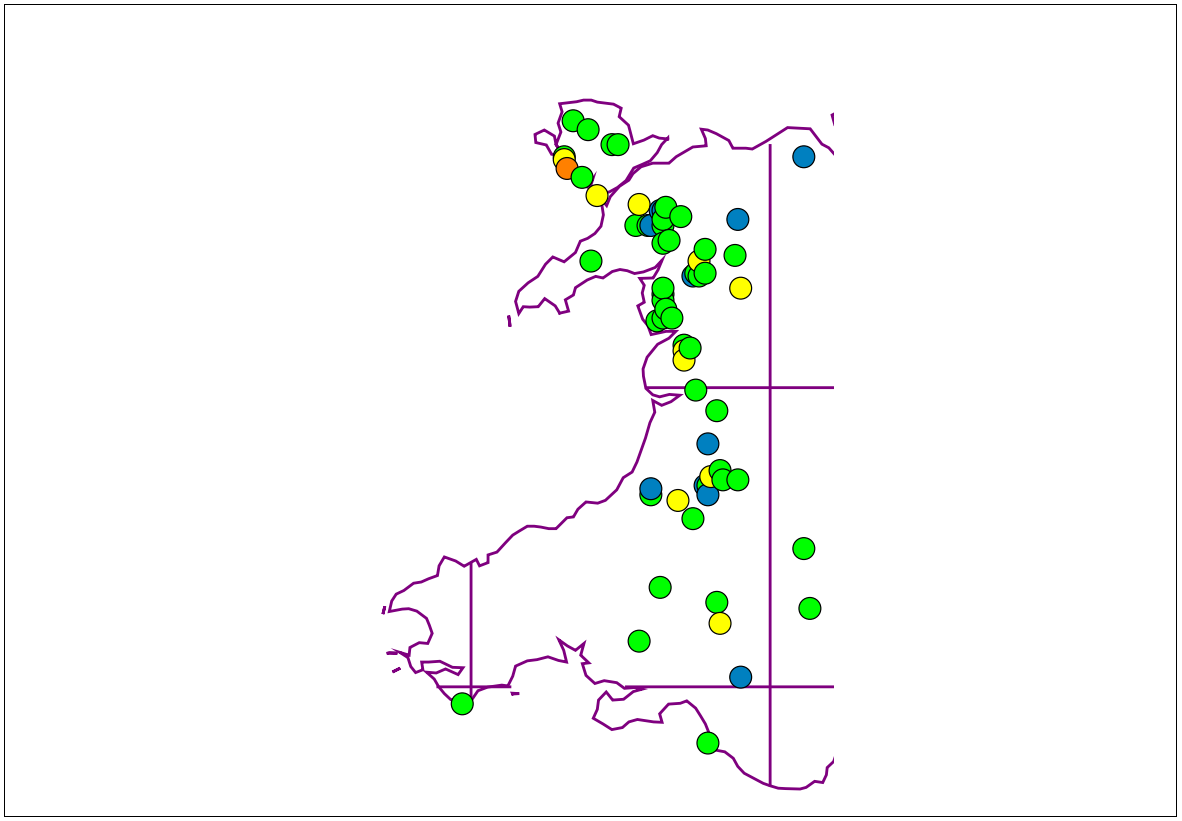


Figure 7.6 Distribution of post-1983 survey lakes by mean class in Wales

7.6.2 Comparison on a geographical basis

On a geographical scale, this classification translates as shown in Figure 7.2. Immediately obvious is the concentration of high and good status sites in the more sparsely populated areas of the north and west of Britain and the high incidence of moderate and poor status sites in lowland England, Northern Ireland and, to a lesser extent, the central belt and north east part of Scotland. The status of brackish water sites as assigned by the current tool should be viewed with caution. The associated figures (Figures 7.3-7.6) serve to illustrate the detail within countries or regions with high concentrations of lakes.

The output of the final classification is summarised on a country-by-country basis in Table 7.7. This demonstrates the generally good ecological status of UK lakes as a whole in numerical terms, but also highlights the relatively impacted nature of lakes in England and Northern Ireland where more than half of lakes fail to achieve good or better status (this figure will be significantly lower when confined to water bodies with 95 per cent confidence of class being below good). In Scotland, the percentage of impacted lakes is low due to the numerical dominance of minimally impacted low-alkalinity lakes in the sparsely populated north and west of the country. However, the tabulated sites represent only eight per cent of the lake resource in Scotland and therefore, while it is invalid to scale up directly from this figure, it is evident that in Scotland the absolute number of failing lakes will not be insignificant.

Table 7.7 Summary of final classification of lakes in UK and Ireland by number and percentage.

		High	Good	Moderate	Poor	Bad	Total
England	Count	37	67	71	32	20	227
	%	16.30	29.52	31.28	14.10	8.81	
Scotland	Count	1,134	596	122	38	9	1899
	%	59.72	31.38	6.42	2.00	0.47	
Wales	Count	11	46	11	1	0	69
	%	15.94	66.67	15.94	1.45	0.00	
Northern Ireland	Count	29	188	278	110	19	624
	%	4.65	30.13	44.55	17.63	3.04	
Republic of Ireland	*Count*	*24*	*49*	*22*	*4*	*0*	*99*
	%	*24.24*	*49.49*	*22.22*	*4.04*	*0.00*	
Total	Count	1,211	897	482	181	48	2819
	%	42.96	31.82	17.10	6.42	1.70	

Lakes in Republic of Ireland are excluded from the final total.

7.6.3 Use of older data for classification purposes

A large amount of 'contemporary' macrophyte survey data from lakes was collected during intensive surveys in Scotland and Northern Ireland 10-20 years ago. While the inclusion of such data for tool development is perfectly valid, it is pertinent to ask whether this inclusion in a synoptic assessment results in a biased view of the current status of UK lakes. Consequently an analysis was undertaken to compare the mean class of lakes sampled at least once in several different time periods.

i. 1985-1994

ii. 1995-2001

iii. 2002-2005

Ninety-four sites were sampled at least once in each of the periods 2002-05 and 1995-2001. Of these sites 88 (94 per cent) had a mean class in one period that was within one class of the mean class in the other period (there was no directional trend in class difference). Seventy-four percent of sites that were on average high or good in 1995-2001 were also high or good in 2002-05.

Fifty-eight sites were sampled at least once in each of the periods 2002-05 and 1985-1994. Of these sites 53 (91 per cent) had a mean class in one period that was within one class of the mean class in the other period (again there was no directional trend in class difference). Eighty-two percent of sites that were on average high or good in 1985-94 were also high or good in 2002-05.

Seventy-seven sites were sampled at least once in each of the periods 1995-2001 and 1985-1994. Of these sites 71 (92 per cent) had a mean class in one period that was within one class of the mean class in the other period (there was no directional trend in class difference). Ninety-four percent of sites that were on average high or good in 1985-94 were also high or good in 1995-2002.

Generally, we should have high confidence that older data, when used on its own, will be a good reflection of the current state of a site. Specifically, there is less than a 20 per cent chance that sites that were high or good 10-20 years ago will not be classified as high or good now.

7.7 Case studies

Two groups of lakes in England, the Norfolk Broads and West Midland Meres have received considerable attention from botanists, ecologists and limnologists over an extended period. Largely due to the endeavours of Victorian naturalists there is a historical archive of macrophyte data for these groups of sites which has been extracted from herbarium specimens, diaries and notebooks. LMNI values for sites from each group of lakes in the historical archive and contemporary period are compared in Figure 7.7. In both cases there has been a highly significant increase in LMNI values over the last century or so.

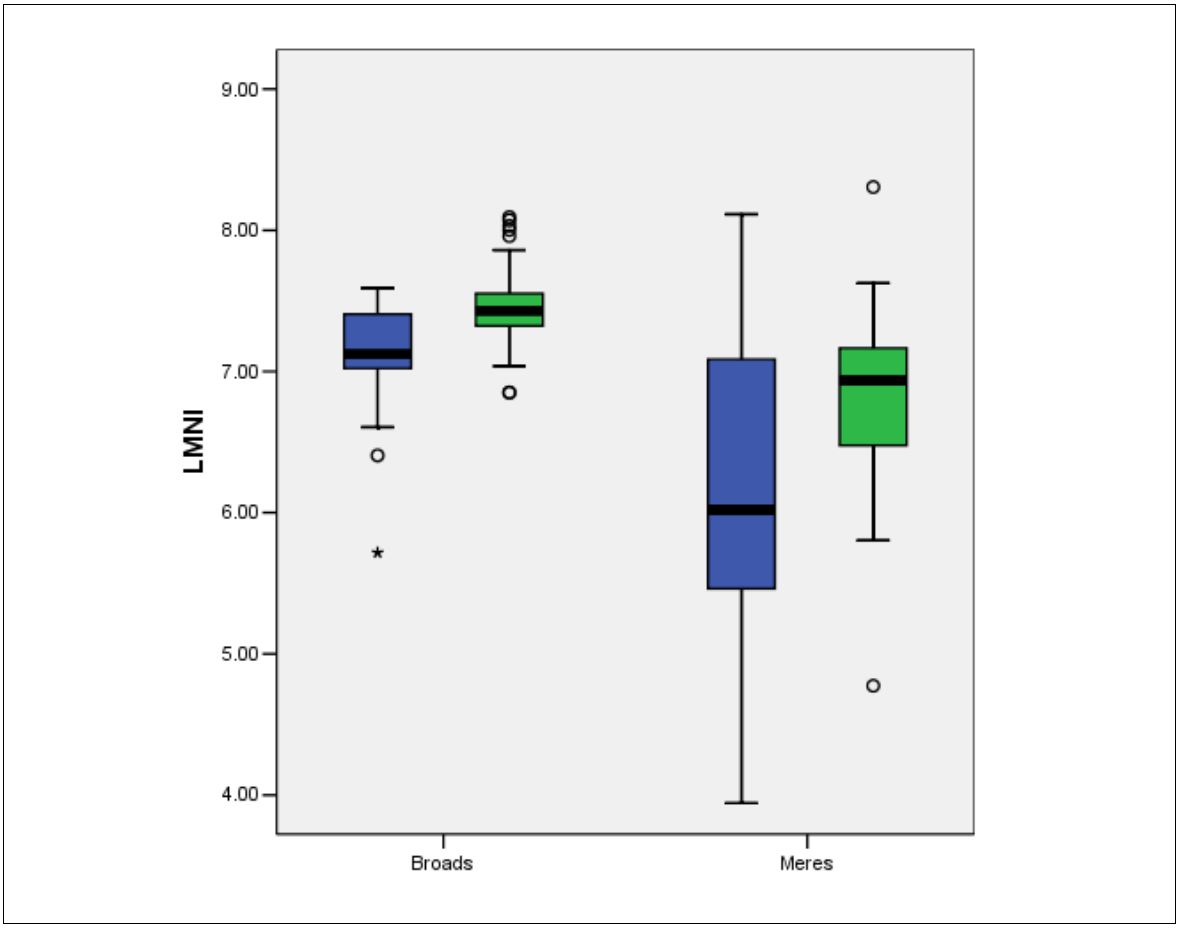


Figure 7.7 Comparison of LMNI values in historical archived data (blue) and contemporary data (green) for two groups of high-alkalinity shallow lakes in England. For the Broads historical data refers to data from the period 1860-1920 and for the Meres, 1820-1870. Historical data for Broads data collated mainly by M. Jackson and for Meres by A. Lockton. Differences between historical and contemporary data are significant at p=0.001 for both groups of sites.

Historical data provides a perspective against which more recent intensive recording for environmental assessment and conservation inventory purposes can be assessed. Figure 7.8 illustrates temporal change in the final EQR for these lakes since the date of the earliest available records. For comparative purposes, contemporary data is also treated as a composite of records within a decadal period. One of the main points to emerge from this comparison is that use of historical data (pre-1900) does not imply capture of reference condition since it is apparent that many sites were already

significantly impacted by the turn of the twentieth century. The decline in EQR during the 1800s is readily apparent in the case of the West Midland Meres.

Annual surveys of 16 Broads undertaken by the Broads Authority indicate that, based on the models developed in this project, the average state of the Broads fluctuated around the M/P boundary during this period (Figure 7.9). However, at an individual site level the EQR can fluctuate markedly between years (over 0.2 EQR units), with an average inter-annual fluctuation over all sites of 0.12 units. These fluctuations tend to be most exaggerated at the poorest quality sites. These surveys reveal weak but generally non-significant trends in EQR over this period, although some high profile sites, notably Hickling Broad, have shown declines since 1990 while other sites, such as Barton Broad or Hoveton Great Broad,which have been the focus of biomanipulation and other restoration measures, have increased EQRs.

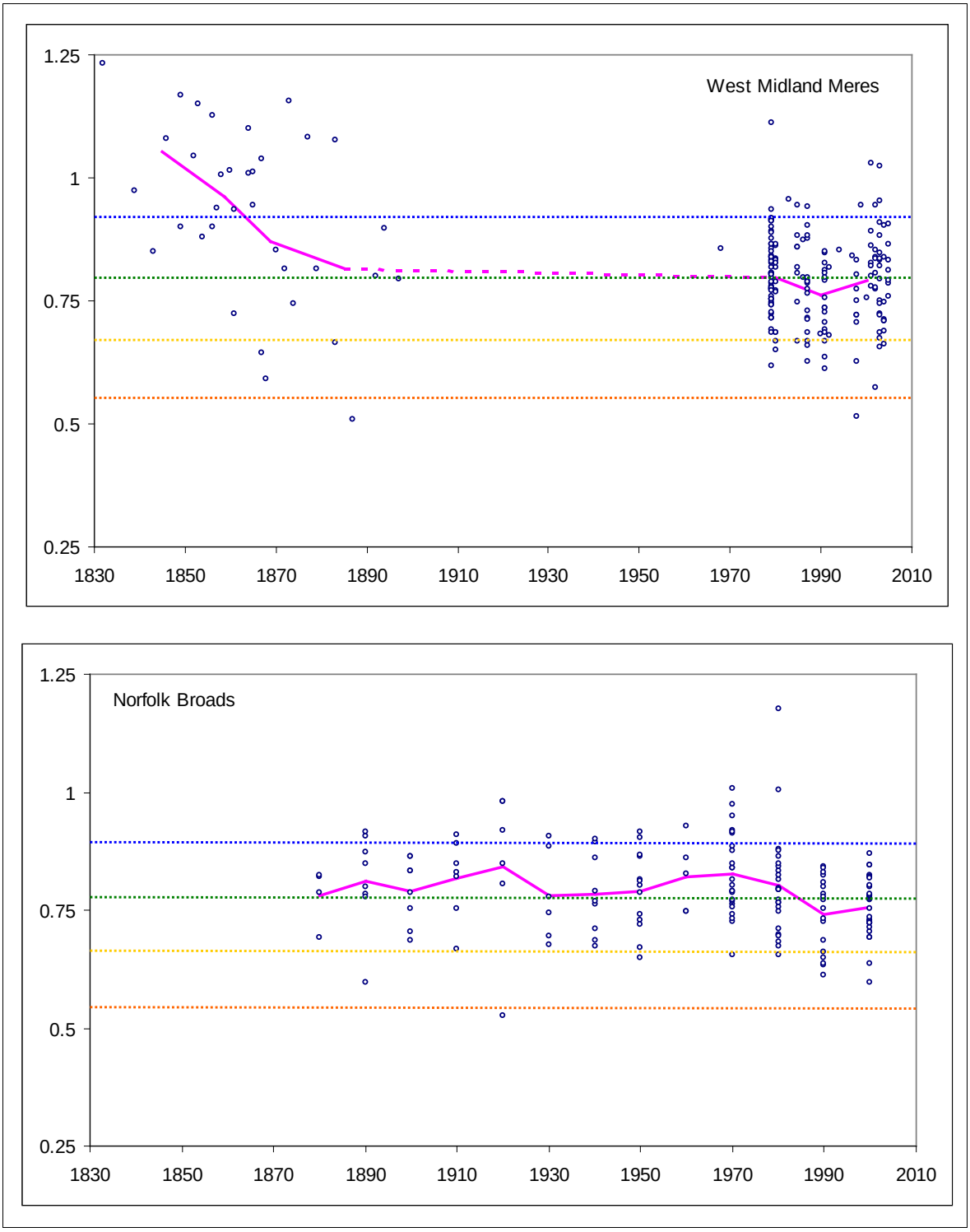
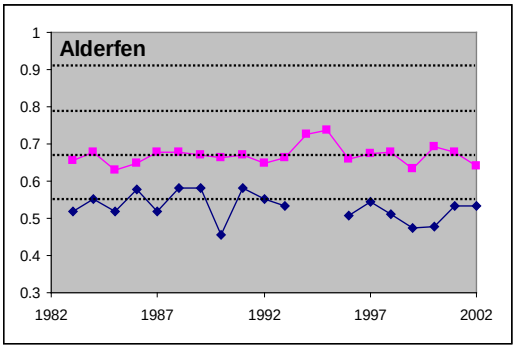


Figure 7.8 Long-term changes in EQR of West Midland Meres and Norfolk Broads based on presence-absence historical archive data aggregated into decadal bands. Note the marked decline in EQR in the Meres during the 1800s and general lack of high status sites in Broadland even in the late 1800s. Note that EQRs calculated for these figures are not directly comparable to final EQRs calculated for contemporary data based on a full suite of metrics.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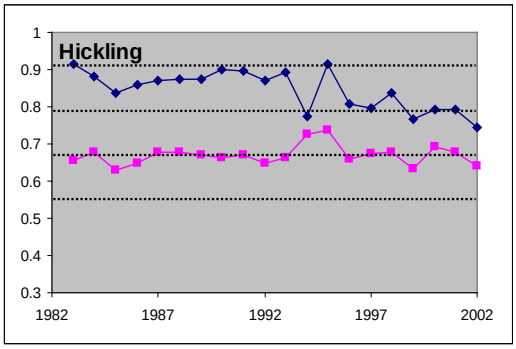
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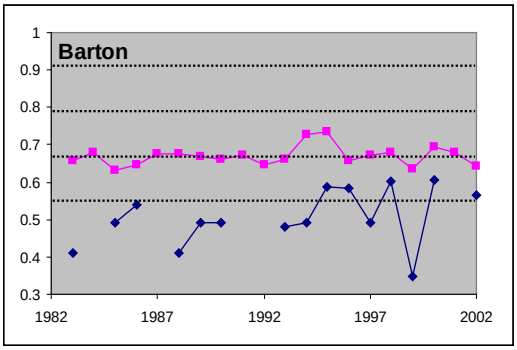
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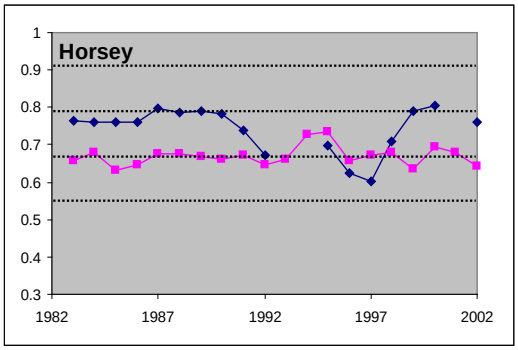
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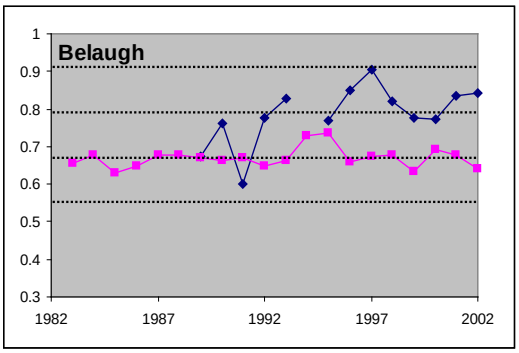
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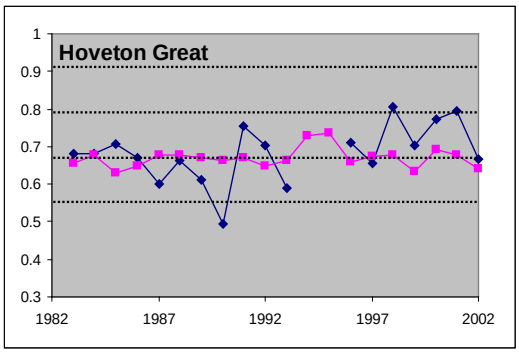
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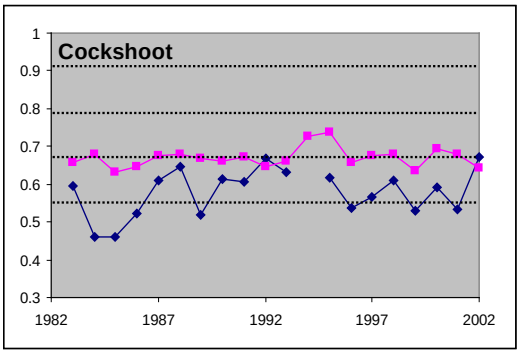
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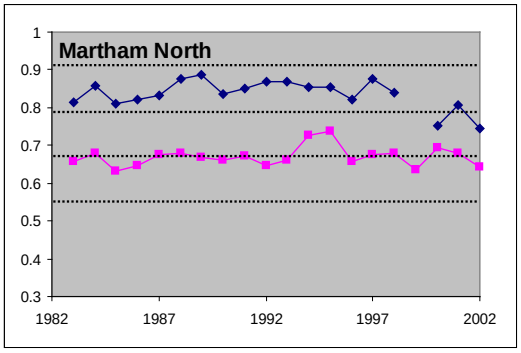
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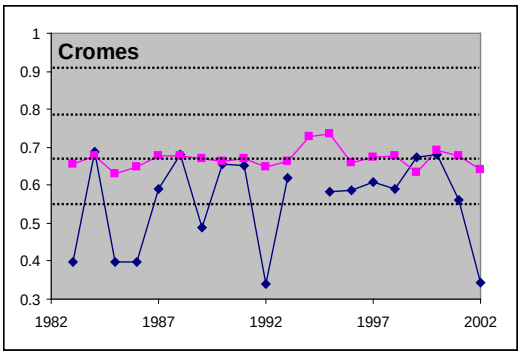
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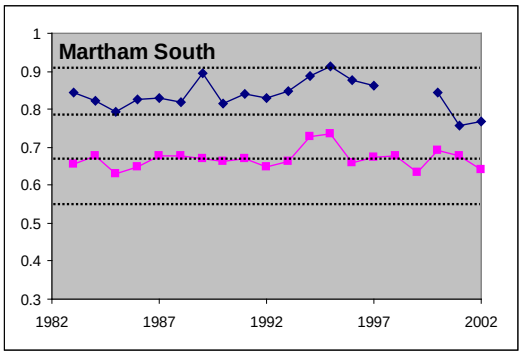
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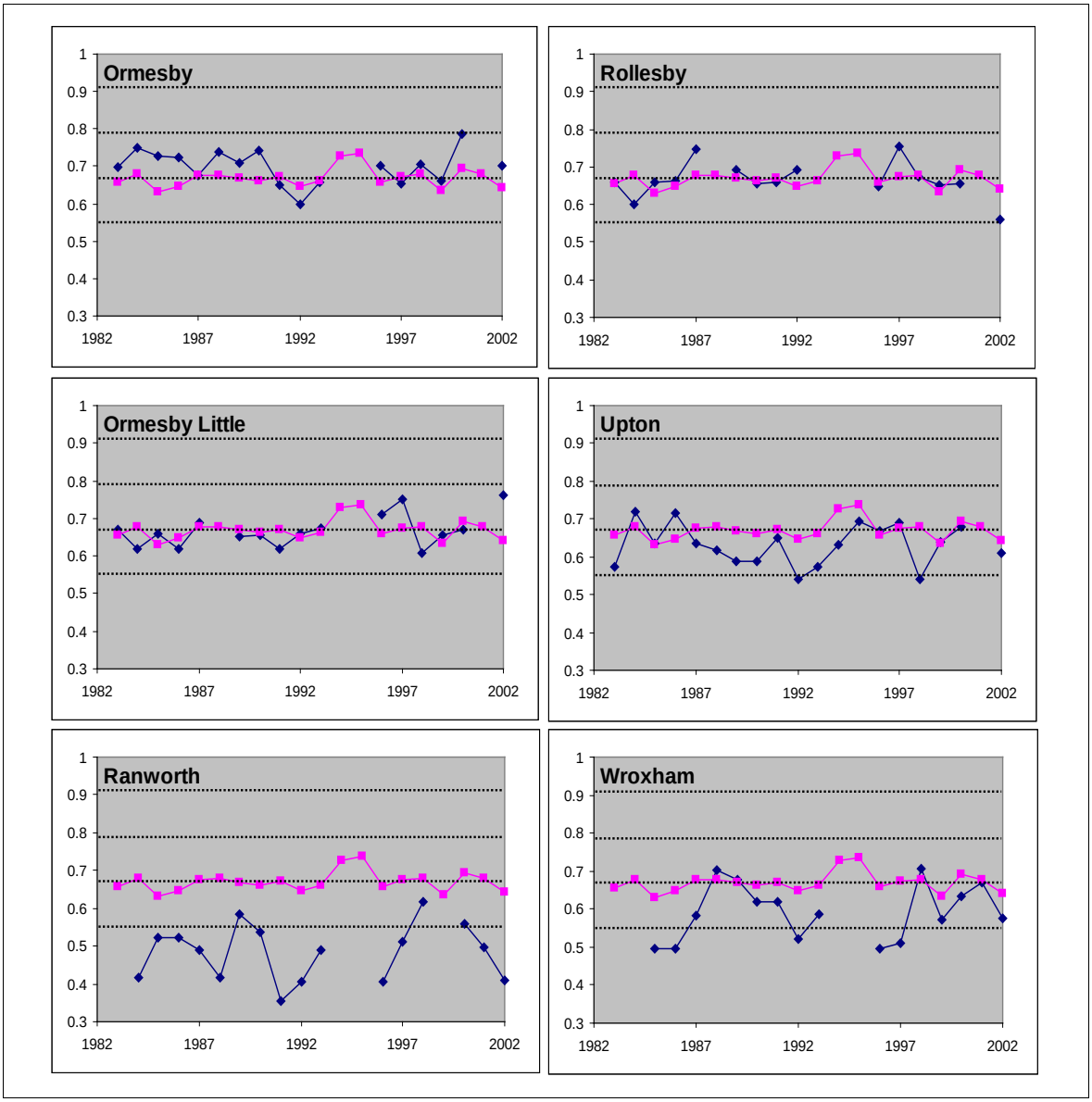

Figure 7.9 Annual fluctuations in EQR in 16 intensively surveyed Broads over a 20-year period. Pink line shows global average EQR each year for all Broadland sites. Dashed lines show class boundaries. On average, over this period the Broads fluctuated around M/P boundary. Underlying long-term trends (post-1990) are significantly negative for Hickling and weakly positive for Belaugh, Hoveton Great and Barton. Note the large degree of inter-annual fluctuation in EQR at some sites (over 0.2 units), especially in the poorest status sites (average fluctuation across all sites of 0.12 units) compared to relative stability of sites in good status. All data from Broads Authority surveys.

7.8 Rescaling final EQR to a range of zero to one

For reporting purposes, combining classifications based on different quality elements, and to estimate uncertainty it is desirable to rescale all EQRs to a common scale from zero to one. Figure 7.10 shows the distribution of final EQRs for all lake surveys in the database and parameters of the upper and lower parts of the frequency distribution are summarised in Table 7.8. Rescaling may be based on the maxima or minima or some slightly less extreme values. Scaling based on absolute maximum (1.24) and minimum (0.11) may be restrictive or misleading since values close to these extremes are rare.

Having identified a population of reference sites it is desirable to reflect their EQR in a value at or close to one. At the opposite end it may be best to take the minimum or a value close to this for scaling to zero since it is unlikely that many sites would be truly devoid of ecological value. However, it is possible that extreme low values reflect incomplete surveys, erroneous predictor data, or simply relate to a lake type (such as brackish lagoons) that is poorly served by the current tool. Conversely one would not wish to take a value dramatically above the minimum or below the maximum and simply cap all values falling below or above this as zero or one respectively, as this is likely to strongly distort the distribution of EQRs. It may also create problems when EQRs are averaged and may suppress the variation within high or bad classes.

