Variability components for macrophyte communities in rivers: summary report

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Integrated catchment science programme

Evidence Directorate

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Director of Evidence

Executive summary

Introduction

This report summarises and synthesises the results from work, undertaken between 2008 and 2009, to investigate the variability in river macrophyte communities, as measured by the LEAFPACS and JNCC methodologies. The project's goal was to increase our understanding of the sources of variability in survey measurements, in order to optimise the macrophyte survey methodologies and sampling strategies used by UK environmental protection and conservation agencies.

Key findings

Any environmental metric is subject to four different types of variation: spatial (among-reach and among-site), temporal (among-year and among-month), spatio-temporal and operator error. The latter relates to measurement errors (i.e. variability among different operators) and not to actual variation in the macrophyte community itself. Unlike the other sources of variation, operator error can, potentially, be controlled.

Spatial variation in macrophyte communities appears to be much higher than temporal variation. It is dominated by variation among reaches, with relatively little variation among the sites within a reach.

Operator variability also appears to be a considerable source of variation. Analysis of LEAFPACS surveys from 2008 revealed that operator variability can contribute significantly to variability in estimates of taxonomic richness and plant cover, and, to a lesser extent, to variability in community metrics such as EQR and RMNI. Operator variability is also a significant source of variation in JNCC surveys. This can frequently produce a biased result at the sub-target level, but is less important at the overall level of the condition assessment.

Temporal variation tends to be low relative to spatial variation and operator variability. Analysis of limited data suggests that inter-annual variation is larger than monthly variation for two of the analysed indicators (EQR and NTAXA), but lower for RMNI.

The most efficient sampling strategy for measuring the ecological status of a water body is to conduct surveys at replicate sites and reaches in replicate months and years. This allows all sources of spatial and temporal variation to be averaged out. However, practical and financial constraints may necessitate a different strategy. In situations where monitoring resources are the limiting factor, a cost-benefit analysis could be undertaken to select the most cost-effective strategy.

During site selection, care must be taken to avoid introducing conscious or sub-conscious bias into the survey programme. As far as possible, sites should be representative of conditions in the wider water body and surveyors should be mindful of the implications of site selection for both the precision and accuracy of the survey results.

There is no simple answer to the question: "How many surveys are adequate to characterise the status of macrophyte communities in a water body?" Critically, the number of surveys that are necessary will depend on the level of precision and confidence required in the results. However, variance components can be used to make an informed decision about the number of surveys per water body that are needed to classify a water body on the basis of mean LEAFPACS EQR.

The JNCC method is more sensitive to survey length than the LEAFPACS method because of this method's dependence on specific taxa listed in the constancy tables. The length of a JNCC survey will depend on the level of error that is acceptable to the organisation conducting the survey. Shorter length surveys will produce a higher rate of “false failures” as fewer taxa are likely to be observed.

Further research needs

There are several options available for reducing operator variability. These include: more rigorous and frequent training and testing of operators, consistent use of equipment (e.g. waders, snorkels), better adherence to current monitoring protocols or the development and use of clearer monitoring protocols, and employing pairs of operators instead of lone operators. The costs of implementing these options differ and it is important that their benefits are quantified so that a cost-effective approach can be adopted. We recommend a review of current surveying practices and a research investigation undertaken to evaluate the effectiveness of a variety of measures in reducing operator variability.

Understanding and quantifying variability in macrophyte communities, at appropriate spatial and temporal scales, is an essential prerequisite for developing cost-effective monitoring programmes and for measuring uncertainty in ecological and conservation status assessments. Environment Agency tools, such as VISCOUS and ROMANSE, already use the results of this project and allow users to make an informed judgement about the level of survey effort required to achieve a given level of confidence in the survey results, and to easily and accurately quantify the uncertainty in status assessments.