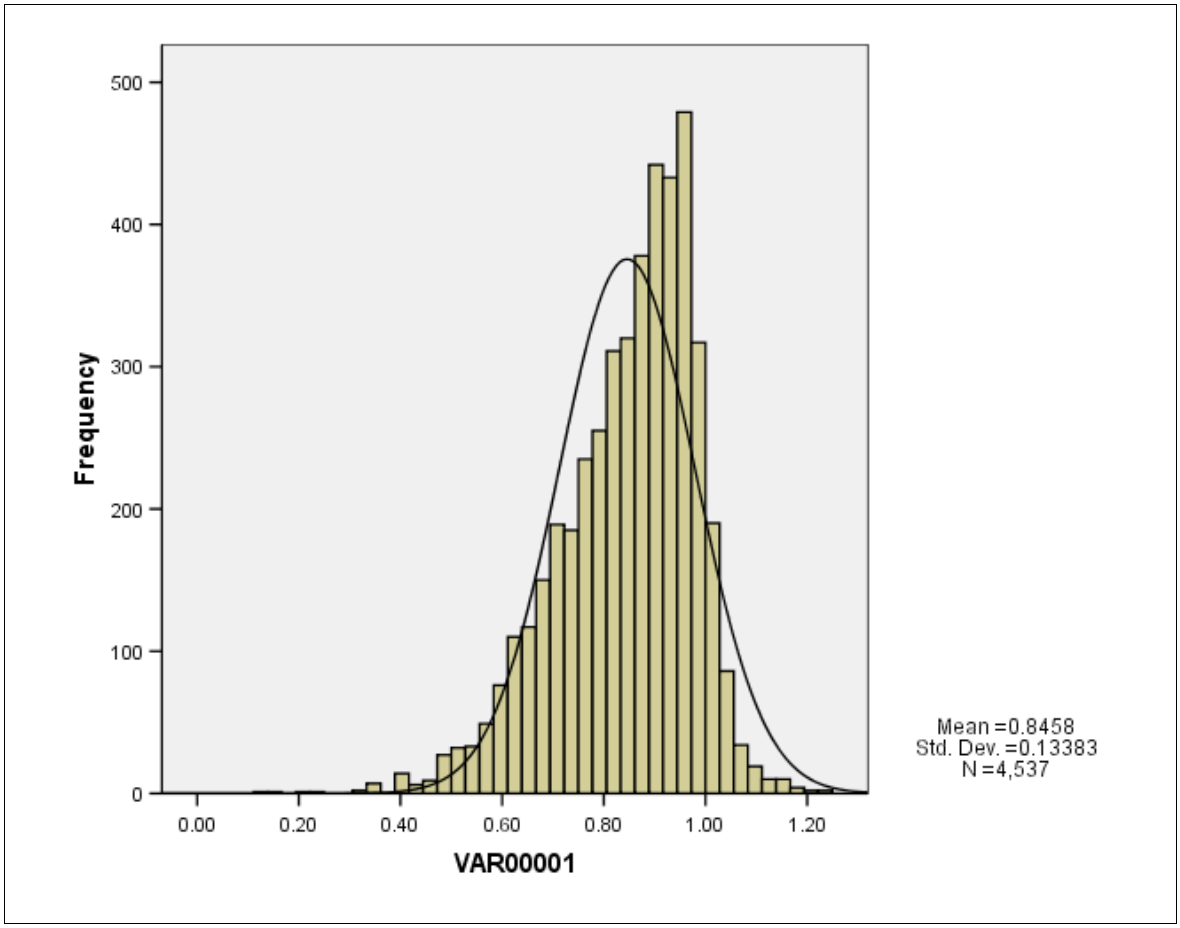

Figure 7.10 Global distribution of water body final EQR

Table 7.8 Characteristics of extremities of final EQR frequency distribution based on currently available surveys

Percentile	EQR
Min	0.11
0.1	0.27
0.5	0.41
1	0.48
2.5	0.54
5.0	0.60
95	1.02
97.5	1.04
99	1.08
99.5	1.13
99.9	1.18
Max	1.24

Rescaling is somewhat arbitrary but was based on the following approach. In addition to the considerations raised above, other rules were imposed for statistical and practical purposes.

i. The upper and lower limits for rescaling should not fall significantly inside the range of mean values for a site obtained from replicate surveys which might be subsequently used for uncertainty assessment.

ii. The values chosen should, for ease of subsequent use, lead to a simple transformation of existing class boundary values to 'whole' numbers (0.91 to 0.80 rather than 0.797371).

iii. The range should be as narrow as reasonably possible to maximise the width of each status class.

iv. Provided the full range of classes can be justified ecologically, transformation to a simple set of equally placed standard boundaries, with H/G at 0.8, would help subsequent amalgamation of classifications for different quality elements. The present step is a convenient point to implement this rescaling.

Taking these criteria into account, rescaling to zero to one was based on

$$\text{Std EQR} = (\text{ObEQR} - \min(0.43))/(\max(1.03) - \min(0.43))$$

and the small number of surveys then returning negative values or above one were capped at zero or one respectively. The implications for the class boundary values are shown below (Table 7.9). In subsequent parts of this report, the EQR values referred to are exclusively in this standardised form.

Table 7.9 Class boundary values following transformation to a zero-one scale

Class boundary	EQR	Std EQR
H/G	0.91	0.8
G/M	0.79	0.6
M/P	0.67	0.4
P/B	0.55	0.2

This process does not change the relative positions of the original class boundaries and the rationale for their placement. If so wished, the rescaling process can be seen merely as a step in the process for expressing confidence of class.

Although this rescaling was carried out subsequent to intercalibration of the UK lake macrophyte method within the Northern and Central-Baltic GIGs, rescaling did not alter the class of a site or the position of the class boundaries. Consequently it had no effect on intercalibration, whether undertaken via Option 3 (CGIG) or Option 2 (NGIG).

7.9 Application to environmental standards for nutrients in lakes

7.9.1 Pressure-response relationships

The relationship between LMNI and lake TP was presented in Figure 4.3. This looks ostensibly like a pressure-response relationship but in fact is somewhat deceptive. This is because the relationship is driven primarily by the natural gradient in macrophyte composition found in lakes distributed across the full natural productivity gradient (where productivity is determined by supply of phosphorus from 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. 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 TP) will never occur at an individual site owing to environmental constraints. Consequently, the true biological response is a much dampened version of that which might be expected from Figure 4.3. As an analogy, one could consider plotting phosphorus concentrations of a large number of lakes against the proportion of tilled land in the catchment of each lake. Although the concentration of phosphorus will undoubtedly increase with increasing tillage, the increasing 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 lake P signal associated purely with 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. This analysis tests whether there is a relationship between a pressure and the biological response once covariation between the response and driving variables, such as alkalinity, has been removed. It seems arbitrary whether this covariation is removed when first developing metrics (for example, by minimising the covariation between a metric, such as LMNI, and alkalinity) or at a later stage (by incorporating alkalinity as a predictor of values of metrics expected at reference condition, and 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 for 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 raise both the

alkalinity and TP of a lake from its background levels. Variance partitioning would exclude this shared element of the response.

Relationships between metrics and pressures are useful for diagnostic purposes. For example, LMNI 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 lake types, thereby overcoming weaknesses inherent in single metric-based classification. Although LMNI might be seen as the primary measure of macrophyte response to nutrient enrichment other metrics, such as richness, can modify the signal provided by composition alone. Thus, species-poor, low-alkalinity lakes with little indication of enrichment may actually be acidified, while high species-richness of base-rich lakes will to some extent mitigate against compositional changes. Consequently, it is most instructive to explore the relationship between a pressure and the 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.

Raw metric values (such as LMNI or N_TAXA) can be evaluated 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, 50ug/l in a high-alkalinity shallow lake but at the same nutrient concentration a low-alkalinity deep lake 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 alkalinity limits of that lake type.

Figure 7.11 attempts to represent the eutrophication pressure-macrophyte response relationship by plotting lake P concentrations relative to the MEI for the lake, with macrophyte classes overlain. As a basic minimum one would expect a worsening of macrophyte class with increasing P for a given MEI value and that macrophyte classes should show some separation and be arranged in a consistent manner. Figure 7.11 largely confirms these patterns and is therefore a simple graphic indication of a likely underlying pressure-response relationship.

To determine statistically whether there was a significant pressure-response relationship the final EQR was used to predict lake TP, having first fit the effect of MEI and GIG type (a dummy variable used to discriminate between high-alkalinity lakes in CB-GIG on soft calcareous geologies from those in N-GIG or A-GIG on hard limestones). A range of other model terms were applied (including lake altitude or area to substitute for the absence of depth data, or separation of MEI into its component terms) but the MEI and GIG type model produced the global model with the lowest prediction error and offered the most logical evolution to the derivation of environmental standards for different lake types. Prior to fitting the model, checks were carried out to identify any major TP outliers by correlating TP and MEI on a class-specific basis. At this point, 32 spuriously high values of TP (more than two times the standard deviation of the residuals) out of a total of 1,022 values were removed from the dataset.

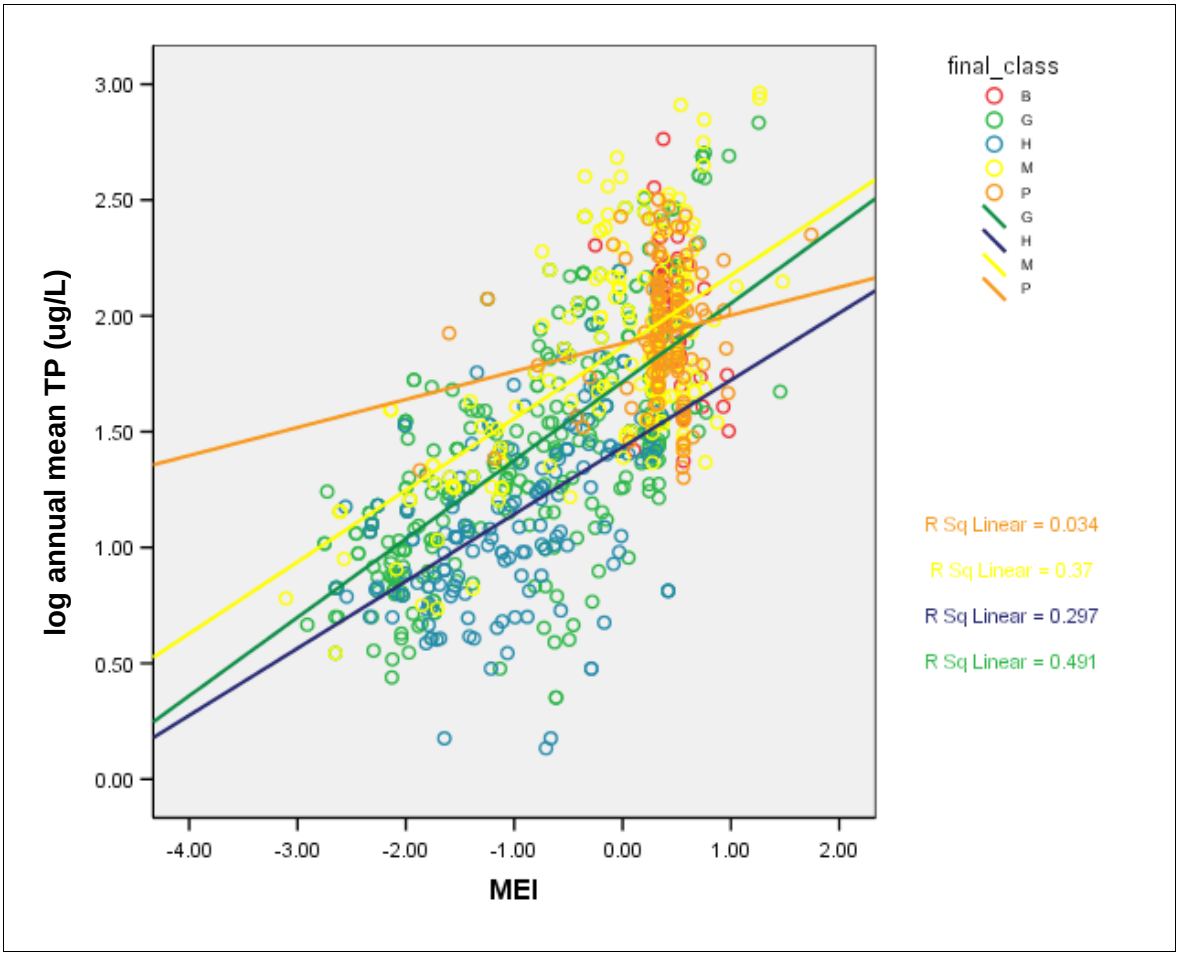

Figure 7.11 Lake TP versus MEI stratified by macrophyte-based ecological status. All plotted lines are significant at p = 0.01. No relationship was found for bad status sites since these were confined to high values of MEI.

The results of the analysis are summarised in Table 7.10. As would be expected MEI is a highly efficient predictor of TP, with the MEI and GIG type together explaining 55 per cent of the variation in TP. Macrophyte EQR explains an additional two per cent of variation, although this is still highly significant (p = <0.0001). While the amount of variation in P 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 that is left unexplained after fitting MEI and GIG type, which are likely to be the drivers of lake P. Given that, at best, two-thirds of the total variation in TP could be explained by a full suite of predictors, the two per cent explained by macrophytes represents about 20 per cent (two per cent of an additional 10 per cent) of the total that could be explained after fitting MEI and GIG type. This two per cent 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 lake but even this variable only explains three per cent of the variation in TP after fitting MEI. Interestingly, the component of variation in TP that can be explained by macrophytes is virtually undiminished after fitting the land cover variable which points to macrophytes having unique indicative value. Macrophytes themselves are not especially poor predictors of nutrient status (or vice versa), as has sometimes been asserted (see Demars and 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 the limited diagnostic potential of macrophytes 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 TP. 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 smaller than that used in this study. While it is possible that such analyses will reveal a stronger link than found here, no relationship at all may be found if, for example, the range of lake types, MEI, phosphorus gradient or EQR is rather narrow.

Having demonstrated statistically the unique and significant relationship between TP and lake macrophytes, the global relationship was refined to derive environmental standards. Two separate models were developed, one based on northern-Atlantic lakes in which MEI, marl (zero or one) and final EQR were used as model terms, and the second for southern-continental type lakes in which MEI and final EQR were the only terms used (marl lakes are restricted to hard limestones in the northern-Atlantic area of the UK). In the second model, low- and moderate-alkalinity lakes across the UK were included to maximise the gradient of TP and macrophyte EQR but northern-Atlantic high- or very high-alkalinity lakes and marl lakes were excluded. Both models were first screened for outliers as described above. Through this analysis, it is apparent from Table 7.10 that the macrophyte-TP signal is much stronger amongst the generally less fertile pool of northern-Atlantic lakes (pure effect of 13 per cent) than in the more fertile southern-continental lakes (pure effect of three per cent). This presumably reflects the range of additional buffering mechanisms that operate most efficiently at higher fertility (such as uptake of P by lake macrophytes, lag responses to water column P due to sediment nutrient pools, maintenance of clear water phase at high P by zooplankton grazing).

7.9.2 Deriving environmental standards for phosphorus in lakes

Having established models relating lake TP to core typing variables (alkalinity and depth via MEI) and shown that macrophyte EQR is a significant additional term in these models, such models can be used to predict lake TP values associated with a particular EQR for any value of MEI. Consequently, these models can be used to predict values of TP associated with class boundaries in each lake type. For example, the median values of alkalinity and depth found across the UK resource of high-alkalinity shallow lakes could be used to derive an MEI value and then used, in conjunction with an EQR of 0.8, to estimate the average TP at the H/G boundary in this lake 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 lake type, since it represents the highest predicted value of TP that a lake 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 G/M boundary. Hence, for a typical very high-alkalinity, very Shallow, southern-continental lake TP concentrations of 52 and 66 µg/l would typically be required to support macrophytes at high or good ecological status respectively. Such standards would be less suitable as targets (for example, for restoration) and the TP associated with the EQR in the middle of a class might be better for this purpose. Thus one would use the predicted TP associated with the middle of high or good status (EQR of 0.9 or 0.7 respectively).

TP predicted by global and GIG-specific models was determined and compared. The GIG-specific models produced a superior spread of TP values across the EQR range for each lake type and slightly more precautionary values at higher EQRs more closely aligned with the environmental standards of TP for lakes based on TP-chlorophyll relationships (UKTAG, 2006). The full set of standards for phosphorus for a range of lake types, based on their macrophyte response, is presented in Table 7.11.

Table 7.10 Models relating lake TP to morpho-edaphic index and macrophyte EQR to test existence of unique relationship between macrophyte ecological status and phosphorus

Terms	Global model			Northern-Atlantic model			Southern-continental model		
	coefficient	s.e.	r^2	coefficient	s.e.	r^2	coefficient	s.e.	r^2
Constant	1.811336	0.052733		2.345146	0.080976		2.001351	0.034485	
MEI	0.229699	0.017857	0.503	0.27205	0.018222	0.268	0.300711	0.012519	0.587
CGIG = 1	0.246121	0.037743	0.545						
Marl = 1				-0.13083	0.054884	0.277			
Overall EQR	-0.41851	0.065482	0.564	-1.09431	0.104823	0.414	-0.50646	0.060947	0.620
SE			0.340583			0.303896			0.302925
n in model			934			464			782
removed at screening			32			11			32

All model terms except marl in northern-Atlantic model are significant at p = 0.0001.

Table 7.11 Standards for total phosphorus (ug/l) in lakes to support macrophytes at different ecological status

Alkalinity	Depth	MEI[1]	GIG	Ref[2]	HG	GM	MP
H	VSh	-0.048	A/N	17	29	47	78
H	Sh	-0.460	A/N	13	22	37	61
L	VSh	-1.398	A/N	7	12	20	34
L	Sh	-1.762	A/N	6	10	16	27
L	Deep	-2.362	A/N	4	7	11	18
M	VSh	-0.640	A/N	12	20	33	54
M	Sh	-1.103	A/N	9	15	24	40
Marl	VSh	0.001	A/N	13	22	36	60
Marl	Sh	-0.294	A/N	11	18	30	50
P	VSh	-1.203	A/N	8	14	23	38
P	Sh	-1.816	A/N	6	9	16	26
VH	VSh	0.137	A/N	19	32	53	88
VH	Sh	-0.107	A/N	17	28	46	76
H	VSh	-0.014	C	31	39	49	62
H	Sh	-0.409	C	24	30	38	47
VH	VSh	0.403	C	41	52	66	83
VH	Sh	-0.179	C	28	35	44	56

[1]Where MEI = $\log_{10}$(alkalinity (as meq/l)/mean depth (m)) and is based on the median alkalinity and depth of lakes in each type for which macrophyte surveys exist.
[2]Where 1.0, 0.8, 0.6 and 0.4 represent the EQR boundaries for Ref, HG, GM and MP.

The purpose of classification tools is not to predict 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. The relationships derived here and used to generate the figures in Table 7.11 are correlational, not causal, and have relatively poor precision. The values should be considered as indicative rather than prescriptive. Used for guidance purposes, increasing deviation above these values represents an increasing risk that a lake of a given type will not achieve high or good ecological status as a result of nutrient enrichment. Conversely achieving values below the thresholds given in Table 7.11 should in no way be interpreted as a guarantee that the desired ecological status will be attained.

7.10 Summary

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

ii. A conceptual basis for combining metrics that reflect structure, diversity and abundance is presented and options for combining metrics are explored.

iii. On the basis of intercalibration at Northern and Central GIG levels, a rule-based approach for combining metrics is set out to ensure harmonisation of UK lake classifications at high-good and good-moderate boundaries.

iv. Rescaling of metrics to a common class boundary system is required before different metrics can be combined.

v. This rule takes compositional information in the form of the metric LMNI and the highest of the EQRs for the two richness metrics to create an interim value. Where the richness EQR is higher than the LMNI EQR, the richness EQR is multiplied by a weighting factor dependent on baseline productivity and then added to the LMNI EQR. This product is then divided by the weighting factor plus unity. If the highest of the richness EQRs is less than the LMNI EQR, the mean of the LMNI EQR and the richness EQR is calculated.

vi. The consequences of the rule-based approach are that: (a) impoverished sites will have a lower final EQR for the same LMNI EQR; (b) sparsely vegetated sites will have a lower EQR than well-vegetated sites with the same composition, and (c) with increasing baseline productivity, diverse sites with an impacted composition will have a higher EQR than impoverished sites with the same LMNI. Low and moderate productivity sites will be unaffected.

vii. Based on the final EQR, high- and very high-alkalinity lakes, especially in England and Northern Ireland, exhibit a relatively high level of impact. In the low- and moderate-alkalinity lakes types, deep lakes appear to be more impacted on average than shallow lakes.

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

ix. Based on comparisons between different time periods, there is a high confidence that sites classified as high or good with survey data 10-20 years old will still achieve that status.

x. Two case studies are presented examining compositional changes in two lake sets since the nineteenth century. These highlight the difficulties of finding reference conditions even in older data and the changes that occurred in some groups of lakes in the early 1800s.

xi. An examination of interannual changes in the final EQR of an intensively monitored group of lakes in the Norfolk Broads highlights the large interannual fluctuations in EQR that occur in the poorest status lakes compared to the stability of good status lakes. Trends in several key lakes indicative of recovery or deterioration are also identified.

xii. A system is presented for rescaling the final EQR to a common boundary system and scale running from zero to one with class intervals of 0.2.

xiii. 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 even when this covariation is removed.

xiv. Using the morpho-edaphic index and subdividing the UK lake resource into northern-Atlantic and southern-continental lakes, standards are derived for TP to support macrophytes at high or good ecological status in a range of lake types.

8 A macrophyte-based guide to the ecological status of lakes in the UK

8.1 Background

The purpose of this chapter is to provide a biological perspective on the changes that occur in different lake types across a gradient of ecological quality. While a multimetric approach provides the necessary sensitivity to a range of pressures at different intensities, the end product may be difficult to visualize in terms of the basic biological features that discriminate poor or bad status water bodies from high or good status ones. Establishing a guiding image for good ecological status is a fundamental stage in planning and measuring the success of restoration measures (Palmer *et al*. 2005). This chapter provides a type-by-type portrait of the floristic changes across a quality gradient as indicated by the overall EQR.

8.2 Explanation of lake type accounts

To provide a measure of the relative importance of different taxa, the percentage frequency of each taxon in surveys belonging to a given class of lake type was determined. Because historical archive data is necessary to establish high status macrophyte assemblages in some lake types, the frequency analysis is based purely on the proportion of lakes in which a species occurs, irrespective of its abundance. In preparing these accounts it was necessary to omit some detail in the interests of presentation. Thus, no reference is made to taxa which occur in a maximum of five per cent of surveys of any class of a particular lake type. Taxa that occur in a maximum of five to ten per cent of surveys of any class of a particular lake type are mentioned in the text only where they are indicative of the upper or lower end of the quality gradient. Graphical presentation is restricted to species that occur in more than 7.5 per cent of surveys of at least one class of a particular lake type. Within this range, species are allocated to four ranges of frequency (below 7.5, 7.5-22.5, 22.5-47.5 and above 47.5 per cent) and colour-coded to give a rapid impression of their relative importance in the different classes of each lake type. In the case of taxa that are widely reported as an aggregate (*Callitriche*, *Nitella*, and *Chara*), the values in the tables for these aggregates refer to the aggregate records themselves *plus* all records that could be assigned to each aggregate. Therefore, the distribution of individual species should provide a basis for interpreting the composition of the aggregate across the quality gradient

For the purposes of preparing these accounts some subtley different lake types were combined, where keeping them separate would have resulted in less than 10 surveys of a given class for that lake type. Thus, the different depth classes of peat and marl lakes were merged, while the shallow (3-15 m average depth) lakes in moderate- and low-alkalinity types were combined with the small number of deep lakes (above 15m) in these types. In both cases, very shallow lakes (under three metres average depth) were kept separate. For high-alkalinity southern-continental lakes it was necessary to aggregate shallow and very shallow lakes, while the same was true for very high-alkainity northern Atlantic lakes. Accounts are provided for a total of 12 lake ‘types’. In

all cases, the number of surveys of water bodies in bad status was small and for the purposes of this exercise, surveys of this type were aggregated with poor status.

The accounts deal primarily with compositional change. As a bridge to the classification system, the median metric values for LMNI, N_TAXA and N_FG in each class in each lake type are presented in Table 8.1. To further evaluate the overall indicator value of each taxon, the average overall EQR of all the surveys in which a taxon was found is indicated. Taxa are ranked in decreasing order of this value. In interpreting the tables for each lake type, it should be borne in mind that the survey data originates mainly from the early to mid-1990s. Consequently, the contribution of invasive alien taxa (such as *Crassula helmsii* and *Elodea nuttallii*) that are now more widely distributed in UK lakes may appear to be underestimated.

8.3 Lake type accounts

8.3.1 Peaty lakes

Peaty lakes, as recognised here, are effectively oligotrophic, low-alkalinity lakes within catchments dominated by blanket peat. Although such sites have the potential to be coloured they are not exclusively so and coloured lakes are not restricted to this lake type. Peaty lakes have a more predictable flora than almost any other lake type. Thus, high status examples of this lake type are characterised by a small number of taxa with very high constancy, such as; *Juncus bulbosus*, *Myriophyllum alterniflorum*, *Potamogeton polygonifolius*, *P. natans*, *Sparganium angustifolium*, aquatic Sphagna and the isoetids *Littorella*, *Lobelia* and *Isoetes lacustris* (Table 8.2).

Few strongly impacted surveys were returned for this type (a single poor status survey was aggregated with the moderate class). Acidification is likely to be the main pressure on this lake type. This is consistent with the marked reduction in richness at lower quality and the greater relative importance of the acidophile *Juncus bulbosus*. However, a reduction in the frequency of key marginal species such as *Menyanthes* and increase in nutrient-tolerant taxa are also suggestive of pressures such as shoreline modification, or enrichment from diffuse sources.

Additional species: Species characteristic of high status sites include *Eriocaulon aquaticum*, *Utricularia minor*, *U. ochroleuca* and *U vulgaris*, *Potamogeton gramineus*, *Sparganium natans* and the alga *Batrachospermum*. At the impacted end of the gradient, species include *Elatine hexandra*, *Ulva* (*Enteromorpha*), *Ruppia maritima*, *Ranunculus peltatus* subsp *baudottii* and *Zannichellia palustris*. This is suggestive of a slight brackish water influence, although this would need to be over and above the effect of distance from coast, since this variable is already used as a term in the prediction of reference LMNI.

Examples: Widely distributed across northern Scotland and the Northern Isles.