Contents

1 Introduction

1.1 Rationale for project

In the UK, the Statutory Agencies are required to survey river macrophyte[1] communities for several reasons. The environmental protection agencies – the Environment Agency (EA) in England and Wales, the Scottish Environment Protection Agency (SEPA) in Scotland and the Northern Ireland Environment Agency (NIEA) in Northern Ireland – monitor macrophytes for the Urban Waste Water Treatment Directive (UWWTD) and the Water Framework Directive (WFD). The conservation agencies – Countryside Council for Wales (CCW), Natural England (NE), Scottish Natural Heritage (SNH) and NIEA – survey and monitor river macrophyte communities as part of their work to select, designate and assess the condition of rivers, of conservation importance, in fulfilment of their duties under the Wildlife and Countryside Act 1981 and the Habitats Directive (HD). Future monitoring needs are likely to be driven mainly by the needs of the HD and WFD.

Macrophyte communities vary both spatially and temporally. Good understanding of these sources of variability is required to optimise the macrophyte survey methods and sampling strategies used by UK environmental protection and conservation agencies. The WFD requires sources of uncertainty in the monitoring programmes of Member States to be quantified. Specifically, estimates of the level of confidence and precision of the results provided by the monitoring programmes must be stated in the river basin management plan, and will be used to guide the development of cost-effective programmes of measures. This requires an understanding of how macrophyte communities vary in space and time, as well as an estimate of the magnitude of measurement error. In addition, UK conservation agencies need to understand variability of macrophyte communities and the uncertainty of resulting metrics, in order to refine guidance on common standards monitoring of SSSI and SAC rivers. In particular, there is a need to establish the minimum survey length required to gauge the conservation status of a river. Ultimately, there is a desire to move towards a common survey method for macrophyte monitoring across the UK

In response to these issues, the UK environmental protection and conservation agencies formed a project steering group to investigate variability and uncertainty in river macrophyte communities. Between 2008 and 2009, a series of three work packages were carried out to:

- analyse components of variability in historical macrophyte survey data (Report SC070051/SR2; Davey et al., 2008);
- address the gaps in the historical survey data through the use of survey data from 2008 (Report SC070051/SR3; Davey & Garrow, 2009); and
- analyse spatio-temporal patterns in macrophyte communities in the River Allen (Report SC070051/SR2A; Davey & Garrow, 2008).

The objectives of this report are:

1. to summarise the results of the three work packages;

[1] Macrophytes are larger plants of freshwater which are easily seen with the naked eye, including all aquatic vascular plants, bryophytes, stoneworts (Characeae) and macro-algal growths.

2. to synthesise the results and to develop practical advice for the design and operation of macrophyte monitoring programmes; and
3. to highlight future research needs.

1.2 Scope of work

1.2.1 Survey methodologies

There are three macrophyte sampling methods used widely in UK: the JNCC, MTR and LEAFPACS methodologies. The taxa listed (for recording) in each method are different, which make comparisons between the methods difficult.

JNCC Method

The method used by the UK conservation agencies for baseline surveys and condition assessments of SSSI and SAC rivers, surveys a 500-m reach, records all macrophyte species present and uses a relatively simple 3-point cover score. It was originally designed primarily for surveying river reaches, to record the maximum number of species present. This method is referred to in this report as the JNCC (Joint Nature Conservation Committee) method. The Common Standards Monitoring Guidance for Rivers (CSM) provides a Favourable Condition Table (FCT) that specifies attributes and targets (JNCC, 2005). There are four mandatory attributes, each with associated measures:

1. Habitat Functioning (Water flow and water quality);
2. Habitat Structure (substrate, channel and banks, structure);
3. Plant Community (species composition and abundance, reproduction); and
4. Negative Indicators (native species, alien/introduced species).

If a surveyed reach meets the specified targets then the condition of the river can be regarded as favourable. Conservation Agencies must report, to the EU, on the condition of SACs on a six year monitoring cycle.

MTR Method

The mean trophic rank (MTR; Holmes et al., 1999) method used by the Environment Agency is intended for use with paired samples upstream and downstream of a potential pollution source. It was originally designed to assess changes following phosphorus removal from sewage effluents. It records only a subset of the macrophyte species present over a relatively short survey length. Typically, cover is recorded on a 9-point scale. The surveyor can choose to survey a 100 or 500-m length of river; the Environment Agency normally surveys 100-m.

LEAFPACS Method

SEPA have stopped using MTR and have used the LEAFPACS protocol (UKTAG, 2009) since 2006 – this is based on surveying 5x100-m sites per water body and uses a 9-point cover scale. LEAFPACS has been designed to detect the impact, on macrophytes, of nutrient enrichment, alterations to river flows and modifications to morphological conditions. It computes a summary metric, called an Ecological Quality Ratio (EQR), which integrates five separate sub-metrics:

1. River Macrophyte Nutrient Index (RMNI);
2. River Macrophyte Hydraulic Index (RMHI);
3. Number of macrophyte taxa which are not helophytes (NTAXA);
4. Number of functional groups of macrophyte taxa that are not helophytes (NFG);
5. Percentage cover of green filamentous algae (ALG).

Each observed sub-metric score is divided by an expected score, which uses measured local physical and chemical conditions to predict the macrophyte community under minimally impacted reference conditions. The sub-metric EQRs are then combined to give an overall EQR between 0 and 1, with high ecological status represented by values close to one and bad ecological status by values close to zero. The EQR scale is divided into five classes ranging from high to bad ecological status.

1.2.2 Macrophyte parameters

The project examined various parameters that indicate or describe the status of the macrophyte community, although not every parameter was considered in every work package. The parameters analysed were:

1. Overall Ecological Quality Ratio (EQR);
2. Number of aquatic taxa (NTAXA);
3. Overall River Macrophyte Nutrient Index score (RMNI);
4. Total percentage cover (%Cover);
5. Squared Chord Distance (SCD).

NTAXA and RMNI are metrics calculated from the survey data. NTAXA is a diversity metric and is simply the number of non-helophyte taxa listed in LEAFPACS that are recorded in the survey. This is based on the main taxa list. RMNI is a compositional metric designed to indicate the extent of eutrophication. Each taxon on the survey list has a nutrient index score, and the associated RMNI for each survey is calculated using the equation:

$$RMNI = \frac{\sum_{j=1}^{n}\left(C_j \times R_j\right)}{\sum_{j=1}^{n} C_j}$$

where:

RMNI = River Macrophyte Nutrient Index value for survey;

R_j = River macrophyte nutrient index score for the j^{th} taxon;

C_j = Cover value for the j^{th} taxon, measured on a scale of 1 to 9;

n = number of LEAFPACS taxa observed in survey.

Total percentage cover of macrophytes is recorded directly by the operator during the survey. This parameter is not a required metric in LEAFPACS surveys and is not routinely recorded by all the contributors. Percentage cover values for individual taxa were also recorded in the River Allen dataset and analysed in Report SC070051/SR2A (Davey & Garrow, 2008).

SCD is an index that measures similarity in community composition between pairs of surveys. It was used to assess inter-operator variability in SC070051/SR3 (Davey & Garrow, 2009).

The SCD values were calculated using the following equation:

$$d_{ij} = \sum_{k=1}^{m}\left(\sqrt{p_{ik}} - \sqrt{p_{jk}}\right)^2$$

where:

d_{ij} = SCD between samples i and j;

m = total number of taxa;

p_{ik} = proportion of k^{th} taxon in sample i;

p_{jk} = proportion of k^{th} taxon in sample j.

Surveys performed using the JNCC methodology were analysed using the number of taxa found. Individual taxa were aggregated, where necessary, to provide a consistent taxa list.