Table 8.1 Summary of changes in LMNI and richness metrics at different status in major UK lake types

Type		P/B	M	G	H
Peaty	n		18	59	121
	LMNI		4.1	4.2	3.9
	N_TAXA		1	4	8
	N_FG		1	3	5
Low alkalinity, very shallow	n	11	24	100	143
	LMNI	6.4	5.1	4.6	4.1
	N_TAXA	4	3	6	10
	N_FG	3	2	4	6
Low alkalinity, shallow-deep	n	13	71	449	785
	LMNI	5.8	5.0	4.4	3.9
	N_TAXA	2	4	9	10
	N_FG	2	3	6	6
Moderate alkalinity, very shallow	n	40	88	100	74
	LMNI	6.7	6.1	5.5	4.6
	N_TAXA	5.5	7	10	9
	N_FG	4.5	6	7	6
Moderate alkalinity, shallow-deep	n	36	113	245	317
	LMNI	6.7	6.0	5.3	4.3
	N_TAXA	6	9	12	12
	N_FG	5	7	8	7
Marl	n	9	64	94	35
	LMNI	7.3	6.8	6.6	6.0
	N_TAXA	4	9	10	13
	N_FG	3	6	7	8
High alkalinity, very shallow, northern-Atlantic	n	43	74	36	27
	LMNI	7.1	6.6	6.3	6.2
	N_TAXA	6	10	13.5	16
	N_FG	5	7	9.5	10
High alkalinity, shallow-deep, northern-Atlantic	n	33	53	66	46
	LMNI	7.0	6.5	6.0	5.0
	N_TAXA	7	8	12.5	11
	N_FG	5	6	9	8
High alkalinity, very shallow-shallow, southern-continental	n	25	59	70	32
	LMNI	7.1	7.0	6.7	5.9
	N_TAXA	3	6	10	10.5
	N_FG	3	5	7	8
Very high alkalinity, very shallow-shallow, northern-Atlantic	n	8	37	58	41
	LMNI	7.3	6.9	6.7	5.5
	N_TAXA	3.5	8	10.5	14
	N_FG	3	6	8	9
Very high alkalinity, very shallow, southern-continental	n	255	202	171	48
	LMNI	7.7	7.3	7.2	6.9
	N_TAXA	3	6	10	13
	N_FG	2	4	6	8
Very high alkalinity, shallow, southern-continental	n	7	26	17	24
	LMNI	7.7	7.3	7.0	6.1
	N_TAXA	2	7.5	13	11.5
	N_FG	2	6	9	7.5

Table 8.2 Compositional changes in vegetation in peaty lakes in the UK across a quality gradient

Class	av EQR	M	G	H
Eleocharis multicaulis	0.91			33.1
Utricularia intermedia sens.lat.	0.88			9.9
Eleogiton fluitans	0.88		1.7	10.7
Nymphaea alba	0.87		3.4	16.5
Lobelia dortmanna	0.87		15.3	68.6
Isoetes lacustris	0.87		15.3	43.0
Potamogeton polygonifolius	0.85	11.1	22.0	58.7
Sparganium angustifolium	0.85	16.7	33.9	57.0
Potamogeton natans	0.84	11.1	16.9	44.6
Potamogeton perfoliatus	0.84		8.5	14.9
Subularia aquatica	0.84		8.5	10.7
Sphagnum (aquatic indet.)	0.84	5.6	44.1	47.1
Littorella uniflora	0.83	16.7	50.8	71.9
Fontinalis antipyretica	0.83	5.6	27.1	33.9
Myriophyllum alterniflorum	0.83	16.7	40.7	62.0
Menyanthes trifoliata	0.83	22.2	33.9	56.2
Juncus bulbosus	0.82	55.6	91.5	100.0
Nitella spp	0.77	5.6	13.6	5.0
Callitriche spp	0.77	27.8	33.9	18.2
Callitriche hamulata	0.77	16.7	20.3	11.6
Callitriche stagnalis	0.76	22.2	18.6	10.7
Hippuris vulgaris	0.76	16.7	8.5	9.1
Nitella flexilis agg.	0.74	5.6	10.2	2.5
Potamogeton berchtoldii	0.68	11.1	1.7	1.7
Nuphar lutea	0.64	11.1	1.7	0.8
Ceratophyllum demersum	0.49	11.1		
Potamogeton pectinatus	0.41	11.1		
Ranunculus hederaceus	0.41	11.1		

Values represent percentage of surveys falling into a given class in which the taxa occurred. Solid black = above 50% frequency, dark grey = 25-50% frequency, light grey = 7.5-25% frequency.

8.3.2 Low- and moderate-alkalinity lakes

At high status there are many similarities with the vegetation of peaty lakes. The core component of *Juncus bulbosus*, *Littorella uniflora*, *Potamogeton polygonifolius*, *P. natans*, *Isoetes lacustris*, *Myriophyllum alterniflorum*, *Menyanthes* and *Sparganium angustifolium* are all highly likely to occur. However, richness is likely to be somewhat higher than the equivalent peaty lakes due to the frequent presence of taxa such as *Eleogiton fluitans*, *Subularia aquatica* and various species of *Utricularia*, *Nitella* and *Callitriche*. In contrast to more base-rich lakes *Chara* species are likely to be rare. At lower quality there is a marked reduction in the frequency of most species with high constancy in high status sites. The main exceptions are *Potamogeton natans* and *Nymphaea alba* which typify a shift to a nymphaeid-dominated community of which *Nuphar lutea*, virtually absent from high status sites, is a key element. Many of the other more frequent associates of lower status sites, such as *Persicaria amphibia*, *Potamogeton obtusifolius*, *P. perfoliatus*, filamentous algae, *Lemna minor* and *Ceratophyllum demersum* would normally be associated with elevated nutrient concentrations, but might also be favoured over isoetid vegetation by stabilisation of water levels. The ability for *Nuphar lutea* to persist almost in isolation in some acidified upland sites, as an alternative to acidophiles such as *Juncus bulbosus*, may also be evidence of a different type of pressure (Tables 8.3 – 8.6).

Additional species: A wide range of additional species are characteristic of high status sites. These include: *Utricularia ochroleuca* and *U. australis*, *Sparganium natans*, *Isoetes echinospora*, *Hypericum elodes*, *Potamogeton praelongus*, *Nitella opaca* and *Chara virgata*. *Ranunculus peltatus*, *Callitriche platycarpa* and *C. hermaphroditica* tend to be associated with lower status sites.

Distribution: Widespread, across a range of altitudes in NW and N Scotland, the outer Isles, Dumfries and Galloway, Cumbria, Pennines and mid and N Wales.

Table 8.3 Compositional changes in vegetation in low-alkalinity, very shallow lakes in the UK across a quality gradient

Class	av EQR	P/B	M	G	H
Utricularia stygia	0.96				7.7
Eleocharis multicaulis	0.92			3.0	35.7
Subularia aquatica	0.89			7.0	23.8
Eleogiton fluitans	0.89		4.2	8.0	25.9
Lobelia dortmanna	0.88			28.0	71.3
Utricularia intermedia sens.lat.	0.87			2.0	11.9
Isoetes lacustris	0.87		4.2	21.0	48.3
Utricularia minor	0.87			9.0	20.3
Batrachospermum	0.86		4.2	3.0	9.1
Potamogeton polygonifolius	0.86		8.3	44.0	79.0
Sparganium angustifolium	0.85			37.0	44.8
Menyanthes trifoliata	0.84	18.2	16.7	34.0	58.0
Utricularia spp	0.84			4.0	7.7
Juncus bulbosus	0.84		45.8	66.0	92.3
Myriophyllum alterniflorum	0.83	9.1	25.0	39.0	60.1
Utricularia vulgaris	0.83			4.0	7.7
Sphagnum (aquatic indet.)	0.83		20.8	37.0	37.1
Potamogeton natans	0.83	27.3	16.7	43.0	62.2
Chara spp	0.83		4.2	5.0	12.6
Littorella uniflora	0.82	9.1	29.2	59.0	74.1
Nitella spp	0.82		8.3	21.0	26.6
Luronium natans	0.82			7.0	7.7
Nitella translucens	0.80		8.3	5.0	7.0
Fontinalis antipyretica	0.80		16.7	29.0	25.2
Potamogeton berchtoldii	0.80	9.1		6.0	6.3
Nitella flexilis agg.	0.79			9.0	9.8
Nymphaea alba	0.79	27.3	12.5	28.0	30.8
Callitriche hamulata	0.76	27.3	16.7	28.0	19.6
Potamogeton perfoliatus	0.76	18.2	4.2	3.0	7.0
Callitriche spp	0.75	27.3	37.5	38.0	24.5
Apium inundatum	0.74		8.3	4.0	4.2
Sparganium emersum	0.73	9.1		2.0	2.8
Callitriche stagnalis	0.72	9.1	12.5	13.0	6.3
Potamogeton alpinus	0.69	9.1		2.0	1.4
Filamentous algae	0.65	36.4	12.5	17.0	6.3
Elodea canadensis	0.64	18.2	4.2	6.0	1.4
Nuphar lutea	0.63	54.5	20.8	15.0	3.5
Ranunculus hederaceus	0.58	9.1		2.0	
Ranunculus aquatilis agg.	0.53	9.1	4.2	1.0	
Potamogeton obtusifolius	0.52	18.2		3.0	
Nymphaea (exotics)	0.50	9.1		1.0	
Persicaria amphibia	0.46	18.2	4.2	2.0	
Lemna minor	0.44	45.5	12.5	4.0	
Potamogeton crispus	0.41	9.1		1.0	
Ceratophyllum demersum	0.23	18.2			
Lemna trisulca	0.18	9.1			
Callitriche obtusangula	0.14	9.1			

Table 8.4 Compositional changes in vegetation in low-alkalinity, shallow-deep lakes in the UK across a quality gradient

Class	av EQR	BP	M	G	H
Eleocharis multicaulis	0.89		4.2	9.8	43.3
Utricularia minor	0.88			6.9	16.6
Utricularia intermedia sens.lat.	0.87			4.9	12.4
Eleogiton fluitans	0.87		4.2	12.0	28.3
Potamogeton polygonifolius	0.85		4.2	49.2	73.5
Lobelia dortmanna	0.85		8.5	54.6	83.6
Subularia aquatica	0.84		2.8	20.5	25.0
Sparganium angustifolium	0.83	7.7	11.3	43.2	51.8
Potamogeton natans	0.83	23.1	21.1	51.4	62.4
Menyanthes trifoliata	0.83	7.7	15.5	38.1	44.7
Isoetes lacustris	0.83	7.7	28.2	51.0	57.7
Myriophyllum alterniflorum	0.82	15.4	29.6	66.6	69.4
Juncus bulbosus	0.82	15.4	47.9	89.3	95.7
Nymphaea alba	0.82	23.1	11.3	24.5	28.3
Littorella uniflora	0.82	23.1	57.7	81.5	87.5
Sphagnum (aquatic indet.)	0.82		19.7	28.5	26.6
Utricularia vulgaris	0.81	7.7	2.8	10.9	8.9
Chara spp	0.80	7.7	2.8	12.5	9.9
Sparganium natans	0.80		7.0	7.6	6.6
Nitella translucens	0.80	7.7		6.0	3.4
Potamogeton lucens	0.79	7.7			0.5
Nitella spp	0.78	7.7	26.8	38.1	21.7
Fontinalis antipyretica	0.78	23.1	49.3	46.1	30.7
Potamogeton alpinus	0.77	7.7	1.4	7.1	3.2
Callitriche spp	0.77	46.2	47.9	48.1	24.7
Callitriche stagnalis	0.77	15.4	14.1	19.4	8.5
Callitriche hamulata	0.77	30.8	38.0	37.9	17.7
Nitella opaca	0.77		7.0	12.0	5.0
Nitella flexilis agg.	0.76		11.3	13.8	5.4
Elatine hexandra	0.75		8.5	11.1	2.9
Potamogeton berchtoldii	0.75	7.7	11.3	13.4	4.6
Nuphar pumila	0.73	7.7	1.4	3.1	0.8
Nuphar lutea	0.72	23.1	12.7	10.5	3.1
Elodea canadensis	0.70	7.7	5.6	4.7	1.3
Sparganium emersum	0.70	15.4	4.2	6.0	1.0
Potamogeton obtusifolius	0.70	7.7	2.8	3.6	0.6
Hippuris vulgaris	0.70	7.7	4.2	2.7	0.8
Filamentous algae	0.69	38.5	23.9	18.9	2.0
Myriophyllum spicatum	0.65	7.7	1.4	0.7	0.4
Ranunculus aquatilis agg.	0.65	7.7	5.6	0.9	0.3
Lemna minor	0.62	7.7	7.0	1.3	0.3
Callitriche hermaphroditica	0.62	7.7		0.2	0.3
Potamogeton crispus	0.59	7.7	2.8	0.9	0.1
Riccia fluitans	0.46	15.4	1.4	0.2	
Potamogeton filiformis	0.46	7.7		0.2	
Elatine hydropiper	0.44	7.7	1.4		
Lemna trisulca	0.39	7.7			
Potamogeton pectinatus	0.37	7.7	1.4		

Table 8.5 Compositional changes in vegetation in moderate-alkalinity, very shallow lakes in the UK across a quality gradient

Class	av EQR	BP	M	G	H
Utricularia minor	0.94			2.0	18.9
Eleocharis multicaulis	0.92			2.0	28.4
Utricularia spp	0.86		2.3	10.0	32.4
Eleogiton fluitans	0.85		2.3	6.0	18.9
Sphagnum (aquatic indet.)	0.85		2.3	10.0	24.3
Lobelia dortmanna	0.85			12.0	31.1
Utricularia australis	0.85			2.0	8.1
Potamogeton polygonifolius	0.82	2.5	4.5	27.0	62.2
Juncus bulbosus	0.81		6.8	40.0	74.3
Sparganium natans	0.80		4.5	7.0	14.9
Hypericum elodes	0.79			9.0	4.1
Isoetes lacustris	0.79		1.1	13.0	17.6
Menyanthes trifoliata	0.79		11.4	28.0	47.3
Nitella opaca	0.78		2.3	9.0	10.8
Potamogeton x nitens	0.77		1.1	5.0	8.1
Potamogeton gramineus	0.77		4.5	7.0	14.9
Sparganium angustifolium	0.76		8.0	26.0	31.1
Myriophyllum alterniflorum	0.76	2.5	15.9	48.0	55.4
Chara virgata	0.74	5.0	3.4	10.0	16.2
Nitella translucens	0.73		3.4	11.0	10.8
Littorella uniflora	0.72	10.0	33.0	58.0	58.1
Chara spp	0.70	17.5	10.2	25.0	29.7
Hippuris vulgaris	0.70		5.7	10.0	4.1
Eleocharis acicularis	0.70		2.3	11.0	2.7
Elatine hexandra	0.70		2.3	11.0	1.4
Nitella spp	0.69	12.5	21.6	38.0	35.1
Potamogeton natans	0.68	25.0	51.1	51.0	67.6
Nymphaea alba	0.68	10.0	29.5	25.0	31.1
Potamogeton berchtoldii	0.68	7.5	23.9	37.0	25.7
Apium inundatum	0.68	2.5	12.5	16.0	10.8
Potamogeton perfoliatus	0.66	12.5	10.2	25.0	14.9
Elodea nuttallii	0.66	2.5	2.3	11.0	1.4
Callitriche hamulata	0.65	15.0	34.1	39.0	28.4
Nitella flexilis agg.	0.65	10.0	13.6	16.0	14.9
Ranunculus peltatus subsp *peltatus*	0.64	5.0	5.7	9.0	5.4
Ranunculus spp	0.63	15.0	21.6	24.0	14.9
Potamogeton alpinus	0.63	5.0	23.9	17.0	10.8
Callitriche stagnalis	0.62	17.5	21.6	24.0	12.2
Potamogeton obtusifolius	0.62	15.0	27.3	28.0	12.2
Fontinalis antipyretica	0.62	20.0	43.2	33.0	20.3
Ranunculus aquatilis agg.	0.62	7.5	11.4	13.0	4.1
Callitriche spp	0.61	57.5	54.5	58.0	35.1
Persicaria amphibia	0.60	22.5	21.6	21.0	10.8
Elodea canadensis	0.59	47.5	28.4	33.0	17.6
Myriophyllum spicatum	0.57	10.0	8.0	9.0	2.7
Potamogeton crispus	0.56	25.0	17.0	14.0	6.8
Nuphar lutea	0.56	37.5	44.3	25.0	9.5
Sparganium emersum	0.56	17.5	38.6	11.0	8.1
Filamentous algae	0.55	25.0	26.1	23.0	2.7
Lemna minor	0.55	60.0	50.0	28.0	14.9
Callitriche hermaphroditica	0.55	10.0	6.8	5.0	2.7

Callitriche platycarpa	0.50	10.0	4.5	2.0	1.4
Lemna trisulca	0.49	30.0	9.1	5.0	2.7

Table 8.6 Compositional changes in vegetation in moderate-alkalinity, shallow-deep lakes in the UK across a quality gradient

Class	av EQR	BP	M	G	H
Utricularia intermedia sens.lat.	0.91			2.4	16.4
Eleocharis multicaulis	0.90		0.9	6.1	43.5
Eleogiton fluitans	0.89		1.8	6.5	34.7
Utricularia minor	0.89		1.8	3.3	16.4
Subularia aquatica	0.88		1.8	6.1	18.6
Utricularia spp	0.86		4.4	16.3	46.4
Potamogeton polygonifolius	0.86	2.8	8.0	27.3	73.2
Lobelia dortmanna	0.86		10.6	27.8	67.5
Sphagnum (aquatic indet.)	0.85	2.8	2.7	6.1	15.5
Juncus bulbosus	0.83	5.6	14.2	62.9	91.5
Isoetes lacustris	0.83		13.3	29.0	43.5
Sparganium angustifolium	0.82		14.2	31.8	50.5
Sparganium natans	0.82		7.1	10.6	16.1
Myriophyllum alterniflorum	0.81	2.8	27.4	68.6	82.6
Chara virgata	0.80	8.3	9.7	23.3	25.9
Menyanthes trifoliata	0.80	11.1	23.9	41.6	55.5
Potamogeton natans	0.80	16.7	44.2	50.2	77.0
Nymphaea alba	0.79	8.3	28.3	31.8	45.1
Littorella uniflora	0.79	13.9	57.5	78.8	90.2
Chara spp	0.78	25.0	22.1	34.7	44.5
Nitella translucens	0.78		8.0	10.2	9.8
Potamogeton gramineus	0.77	8.3	11.5	24.1	19.9
Nitella spp	0.76	16.7	40.7	49.0	43.2
Nitella opaca	0.76	5.6	8.8	15.1	11.4
Potamogeton x nitens	0.75		8.0	13.1	9.5
Fontinalis antipyretica	0.75	11.1	31.9	49.4	34.4
Apium inundatum	0.75	2.8	7.1	15.1	8.2
Potamogeton perfoliatus	0.74	13.9	33.6	39.2	30.3
Potamogeton alpinus	0.74		21.2	24.1	12.0
Nitella flexilis agg.	0.73	5.6	24.8	23.3	15.5
Callitriche hamulata	0.72	11.1	35.4	40.4	19.6
Lythrum portula	0.72		8.8	13.9	4.4
Elatine hexandra	0.71	2.8	13.3	11.4	5.0
Callitriche spp	0.70	47.2	61.9	59.2	31.5
Callitriche stagnalis	0.70	30.6	31.0	32.7	17.4
Hippuris vulgaris	0.70	5.6	7.1	13.9	3.5
Sparganium emersum	0.69	5.6	13.3	13.5	6.3
Potamogeton berchtoldii	0.69	13.9	40.7	38.4	16.1
Ranunculus spp	0.68	16.7	26.5	24.1	7.9
Potamogeton obtusifolius	0.68	11.1	23.9	15.9	6.0
Ranunculus peltatus subsp *peltatus*	0.66	11.1	15.9	10.6	3.8
Persicaria amphibia	0.65	22.2	23.9	20.4	5.4
Nuphar lutea	0.64	30.6	38.1	22.0	7.3
Myriophyllum spicatum	0.63	19.4	11.5	6.5	3.2
Elodea canadensis	0.63	44.4	46.0	33.1	7.6
Callitriche hermaphroditica	0.62	27.8	23.9	14.3	4.4
Potamogeton pectinatus	0.60	19.4	7.1	5.3	1.3
Lemna minor	0.60	41.7	37.2	23.3	4.1
Elodea nuttallii	0.60	5.6	14.2	6.1	0.3
Potamogeton crispus	0.60	27.8	25.7	13.9	1.9
Potamogeton pusillus	0.58	27.8	20.4	7.3	1.9
Lemna trisulca	0.57	11.1	10.6	4.9	0.3
Filamentous algae	0.57	16.7	31.0	10.2	0.9
Zannichellia palustris	0.51	22.2	11.5	0.8	0.6
Ceratophyllum demersum	0.50	11.1	0.9	1.6	

8.3.3 High-alkalinity, shallow-very shallow, southern-continental lakes

The most obvious feature of high status sites is the presence of a diverse, shallow water flora, often characteristic of mesotrophic sites with naturally fluctuating water levels, and typified by *Littorella uniflora, Myriophyllum alterniflorum* and *Potamogeton gramineus*, growing alongside a range of more widely distributed floating-leaved and submerged open water species, such as *Nuphar lutea*, *Nymphaea alba* and *Potamogeton natans*, various stoneworts (especially *Chara virgata*), *Myriophyllum spicatum* and *Potamogeton perfoliatus*. Deeper marginal areas are likely to feature species such as *Persicaria amphibia* or *Menyanthes trifoliata*. The more characteristic species of the northern-Atlantic equivalent of this lake type, such as *Isoetes lacustris*, *Lobelia dortmanna* and *Sparganium angustifolium* are rare or absent. Many of those species characteristic of high status sites have decreased markedly across lowland England and Wales over the past century. Maps provided by the Botanical Society of the British Isles (http://www.bsbimaps.org.uk/atlas/main.php) offer a range of examples. The major representatives in impacted sites are likely to be a small subset of species which were relatively frequent in high status sites. These include *Lemna minor*, *Elodea canadensis*, *Potamogeton crispus*, *Nuphar lutea* and *Zannichellia palustris*. The lack of a more distinctive poor quality endpoint in this lake type may reflect the wide range of pressures to which these water bodies are exposed (eutrophication, water level regulation, tree encroachment, invasive species, shoreline modification). It may also reflect a strong local geographical influence on the species pool which explains the wide-ranging composition of high status sites (Table 8.7).

Additional species: *Lobelia dortmanna*, *Sparganium natans*, *Pilularia globulifera*, *Chara hispida*, *Potamogeton x nitens* and *Eleocharis multicaulis* are associated with high ecological status. By contrast *Elodea nuttallii*, *Callitriche platycarpa* and *C. obtusangula* are associated with moderate ecological status.

Distribution: The West Midlands Meres provide the classic example of this lake type. High status conditions in the majority of cases are derived from nineteenth century records.

Table 8.7 Compositional changes in vegetation in high-alkalinity, southern-continental lakes in the UK across a quality gradient

Class	av EQR	P	M	G	H
Eleogiton fluitans	1.00				9.4
Baldellia ranunculoides	1.00				12.5
Potamogeton gramineus	0.94				31.3
Luronium natans	0.91				9.4
Myriophyllum alterniflorum	0.90			5.7	25.0
Potamogeton polygonifolius	0.86		1.7	1.4	9.4
Apium inundatum	0.85		1.7	5.7	18.8
Juncus bulbosus	0.84			7.1	21.9
Potamogeton filiformis	0.82			4.3	21.9
Sphagnum (aquatic indet.)	0.82			2.9	12.5
Elatine hexandra	0.80		3.4	2.9	15.6
Chara aspera	0.79		1.7	7.1	12.5
Littorella uniflora	0.78		6.8	24.3	56.3
Potamogeton natans	0.78		5.1	18.6	46.9
Potamogeton perfoliatus	0.75	4.0	5.1	11.4	21.9
Menyanthes trifoliata	0.75	4.0	6.8	32.9	46.9
Lythrum portula	0.74		1.7	2.9	9.4
Nitella flexilis agg.	0.74		6.8	11.4	15.6
Chara virgata	0.73	4.0	8.5	17.1	28.1
Nitella spp	0.72	4.0	10.2	20.0	28.1
Eleocharis acicularis	0.72	8.0	1.7	31.4	18.8
Nitella opaca	0.71	4.0	1.7	5.7	12.5
Ranunculus peltatus	0.71		3.4	4.3	12.5
Chara spp	0.71	8.0	27.1	42.9	59.4
Sparganium emersum	0.71		3.4	8.6	9.4
Potamogeton berchtoldii	0.70		11.9	32.9	25.0
Hippuris vulgaris	0.69		10.2	5.7	15.6
Fontinalis antipyretica	0.68	4.0	10.2	11.4	25.0
Callitriche stagnalis	0.68	4.0	13.6	31.4	28.1
Ranunculus aquatilis agg.	0.67	4.0	5.1	7.1	9.4
Potamogeton obtusifolius	0.66	8.0	6.8	21.4	15.6
Persicaria amphibia	0.65	20.0	45.8	70.0	46.9
Myriophyllum spicatum	0.65	16.0	22.0	31.4	31.3
Nymphaea alba	0.65	16.0	37.3	48.6	31.3
Lemna trisulca	0.65	8.0	18.6	35.7	15.6
Callitriche spp	0.64	16.0	28.8	38.6	37.5
Callitriche hermaphroditica	0.64	4.0	22.0	28.6	15.6
Elodea canadensis	0.64	20.0	32.2	58.6	25.0
Ranunculus hederaceus	0.63	4.0	3.4	8.6	
Nuphar lutea	0.63	24.0	40.7	51.4	28.1
Callitriche hamulata	0.63		10.2	8.6	6.3
Butomus umbellatus	0.62	8.0	1.7	4.3	6.3
Potamogeton crispus	0.62	28.0	22.0	27.1	18.8
Ulva (Enteromorpha)	0.61		6.8	8.6	3.1
Ranunculus circinatus	0.61	16.0	15.3	25.7	6.3
Potamogeton pusillus	0.61	16.0	27.1	20.0	18.8
Potamogeton pectinatus	0.61	12.0	37.3	30.0	15.6
Potamogeton lucens	0.60	8.0	5.1	4.3	6.3
Chara globularis	0.60	4.0	8.5	2.9	6.3
Zannichellia palustris	0.60	20.0	28.8	34.3	3.1
Chara vulgaris	0.59	8.0	1.7	1.4	3.1
Lemna minor	0.58	44.0	50.8	50.0	18.8
Oenanthe aquatica	0.57	8.0	6.8	1.4	3.1
Filamentous algae	0.57	12.0	23.7	12.9	6.3
Ceratophyllum demersum	0.56	12.0	20.3	7.1	3.1
Spirodela polyrhiza	0.30	12.0			

8.3.4 Very high-alkalinity, very shallow, southern-continental lakes

The overriding impression of vegetation change in this lake type is of the large scale erosion of diversity due to the progressive deletion of the large pool of mostly nutrient-tolerant, but competitively inferior species. Ultimately, all that remains is a combination of macroalgae, large nymphaeids, lemnids and fast-growing canopy-forming eloeids and pondweeds. Nevertheless, most of these tolerant taxa occur at a relatively high frequency even in the highest status sites. The most discriminating feature at high status is the presence of a diverse assemblage of charophytes (principally *Chara aspera*, *C. hispida, C. vulgaris and Nitellopsis*), alongside a small number of typically shallow water, floating-leaved or semi-emergent species (*Persicaria amphibia*, *Hippuris vulgaris*, *Potamogeton natans, Callitriche* spp) that are replaced in open water by submerged species such as *Myriophyllum spicatum* or *Najas marina*. The presence of bladderworts (most notably *Utricularia vulgaris*), a range of fine-leaved pondweeds, *Myriophyllum verticillatum*, *Ranunculus circinatus*, *Lemna trisulca*, *Stratiotes* and *Hydrocharis*, all of which would be more typically associated with small sheltered pools or ditches, points to a potentially key role of habitat 'engineering' by other macrophytes or larger emergent species (such as *Schoenoplectus lacustris* or *Typha angustifolia*) in providing suitable habitat. Collectively the picture is of a plant assemblage that is structurally highly complex. Only a small pool of characteristically nutrient-sensitive and typically amphibious species show a strict association with high status sites (*Littorella uniflora*, *Potamogeton gramineus*, *Apium inundatum*). According to the historical archive and macrofossil record, these species were never frequent. They represent the transition with somewhat lower alkalinity shallow lakes and may have been a more characteristic feature of small shallow pools that were periodically connected to the major water bodies (Table 8.8).

Additional species: *Myriophyllum alterniflorum*, *Eleogiton fluitans*, *Hypericum elodes*, *Baldellia ranunculoides*, *Juncus bulbosus*, *Eleocharis acicularis*, *Nitella flexilis* agg., *Chara virgata* and aquatic sphagna are all strongly associated with high ecological status. *Spirodela polyrhiza* is the only unlisted species to be associated predominantly with moderate or lower ecological status.

Distribution: The Norfolk Broads provide classic examples of this lake type in the UK.