1.3 Structure of report

The remainder of this report is divided into three sections. Section 2 provides an overview of the three work packages; Section 3 summarises the main findings and translates the results into practical advice for the design and operation of monitoring programmes; Section 4 identifies remaining gaps in our understanding of variability in macrophyte communities and makes recommendations for future research.

2 Project Overview

This Section provides an overview of the three work packages undertaken by WRc, plus a recent CCW investigation of inter-operator differences in the JNCC surveys and differences between JNCC and LEAFPACS surveys using the CSM methodology.

2.1 Components of variability in historical macrophyte survey data

The first work package analysed components of variability in historical macrophyte survey data. The study examined and quantified three sources of variability: (i) spatial variability (within- and between-reach); (ii) temporal variability (seasonal and annual); and (iii) operator variability (measurement error). The analysis focused on three community parameters – EQR, %Cover and NTAXA, it also considered spatial variability in two common macrophyte taxa.

Three data sources were used: the LEAFPACS development database, a SEPA database and a smaller Countryside Council for Wales (CCW) database. The LEAFPACS and SEPA databases were combined into a single analysis dataset. This included surveys conducted between 1976 and 2007. The CCW dataset was kept separate and included MTR and JNCC surveys performed between 1999 and 2007.

The study was successful in estimating spatial variability but, due to the limitations of the data available, was less successful in estimating temporal variability and operator variability. Unlike later studies, this study focused on spatial variation at a 3 km scale. The results suggested that surveys from a single 3 km reach could be representative of the conditions within the water body as a whole. However, there was inconsistency in the number and spacing of individual surveys within these 3 km stretches and variability on a smaller spatial scale could not be analysed.

The findings of this study were presented in SC070051/SR2 (Davey et al., 2008).

2.2 Components of variability in macrophyte survey data: 2008 survey

In response to the issues identified in the first study (Davey et al., 2008), a co-ordinated macrophyte survey programme was organised and carried out by the various agencies over the summer of 2008. This second study examined the following sources of variability: (i) spatial variability (variability among rivers, among reaches and among sites); (ii) temporal variability (variability among months); and (iii) operator variability (inter-operator and within-operator variability). The same three parameters considered in the earlier work (EQR, NTAXA and %Cover) were examined here along with the River Macrophyte Nutrient Index (RMNI) scores for the surveys.

Four separate datasets, from surveys carried out throughout the summer and autumn of 2008 by the EA, SEPA, CCW and NE, were combined into a single dataset for this study. These surveys were all designed to study variability and consequently this dataset was more focused than that used in the earlier work.

This study analysed spatial, temporal and inter-operator variability in greater detail than in the earlier report SC070051/SR2 (Davey et al., 2008). The contribution of each component to total variability was also determined. Surveys were only available for a single year, so inter-annual temporal variation could not be analysed.

The findings of this study were presented in SC070051/SR3 (Davey & Garrow, 2009).

2.3 Spatio-temporal patterns in macrophyte communities in the River Allen

For several years an intensive macrophyte monitoring programme has been undertaken on the River Allen, a chalk stream in Dorset. This study examined the natural spatial and temporal components of variation in the macrophyte community recorded along a 21 km length of river, between 1998 and 2008. The analysis used three parameters: total macrophyte cover and cover of the two most abundant taxa, *Ranunculus spp.* and *Scirpus/Sparganium*.

The data was from surveys performed by the same operator in the month of May for the period 1998-2008 (with the exception of 2001). The data consisted of percentage cover values for selected taxa, in each of 212 contiguous 100-m sites. This data is high resolution, but is not compatible with the datasets used in the other parts of the project (Sections 2.1 and 2.2). Therefore, this analysis was conducted independently from the rest of the study.

The analysis modelled spatio-temporal patterns in the macrophyte community of the River Allen. The changing spatial variability in the data was modelled over a continuous 20 km length of the river, which represents a greater spatial resolution than could be achieved in the other work packages. In addition, a simulation exercise compared the precision of different surveying strategies and numbers of surveys.

Although successful in elucidating scale-dependent patterns of spatial variation and their consistency over time, this study was based on data from just one river and it is not certain to what extent the results may be applicable to other rivers.

The findings of this study were presented in SC070051/SR2A (Davey & Garrow, 2008).

2.4 CCW analysis of River Dee surveys

In addition to the three main work packages described above, CCW made a detailed study of survey results from the River Dee in 2008. In particular, this explored the inter-operator differences in JNCC surveys on the same site and differences between JNCC and LEAFPACS surveys using the CSM methodology.

The Plant Community and Negative Indicators attributes were considered in this exercise. The Plant Community: Species Composition and Abundance attribute is split into three sub-targets, which all have to be met for the survey to pass:

1. species composition: The species recorded during a survey are compared with those in the constancy table for the relevant River Community Type. The constancy table gives a list of species expected to occur for that River Community Type at an abundance of I to V. A minimum of 60% of species with abundance V, 60% of species with abundance IV and 25% of species with abundance III, in the constancy table, should be present.

2. loss of species: The species recorded during a survey are compared to a baseline survey previously conducted on that reach. At least 60% of the species that had a cover value greater than 1 in the initial baseline survey should be present.

3. abundant species: At least 25 to 35% of the species recorded as dominant in the initial baseline survey should still be recorded as dominant.

The Plant Community: Reproduction attribute requires that a sufficient proportion of all aquatic macrophytes should be allowed to reproduce in suitable habitat, unaffected by river management practices. As no control measures, such as weed cutting, are undertaken at the survey locations on the Dee, this measure passes for all surveys.

The Negative Indicator: Negative Species attribute has two sub-targets that must be met:

1. for blanketweed, epiphytic or other algae, *Potamogeton pectinatus* or *Zannichellia palustris* cover values over 25% should be considered unfavourable. Cover values should not increase significantly from an established baseline.

2. for River Community Type VI (all the surveys are type VI) cover values over 25%, for taxa with Species Trophic Ranks (STRs) of 1 to 3, should be considered unfavourable. Cover values should not increase significantly from an established baseline.

CCW determined the pass/fail results for the surveys conducted by different operators and compared the results at sub-target and overall level.

3 Key Questions

This section summarises and synthesises the main findings of the three work packages by presenting answers to seven key questions.

3.1 What are the sources of variation in macrophyte monitoring data?

Any environmental metric is subject to four broadly different types of variation:

1. *Spatial variation* – at any given point in time, macrophyte communities vary from place to place. This spatial variation can be considered at several scales in an overall hierarchy. The spatial scales considered in this project were:

   (i) variation among rivers;

   (ii) variation among 500-m reaches within a river;

   (iii) variation among 100-m sites within a 500-m reach.

   (Note: The three work packages analysed spatial variation at different scales because of the different data structures that were used. The terminology used necessarily differs between the three reports. Notably, Report SC070051/SR2 (Davey et al., 2008) defined a reach as a 3 km length of river; to avoid confusion, a 3 km length is referred to as a *stretch* in this report. Note also that the JNCC method surveys 500-m *reaches*, whereas the LEAFPACS method surveys 100-m *sites*). The spatial hierarchy, as used in SC070051/SR3 (Davey & Garrow, 2009) is shown in Figure 3.1.