Table 8.8 Compositional changes in aquatic vegetation in very high-alkalinity, very shallow, southern-continental lakes in the UK, across a quality gradient

Class	av EQR	P/B	M	G	H
Utricularia minor	1.07				8.3
Potamogeton polygonifolius	1.05				10.4
Potamogeton gramineus	1.03				8.3
Littorella uniflora	1.01		0.5	0.6	8.3
Apium inundatum	0.98		0.5		8.3
Menyanthes trifoliata	0.91		1.0	1.8	20.8
Callitriche hamulata	0.90			1.2	8.3
Hottonia palustris	0.80		1.0	2.3	10.4
Sagittaria sagittifolia	0.79		1.0	4.1	14.6
Utricularia spp	0.78		1.5	1.8	16.7
Ranunculus trichophyllus	0.77		1.0	1.2	8.3
Potamogeton natans	0.76		4.0	6.4	31.3
Callitriche stagnalis	0.75	0.4	4.0	19.3	43.8
Butomus umbellatus	0.72	0.8	0.5	1.2	8.3
Hydrocharis morsus-ranae	0.72	0.8	3.0	6.4	18.8
Utricularia vulgaris	0.72	1.6	5.0	6.4	33.3
Potamogeton obtusifolius	0.71	0.4	2.0	8.2	10.4
Chara aspera	0.71	0.4	5.0	22.2	27.1
Callitriche spp	0.71	1.2	9.4	41.5	54.2
Sparganium emersum	0.71	0.4	2.0	1.8	8.3
Chara pedunculata	0.70		3.0	8.8	12.5
Potamogeton lucens	0.69	0.8	2.5	11.1	8.3
Myriophyllum verticillatum	0.68	1.6	3.5	7.6	16.7
Myriophyllum spicatum	0.67	1.2	15.3	45.0	43.8
Hippuris vulgaris	0.67	1.6	18.8	42.1	47.9
Ranunculus aquatilis agg.	0.67	0.4	3.0	3.5	10.4
Ranunculus circinatus	0.67	2.0	4.5	19.3	16.7
Chara hispida	0.67	1.2	9.9	29.8	22.9
Potamogeton perfoliatus	0.66		5.9	9.4	8.3
Nitellopsis obtusa	0.66	1.2	9.4	22.8	22.9
Persicaria amphibia	0.66	1.6	11.9	17.0	35.4
Chara globularis	0.65	2.4	8.9	27.5	22.9
Chara baltica	0.65		4.5	5.8	10.4
Chara intermedia	0.64	1.2	11.4	21.6	18.8
Nymphaea alba	0.64	4.3	36.1	39.8	58.3
Potamogeton berchtoldii	0.64	1.6	5.4	10.5	12.5
Stratiotes aloides	0.62	2.0	5.9	7.0	10.4
Nitella spp	0.62	2.7	6.4	7.0	16.7
Chara connivens	0.62	2.7	9.9	16.4	16.7
Chara vulgaris	0.60	5.5	9.9	17.0	22.9
Lemna trisulca	0.59	4.7	17.3	14.6	27.1
Chara spp	0.59	11.8	45.0	56.1	43.8
Elodea canadensis	0.59	16.5	27.7	45.6	45.8
Potamogeton friesii	0.59	4.3	11.4	11.1	14.6
Potamogeton pusillus	0.58	7.5	17.8	24.0	20.8
Fontinalis antipyretica	0.58	6.7	14.9	14.6	22.9
Nuphar lutea	0.58	22.4	44.6	60.2	58.3
Potamogeton pectinatus	0.56	22.4	44.1	58.5	52.1
Najas marina	0.55	17.6	21.8	40.4	35.4
Potamogeton crispus	0.54	13.7	25.7	26.9	31.3
Lemna minor	0.54	20.4	37.6	34.5	39.6
Elodea nuttallii	0.52	4.7	8.9	6.4	6.3
Zannichellia palustris	0.51	29.8	34.7	37.4	31.3
Filamentous algae	0.46	56.1	52.0	46.2	18.8
Ceratophyllum demersum	0.45	49.0	41.6	37.4	33.3
Ulva (Enteromorpha)	0.41	29.8	24.3	12.9	16.7

9 Uncertainty in ecological status assessments using lake macrophytes

9.1 Introduction

Macrophyte surveys generate data that can be used to derive certain metrics. These metrics, their value, how they can be predicted under reference conditions and how they are combined to provide a final class are described in previous chapters. If one was prepared to classify only the part of a site surveyed, at the time it was surveyed, and the data used could be collected without any measurement error, the EQR reported would have no uncertainty associated with it. In reality, when one wishes to classify a whole water body across a longer period of time and surveys are undertaken by different personnel under different conditions, a degree of uncertainty must be associated with the EQR reported. Thus, different surveyors may obtain subtly different data even from the same points in a lake on the same date. Different data may be obtained from a lake on a given date if transects were assigned to one set of locations as opposed to another. Different data may be obtained from a lake in two different years, even if those data were collected at the same locations and by the same surveyors. Consequently, once the variability associated with a face value EQR is taken into account, there is a risk that a site will be misclassified due to the difference between the 'true' EQR and the sample EQR. This risk depends on the scale of the different sources of error associated with that EQR.

From an investment perspective the risk of misclassification is important, especially close to the good-moderate boundary, because there is a risk, depending on the scale of error relative to the width of class, that a programme of measures may be initiated in a good status site that has been misclassified as moderate. To reduce this risk, it may be necessary to stipulate a high level of confidence (such as 95%) that the observed status is indeed the true status. Similarly, there is a risk that a water body classified as good may in reality be no better than moderate, and therefore requires restoration. Confidence of class also extend to the rules relating to no deterioration in class and the need to improve the class of failing sites.

For classification, a key task is to manage uncertainty to maximise the ratio of confidence of classification to the resource required for sampling. The primary requirement is therefore to establish the relative sources of variation in a final EQR and design a sampling protocol to ensure the main sources of variation are minimised. If sampling itself is a major source of variation independent of spatial and temporal sources, steps such as training, accreditation, paired working and quality assurance procedures may need to be introduced. If temporal sources account for significantly more variation than spatial sources, uncertainty can best be managed by conducting surveys in several different years than by carrying out intensive sampling at multiple locations within a water body at a given point in time.

Macrophyte composition and cover varies within and between sites according to parameters that are frequently not quantified and fall outside the normal suite of environmental predictors. These factors include shading, wave fetch, substrate characteristics, shoreline gradient and herbivory. The approach followed in the standard methodology is to survey transects in different lake sectors and to present

data for the water body as a whole based on a composite of these sectors. This mitigates local spatial variation in, for example, physical habitat. In terms of temporal variation, macrophytes are often relatively long-lived and form extensive vegetative clones. In this respect they are probably less responsive to short-term variations in factors such as nutrient supply than diatoms or phytoplankton. However, significant interannual fluctuations in macrophyte cover, as opposed to composition, may arise due to differences in climatic factors, including spring temperatures, winter ice cover, or the effect of these on feeding by flocks of waterfowl.

The derivation of uncertainty estimates for macrophyte surveys is beset with difficulties. The standard survey method has existed for a relatively short period and the water bodies surveyed do not adequately cover the EQR gradient. To provide coverage of sites of different type and status over a reasonable time frame requires the use of additional data from various sources, collected by slightly different methods and designed for different purposes. This is very different to obtaining data via a carefully planned, nested hierarchical experimental design of the sort reported by Jones *et al*. (2006) for assessing sources of variability associated with assessment of littoral invertebrates. Consequently a rigorous initial treatment of the data is required, while the estimates of error distribution obtained may need to be confirmed using more standard data at some stage in the future.

As a final point, error associated with expected metric values that form the basis for calculating an EQR is not considered here. As is the case with RIVPACS (Clarke, 2000) we regard the choice of reference sites, predictor variables and prediction method as an integral part of the definition of expected metric values. For variation in expected metric values associated with variation in predictor variables, we assume that this is negligible because predictor values can be measured without significant error (such as lake area and altitude) or for parameters such as alkalinity are (or should be) based on a long-term mean.

This chapter is therefore devoted to:

i. Deriving reasonable estimates of the different sources of error associated with a face-value EQR.

ii. Determining how this error is manifested in confidence of classification.

iii. Assessing how the resource available for macrophyte surveys is best deployed to minimise uncertainty.

9.2 Treatment of data and preliminary analysis

9.2.1 Treatment of data

The data available are not of an ideal standard derived from studies designed specifically to assess uncertainty. Thus, the replicate survey data for macrophytes covers, inter alia, spatial variability within some water bodies on a given date based on a variable number of samples, or variability in other lakes over time intervals of non-standard length and sampling density. In many cases replicate samples have not been collected by identical methods, the methods used may differ (usually quite subtly) from the standard protocol, and personnel undertaking the surveys differ widely. The data available and types of variation covered are listed in Table 9.1 below. To retain as much compatible data as possible whilst minimising the risk of overestimating uncertainty, several investigations of separate datasets or classes of data were carried out. These are described below.

Table 9.1 Description of groups of data, sample sizes and number of replicates according to different types of relevant variation

Description of data	No. sites	Samples per site	Sampling	Temporal (between years)	Local Spatial	Spatial	Between methods
Surveys by ENSIS to standard SCM protocol, July and September 2005	18	2	x				
Surveys by BA and for JNCC that have considered several basins separately within a WB on a given date	46	2-5	x			x	
Surveys by a single method that have assessed the same locations in a WB in two + years	48	2-5	x	x			
Surveys by a single method that have assessed the whole WB body in two + years	51	2-5	x	x	x		
Surveys that have assessed the whole WB in two + years but using different survey methods	104	2-6	x	x	x		x

In this study, only variation in the 'final EQR' derived from the contribution of different metrics is considered, not the variation in each of the metrics. Part of the justification for this is that the metrics are effectively derivatives of a common set of data collected by a standard survey approach while the variation in individual metrics is reflected in the placement of class boundaries. Different final EQRs may reflect a different blend of metrics, each with different error properties but given that the final EQR is heavily dependent on both LMNI and richness metrics in almost all cases, this is considered to be a minor additional source of variation.

9.2.2 Effect of time window on temporal variability

The current dataset covers sites where replicate samples have been collected over time periods ranging from two to 30 years. In fact, of 155 samples available for investigating temporal variability half concern sampling carried out over a time frame exceeding 10 years – well beyond the length of a WFD monitoring cycle. Moreover in 74 per cent of all cases these replicates comprise only duplicate samples. While this may reflect the availability of resources for repeat surveys, it may also reflect an underlying impression of the stability of lake macrophyte communities. Only one dataset, that collected by the Broads Authority, constitutes a true time-series dataset, with more or less annual sampling of 20 sites over a 20-year period.

The longer the time period over which samples are collected, the greater the likelihood that sample variability will capture a long-term underlying trend that extends beyond the variability one might expect within a single monitoring cycle. The effect of time window on sample variability was assessed using linear regression in which we assessed whether the standard deviation for a given mean EQR was dependent upon the time span of the window over which samples were averaged (that is, is the residual variation after relating EQR standard deviation to mean EQR dependent on time span). In the initial analysis of the global dataset (average time window of data was 12 years) the effect of time was highly significant ($p = 0.001$), see Figure 9.1. However, by filtering the data to sequentially reduce the time window over which repeat samples were averaged, the effect of time ceased to be significant ($p = 0.06$) once the averaging window was 11 years or smaller (average time window of dataset was six years). This dramatically reduced the data available for analysis, although where more than two samples existed within an averaging window above 11 years, it was possible to retrieve some 'lost' data by shortening the averaging window. This left a dataset of 109 sites with replicate sampling undertaken over a time period of two to 11 years (mean of six years), of which 90 per cent comprised duplicate samples only.

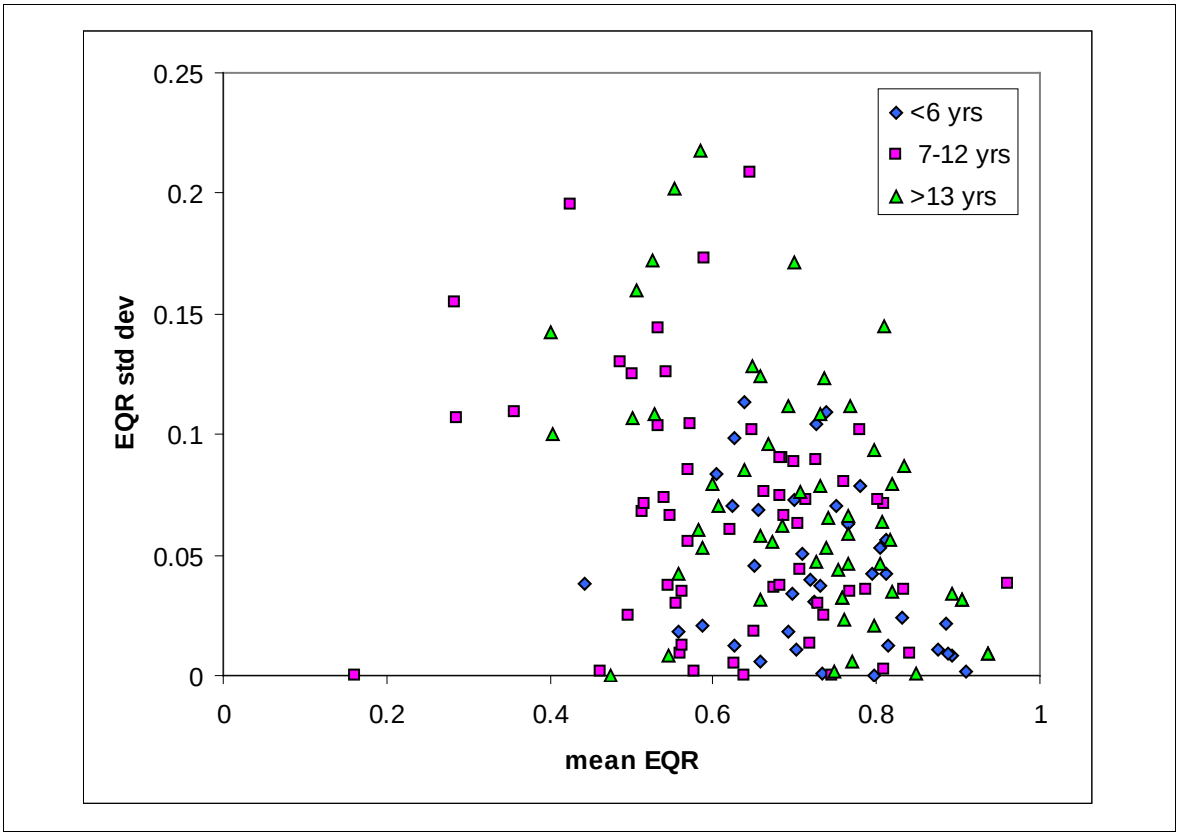

Figure 9.1 Effect of length of time window over which averages are derived on variation associated with a given mean EQR. Note that all cases of high variability for roughly the same EQR are associated with means based on samples collected from a period spanning more than seven years.

9.2.3 Effect of number of samples on temporal variability

Ideally the number of replicate surveys per lake would be fixed and preferably large (for example, each year in a monitoring cycle). Using this type of data one could assess the

effect on uncertainty of increasing the number of surveys of a given lake in a monitoring cycle. In reality, the number of replicates per lake is variable and typically confined to two. Prior to pooling data based on variable numbers of replicates, we assessed whether EQR SD for a given mean EQR was influenced by the number of replicates on which the mean was based. This analysis showed that EQR SD for a given mean EQR is not affected by n when it ranges between two and five ($p = 0.07$). Consequently all replicate data was considered in this analysis irrespective of the number of samples on which the average was based. From a statistical point of view, the uncertainty in the SD for a given EQR is considerable when the mean is derived from a small number of samples and the relationships derived below therefore depend on the weight of evidence provided by a relatively large number of independent data points. When a mean is based on just two samples, the only confidence one can take from the data is that for such sites the data, firstly, is rarely derived from consecutive years (11 per cent of data points in consecutive years; average time span between duplicate surveys is six years), and is thus likely to be truly independent. Secondly, a pair of sites are as likely to be similar as they are to be different and therefore equally likely to under- or overestimate the true standard deviation. While the retention of lakes in which the mean is based on two samples is not ideal it is necessary to achieve representation across the mean EQR gradient, while removing such data would reduce the data available by 80 per cent. See figure 9.2.

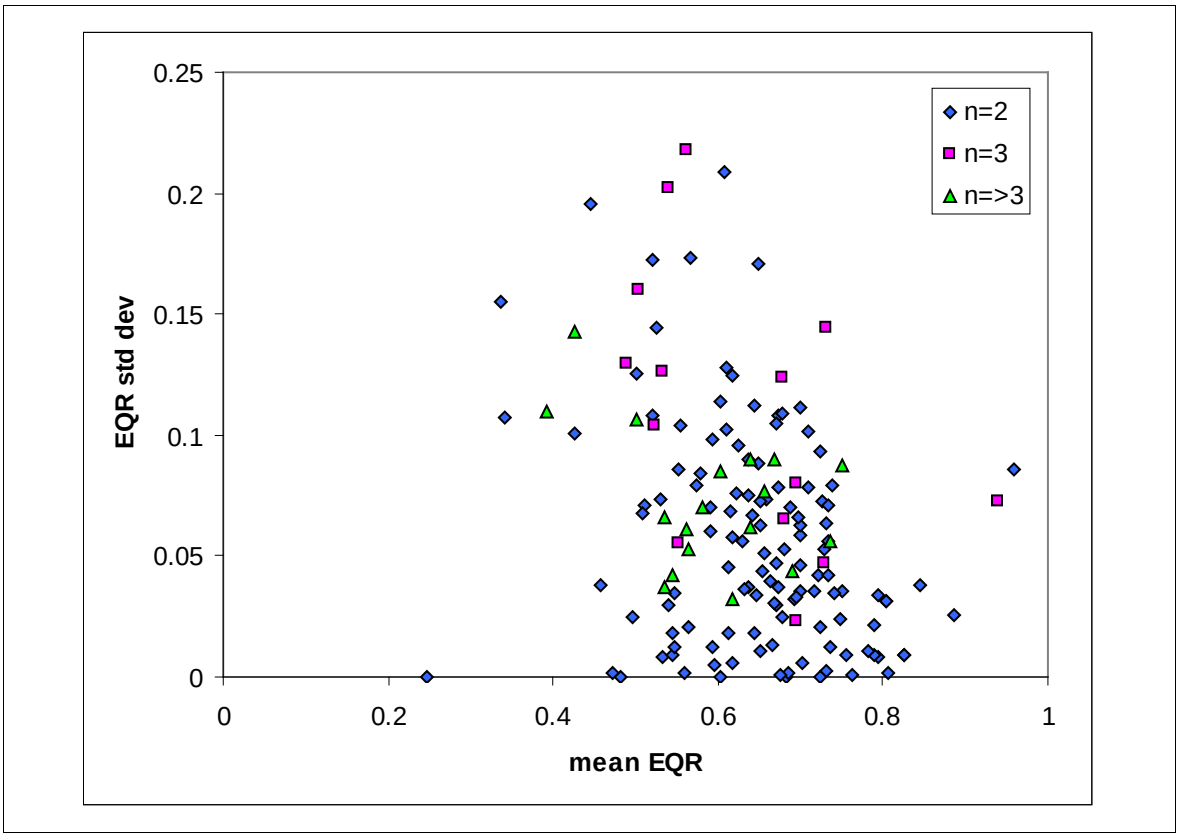

Figure 9.2 Influence of number of replicate surveys per lake on temporal variability associated with a given EQR

9.2.4 Effect of survey method on temporal variability

Many replicate surveys have employed a slightly different method for surveying the same lake rather than closely following the approaches used previously. If the variability between replicates obtained by different methods is significantly greater than the variability associated with replicates obtained with the same method, this would inflate the temporal variability beyond what we might expect by following a standard protocol. On the other hand if different methods are essentially capturing the same information in a slightly different way, the influence of method *per se* may be negligible (generally this is what one would hope, since one of the assumptions in developing this tool was the legitimacy of combining data from different sources where slightly different survey methods were employed). The effect of method was assessed using linear regression in which the EQR SD associated with a given mean derived from repeat sampling was compared with repeat samples using different methods. This confirmed that the effect of method is non-significant (p=0.574). In other words the EQR SD is no higher for a given mean when replicates are obtained by different methods than by the same method. Consequently, temporal data collected by different methods were pooled for the purposes of the analysis. See figure 9.3.

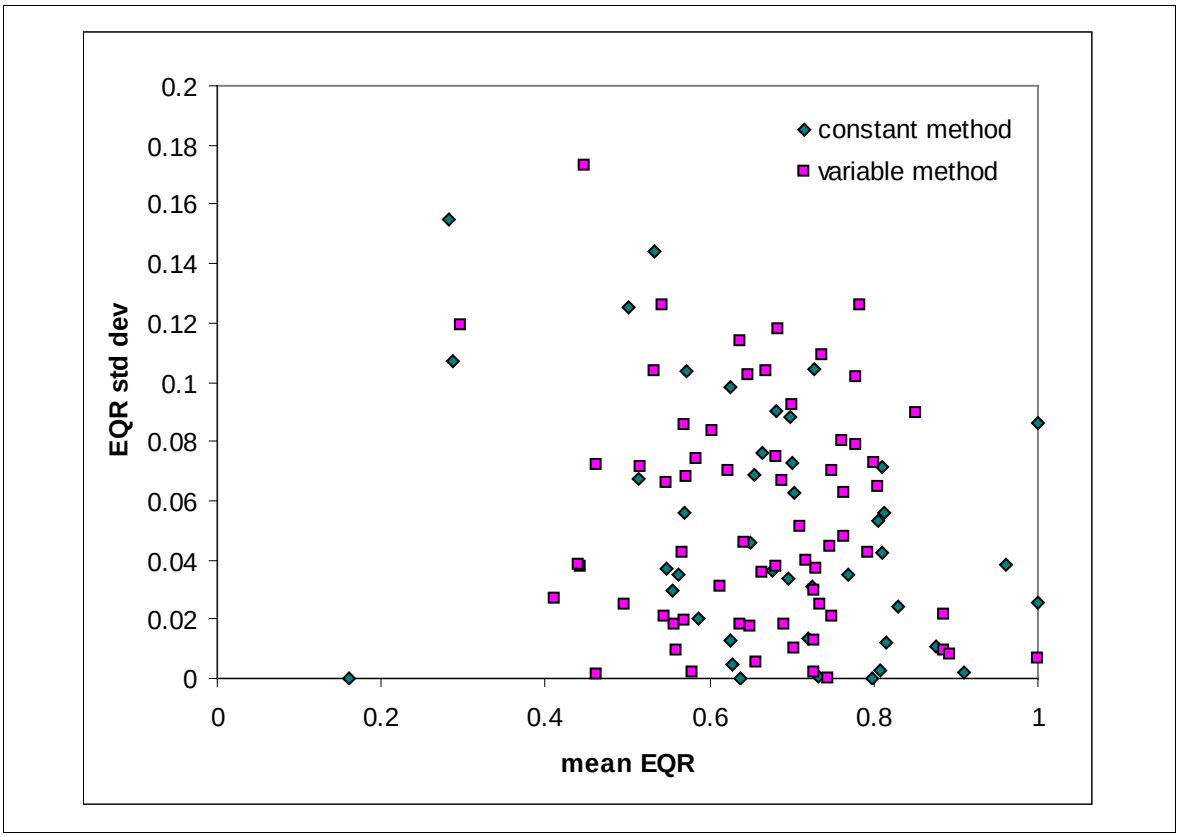

Figure 9.3 Influence of constant or variable survey method on temporal variability in lake EQR. There is no evidence that variability is higher for a given EQR when the survey methods contributing data for a site vary between dates than when these methods are constant. One data point at SD = 0.27 was removed since one survey in the comparison period was incomplete.

9.2.5 Defining anchor points in the mean versus standard deviation relationship

When assessing the relationship between mean EQR and EQR SD it is necessary to anchor the SD values at an EQR of zero and one. The SD at these extremes is not required to be zero since the x-axis is intended to represent the *true* (unknown) mean EQR at a site, not the *observed* mean EQR. Therefore it is possible to have a non-zero standard deviation even at a site whose true mean EQR is zero or one, because of sampling and measurement error.

Part of the problem in setting a value for the low end of the EQR range is the lack of sites with replicate data in this range, or the fact that averaging over a three- to six-year time window eliminates very low EQRs because these are lost within the average. As a solution, data from the Broads, which includes sites with replicate surveys with low EQR values, was subsampled focusing on sites and years with very low EQR values. This is equivalent to selecting all the lowest EQR surveys from a site, averaging these and considering the variability associated with each mean to be typical of what one would find if surveys were always carried out in 'bad' years. Effectively this approach asks 'what would the population of surveys need to look like to give a mean EQR as low as that observed' and 'what is the SD of that survey population'?

The anchor points at EQR of zero and one were then based on the 10th percentile of the distribution of EQR SDs in the upper and lower half of the EQR gradient. These values were 0.005 and 0.01 respectively. The greater variability in the lower half of the gradient probably reflects the fact that these sites are degraded by various pressures and there is little buffering capacity left in the system. This means that these sites are more responsive to external fluctuations (such as water temperature or level).

9.3 Analysis of sources of variability

9.3.1 Sampling variability (Sv)

Sampling variability is the variability associated with a standard sample from a given location collected at a given point in time. Factors which contribute to variability include:

- fine-scale differences in transect relocation;
- variations between observers in visual assessments of cover;
- differences in identification or detection of sampled taxa;
- variations due to small differences in the placement of rakes, grapnels and so on which reflect fine-scale spatial variation;
- differences in conditions (such as turbidity, wind) at the time of survey which may accentuate sources of observer-related variability.

Sampling variability contributes to variation in all datasets. The closest that can be used to assess the unique contribution of sampling variability are surveys done to standard protocol at 18 sites in July and September 2005 (Figure 9.4, 9.5). These were carried out by the same personnel at the same locations in the water body. The seasonal component may slightly inflate the variability but given that there is no consistent difference between samples taken in July or September (paired t test, $p = 0.172$) the data can be used safely to estimate sampling variability. The observer-based component of sampling variability is probably minimal since surveys are

generally undertaken by pairs of surveyors which will buffer the recording idiosyncrasies of individuals.