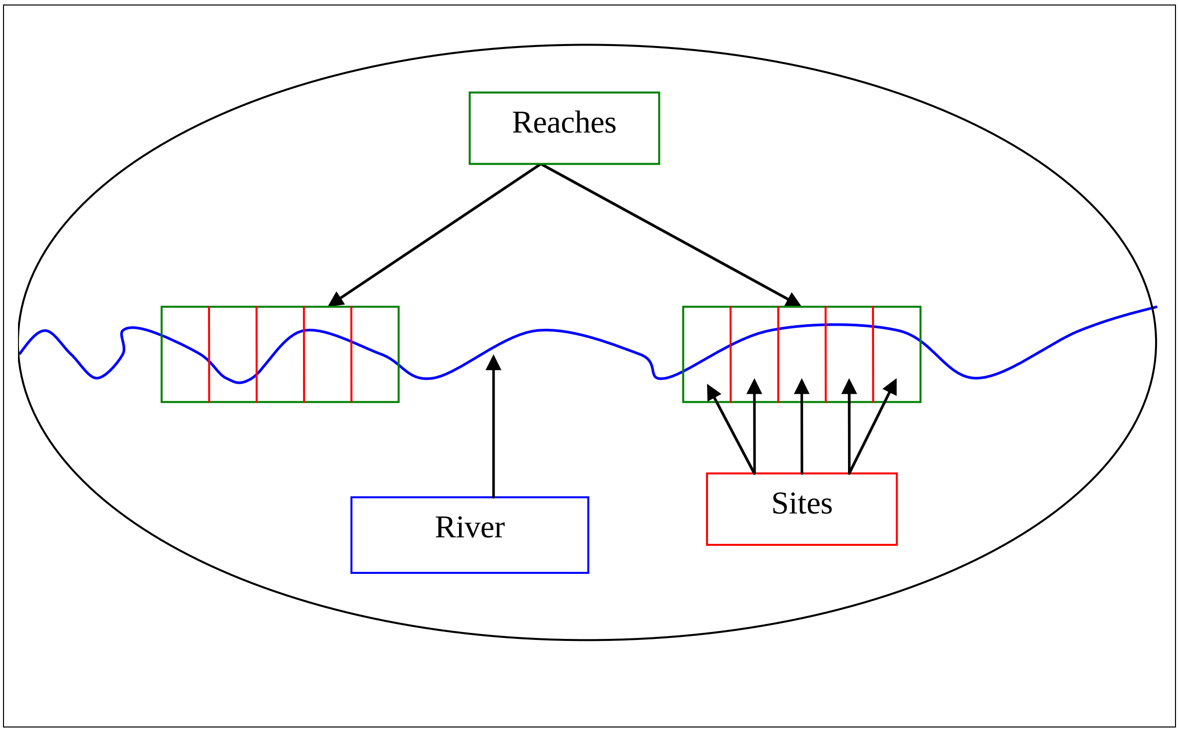

Figure 3.1 Spatial hierarchy of data used in analysis of 2008 survey data.

2. *Temporal variation* – at any given point in space, the macrophyte community will change over time This project considers the variation at two temporal scales:

    (i) variation among years (annual variation);

    (ii) variation among months within a year (monthly variation).

3. *Spatial-temporal interaction* – this occurs when a particular temporal effect operates differently in some locations than others. Temporal variation may be greater at some locations within a river than others. It can be distinguished from operator variability only if replicate surveys are undertaken at a number of locations on a number of occasions. This was partially possible in Report SC070051/SR3 (Davey & Garrow, 2009), but for simplicity spatio-temporal variation was pooled with temporal (monthly) variation.

4. *Operator variability* – this is variation generated by the measurement process and does not reflect actual variation in the macrophyte community itself. Operator variability is the difference between the true metric value and the value recorded on a particular sampling occasion. It is made up of both inter-operator variability, where different operators may produce different results for the same survey, and within-operator variability, where the same operator may produce different results when repeating the same survey.

Potentially operator variability can be reduced through improved training and methodologies. Natural sources of variation cannot be reduced, but their impact on uncertainty can be reduced by careful survey design.

3.2 What is the relative importance of these sources of variation?

A key aim of each of the three reports was to study the components of variation in survey results. SC070051/SR2 (Davey et al., 2008) found that spatial variation in macrophyte communities within a water body appears to be driven, predominantly, by variation between 100-m sites within a 3-km stretch, with relatively little additional variation among these stretches. However, it was not possible to distinguish between among-reach and among-site variation.