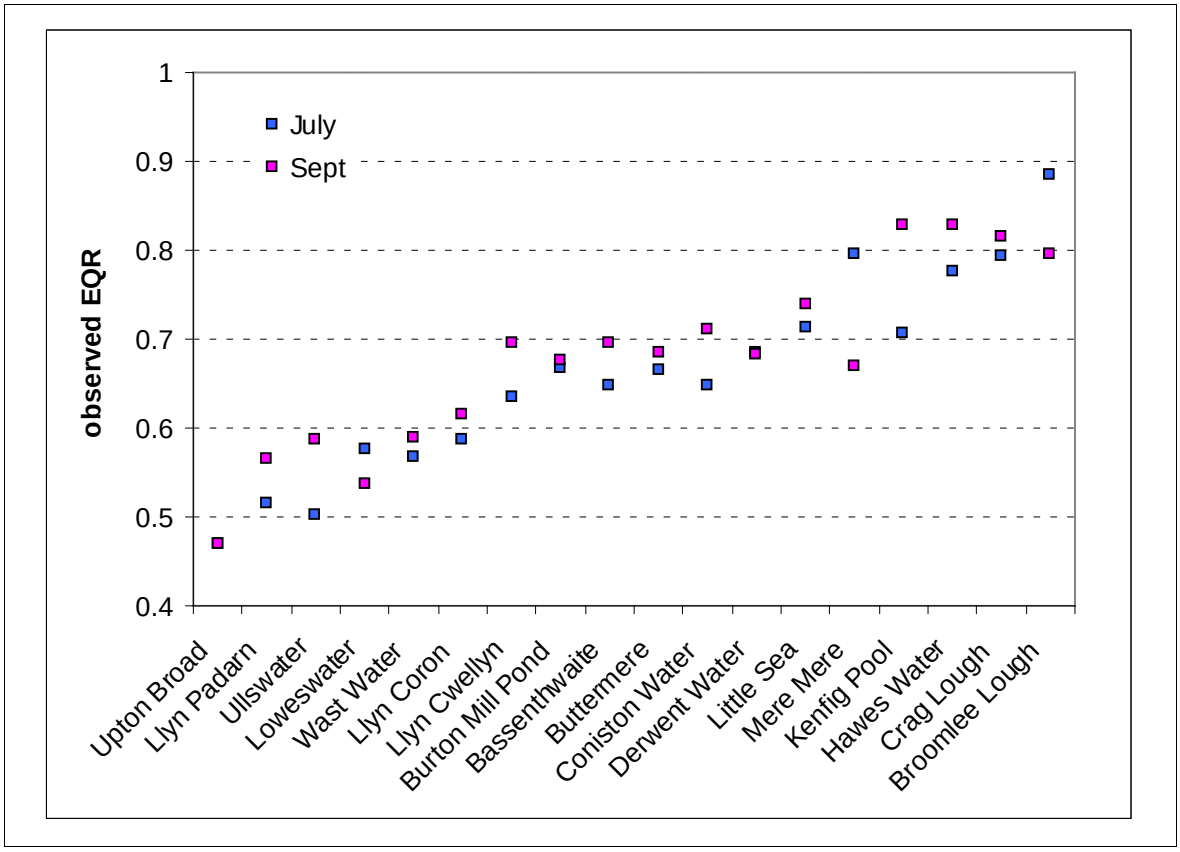

Figure 9.4 Sampling variability illustrated as difference between EQR of sites surveyed in July and September 2005 at same locations and by same personnel

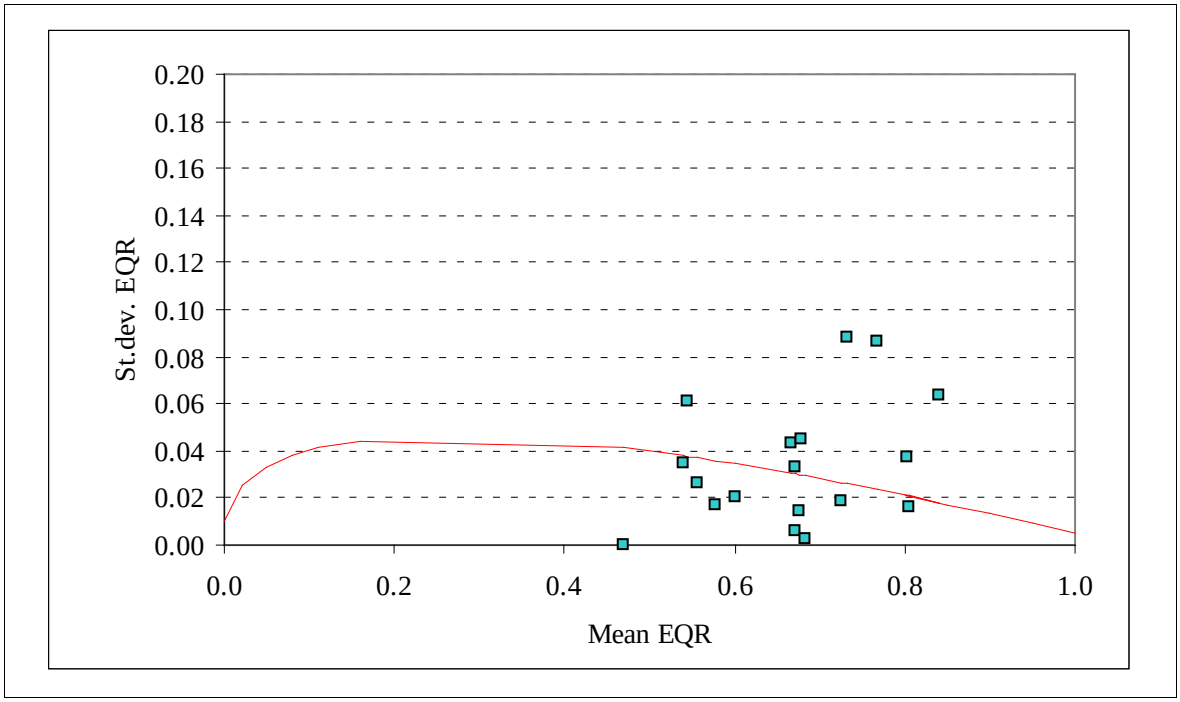

Figure 9.5 Sampling variability in UK lakes based on surveys carried out at 18 sites in July and September 2005 in which the same personnel returned to the

same locations in each water body. The red line is a power function (power = 0.6), anchored at EQR 0 = 0.01 SD and EQR 1 = 0.005 SD.

The other sources of variation discussed below may also be tainted by variation associated with measurement error and fine-scale variation of the type described above. Having obtained a separate estimate of pure sampling error, it is possible to partition the various contributions of different sources of error (see Section 9.3.6).

9.3.2 Spatial variation (Lv)

The ideal assessment of unique spatial variation would use replicate surveys on a given date following the standard protocol, in which different sectors of the same lake would be sampled with the same effort, or the same sectors sampled while offsetting shoreline and boat transects by several 100 metres from their original positions. This would determine the error associated with a single survey at a given point in time (literally, how representative a single survey is of a larger number of surveys or the lake as a whole).

At present, the standard survey protocol has been applied to a relatively small number of sites. Repeat surveys with this approach can be used to derive an estimate of temporal variability but no surveys have assessed more than the minimum number of transects needed for a water body assessment at a single point in time. Consequently, the spatial uncertainty associated with the method (variation that would arise from assessing a water body two or more times using the standard approach on a given date) can only be estimated using data from surveys carried out to a similar format. Generally, pure spatial variability should be relatively small since boat and shoreline transects are located partly to capture a representative cross-section of the variation in physical habitat within a lake. Repeat surveys following this principle should therefore reflect mainly the local-scale variability within broad physical habitats, rather than coarse-scale variability that may exist between contrasting habitats when assessed independently.

One solution to the lack of suitable data is to use other surveys that have considered the different basins or sub-bodies of a given water body on a given date. Note that these separate assessments are akin to ‘complete’ surveys of a particular area and cannot be equated to a survey of an individual lake sector, using the standard protocol, in which single transect lines are used. In using this data to estimate spatial variation, it is assumed that the separate basins or sub-bodies assessed in a lake are themselves comparable (for example, contain the same broad collection of physical habitats and are subject to the same pressures). If this is not the case, the resulting estimate of spatial variation will prove excessive.

The graph below (Figure 9.6) illustrates the variation associated with the mean EQR from 48 sites in which the mean is based on two to four separate assessments of basins or sub-basins of that water body on a single date. Although there must be significant uncertainty associated with the relationship itself, values above 0.1 are rare and in general, large-scale variation is likely to be a small source of error compared to variation between years.

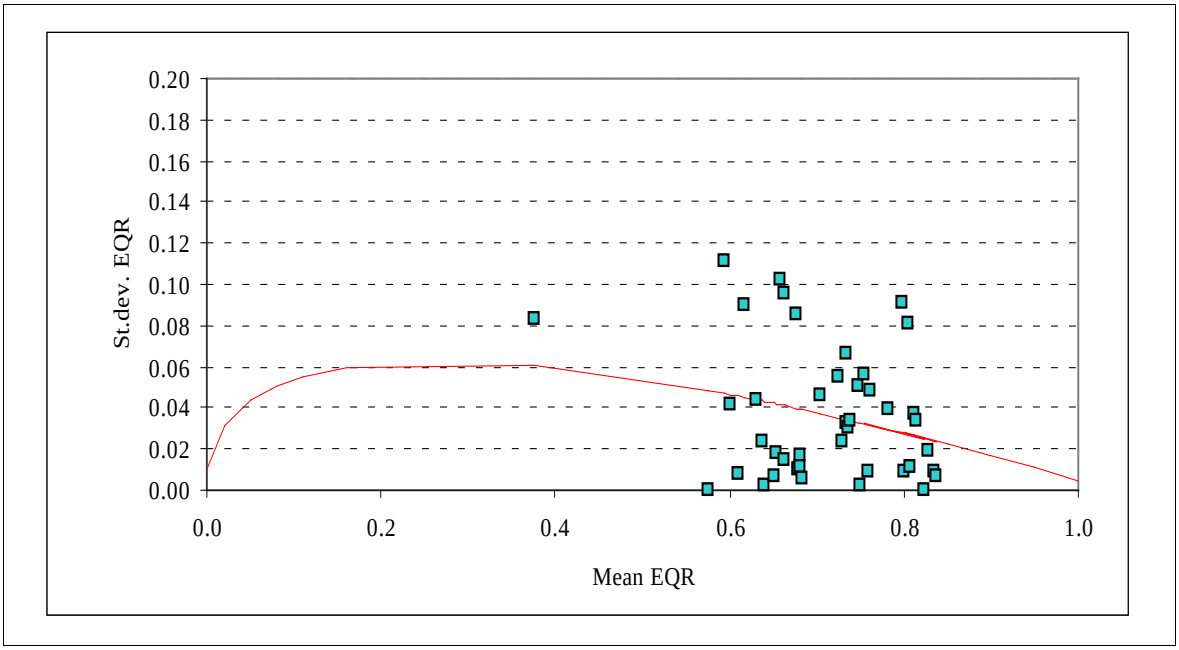

Figure 9.6 Between-basin variability in UK lakes where temporal variation is eliminated by sampling on a fixed date. The red line is a power function (power = 0.6), anchored at EQR 0 = 0.01 SD and EQR 1 = 0.005 SD. One sample point with SD 0.19 was removed due to known differences between sub-basins.

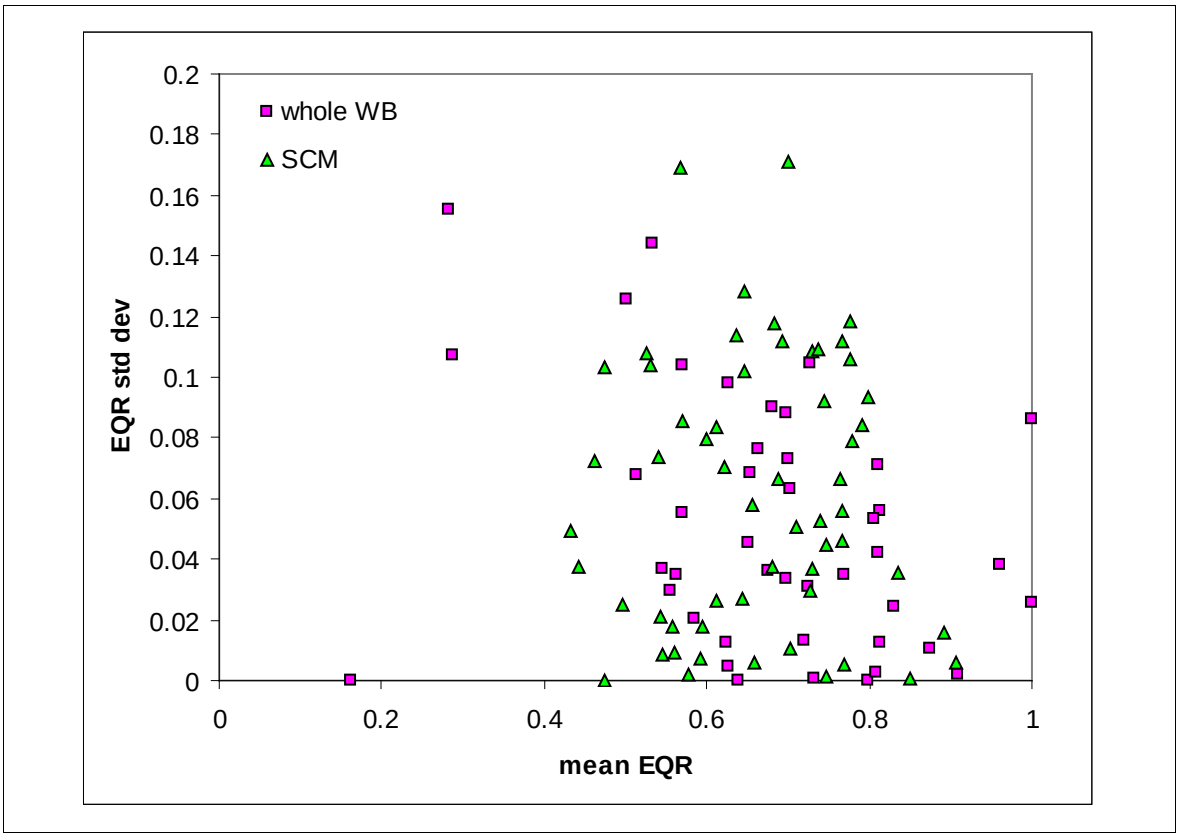

Figure 9.7 Standard deviation between replicate surveys undertaken at a whole water body level or between whole body and subsample (SCM). There is no evidence in the latter that variation is higher between replicates for a given mean EQR, suggesting that spatial variation is a minor source of variation between repeat surveys.

A second way to derive an estimate of the scale of spatial variation between surveys is to compare repeat surveys from different dates where the whole lake was surveyed on both occasions, with repeat surveys in which the second survey (by the standard protocol) only covered a sub-sample of the lake. A significantly larger standard deviation between samples in the second case could be used to infer a significant additional spatial component to the variation.

However, if a series of repeat surveys at a whole-lake level is compared with a series of 57 surveys first conducted at a whole-lake level and then a similar time period later based on the standard SCM survey protocol, there is no indication of greater between-survey variability in the latter case (Figure 9.7). This strongly suggests that the spatial error associated with the standard protocol is small; in other words, a single survey is likely to show little departure from the true mean due to spatial variation.

9.3.3 Temporal variation (Tv)

It is possible to isolate the pure temporal component of variation if repeat surveys on different dates are carried out at the same locations. Thus, 48 surveys conducted to the standard protocol (or a close variant, such as resurveying a larger number of boat transect lines) were carried out in 2003 and repeated in 2005. The relationship between inter-survey variability and mean EQR across these sites is described in Figure 9.8 The EQR SD is relatively small, peaking at 0.05 at a true EQR of 0.3 (middle of 'poor' status).

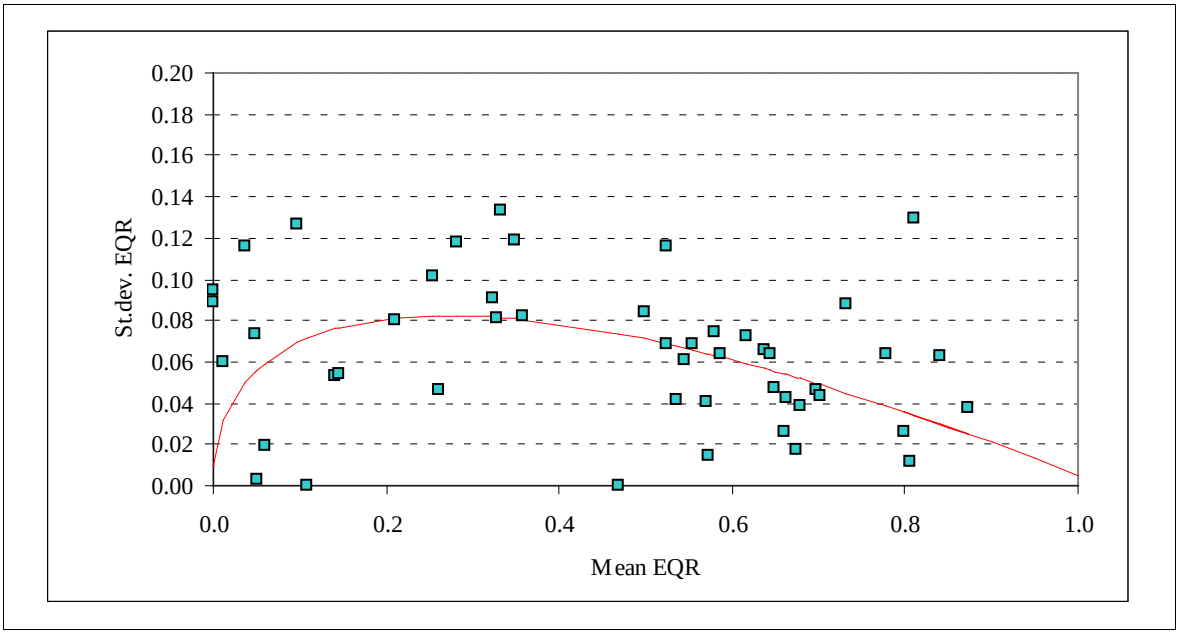

Figure 9.8 Between-year variability in UK lakes where spatial variation is eliminated by re-sampling at the same locations. The red line is a power function (power = 0.6), anchored at EQR 0 = 0.01 SD and EQR 1 = 0.005 SD.

9.3.4 Combined spatial + temporal variation (Lv + Tv)

Repeat surveys of the same lake over time provide a perspective on temporal variation but if these surveys are conducted at a whole-lake level, this variation might be expected to contain an element of spatial variation beyond that due to sampling sources. Even when the whole water body is being surveyed, in practice different

surveys are likely to consider slightly different parts of the same lake each time. An indication of the scale of this additional spatial component could be obtained by comparing the EQR versus EQR SD relationship with that in Figure 9.8.

Having controlled for the variable time window effect, Figure 9.9 shows that the pattern of variation in EQR SD over the EQR gradient is remarkably similar to that associated with pure temporal variation described in Figure 9.8 above. The EQR SD again peaks at 0.05 in a similar area of the EQR gradient. Thus, when these relationships are overlain (Figure 9.10) there is nothing to distinguish them. This again suggests that the spatial component to variation in lake surveys is very small.

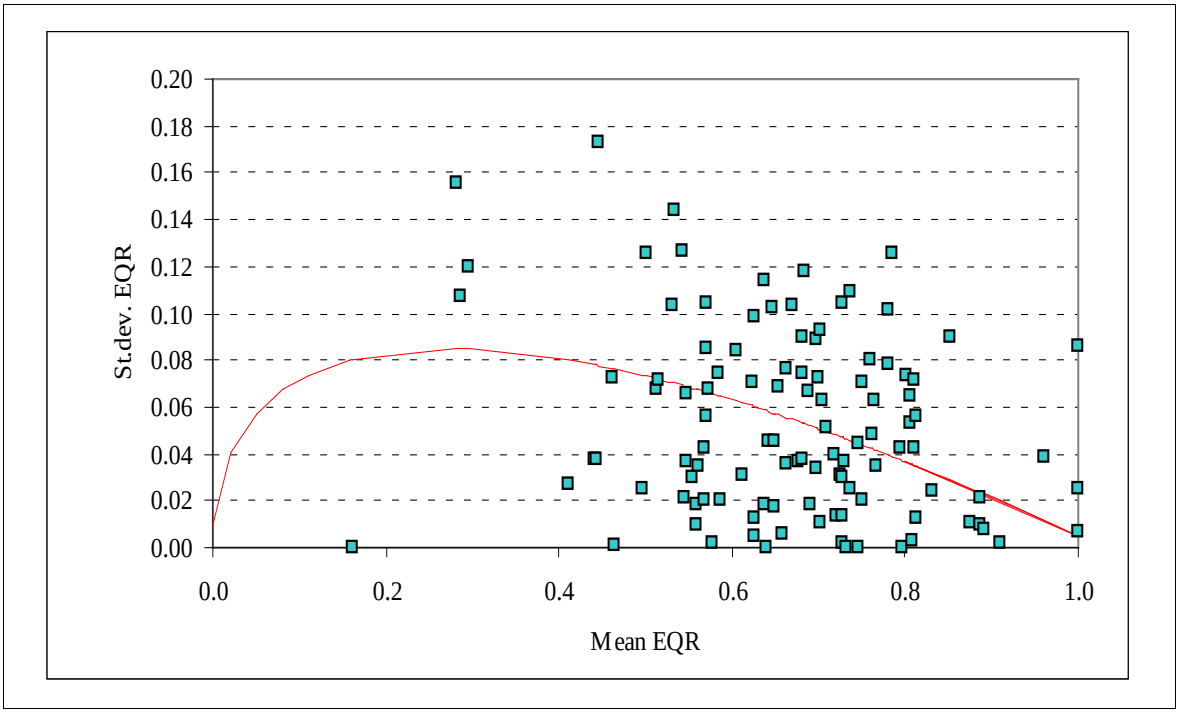

Figure 9.9 Between-year variability in UK lakes in which whole-lake surveys were conducted up to 11 years apart. The red line is a power function (power = 0.6), anchored at EQR 0 = 0.01 SD and EQR 1 = 0.005 SD.

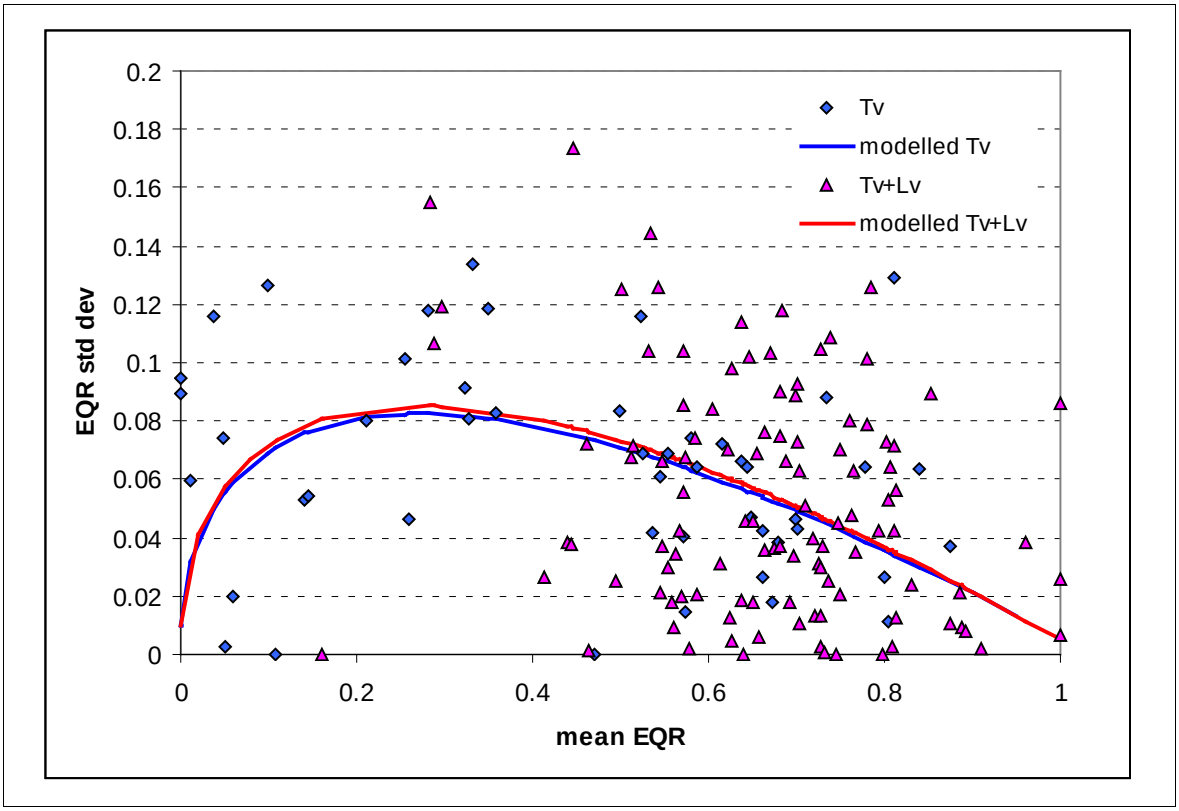

Figure 9.10 Overlay of Figures 9.8 and 9.9 showing the close match in EQR SD relationship with mean EQR when data is derived from replicate whole-lake surveys or replicate surveys in which the spatial component of variation has been eliminated

9.3.5 Fine-scale spatial variation associated with the standard sampling protocol

This analysis does not consider the variability and subsequent uncertainty of classification associated with aspects of the standard protocol. Thus, for example, issues such as the extent to which samples from a variable number of sectors capture the species pool of the entire lake, or what the consequences would be of reducing the numbers of sectors sampled from four to three or fewer are not considered here. The surveys represent an aggregate view of a lake in the same way that, for example, a three-minute kick sample represents an aggregate view of the invertebrate fauna of the various mesohabitats in a given reach of river, or a diatom sample from five stones in a lake represents a pooled sample of the flora of that lake. It is acknowledged in this sampling design that the differences in macrophyte flora between, for example, eroding and depositional shorelines or inflow and outflow are potentially marked, and one would not seek to represent an entire lake by sampling only one or other habitat type. Thus, in all cases local spatial variability is recognised as being *so* important that it is 'neutralised' by the sampling method itself. Moreover the allocation of sectors reflects, to some extent, differences in coarse-scale physical habitat, and consequently the variation between sectors is likely to be high compared to say the variation between several basins in which a constant set of physical habitats are present and sampled. It is also impossible to demonstrate that a survey of a tiny percentage of a given sector is truly representative of that sector or that surveys of several sectors can be considered truly independent. Moreover, this tool is built on the basis of whole-lake assessment, is populated with data from reference sites assessed on a whole-lake basis and has been

inter-calibrated against other systems based on whole-lake assessments. Comparing the EQR of separate sectors within a lake where the reference is based on whole-lake data would be misleading and likely to generate a mean EQR lower than that for the lake as a whole.

Generally, one would not seriously consider attempting to represent the macrophytic vegetation of lakes above 10 ha by sampling any fewer than four sectors and, indeed, the method development trials suggest that this approach captures the essential characteristics of lake macrophytes as effectively as whole-lake surveys (and is endorsed by Section 9.3.2), whilst ensuring high repeatability and retaining control over large-scale spatial variability. Technically one could assess the degree of departure from a whole-lake survey associated with sampling an ever smaller number of sectors. However, such an exercise is largely academic. With four sectors sampled the departure from a whole-lake survey is known to be small, thus indicating that this sampling regime is adequate, while the unit costs of sampling four rather than three or two sectors are minimal compared to the cost of deploying a survey team at a lake.

9.3.6 Combining estimates of different sources of variation

Through the above analysis several discrete sources of variation were quantified, all of which contribute to variation in a face value EQR. These comprise:

i. The pure sampling effect attributable to fine-scale spatial variation and observer artefacts (Sv).

ii. Temporal (Tv) + sampling variation (Sv) derived from a spatially controlled sampling, which could not be separated from the temporal variation apparent from whole water body resurveys. Uncertainty caused by temporal variation could be reduced by increasing the frequency of sampling.

iii. Large-scale spatial (Lv) + sampling variation (Sv) derived from comparing surveys of different basins or sub-basins of a single water body. This is the closest approximation to repeat surveys of a given lake using a standard protocol at a fixed time. This analysis and inferences from other analyses suggest that the large-scale spatial component of variation is very small.

The total variation that it is possible to itemise might therefore be visualised as:

$$\Sigma v = Sv + (Tv - Sv) + (Lv - Sv)$$

Since it is possible to model each of these sources of variation for a given mean EQR and to quantify each component independently, it is therefore possible to derive a measure of the total relevant variation for a given EQR. This is illustrated in Figure 9.11. This is preferable to deriving a single relationship by pooling data representing different types of variation.