SC070051/SR3 (Davey & Garrow, 2009) was more successful in determining the absolute magnitude and relative importance of each component, and in quantifying both among-reach and among-site variation. The results of this analysis, for each of the studied parameters, are shown in Table 3.1 (as absolute variances) and Table 3.2 (as a % of the total variance).

Table 3.1 Components of variation (variances) from 2008 LEAFPACS surveys.

Component of variation	EQR	RMNI	NTAXA	%Cover
River	0.0182	1.849	3.951	450.5
Reach (500-m)	0.0083	0.196	4.058	289.2
Site (100-m)	0.0028	0.038	1.595	-209.6
Month[2]	0.0014	0.041	-0.76	NA[1]
Operator	0.0052	0.086	5.937	229.4
Total	**0.0359**	**2.210**	**14.781**	**759.5**

[1] Could not be estimated due to insufficient data.
[2] Includes spatio-temporal variation.

Table 3.2 Components of variation (as % of total variance) from 2008 LEAFPACS surveys.

Component of variation	EQR (%)	RMNI (%)	NTAXA (%)	%Cover (%)
River	50.7	83.6	25.4	46.5
Reach (500-m)	23.2	8.9	26.1	29.8
Site (100-m)	7.7	1.7	10.3	0.0[3]
Month[2]	3.9	1.9	0.0[3]	NA[1]
Operator	14.4	3.9	38.2	23.7
Total	**100.0**	**100.0**	**100.0**	**100.0**

[1] Could not be estimated due to insufficient data.
[2] Includes spatio-temporal variation.
[3] Negative variances set to zero.

Relative standard deviations (RSD), in which the standard deviation of the dataset is expressed as a percentage of the mean, were also calculated. These quantify the relative magnitude of variation for each of the studied parameters. EQR and RMNI were the least variable parameters, with RSDs of 25.6% and 22.0% respectively. %Cover was the most variable, with an RSD of 85.8%. NTAXA had an RSD value of 49.0%.

There are five key findings from these results:

- Derived parameters, such as EQR and RMNI, are less variable than raw parameters, such as %Cover and NTAXA. Therefore, EQR and RMNI require fewer replicate surveys to classify a water body with a given level of precision and confidence (see Section 3.7);
- the majority of the variation, in three out of the four parameters, is due to variation among rivers. Perhaps this is not surprising given the wide geographic spread of the rivers in this study, with data from rivers across Scotland, England and Wales. NTAXA has the lowest % variation among rivers at just 25%, which might reflect to some degree the high operator variability. In contrast, RMNI has the highest % variation among rivers (84%), which may reflect contrasting levels of eutrophication among rivers;
- spatial variation within each river is much greater among reaches than among sites within a reach. This shows that spatial variation increases with the distance between survey locations, as might be expected. This means that replicate surveys undertaken in separate reaches will give a better assessment of ecological status than the same number of replicate surveys undertaken in the same reach;

- all four parameters show very low variation among months in the summer survey period (July-September). Although monthly variation will almost certainly be higher among surveys conducted in different seasons of the year, status is defined only by conditions prevailing during the summer. The results therefore suggest that surveying a river in a single month is sufficient to represent the macrophyte community during the summer;
- operator variability is a considerable source of variation in all four parameters. Interestingly, operator variability is consistently larger than among-site variation by at least a factor of two, and in some cases it is comparable with among-reach variation.

In SC070051/SR3 (Davey & Garrow, 2009) the CCW study of the JNCC 2008 survey results from the River Dee found no variation in the overall condition assessment for the July and September surveys, as all surveys were unfavourable. There was very little variation evident at the attribute level for these surveys. Only at a single reach was there a difference in the sub-target result (for dominant species) between the June and September surveys performed by the same operator. There were noticeable differences in the number of constancy table species recorded for each abundance level and the cover values assigned to certain taxa, but these did not alter the sub-target pass/fail result (Rhian Thomas, CCW, *pers. comm.*).