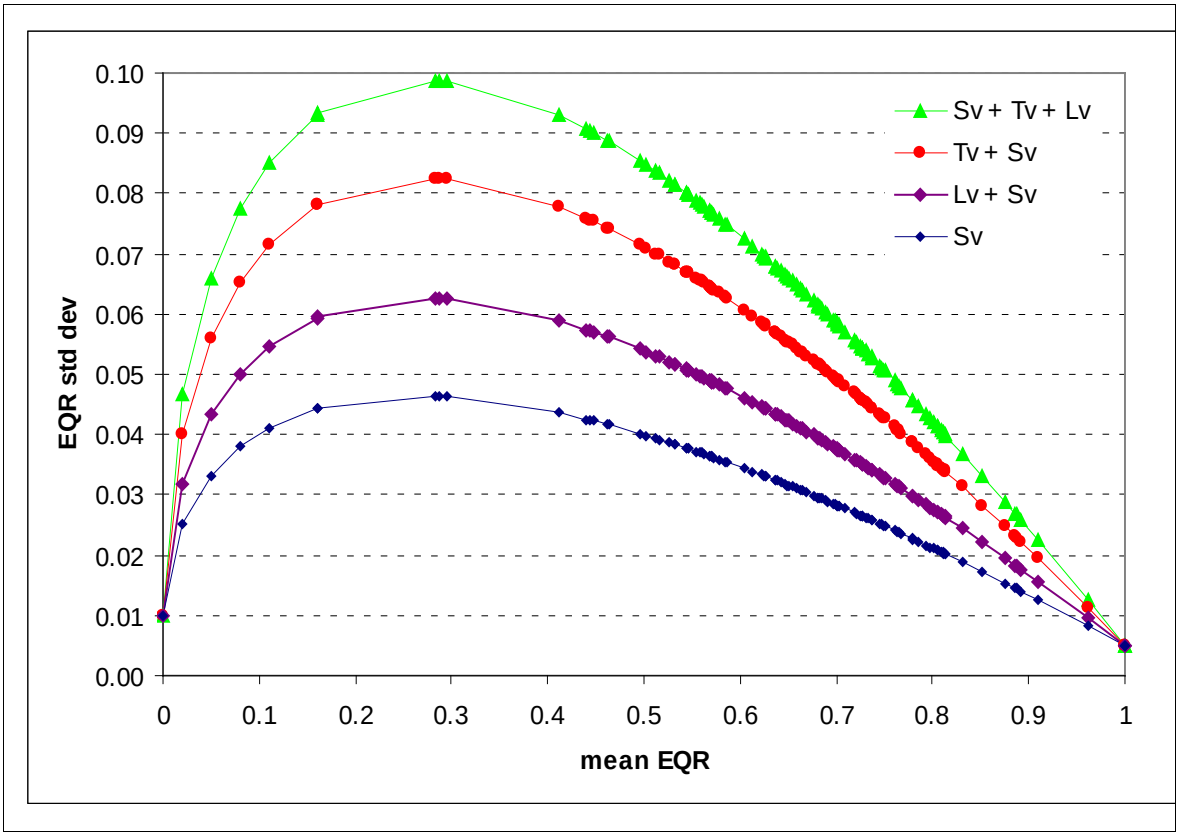

Figure 9.11 Using modelled sources of individual components of variation to derive an overall estimate of the total variation in a single face value lake EQR. Note the small increase between Tv + Sv and Sv + Tv + Lv reflecting the minor additional contribution of spatial variation.

The modelled line for Sv + Tv + Lv = Σv. In reality, the total relevant variation probably lies somewhere between this line and the next line (Tv + Sv) since it is likely that the large-scale spatial component is somewhat overestimated. This being the case, the average distance between Σv and Tv + Sv is therefore used to predict variation associated with a given EQR and from this the confidence of any given classification. This represents the best estimate of compound variation that can be derived given the constraints of the data available. However, given the distribution of SD values associated with observed mean EQRs (90th percentile = 0.11) a large pool of extra data would need to have a markedly different error distribution to require this estimate to be revised significantly.

The function used in this study to estimate error is therefore as follows:

SE = 0.01 [anchor at EQR =0] + -0.447 * EQR + 0.442 * EQR^0.6

The analysis clearly indicates that, in terms of 'managing variability', if a second survey was carried out this would be better timed to consider temporal variation at a water body scale since the average 'within-habitat' variation that a repeat survey of a lake would address is comparatively small. The more important between-habitat variation (that cannot be itemised) is integrated by the method of data collection.

9.4 Implications for classification

9.4.1 Confidence of class

If the modelled relationship between observed mean EQR and EQR SD, taking into account sampling, temporal and spatial sources of variation, is accepted as the best available estimate of the error associated with a given EQR we can combine this with information on class boundaries and predict the confidence with which a site can be assigned to a given class. This approach assumes that the errors associated with a given EQR are normally distributed about that mean with a distribution equivalent to the modelled EQR SD. Given this information, one can assess the impact of different survey frequencies on confidence of class. The procedure for calculating confidence of class is outlined by Ellis (2006). The risk of face-value misclassification (assigning a site to the wrong class) is then computed as the sum of confidences of membership of all classes except for the observed class. The risk of misclassification will always be at least 50 per cent for an EQR that lies exactly on a class boundary but will fall to a minimum moving towards the middle of that class. This approach differs slightly from that trialled previously using the STARBUGS software (Clarke, 2005). In STARBUGS the EQR SD is considered constant and confidence of class is based on the result of multiple simulations in which a random error derived from the distribution defined by the SD is added to each observed EQR. The probability that a site belongs to a specific class is based on the statistical distribution of these simulated values.

The following diagram (Figure 9.12) considers the confidence that a site belongs to the observed class when a single survey is carried out per monitoring cycle. Thus, in the middle of the good class the confidence that the lake belongs to that class is close to 95 per cent but this confidence falls to 75 per cent in poor status, reflecting the asymmetric shape of the SD versus EQR relationship. At each class boundary a site has 50 per cent probability of belonging to either adjacent class whereas in the middle of a class, a site has two to 12 per cent probability of belonging to either adjacent class.

By manipulating the number of surveys one can adjust the predicted error associated with a given mean EQR. Therefore:

SE = (0.01 + -0.447 * EQR + 0.442 * EQR^0.6)/square root n

Where n = the number of surveys in a monitoring cycle.

Thus, if the number of surveys is increased to two the error is reduced by the square root of two. A worked example is given at the end of Section 9.4.2. With two surveys per monitoring cycle (Figure 9.13), classes in which there is least confidence with a single survey are elevated to the same standard as other classes. With two surveys, the confidence that a site belongs to its assigned type is about 95 per centacross the central third to half of the width of any class except poor, where confidence is limited to 90 per cent.

If the number of surveys is increased to three per monitoring cycle (Figure 9.14), the confidence of class in the middle of all classes is above 95 per cent. However, this level of survey effort is necessary only in sites that fall in the middle of poor status (EQR = 0.27-0.33) if there is a desire to classify sites in this range with 95 per cent confidence. For all other classes, two surveys per cycle will provide the necessary confidence.

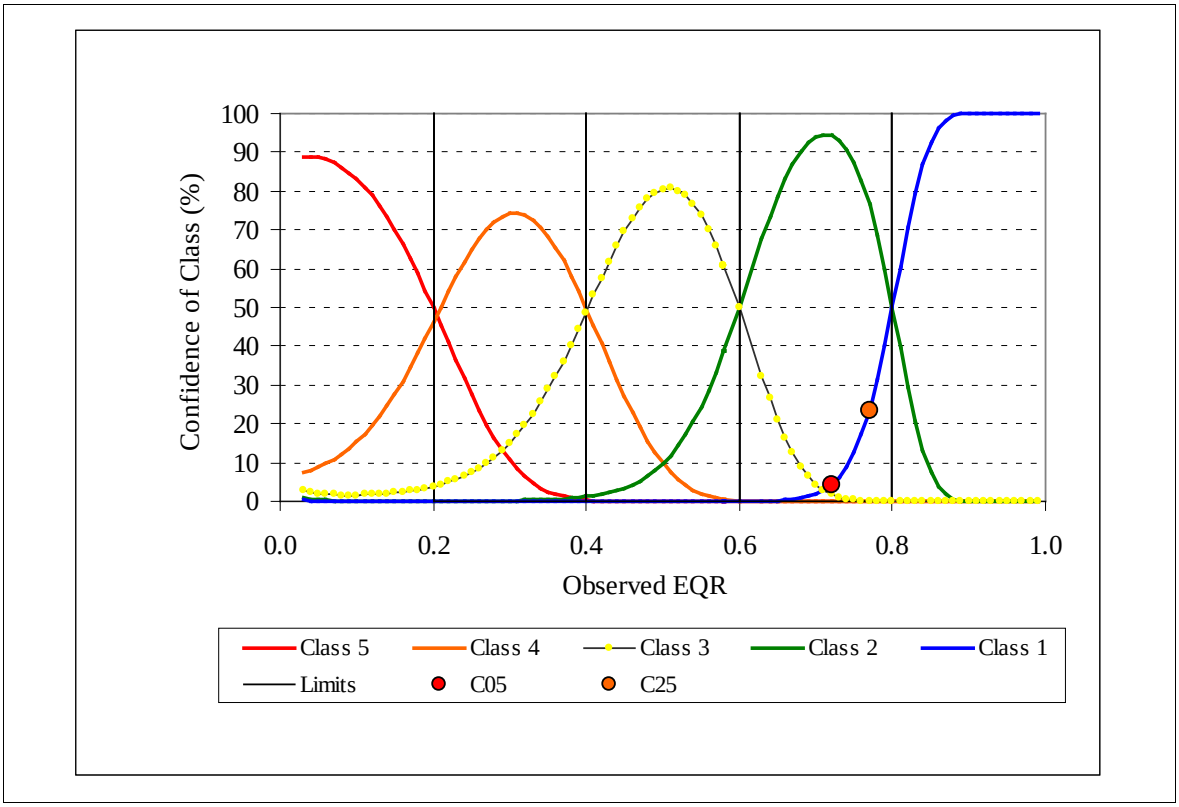

Figure 9.12 Confidence of correctly placing a site according to its face value ecological status, based on a single survey per monitoring cycle

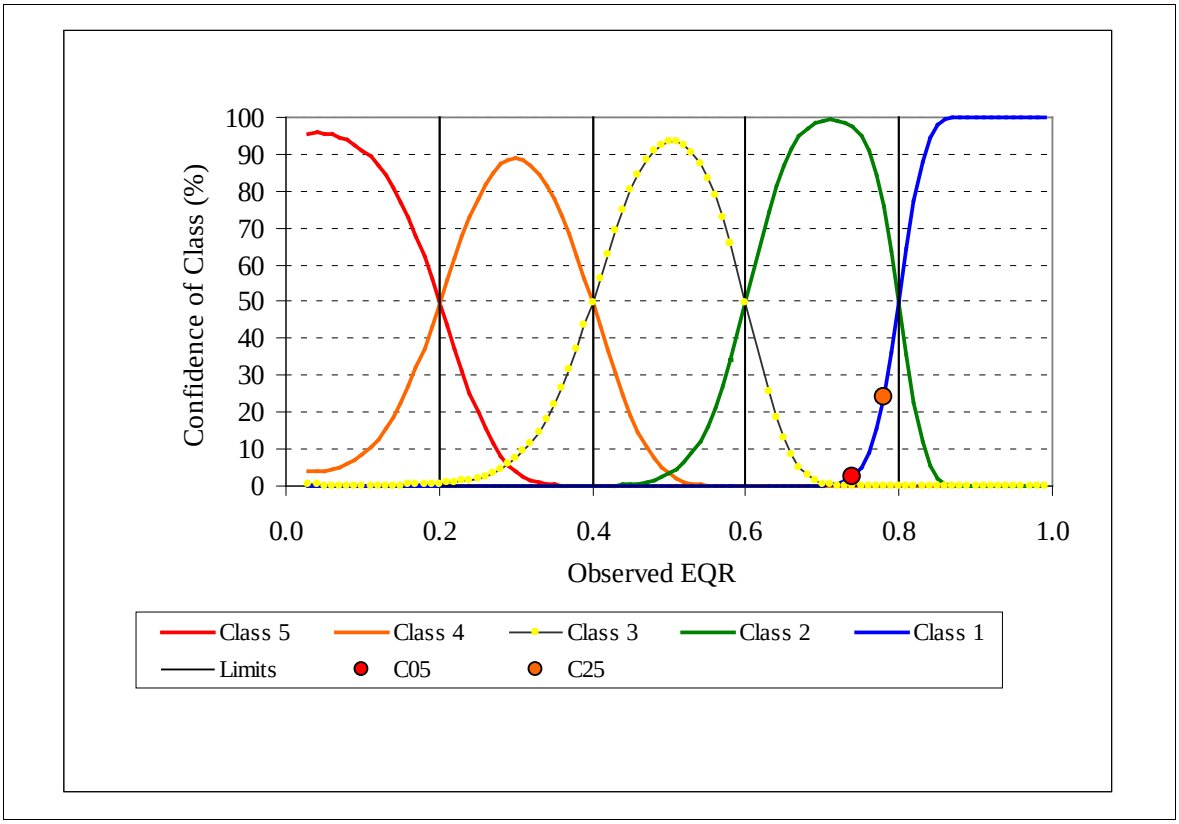

Figure 9.13 Confidence of correctly placing a site according to its face value ecological status, based on two surveys per monitoring cycle

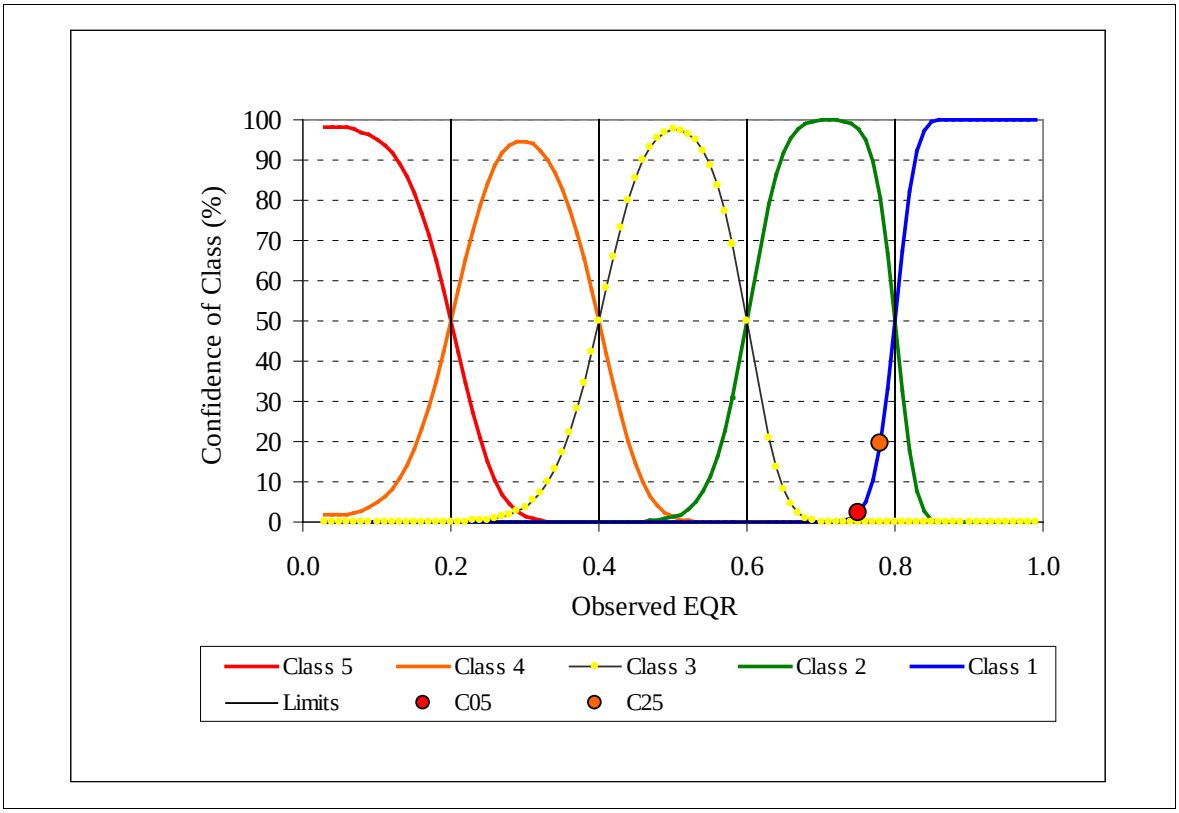

Figure 9.14 Confidence of correctly placing a site according to its face value ecological status, based on three surveys per monitoring cycle

9.5 Misclassification and the effect of sample size

The risk of misclassifying a water body in a class other than its true class is summarised in Figure 9.15, combined with the influence of the number of surveys undertaken.

Thus, in the case of a single survey, a site with an EQR of 0.7 that would be classified as good has a negligible risk of being misclassified. If the EQR falls to 0.65 a site classified as good has a 15 per cent chance of being misclassified as moderate. The greatest risk of misclassification occurs in the middle of poor status where a site would still have a 27 per cent risk of being misclassified as moderate or bad.

If the number of surveys per cycle is increased to two the risk of misclassification in the middle of any class, especially moderate or poor, decreases dramatically. The greatest effect of the number of surveys is on the risk of misclassification in the outer third of the width of each class. Thus, in each status class except high, the risk of misclassifying a site drops from between 25-35 per cent at one-third of a class width from the nearest boundary to six to 10 per cent when the number of surveys is increased from one to three per cycle.

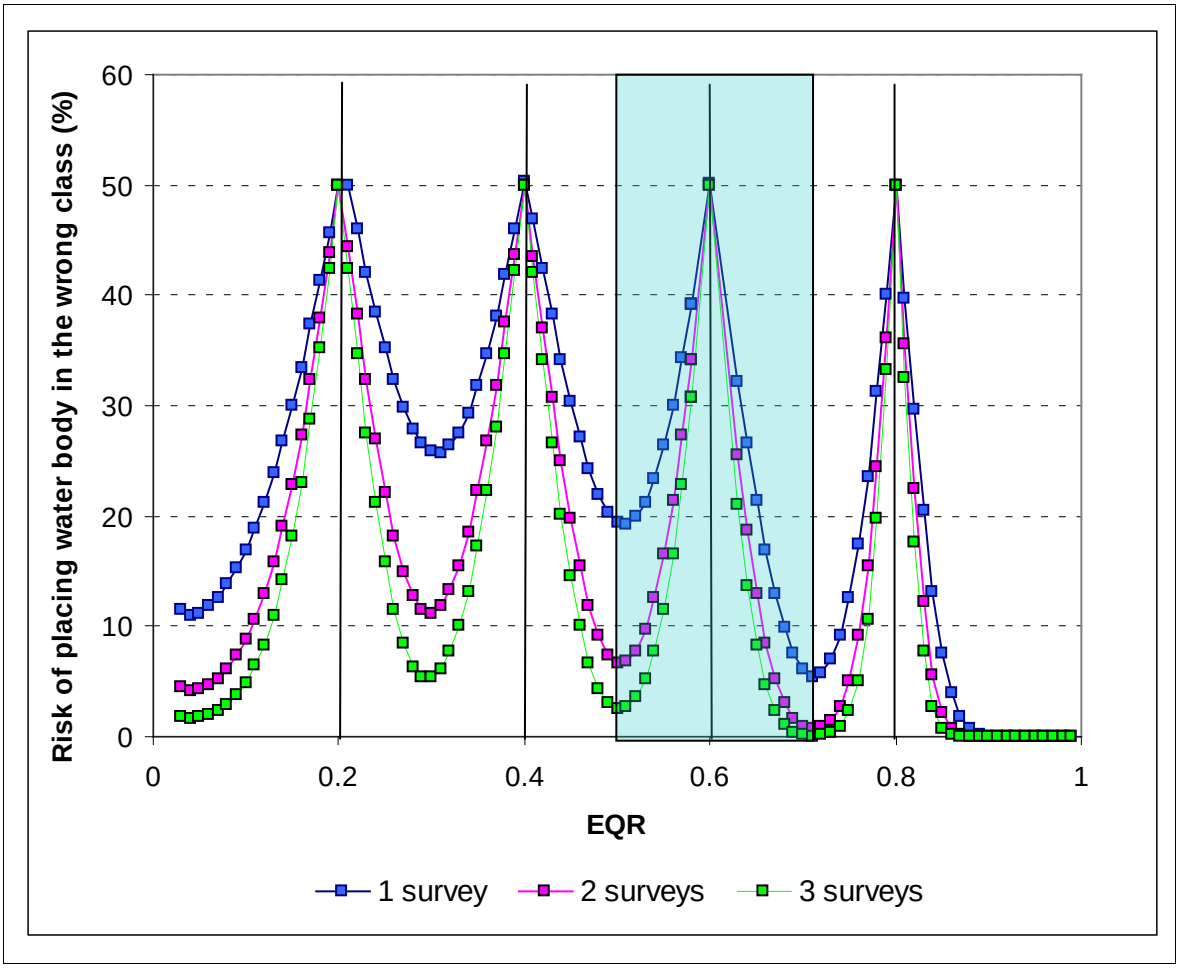

Figure 9.15 Risk of face value misclassification for UK lakes based on different numbers of surveys in a monitoring cycle. The blue box encloses a region of the EQR gradient of particular significance in which, based on a single sample, there would be a high (>5%) risk of misclassifying a moderate status site as good and vice versa.

For sites classified as moderate status or worse, it is likely that programmes of measures (PoM) will only be implemented where there is a low (less than five per cent) risk of misclassification. The following diagram (Figure 9.16) assesses the risk of misclassifying a good site as moderate or worse, or a moderate or worse site as good or better at two different levels of sampling.

A site with an EQR of 0.69 has only a five per cent risk of being misclassified as moderate or worse with a single survey per cycle; the risk is negligible with two or more surveys. To achieve the same risk with two surveys the EQR can fall to 0.67 or 0.66 with three surveys (about one-third of the class width when the class runs from 0.6 to 0.8). A site with an EQR of 0.47 has a five per cent risk of being misclassified as good or better (there is 95 per cent confidence that the site is indeed moderate status or lower) with one survey. The risk of misclassification for the same EQR is negligible with two surveys per cycle. To achieve the same risk with two surveys per cycle, the EQR can rise to 0.51 or 0.53 for three surveys (about one-third of the class width in the latter case). Thus, depending on the sampling intensity, there is a window extending across 13-22 per cent of the total EQR gradient in which it is impossible to state with 95 per cent confidence whether a site is correctly classified as moderate or worse, or good or better.

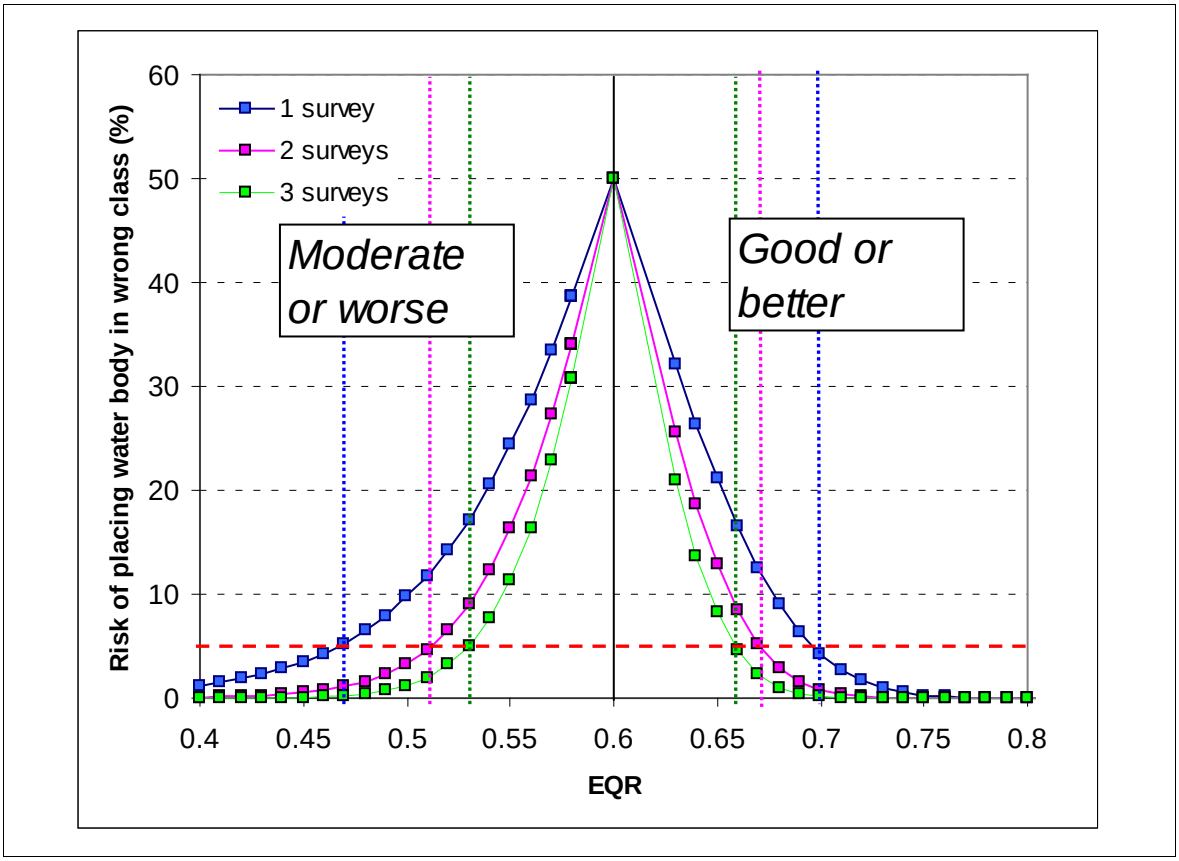

Figure 9.16 Risk of face value misclassification for UK lakes focussing on good and moderate classes and the risk of misclassifying a good site as moderate or worse, or a moderate or worse site as good or better

The box below provides a worked example of the calculation of confidence of class and the risk of misclassification, based on spreadsheets prepared for this purpose by Julian Ellis, WRc.

Confidence of class and risk of misclassification: worked example

1. *A site has an EQR of 0.65 based on a single survey which places it in good status.*
2. *The predicted error associated with this EQR is therefore:*

   *SE = (0.01 + -0.447 * 0.65 + 0.442 * 0.65^0.6)/square root n = 0.061*
3. *Two repeat surveys of this site produce a mean EQR of 0.65.*
4. *The predicted error associated with this mean EQR for this survey effort is:*

   *SE = (0.01 + -0.447 * 0.65 + 0.442 * 0.65^0.6)/square root 2 = 0.043*
5. *Three repeat surveys of this site produce a mean EQR of 0.65.*
6. *The predicted error associated with this mean EQR for this survey effort is:*

   *SE = (0.01 + -0.447 * 0.65 + 0.442 * 0.65^0.6)/square root 3 = 0.035*
7. *Based on the mean EQR and associated error for each level of survey effort, and assuming that the error is normally distributed about the mean, the following percentage Confidence of Class (CoC) statistics can be computed.*
8. *The risk that the site is misclassified (is actually not good status) is calculated as 100 – % CoC for good.*
9. *Therefore as survey effort increases from one to three surveys in a cycle the confidence that the site is indeed good increases from 79 to 92 per cent and the risk that it is not good decreases from 21 to eight per cent.*

			Confidence of Class (%)					Risk of misclassification
n	mean EQR	exp SE	*Bad*	*Poor*	*Mod*	*Good*	*High*	Sum confidence ≠ G
1	0.65	0.061	0.000	0.006	21.195	78.595	0.204	21.405
2	0.65	0.043	0.000	0.000	12.911	87.086	0.002	12.914
3	0.65	0.035	0.000	0.000	8.307	91.693	0.000	8.307

9.5.1 Implications for classification of the UK lake resource and survey effort

To assess the relevance of different levels of survey effort per monitoring cycle for classification, one can determine the difference in the number of lakes in the mean EQR range bounded by the window of uncertainty that applies with different numbers of surveys (0.47 to 0.69; 0.51 to 0.67 and 0.53 to 0.66 for one, two and three surveys respectively). This information is summarised in Table 9.2. If one considers the UK lake resource for which a minimum quality of macrophyte data is available since 1983, the number of lakes that cannot be classified as good or better or moderate or worse with more than 95 per cent confidence falls from 703 to 408 when the number of surveys per cycle increases from one to three. This represents a reduction from 25 to 14 per cent of the resource covered. The difference between two and three surveys is relatively marginal. Since the emphasis is, perhaps questionably, purely on not implementing a PoM when there is a significant risk that a site classed as moderate or worse may actually be good or better (as opposed to ensuring a site classified as good is not actually moderate or worse) it is possible to identify those lakes with an EQR lying between 0.47, 0.51 or 0.53 and 0.6 (the G/M boundary) where the decision to operate one, two or three surveys would have an influence on classification. This

reveals that 337 out of 2,835 lakes have an EQR falling in the range 0.47 to 0.60, this number falling to 240 in the range 0.51 to 0.60 and 197 in the range 0.53 to 0.6. Thus, two surveys per cycle might allow about 100 more lakes to be classified as moderate or worse status with 95 per cent confidence than a single survey. This increases to 140 when three surveys are implemented. Thus, 197-337 lakes in this sample with a face value EQR that would place them as moderate or worse can only be seen to fail with less than 95 per cent confidence. While this amounts to only seven to 12 per cent of all surveyed lakes, more significantly it concerns 27-47 per cent of all lakes classified as moderate or worse based on their face value EQR. To put these figures in context, a certain proportion of lakes will not be classifiable with more than 95 per cent confidence regardless of the survey effort. Thus, even if surveys were conducted every year during a monitoring cycle (n=6) sites with an EQR between 0.56 and 0.64 could not be classified as less than or greater than good with more than 95 per cent confidence. As a baseline figure, in the above sample 114 lakes (16 per cent) classified as moderate or worse could *never* be classified as moderate or worse with 95 per cent confidence. Data for six surveys is illustrative only since there is a risk that samples collected at this frequency are arguably prone to pseudoreplication (such as lack of independence of data collected in consecutive years).

Table 9.2 Implications of different levels of survey effort for the classification of a large sample of UK lakes (n=2,835)

	Number of surveys per cycle			
	One	**Two**	**Three**	**Six**
Confidence at middle of M that WB is below G	90	97	99	100
Confidence at middle of G that WB is G or above	96	99	100	100
EQR where confidence below G = 95%	0.47	0.51	0.53	0.56
EQR where confidence of G or above = 95%	0.69	0.67	0.66	0.64
No. (%) of WBs classified as below G with <95% confidence	337 (47)	240 (33)	197 (27)	114 (16)
No. (%) of WBs classified as G or above with <95% confidence	366 (17)	250 (12)	211 (10)	137 (6)
No. (%) of all WBs classified as below G or as G or above with <95% confidence	703 (25)	490 (17)	408 (14)	251 (9)

Thus, two surveys per monitoring cycle will normally be adequate for the classification of a lake based on macrophytes. In a small number of cases a third survey might be considered for sites with a face value EQR close enough to the centre of the moderate class (between 0.51 and 0.53) for a third survey to make the difference between a 'confident' and 'risky' classification. Lakes that have an EQR above 0.7 on the basis of a single survey can be assumed to achieve good status with more than 95 per cent confidence. In lower status classes where EQR variability is higher, two to three surveys per cycle may be required to confirm no deterioration in status.

9.5.2 Comparison with other quality elements

The peak variability associated with a face value EQR is somewhat less for lake macrophytes than other quality elements such as diatoms, fish and macroinvertebrates. This may reflect a number of factors, including the relative homogeneity of lakes, the tendency for macrophytes to be widely distributed in lakes wherever suitable habitat occurs, and the ability of the survey method to minimise large-scale spatial variation. It is also likely that multimetric systems, such as the one developed for lake macrophytes, buffer the variation in a single metric and are thus less variable overall than systems which rely on just one metric, such as TDI for diatoms.

Survey costs, in terms of time and number of personnel required in the field, coupled with other costs such as accommodation and boat hire, are relatively large for macrophytes; typically it will require five person-days to plan and survey a water body twice in a single monitoring cycle. These time costs are concentrated on the period spent in the field recording. In contrast, for other quality elements such as phytoplankton, diatoms, or littoral invertebrates the costs associated with sample collection are relatively small (although more frequent sampling of diatoms and phytoplankton would be required), and may only involve a single individual, but the laboratory time spent processing samples, preparing and counting slides and identification is significant. A properly timed and costed comparison of the resource required to deliver classification with 95 per cent confidence for different quality elements would be a useful exercise.

9.6 Summary

i. Large-scale spatial error associated with macrophyte surveys appears to be comparatively small. There is no suggestion that the difference between surveys in different years is greater when a water body is subsampled using the standard protocol than when whole-lake surveys are conducted. Temporal variation in EQR between years is much more significant than the variation between sub-basins on a given date. Consequently surveys are better deployed in different years to reduce uncertainty.

ii. Because variation due to sampling, temporal and spatial sources can be itemised it is possible to derive an estimate of the total variation associated with a given EQR. At the point where the total estimated error reaches its maximum (about 0.10 EQR SD units), sampling error accounts for 45 per cent, temporal variation for 40 per cent and large-scale variation for 15 per cent of the total variation.

iii. Estimates of the variation in EQR are associated with a rigorous analysis of the available data. In some cases the structure of this data is not ideal and the modelled error distribution should be checked against data collected to a standard survey format and frequency once this becomes available.

iv. Error is not distributed symmetrically and, on the basis of the available data, is higher in the region of the EQR gradient associated with poor status. This may reflect the reduction in buffering capacity and greater sensitivity to external perturbations when, for example, the number of taxa at a site is very low.

v. Various factors probably dampen the error associated with classifications based on lake macrophytes. These include a well-designed survey method that mitigates against spatial variation, homogeneity of lake environments,

widespread distribution of macrophyte species in lakes wherever suitable habitat occurs, and ability of a multimetric approach to cancel out large variations in individual metrics.

vi. Confidence of classification increases with survey effort because the standard error associated with a given mean EQR is reduced. However, at a class boundary it will always be impossible to discriminate between classes. Even with the maximum possible survey effort (one survey per year) it will not be possible to classify about 10 per cent of sites as good or better or moderate or worse with the desired 95 per cent confidence.

vii. Two surveys in separate years within a monitoring cycle will normally be sufficient to classify a lake with more than 95 per cent confidence based on macrophytes when the mean EQR lies in the middle of a class. This survey frequency is adequate to control the number of sites that cannot be assessed as moderate or worse or good or better with 95 per cent confidence. Increasing the survey frequency to three surveys per cycle has relatively little effect on the number of sites that could be classified with 95 per cent confidence.

viii. The cost element of macrophyte surveys is concentrated on time spent in the field. Post-survey costs are minimal and therefore the overall resource required to deliver classification with 95 per cent confidence is likely to compare favourably with other biological quality elements.

10 Future perspectives

10.1 Introduction

The LEAFPACS classification tool for lakes provides a holistic assessment of lake ecological status based on macrophytes. Macrophytes play a pivotal role in lake ecology and the different metrics employed in the classification system reflect the different elements of this role. The tool relies on a large database of lake macrophyte surveys that provides comprehensive coverage of the types, geographical distribution and quality of lakes in the UK, and is underpinned by an intensive analysis of macrophyte-environment relationships. LEAFPACS appears to deliver classifications of UK lakes that are compatible with those provided by other quality elements (Phillips, 2007). The classifications themselves have been subject to testing against common datasets of European lakes and found to be in line with classifications by other member states participating in the inter-calibration exercises within NGIG or CGIG. This chapter addresses possible limitations, suitable refinements and perspectives on the use of the tool in the near future.

10.2 Refinements to sampling protocol

The recommended sampling protocol, based on perimeter, shoreline and boat transects repeated within multiple sectors has been well trialled and now used in around 500 lake surveys. The results of these surveys should be collated and considered alongside previous whole-lake surveys of the same water bodies. By considering individual sampling points on transects as individual records, it would be possible to explore species-accumulation curves in different lake types. This could be used to assess whether the current sampling effort is sufficient to reflect the species pool of a lake, or, equally, if there are opportunities for scaling down effort in particular lake types.

Lake macrophyte surveys are a comparatively labour-intensive exercise and are likely to require more than three man days for many WFD water bodies. There may be opportunities to adopt a more strategic approach to monitoring through savings on the survey effort invested in sites likely to be high status, or by applying an abbreviated version of the protocol in a proportion of surveys during a monitoring cycle. This would allow proportionally greater monitoring of moderate or poor status waters to identify signs of ecological improvement or further deterioration. Possible economies in survey design include the use of intensive strandline surveys to establish a species list for a water body, without the use of other survey methods, although this would require trialling. A second possibility relates to the use of key indicators of ecological quality. These might be species that are easily identified and which show a strong association with a particular status in water bodies of a certain type. For example, *Lobelia dortmanna*, which is easily identified and can be spotted from some distance when flowering, is a characteristic species of low-alkalinity, Atlantic lakes. Of 1,500 surveyed lakes in the UK that contained *Lobelia* 79 per cent were of high status and 99 per cent of good or better status. Consequently, *Lobelia* may be a rapid and reliable indicator of good ecological status. Such an approach could offer only a simple ‘health check’ on the status of a site, rather than proper confirmation of an EQR, and therefore would be unsuitable for surveillance monitoring.

Other potential refinements relate to the level of identification in some plant groups. Widespread use of the term ‘filamentous algae’ as a catch-all for almost all macroalgae

undoubtedly leads to a loss of discriminatory potential. Similarly, bryophytes in shallow waters or in the vicinity of inflows are almost certainly under-recorded. It is questionable whether genus level identification of all macroalgae in lakes would bring sufficient reward to justify the time and expense, yet some basic level of discrimination between, for example zygnematalean algae, small unbranched filamentous greens (such as *Ulothrix*) and large trailing growths of *Cladophora*, would be potentially useful.

10.3 Limitations

LEAFPACS is designed to assess the ecological status of a range of lake types and geographical distribution. Although it should give results in line with a common view of the majority of lakes (and certainly lakes of the size or designated status targeted by WFD), in some instances the classification will not resound fully with the views of those who have in-depth knowledge of a given lake. Sometimes this will be because the tool is applied to a situation it was not designed for or is simply beyond the limits of its usefulness. Examples of these situations include:

- Brackish lakes: information on additional environmental factors, such as extent of tidal exchange, is required to classify these water bodies, and to ensure separation from transitional and coastal waters. They also need to be considered as an independent lake type, rather than treated as part of a continuous gradient from low-alkalinity, deep lakes to very high-alkalinity shallow lakes. The presence of characteristic type-specific species (such as *Ruppia* spp, or certain charophytes, such as *Lamprothamnion*) should contribute to the classification of brackish lakes. Current evidence suggests that a shift from characteristic angiosperms (such as *Ruppia* spp) to opportunistic macroalgae (such as *Ulva*) and cyanobacteria is a cost-effective approach for evaluating coastal lagoons and identifying the impacts of increased nitrogen loading (Orfanidis *et al*., 2008).

- Very shallow lakes where much of the water body is shallower than 0.8 m: when the entire littoral zone lies inside the maximum rooting depth of most emergent plants, these are likely to dominate over submerged or floating leaved vegetation, unless strong additional environmental controls (such as wave exposure) are operating. Consequently ,the extent and richness of aquatic plant assemblages will appear compromised.

- Highly dystrophic lakes: lakes which are strongly coloured by humic substances appear to be unusual in the UK compared to northern Scandinavia where they are the norm. Humic lakes will potentially support a lower cover and less diverse aquatic vegetation than a clear water equivalent (Toivonen and Huttunen, 1995), and may also appear slightly enriched as a result of greater supply of organic forms of phosphorus. Consequently, since lake colour was not available for inclusion as a model term, LEAFPACS might be expected to discriminate against the most strongly humic lakes in the UK. A recent application of the ECOFRAME classification scheme to Finnish lakes concluded that performance improved significantly after stratifying lakes based on their humic content (Nykanen *et al*., 2005).

- Temporary water bodies (such as vernal pools, dune slacks, turloughs): none of the water bodies used in model development have water level fluctuations which extend as far as a complete loss of water during the growing season. Desiccation is a driving influence on the composition and life cycles of the aquatic flora of temporary waters (see Fernadez-Alaez *et al.*, 1999). For this reason, and since temporary water bodies often have

high conservation value (Mediterranean temporary pools are a priority habitat type under the EU Habitats Directive), it would be inappropriate to use LEAFPACS for their assessment. Other tools might be suitable for the assessment of temporary ponds (such as PSYM, Williams *et al*., 1999), although these might need some modification to be made WFD compliant. Alternatively, specialist WFD tools designed for wetland assessment might be applied to temporary water bodies. In the UK almost all temporary water bodies are well below the size threshold considered by the WFD.

- Recently created water bodies: some recently created water bodies (such as gravel pits that have been inactive for less than 10 years) maybe in a sufficiently early successional state that they cannot be considered in any form of equilibrium with environmental conditions. Charophytes are common early colonists of recently disturbed sites, such as gravel pits (Stewart and Church, 1992), and depending on those species present might imply a level of ecological quality that would not be sustained in the future, even without further environmental change.

- Water bodies subject to some forms of hydromorphological disturbance in the absence of nutrient enrichment: currently it is unclear how well the classification system can deal with water bodies, such as reservoirs, where water level fluctuations may be a significant influence on the ecology, but nutrient-related pressures may be weak or lacking. The potential to develop a simple index of disturbance, based on sensitivity of hydrophytes to water level fluctuations, is currently under consideration. Applying the method to heavily modified water bodies is considered further in Section 10.9.

In other instances a lake may have unusual attributes that are not adequately reflected in the environmental variables used to predict the biology under reference conditions, or which place that lake outside the envelope of conditions represented by the available population of reference surveys. Also, the perception of the status of a lake may integrate information from several biological quality elements into an expert view, it may incorporate a historical dimension of improvement or deterioration that cannot be reflected by a single face value survey, or it may reflect a view based on a different environmental policy driver, such as the Habitats Directive (Mainstone, 2008).

10.4 Devising additional metrics

LEAFPACS is a multimetric system for assessing lake macrophytes that takes account of collective responses (empirical or assumed) to a range of pressures. However, hydromorphological impacts are not explicitly covered within the assessment approach reported here. In so far as the complexity, stability and connectivity of the littoral zone are all affected by hydromorphological pressures (see Moss, 2008) it should be safe to assume that the existing suite of metrics will cover more severe cases of hydromorphological pressure. Conversely, the more benign effects of, for example, small or phased fluctuations in water level on macrophyte composition are likely to be subtle and beyond the reach of the present tool. There is a reasonable scientific understanding of the effects of water level fluctuations on macrophyte distribution in lakes, based on a combination of experimental work and observation (see Hellsten and Riihimaki, 1996; Riis and Hawes, 2002; Peintinger *et al.*, 2007). Consequently the raw material exists to build a compositional metric similar to LMNI that covers hydrological regime. A possible template exists in the form of the index recently proposed by Hellsten and Mjelde (2009) for Scandinavian lakes, although differences in extent of ice scour and in water level regime in relation to climate and use would probably necessitate some adjustments before this index could be applied in the UK. A possible

index might be based purely on hydrophytes, or could integrate information from emergent macrophytes, since these are likely to be affected by factors such as duration of inundation, sediment supply or ice scour. Alternatively, emergent macrophytes could be considered independently (in terms of composition or richness) as a more general indicator of hydromorphological pressures that could integrate factors such as shoreline modification. Currently the major shortcoming for hydromorphological indices is the difficulty in validation, since the number of UK lakes with both water level and macrophyte survey data is small (tens of lakes). A pragmatic short-term solution to progress the biological assessment of hydromorphological pressures using macrophytes would be to link Lake Habitat Survey and macrophyte survey databases and use this to explore the sensitivity of macrophyte structure to specific or general aspects of hydromorphological impact.

A major opportunity exists to refine the abundance metric by using, in its place, the maximum depth of macrophyte colonisation (or the ratio of this value to lake maximum depth). This is potentially the most useful and unambiguous measure of the extent of macrophyte growth in a lake and this information will now be available from around 400 surveys of a range of lake types. Maximum depth of colonisation is already integrated in some WFD lake macrophyte classification tools (see Free *et al*., 2006, Coops *et al*., 2007). The total percentage volume of the water column infested (PVI) would also be useful but is more subjective and reflects the product of spatial extent of vegetation and plant growth form which is already covered under other metrics. Because the depth of colonisation reflects light regime, which in turn partly reflects chlorophyll concentrations (Canfield *et al*., 1985), it could be seen as a simple vehicle for connecting impacts of nutrient enrichment on primary producers to secondary effects on consumers. Logically the influence of macrophytes on a range of biotic, metabolic and limnological processes in lakes depends more on the abundance and productivity of the vegetation than on its composition (Gasith and Hoyer, 1997) and in this respect a robust measure of macrophyte abundance might merit greater weighting in macrophyte-based assessments of lake ecological status. In deep (and generally large) lakes (where the majority of the bed lies below the compensation depth of macrophyte growth) it is questionable whether macrophytes naturally reach the abundance needed to influence limnological and metabolic processes, although even limited cover or isolated plant patches can still influence biotic interactions (see Conrow *et al*., 1990). Consequently, use of the maximum depth of colonisation as an additional macrophyte metric might best be reserved for lakes with maximum depths of under 15 m (shallow or very shallow lakes in the present typology, based on mean, not maximum, depth).

10.5 Refining the selection of reference sites

Most biological classification tools have been developed independently and have derived their own concept of reference condition, albeit sometimes adopting similar rules for pressure screening. Given that different classification tools deliver similar classification outcomes when applied to the same water body (see Section 10.8) it is unlikely that concepts of reference state are widely adrift. However, the WFD concept of reference strictly requires that all quality elements, including physicochemical and hydromorphological, would need to satisfy the normative definitions for high status before a water body could be regarded as being in reference condition. It is possible that a water body considered as being in reference condition for macrophytes might not be for diatoms, for example. Subsequent refinement of the pool of reference sites may therefore be required to take account of the classifications based on other quality elements. Comparisons with lake diatoms, where reference conditions were based on comparison of valve composition at the surface and base (around 1850) of sediment cores (Kelly *et al*., 2008), would be especially worthwhile, although the number of water bodies with overlapping data is comparatively small.

Because reference sites are likely to be located in areas of low human population density they mostly fall outside existing environmental monitoring networks. Indeed, we generally know far more about environmental conditions in relatively impacted sites and consequently have little understanding of conditions and natural variability within potential reference sites (Irvine, 2004). A future stratified sampling of chemistry in a subset of lakes of different types used as reference sites here would be beneficial for validation purposes. It would also extend the gradients for testing performance of different metrics and for devising the best rules for combining metrics.

10.6 Improving predictive models

Models for improving the prediction of the biota under reference conditions might be refined in one of three main ways: (1) more rigorous screening of reference sites to exclude unexplained outliers; (2) the addition of other environmental variables as predictors; (3) assessment of model sensitivity to measurement error in predictor variables; or (4) the adoption of different modelling approaches. The second two are considered below in greater detail.

Various environmental variables might help explain variation in metric values in reference sites. It is unlikely that the performance of models for predicting LMNI can be much improved. However, variables such as water colour, plus different aspects of lake morphometry such as shoreline configuration and basin slope (Hakanson, 1981), could have additional effects on metrics such as macrophyte richness and abundance, because of their influence on light, substrate characteristics and wave erosion. A more spatially explicit approach that considered connectivity between different water bodies or the density of lakes in a given region or catchment might also improve the prediction of richness metrics. Application of existing models to some lakes (such as large or very deep lakes) might push these models beyond the envelope of conditions for which they were designed. Consequently, when values for predictor variables fall outside the range of values for reference sites, model predictions should be interpreted with caution.

Existing models are highly sensitive to measured alkalinity. In this context alkalinity is used as a continuously varying surrogate for geology. Alkalinity is treated as invariant and independent of anthropogenic influence, although in reality enrichment of lakes is likely to elevate alkalinity, either indirectly through effects on primary production, or directly via the mode of fertiliser application. A model to determine reference alkalinity from drift and solid geology would therefore be a useful improvement, both to the our classification tool and tools for other BQEs that use alkalinity as a predictor. Some models use a relatively large number of predictors and might be made more parsimonious with only a small cost in terms of greater prediction error.

This project uses relatively simple generalised linear models to directly predict metric values. Alternative modelling approaches might reduce prediction error or provide an alternative framework in which to develop classification tools. Other modelling approaches employed successfully to predict expected metric values directly include back propagation neural networks (Walley and Fontama, 1998) which offer superior predictive power and lower model bias than the existing route to prediction (cluster analysis followed by multiple discriminant analysis) employed within RIVPACS. Multi-response neural networks for predicting entire community membership have also been found to significantly outperform species-by-species logistic models and conventional approaches for modelling invertebrate assemblage types (Olden *et al*., 2006) and consequently could be a powerful quantitative tool for future biomonitoring. Tison *et al*. (2007) found that community ordination of diatoms using self-organising maps, followed by prediction of community types from environmental variables using a multi-layer perceptron was superior to traditional approaches based on discriminant analysis.

Bayesian reasoning (see Ellison, 1996) lends itself naturally to the direct assessment of status based on deviation from reference conditions, and uncertainty in that assessment, circumventing the need for an EQR. Such approaches are relatively untested and merit future consideration. Bayesian reasoning has been incorporated into some prototype river invertebrate classification tools and shows promise in diagnosing sources of stress and predicting biological responses (Walley *et al*., 2002).

Species-based models offer a different approach to tool development. This would require a different philosophy in which observed species were assessed against the individual probabilities of occurrence of expected species. Generalized additive models (GAMs) exhibit considerable promise (as demonstrated in this study using a range of Charophytes) but tend to perform best for species with high prevalence (Meynard and Quinn, 2007) and therefore may not be suited to most macrophytes. Recent studies have also emphasised the need for spatially explicit approaches (for example, taking account of spatial dependence, spatial hierarchies and irregular sampling intensities) in building species-based models (Latimer *et al*., 2006; Diez and Pulliam, 2007). There has been some success in the development of species-specific models for aquatic plants in lakes (Heergard *et al*., 2001) and rivers (Barendregt and Bio, 2003). Species-based models are, however, reliant on imperfect survey data, due partly to the scale of detection bias (Royle *et al*., 2007). Such forms of bias are potentially acute for aquatic macrophytes given the difficulty of observation and patchiness of distribution, and hence present a considerable challenge to species-based modelling.

10.7 Modifying rules for combining metrics

The rule to combine metrics to provide an overall EQR for a water body takes into account the shape of the relationship between each metric and pressure data. On this basis a variable weighting is given to richness metrics, which means that high EQRs for richness relative to LMNI have a neutral influence in moderate to low productivity sites, but would elevate the EQR of high productivity sites. Some of the type-specific exploration of other metrics suggests that this approach could be extended to the abundance, algal and invasives metrics so that they have lower weight in naturally less productive lakes. At lower productivity there are various sources of stress or disturbance (see wave exposure, water level fluctuation) that might naturally distort the distribution of cover values, and hence the abundance metric. Similarly, exposed shorelines with extensive hard substrate in oligotrophic lakes might be naturally prone to late season flushes of filamentous algal growth. This metric also appears to merit lower weight in less productive lakes than it is currently given. For example, most quality elements place the oligotrophic lake Wastwater (WBID 29183) at high or good status while the latest macrophyte survey data suggests that it is moderate status, largely by virtue of extensive filamentous algal growths recorded at the time of survey.

In previous intercalibration exercises, the quality of survey data from other member states has precluded calculation of metrics such as abundance and algal cover that feature in the UK method. Consequently, it has not been possible to assess their wider influence on classification.

10.8 Integration with diatoms

Based on an intercalibration Option 3 approach, Phillips (2007) found close agreement between classifications of lakes using phosphorus, chlorophyll, macrophytes and diatoms independently. This suggested that all lake classification tools sensitive to eutrophication have a broadly similar view of ecological change at class boundaries. Classifications based on macrophytes were within one class of those based on other

quality elements in 76 per cent of cases. In general, the macrophyte classification was slightly more precautionary (by 0.14 class units) than other quality elements, which might reflect the multimetric basis of the macrophyte tool or wider sensitivity of the classification to pressures other than eutrophication (such as invasive species).

The need to consider classifications based on diatoms and macrophytes is more acute since, ultimately, both are components of the same quality element and therefore at some stage their classifications will need to be resolved in deriving overall ecological status for a water body. Curiously, macrophyte-based classifications of lakes are in general more precautionary than those using diatoms (by about 0.2 class units) which is the inverse of the situation for classifications on rivers. The pattern of differences seems most inconsistent in high-alkalinity lakes. Consideration of the sites affected indicates that some are densely vegetated shallow lakes in which nutrient sequestration by macrophytes might favour less nutrient availability and consequently a higher status diatom than macrophyte assemblage. Other sites tend to be shallow reservoirs. In this case it is possible that water level fluctuations buffer the effects of nutrient availability on macrophytes by imposing sufficient disturbance to prevent dominance by tolerant species, whereas diatoms do not benefit from this buffer (Van Geest *et al*., 2005).

Given that the macrophyte classification tool provides a holistic assessment whereas the diatom tool is based on a single metric, TDI, it is likely that future efforts to combine diatom and macrophyte classifications will need to consider a wider range of options than merely averaging or worst case.

10.9 Application to artificial and heavily modified water bodies

All water bodies considered in this project have been assessed against a basic environmental objective of good ecological status. However, some of these water bodies will be designated as artificial (AWB) or heavily modified water bodies (HMWB), for which the basic objective is good ecological potential. In the case of lakes, use as reservoir for water supply or hydropower purposes will be the major reason for designating a site as an HMWB. Maximum ecological potential (MEP) is defined as: *"The values of the relevant biological quality elements (that) reflect, as far as possible, those associated with the closest comparable surface water body type, given the physical conditions which result from the artificial or heavily modified characteristics of the water body* (WFD, European Union, 2000).

In such cases it is necessary to assess the aquatic vegetation against an alternative benchmark that essentially allows for any differences from high ecological status that can be attributed to hydromorphological modifications to sustain the designed use. To minimise the departure of the biology from the closest comparable surface water type at high status, hydromorphological modifications must take account of the range of possible mitigations that ensure the best approximation to an ecological continuum (for example, maximise lateral connectivity between water body and riparian zone). The current LEAFPACS approach would allow some metrics to be 'unplugged' from the classification system, if these were deemed unsuitable to assess reservoirs. It would also be possible, using those metrics likely to be most sensitive to physical modifications to serve the design purpose (such as increased water level fluctuations), to extract a pool of water bodies most likely to satisfy the requirements of MEP. These could then act as the reference point to assess other water bodies modified for the same purpose. Thus, for reservoirs, the pool of water bodies with the highest EQR for a hydromorphological metric could be used to identify HMWBs or AWBs minimally impaired by modifications to hydrological regime or shoreline habitat. Because these

represent the best available biology, it would be assumed that the range of effective mitigation strategies are in place. These sites could then serve as the basis for measuring the ecological potential of other similar water bodies. This template was used by Willby (2008) to develop a system to assess ecological potential of canals in the UK.

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, 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 to avoid overrepresentation or duplication. Data can be cover values on a standard scale or can be expressed as presence-absence. If large amounts of data from different sources obtained by different methods are being pooled, analyzing presence-absence data is likely to be more reliable. 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 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 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 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 also be used during intercalibration if 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. 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 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 taxon in the dataset.

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 species with more than 10 occurrences is used to generate 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).

References

BARENDREGT, A. AND BIO, A.M.F., 2003. Relevant variables to predict macrophyte communities in running waters. *Ecological Modelling*, 160, 205-217.

BENNION H, HILTON J, HUGHES M, CLARK J, HORNBY D, FOZZARD I, PHILLIPS G AND REYNOLDS C, 2005. The use of a GIS-based inventory to provide a national assessment of standing waters at risk from eutrophication in Great Britain. *Science of the Total Environment*, 344, 259-273.

BENNION, H. AND BATTARBEE, R., 2007. The European Union Water Framework Directive: opportunities for palaeolimnology. *Journal of Palaeolimnology*, 38, 285-295.

BIGGS J., WHITFIELD M., WILLIAMS P., CORFIELD A., FOX G. AND ADARE K., 1998. *Biological techniques of still water quality assessment: Phase 2 method development.* Draft report to Environment Agency of R&D project A05(94).

BRAZNER JC, DANZ NP, TREBITZ AS, NIEMI GJ, REGAL RR, HOLLENHORST T, HOST GE,, REAVIE ED, BROWN TN, HANOWSKI JM, JOHNSTON CA, JOHNSON LB, HOWE RW AND CIBOROWSKI JJH, 2007 Responsiveness of Great Lakes wetland indicators to human disturbances at multiple spatial scales: A multi-assemblage assessment. *Journal of Great Lakes Research*, 33, 42-66.

BRYANT, J.A., STEWART, N.F. AND STACE, C.A., 2002. A checklist of Characeae of the British Isles. *Watsonia*, 24, 203-208.

CANFIELD DE, 1985. Relations between water transparency and maximum depth of macrophyte colonization in lakes. *Journal of Aquatic Plant Management*, 23, 25

CARBIENER R, TREMOLIERES M, MERCIER JL AND ORTSCHEIT A, 1990. Aquatic macrophyte communities as bioindicators of eutrophication in calcareous oligosaprobe stream waters (Upper Rhine Plain, Alsace). *Vegetatio*, 86, 71-88.

CEN, 2003. *Water quality – Guidance standard for the surveying of macrophytes in lakes* - CEN/TC230/WG2/TG3. Geneva: Comité European de Normalisation.

CHANDLER, J.R., 1970. A biological approach to water quality management. *Water Pollution Control*, 69, 415-422.

CLARKE, R.T., 2005. *Error/Uncertainty module software STARBUGS. STAR Bioassessment Uncertainty Guidance Software Release 1.2* (July 2005) User Manual. Centre for Ecology and Hydrology.

CLARKE, R.T., 2000. Uncertainty in estimates of biological quality based on RIVPACS. In: Wright, J.F., Sutcliffe, D.W. and Furse, M.T. Editors. *Assessing the biological quality of fresh waters: RIVPACS and other techniques*. Freshwater Biological Association. 39-54.

CLARKE, R.T., FURSE, M.T. *et al*., 1996. Derivation of a biological quality index for river sites: comparison of the observed with the expected fauna. *Journal of Applied Statistics* 23, 311-332.

CONROW, R., ZALE, AV. AND GREGORY, RW., 1990. Distributions and abundances of early life stages of fishes in a Florida lake dominated by aquatic macrophytes. *Transactions of the American Fisheries Society*, 119, 521-528.

COOPS H, KERKUM FCM, VAN DEN BERG MS AND VAN SPLUNDER I, 2007. Submerged macrophyte vegetation and the European Water Framework Directive: assessment of status and trends in shallow, alkaline lakes in the Netherlands. *Hydrobiologia*, 584, 395-402.

CROFT M.V. AND CHOW-FRASER P., 2007. Use and development of the wetland macrophyte index to detect water quality impairment in fish habitat of great lakes coastal marshes. *Journal of Great Lakes Research*, 33, 172-197.

DEMARS, B.O.L. AND EDWARDS, A.C., 2009. Distribution of aquatic macrophytes in contrasting river systems: A critique of compositional-based assessment of water quality. *Science of the Total Environment*, 407, 975-990.

DEMARS, B.O.L. AND THIEBAUT, G., 2008. Distribution of aquatic plants in the Northern Vosges rivers: implications for biomonitoring and conservation. *Aquatic Conservation-Marine and Freshwater Ecosystems*, 18, 619-632.

DEN HARTOG, C. AND SEGAl, S., 1964. A new classification of the water plant communities. *Acta Botanica. Neerlandica*. 13, 367-393.

DIEHL, S. AND KORNIJOW, R., 1998. Influence of submerged macrophytes on trophic interactions among fish and macroinvertebrates. In: Jeppsen, E. *et al.* (Editors) *Structuring Role of Submerged Macrophytes in Lakes*. 24-46.

DIEKMANN, M, 2003. Species indicator values as an important tool in applied plant ecology - a review. *Basic and Applied Ecology*, 4, 493-506.

DIEZ, JM. AND PULLIAM, HR., 2007. Hierarchical analysis of species distributions and abundance across environmental gradients. *Ecology*, 88, 3144-3152.

DODKINS I., RIPPEY B. AND HALE P., 2005. An application of canonical correspondence analysis for developing ecological quality assessment metrics for river macrophytes. *Freshwater Biology*, 50, 891-904.

DUIGAN, C., KOVACH, W. AND PALMER, M., 2006. *Vegetation communities of British lakes: a revised classification*. JNCC.

EATON, J.W. AND WILLBY, N.J., 2003. *Monitoring Protocol for the Waterplant Conservation Interest of Canals*, Report to English Nature.

ECOSTAT, 2005. *Eutrophication assessment in the context of European water policies*. ECOSTAT WG2A. v 7.1.

ELLIS, JC, 2006. *Uncertainty estimation for monitoring results by the WFD biological classification tools.* Environment Agency Science Report.

ELLISON, AM, 1996. An introduction to Bayesian inference for ecological research and environmental decision-making. *Ecological Applications*, 6, 1036-1046.

ERTSEN, ACD, ALKEMADE, JRM AND WASSEN, MJ, 1998. Calibrating Ellenberg indicator values for moisture, acidity, nutrient availability and salinity in the Netherlands, *Plant Ecology*, 135, 113-124.

EUROPEAN UNION, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 20000 establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities* L327: 1-73.

EWALD, J., 2003. The sensitivity of Ellenberg indicator values to the completeness of vegetation releves. *Basic and Applied Ecology*, 4, 507-513.

FERNANDEZ-ALAEZ, C., FERNANDEZ-ALAEZ, M. AND BECARES, E., 1999. Influence of water level fluctuation on the structure and composition of the macrophyte vegetation in two small temporary lakes in the northwest of Spain. *Hydrobiologia*, 415, 155-162.

FERREIRA MT, RODRIGUEZ-GONZALEZ PM, AGUIAR FC AND ALBUQUERQUE A, 2005. Assessing biotic integrity in Iberian rivers: Development of a multimetric plant index. *Ecological Indicators*, 5, 137-149.

FORBES, RS, 2000. Assessing the status of *Stratiotes aloides* L. (Water Soldier) in Co. Fermanagh, Northern Ireland (v.c. 33). *Watsonia*, 23, 179-196.

FREE, G., LITTLE, R., TIERNEY, D., DONNELLY, K. AND CARONI, R., 2006. A multimetric macrophyte lake index. In: *A reference-based typology and ecological assessment system for Irish lakes - preliminary investigations*. Irish EPA draft report.

GASITH, A. AND HOYER, MV., 1998. Structuring role of macrophytes in lakes: Changing influence along lake size and depth gradients. In: Jeppsen, E. *et al.* (Editors) *Structuring Role of Submerged Macrophytes in Lakes*. 381-392.

GASTON, K.K. AND BLACKBURN, T.M., 2000. *Pattern and process in macroecology*. Blackwell, Oxford.

HAKANSON L., 1981. Determination of characteristic values for physical and chemical lake sediment parameters. *Water Resources Research*, 17, 1625-1640.

HATZENBELER GR, KAMPA JM, JENNINGS MJ *et al*., 2004. A comparison of fish and aquatic plant assemblages to assess ecological health of small Wisconsin lakes . *Lake and Reservoir Management*, 20, 211-218.

HEEGAARD E, BIRKS HH, GIBSON CE, SMITH SJ, AND WOLFE-MURPHY S, 2001. Species-environmental relationships of aquatic macrophytes in Northern Ireland. *Aquatic Botany*, 70, 175-223.

HELLSTEN S AND MJELDE M, 2009. Macrophyte responses to water level fluctuation in Fennoscandinavian Lakes - Applying a common index. *International Association of Theoretical and Applied Limnology – Proceedings*, 30, 765-769.

HELLSTEN, S. AND RIIHIMAKI, J., 1996. Effects of lake water level regulation on the dynamics of littoral vegetation in northern Finland. *Hydrobiologia*, 340, 85-92.

HILL, M.O. AND ŠMILAUER, P., 2005. TWINSPAN for Windows version 2.3. Centre for Ecology and Hydrology and University of South Bohemia. Huntingdon and Ceske Budejovice.

HILL, M.O., MOUNTFORD, J.O., ROY, D. AND BUNCE, R.G., 1999. *ECOFACT 2a Technical Annex - Ellenberg's indicator values for British Plants.* NERC/DETR.

HILL, M.O., ROY, D.B., MOUNTFORD, J.O. *et al.*, 2000. Extending Ellenberg's indicator values to a new area: an algorithmic approach. *Journal of Applied Ecology,* 37, 3-15.

HILLI, M, KUITUNEN, MT AND SUHONEN, J, 2007. The effect of land use change on the vascular plant species turnover in boreal lakes. *Biodiversity and Conservation*, 16, 3951-3962.

HOLMES NTH, 1995. *Macrophytes for Water and other River Quality Assessments.* National Rivers Authority, Anglian Region.

HOOPER, DU, CHAPIN, FS, EWEL, JJ *et al*., 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecological Monographs*, 75, 3-35.

HORNUNG, M., BULL, K., CRESSER, M., ULLYETT, J., HALL, J., LANGHAM, S., LOVELAND, P. AND WILSON M., 1995. The sensitivity of surface waters of Great Britain to acidification predicted from catchment characteristics. *Environmental Pollution*, 87, 204-214.

HUGHES R.M., KAUFMANN, P.R., HERLIHY, A.T., KINCAID, T.M., REYNOLDS L, AND LARSEN D.P., 1998. A process for developing and evaluating indices of fish assemblage integrity. *Canadian Journal of Fisheries and Aquatic Sciences*, 55, 1618-1631.

HUSTON, MA, 1994. *Biological Diversity: the coexistence of species*. Cambridge Studies in Ecology. Cambridge University Press.

HUTCHINSON, GE, 1975. *A Treatise on Limnology. Volume 3. Limnological Botany*. Wiley, New York.

JAMES, C, FISHER, J, RUSSELL, V, COLLINGS, S AND MOSS, B, 2005. Nitrate availability and hydrophyte species richness in shallow lakes. *Freshwater Biology*, 50, 1049-1063.

JEPPESEN, E, SONDERGAARD, M, SONDERGAARD, M AND CHRISTOFFERSEN, K, 1998. *The structuring role of submerged macrophytes in lakes*. Springer-Verlag. New York.

JNCC, 2005. *Common Standards Monitoring Guidance for Freshwater Habitats and Species*. First version – March 2005, ISSN 1743-8160 (Online).

JONES, J.I., CLARKE, R.T., BLACKBURN, J.H., GUNN, R.J.M., KNEEBONE, N.T. AND NEALE, M.W., 2006. *Biological quality of lakes: Phase II: Quantifying uncertainty associated with macroinvertebrate sampling methods for lake benthos*. Environment Agency R&D Technical Report (13765).

JONES, JI, LI, W AND MABERLY, SC, 2003. Area, altitude and aquatic plant diversity. *Ecography*, 26, 411-420.

KELLY, M.G., JUGGINS, S., BENNION, H., BURGESS, A., YALLOP, M., HIRST, H., KING, L., JAMIESON, J., GUTHRIE, R. AND RIPPEY, B., 2008. *Use of diatoms for evaluating ecological status in UK freshwaters*. Environment Agency Science Report.

KOHLER, A, WONNEBERGER, R AND ZELTNER, G, 1973. Chemical data and aquatic vascular plants as indicators for pollution in the Moosach river system near Munich. *Archive fur Hydrobiologie*, 72, 533-549.

LANSDOWN, RV, 2007. *Assessment of the condition of the water courses of plain to montane levels with the Ranunculion fluitantis and Callitricho-Batrachion vegetation of the River Wye SAC*. Countryside Council for Wales.

LATIMER, AM, WU, SS, GELFAND, AE *et al*., 2006. Building statistical models to analyze species distributions. *Ecological Applications*, 16, 33-50.

MABERLY, SC, DAWSON, FH, WRIGHT, JF AND HILTON, J, 1999. *PLANTPACS (Phase 1): A study into the feasibility of producing a predictive system to assess river water quality and ecological status using macrophytes*. Environment Agency R&D report W1/017/01.

MACK, JJ, 2007. Developing a wetland IBI with statewide application after multiple testing iterations. *Ecological Indicators*, 7, 864-881.

MAINSTONE, CP, 2008. The role of specially designated wildlife sites in freshwater conservation – an English perspective. *Freshwater Reviews: A Journal of the Freshwater Biological Association*, 1, 89-98.

MCCORMICK FH, HUGHES RM, KAUFMANN PR *et al*., 2001. Development of an index of biotic integrity for the Mid-Atlantic Highlands region. *Transactions of the American Fisheries Society*, 130, 857-877.

MEYNARD, CN AND QUINN, JF, 2007. Predicting species distributions: a critical comparison of the most common statistical models using artificial species. *Journal of Biogeography*, 34, 1455-1469.

MILLER SJ, WARDROP DH, MAHANEY WM *et al*., 2006. A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. *Ecological Indicators*, 6, 290-312.

MOSS B *et al.*, 2003. The determination of ecological status in shallow lakes - a tested system (ECOFRAME) for implementation of the European Water Framework Directive. *Aquatic Conservation-Marine and Freshwater Ecosystems*, 13, 507-549.

MOSS B., WRIGHT J.F., FURSE M.T. *et al*., 1999. A comparison of alternative techniques for prediction of the fauna of running-water sites in Great Britain. *Freshwater Biology*, 41, 167-181.

MOSS, B, 2008. The kingdom of the shore: achievement of good ecological potential in reservoirs. *Freshwater Reviews: A Journal of the Freshwater Biological Association*, 1, 29-42.

MURPHY, K.J., 2002. Plant communities and plant diversity in softwater lakes of northern Europe. *Aquatic Botany*, 73, 287-324.

NAEEM, S. AND LI S.B., 1997. Biodiversity enhances ecosystem reliability. *Nature*, 390, 507-509.

NATURE CONSERVANCY COUNCIL, 1989. *Guidelines for selection of Biological SSSIs; Rationale, Operational approach and criteria*. NCC, Peterborough.

NEWBOLD, C AND PALMER, M, 1979. *Trophic adaptations of aquatic plants*. Chief Scientists Team Notes, No. 18. Nature Conservancy Council, Peterborough.

NYKANEN, M, KAIRESALO, T, MAKELA, S *et al*., 2005. A typology and ecological classification system for Finnish lakes: applicability of the ECOFRAME scheme. *Boreal Environment Research*, 10, 159-179.

O'CONNOR, RJ, WALLS, TE AND HUGHES RM, 2000. Using multiple taxonomic groups to index the ecological condition of lakes. *Environmental Monitoring and Assessment*, 61, 207-228.

OLDEN JD, JOY MK AND DEATH RG, 2006. Rediscovering the species in community-wide predictive modeling. *Ecological Applications*, 16, 1449-1460.

ORFANIDIS S, PINNA M, SABETTA L. *et al*., 2008. Variation of structural and functional metrics in macrophyte communities within two habitats of eastern Mediterranean coastal lagoons: natural versus human effects. *Aquatic Conservation-Marine and Freshwater Ecosystems*, 18, S45-S61.

PAAL, J, TREI, T AND VIIK, M, 2007. Vegetation of Estonian watercourses, III. Drainage basins of the Moonsund Sea, the Gulf of Riga and Saaremaa Island. *Annales Botanici Fennici*, 44, 321-344.

PALMER MA, BELL SL, BUTTERFIELD I, 1992. A botanical classification of standing waters in Britain: applications for conservation and monitoring. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 2, 125-143.

PEARSALL, WH, 1918. On the classification of aquatic plant communities. *Journal of Ecology*, 6, 75-84.

PEARSALL, WH, 1920. The aquatic vegetation of the English lakes. *Journal of Ecology*, 8, 163-201.

PEINTINGER, M., PRATI, D. AND WINKLER, E., 2007. Water level fluctuations and dynamics of amphibious plants at Lake Constance: Long-term study and simulation. *Perspectives in Plant Ecology Evolution and Systematics*, 8, 179-196.

PENNING, WE *et al.,* 2008. Classifying aquatic macrophytes as indicators of eutrophication in European lakes. *Aquatic Ecology*, 42, 237-251.

PHILLIPS, G, 2007. *Initial evaluation of lake classification tools sensitive to eutrophication*. TAG/LTT paper 123.

PHILLIPS, G., CARVALHO, L., MABERLY, S., REYNOLDS, C. AND WILLBY, N.J., 2003. The assessment of ecological quality of lakes in the Great Britain ecoregion: an update on current thinking and a possible approach for phytoplankton. *TeemaNord* 547, 35-39.

PRESTON, C.D., PEARMAN, D.A. AND DINES, T.D. 2002. *New Atlas of the British and Irish Flora*. Oxford University Press.

RATCLIFFE, D, 1977. *A Nature Conservation Review*. Cambridge University Press.

REAVIE E.D., KIRETA A.R., KINGSTON J.C. *et al*., 2008. Comparison of simple and multimetric diatom-based indices for Great Lakes coastline disturbance. *Journal of Phycology*, 44, 787-802.

RICKLEFS, R.E. AND SCHLUTER, D., 1994. *Species Diversity in Ecological Communities*. University of Chicago Press.

RIIS, T AND HAWES, I, 2002. Relationships between water level fluctuations and vegetation diversity in shallow water of New Zealand lakes. *Aquatic Botany*, 74, 133-148.

RINTANEN, T, 1996. Changes in the flora and vegetation of 113 Finnish lakes during 40 years. *Annales Botanici Fennici*, 33, 101-122.

RORSLETT, B, 1991. Principal determinants of aquatic macrophyte richness in northern European lakes. *Aquatic Botany*, 39, 173-193.

ROTHROCK PE, SIMON TP AND STEWART PM, 2008. Development, calibration, and validation of a littoral zone plant index of biotic integrity (PIBI) for lacustrine wetlands. *Ecological Indicators*, 8, 79-88.

ROUND, FE, 1991. Diatoms in river water-monitoring studies. *Journal of Applied Phycology*, 3, 129-145.

ROYLE JA, KERY M, GAUTIER R AND SCHMID H, 2007. Hierarchical spatial models of abundance and occurrence from imperfect survey data. *Ecological Monographs*, 77, 465-481.

RYDER, RA, 1965. A method for estimating potential fish production of north-temperate lakes.*Transactions of the American Fisheries Society*, 94, 214.

SAND-JENSEN K, RIIS T, VESTERGAARD O, LARSEN SE, 2000. Macrophyte decline in Danish lakes and streams over the past 100 years. *Journal of Ecology*, 88, 1030-1040.

SAYER C, ROBERTS N, SADLER J, DAVID C AND WADE PM, 1999. Biodiversity changes in a shallow lake ecosystem: a multi-proxy palaeolimnological analysis. *Journal of Biogeography*, 26, 97-114.

SCHAUMBURG J, SCHRANZ C, HOFMANN G, STELZER D, SCHNEIDER S AND SCHMEDTJE U, 2004. Macrophytes and phytobenthos as indicators of ecological status in German lakes - a contribution to the implementation of the Water Framework Directive. *Limnologica*, 34, 302-314.

SPENCE, DHN, 1967. Factors controlling distribution of freshwater macrophytes with particular reference to lochs of Scotland. *Journal of Ecology*, 55, 147-161

STELZER D, SCHNEIDER S AND MELZER A, 2005. Macrophyte-based assessment of lakes - a contribution to the implementation of the European Water Framework Directive in Germany. *International Review of Hydrobiology*, 90, 223-237.

STEWART, NF AND CHURCH, JM, 1992. *Red data books of Britain and Ireland: Stoneworts*. JNCC, Peterborough.

THIEBAUT, G., GUEROLD, F. AND MULLER, S., 2002. Are trophic and diversity indices based on macrophyte communities pertinent tools to monitor water quality? *Water Research*, 36, 3602-3610.

TISON J, PARK YS, COSTE M, WASSON JG, RIMET F, ECTOR L AND DELMAS F, 2007. Predicting diatom reference communities at the French hydrosystem scale: A first step towards the definition of the good ecological status. *Ecological Modelling*, 203, 99-108.

TOIVONEN H AND HUTTUNEN P, 1995. Aquatic macrophytes and ecological gradients in 57 small lakes in southern Finland. *Aquatic Botany*, 51,197-221.

UKTAG, 2006. *UK Environmental standards and conditions*. UK Technical Advisory Group on the Water Framework Directive. January 2006.

UKTAG, 2007. *UK Classification Scheme for Surface Waters*. UK Technical Advisory Group on the Water Framework Directive. Draft report, September 2007.

UKTAG, 2003. Work Programme Task 2.a (i) – *Typology for Lakes for the UK*; Guidance on Typology for Lakes for the UK. Summer 2004. UKTAG.

UKTAG, 2004. *Guidance on the assessment of alien species pressures*. United Kingdom Technical Advisory Group on the Water Framework Directive.

VAN GEEST GJ, COOPS H, SCHEFFER M AND VAN NES EH, 2007. Long transients near the ghost of a stable state in eutrophic shallow lakes with fluctuating water levels. *Ecosystems*, 10, 36-46.

VESTERGAARD O AND SAND-JENSEN K, 2000. Aquatic macrophyte richness in Danish lakes in relation to alkalinity, transparency, and lake area. *Canadian Journal of Fisheries and Aquatic Sciences*, 57, 2022-2031.

VIROLA T, KAITALA V, KUITUNEN M, LAMMI A, SIIKAMAKI P, SUHONEN J AND VIROLAINEN K, 1999. Species immigration, extinction and turnover of vascular plants in boreal lakes. *Ecography*, 22, 240-245.

WALLEY W.J., O'CONNOR M.A., TRIGG D.J. AND MARTIN R.W., 2002. *Diagnosing and predicting river health from biological survey data using pattern recognition and plausible reasoning*. Environment Agency Technical Report E1-056/TR2.

WALLEY, W.J. AND HAWKES, H.A., 1997. A computer-based development of the Biological Monitoring Working Party score system incorporating abundance rating, biotope type and indicator value. *Water Research*, 31, 201-210.

WALLEY, W.J. AND FONTAMA, V.N., 1998. Neural network predictors of average score per taxon and number of families at unpolluted river sites in Great Britain. *Water Research*, 32, 613-622.

WALLIN, M., WIEDERHOLM, T. AND JOHNSON, R.K., 2005. *Guidance on establishing reference conditions and ecological status class boundaries for inland surface waters*. CIS working group 2.3 – REFCOND.

WARNELINK, G.W.W., GOEDHART, P.W., VAN DOBBEN, H.F. *et al*., 2005. Plant species as predictors of soil pH: Replacing expert judgement with measurements. *Journal of Vegetation Science*, 16, 461-470.

WEST G, 1905. A comapartive study of the dominant phanerogamic and higher cryptogamic flora of aquatic habit in three lake areas of Scotland. *Proceedings of the Royal Society of Edinburgh*, 25, 967-1023.

WEST G, 1910. A further contribution to a comparative study of the dominant phanerogamic and higher cryptogamic flora of aquatic habit in Scottish lakes. *Proceedings of the Royal Society of Edinburgh*, 30, 65-181.

WIEGLEB G AND BRUX H, 1991. Comparison of life-history characters of broad-leaved species of the genus *Potamogeton*. 1. General characterization of morphology and reproductive strategies. *Aquatic Botany*, 39, 131-146.

WILLBY, N.J. AND EATON, J.W., 2002. *Plant conservation in the restoration of the Montgomery Canal to navigation*. British Waterways North West Region, Northwich, Cheshire. Research Contract: Interreg IIc for British Waterways.

WILLBY, N.J., 2005. Macrophyte inferred reference conditions and supporting chemistry for marl lakes. Advice note to Lakes Task Team.

WILLBY, N.J., 2007. Managing invasive aquatic plants: problems and prospects *Aquatic Conservation-Marine and Freshwater Ecosystems* 17, 659-665.

WILLBY, N.J., 2008. *WFD 61: Canal Classification Tool*. SNIFFER Final Report.

WILLBY, N.J. AND CASAS-MULET, R., 2009. *European beaver trial reintroduction to Knapdale: baseline survey of the aquatic and semi-aquatic macrophytes of the lochs, 2008*. Scottish Natural Heritage.

WILLBY, N.J., ABERNETHY, V.J. AND DEMARS, B.O.L., 2000. Attribute-based classification of European hydrophytes and its relationship to habitat utilization. *Freshwater Biology*, 43, 43-74.

WILLBY, NJ, PITT, JA AND PHILLIPS, G., 2009. *Ecological classification of UK rivers using aquatic macrophytes*. Environment Agency report. Project SC010080/SR1.

WRIGHT, J.F., 2000. An introduction to RIVPACS. In: Wright, J.F., Sutcliffe, D.W. and Furse, M.T. Editors. *Assessing the biological quality of fresh waters: RIVPACS and other techniques*. Freshwater Biological Association. 1-24.

WRIGHT, JF, FURSE, MT AND MOSS, D, 1998. River classification using invertebrates: RIVPACS applications. *Aquatic Conservation-Marine and Freshwater Ecosystems*, 8, 617-631.

YACHI S. AND LOREAU M., 1999. Biodiversity and ecosystem productivity in a fluctuating environment: The insurance hypothesis. *Proceedings of the National Academy of Sciences of the United States of America*, 96, 1463-1468.

List of abbreviations

ALG	Metric describing relative cover of green filamentous algae
ANOVA	Analysis of Variance
ASPT	Macoinvertebrate metric: Average Score per Taxon
AWIC	Macoinvertebrate metric: Acid Waters Indicator Community
AWMN	Acid Waters Monitoring Network
BGS	British Geological Survey
BMWP	Biological Monitoring Working Party
BSBI	Botanical Society of the British Isles
CB-GIG	Central Baltic Geographical Intercalibration Group
CCA	Canonical Correspondence Analysis
CCW	Countryside Council for Wales
CEH	Centre for Ecology and Hydrology
CEN	Comité European de Normalisation (European Committee for Standardisation)
CoC	Confidence of Class
COV	Metric describing mean cover per taxa of hydrophytes in a lake
CSM	Common Standards Monitoring
DAFOR	Abundance scoring system – Dominant, Abundant, Frequent, Occasional, Rare
DCA	Detrended Correspondence Analysis
ECRC	Environmental Change Research Centre
ENSIS	Environmental Scientific
ESG	Ecological State Group
EQI	Ecological Quality Index
EQR	Ecological Quality Ratio
EU	European Union
FBA	Freshwater Biological Association
FSC	Freshwater Sensitivity Class
GAM	General Additive Model
GES	Good Ecological Status
GIG	Geographical Intercalibration Group
GIS	Geographical Information System
GLM	Generalised Linear Model

IBI	Index of Biotic Integrity
INV	Metric describing relative cover of invasive hydrophytes in a lake
Irish EPA	Irish Environmental Protection Agency
JNCC	Joint Nature Conservation Committee
LIFE	Lotic Invertebrate Index for Flow Evaluation
LMNI	Lake Macrophyte Nutrient Index
MDA	Multiple Discriminant Analysis
MEI	Morpho Edaphic Index
MTR	Mean Trophic Rank
NCC	Nature Conservancy Council
NIEA	Northern Ireland Environment Agency
NILS	Northern Ireland Lake Survey
N_FG	Number of plant functional groups
N_TAXA	Number of hydrophyte taxa
N-GIG	Northern Geographical Intercalibration Group
PSYM	Predictive System for Multimetrics
PVI	Plant Volume Infested or Inhabited
RIVPACS	River Invertebrate Prediction and Classification System
SAC	Special Area for Conservation
SCM	Site Condition Monitoring
SDI	Shoreline Development Index
SEPA	Scottish Environment Protection Agency
SNH	Scottish Natural Heritage
SRP	Soluble Reactive Phosphorus (orthophosphate)
SSSI	Site of Special Scientific Interest
TDI	Trophic Diatom Index
TON	Total Oxidised Nitrogen
TP	Total Phosphorus
TRS	Trophic Ranking Score
TWINSPAN	Two Way Indicator Species Analysis
UCL	Univesity College London
UKTAG	UK Technical Advisory Group on the WFD
UWWTD	Urban Waste Water Treatment Directive
WBID	Water Body Identifier (unique code in GB Lakes Inventory)

WFD	Water Framework Directive

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