Estimating the magnitude of year to year variation has proved difficult. The report SC070051/SR2 (Davey et al., 2008) attempted to examine annual variation but could not isolate year to year variation from other sources of variation, while Report SC070051/SR3 (Davey & Garrow, 2009) was restricted to data from just one calendar year. CCW had intended to repeat surveys conducted in 2007 on the Afon Gwyrfai in 2008 but were prevented from doing so by high river flows. Report SC070051/SR2A (Davey & Garrow, 2008) suggested that there may be marked year to year variation in %Cover, but this was based on data from just one river.

More recently, SEPA re-surveyed seven of the 2008 sites again in 2009. Analysis of this extended dataset showed that inter-annual variation was larger than monthly variation for two of the analysed indicators (EQR and NTAXA), but lower for RMNI. The inter-annual variation for the overall EQR value was estimated at 8% of total variation in the dataset, or about 16% of the variation within a single waterbody (Garrow & Davey 2009).

In summary, spatial variation in macrophyte communities appears to be much higher than temporal variation and is dominated by variation among reaches, with relatively little variation among sites within a reach. Operator variability also appears to be a considerable source of variation.

3.3 How significant is operator variability and how can it be minimised?

Operator variability is variation generated by the measurement process and does not reflect actual variation in the macrophyte community. It is the difference between the true value for the metric and the value recorded on a particular sampling occasion by a particular operator. It includes both inter-operator variability, where different operators may produce systematically different results for the same survey (bias) and within-operator variability, where the same operator may produce different results when repeating the same survey (repeatability).

Within-operator variability is difficult to study. Any operator repeating the same survey within a short space of time is likely to be influenced by their experience during the first survey. Anecdotal evidence from CCW's survey work on the River Dee suggests that operators are likely to find it difficult to treat the two surveys as completely independent. They may subconsciously search for, or record without directly observing, taxa that they observed in their previous survey on that survey reach (Clarke et al., 2009). Surveyors present at the project workshop (4-5th June 2009) confirmed that this was an issue. Furthermore, the intensity of the 2008 surveying programme led to surveyor fatigue, which may have increased within-operator variability.

Inter-operator variability can be studied more easily than within-operator variability and has been the focus of the analyses of operator variability carried out in this project. The analyses used LEAFPACS data because there was not enough JNCC data to use in a detailed study of operator variability.

Report SC070051/SR3 (Davey & Garrow, 2009) considered operator variability in detail and found that it was a considerable source of variation for all four parameters, accounting for between 4 and 38% of total variation in the data examined (Table 3.2). Operator variability was most important for NTAXA, which suggests that missing, and mis-identification of, taxa can be an important source of error. It was also high for %Cover, which may reflect the fact that total cover is not routinely recorded in LEAFPACS surveys. Operator variability was least important for EQR and RMNI, which suggests that these metrics are less sensitive to inaccurate survey data or different interpretations of the area of channel and bank to be surveyed. Indeed, it is possible for inaccurate identification, or over-looking of taxa, to balance out over the surveyed area. Nonetheless, an analysis of a reduced dataset, which contained surveys at the same sites in the same months by different operators, showed that operator variability can produce differences of ±20% in EQR scores.

Significantly, operator variability was consistently larger than among-site spatial variation by at least a factor of two. In some cases it was comparable with among-reach spatial variation. This means that assessments of ecological status, at a water body level, may be prone to a high degree of uncertainty, unless operator variability is reduced or replicate surveys are used to average out operator variability.

Detailed examination of survey data from the River Dee showed that variation between operators was not systematic, i.e. no surveyor recorded consistently higher or lower values than any other surveyor. This is perhaps surprising, given that a mix of highly experienced and less-experienced surveyors were active in this programme.

There are a number of factors that could contribute to the high variation observed among operators:

- mis-recording of taxa. Mis-identification of taxa is an obvious potential problem, but so long as the confusion is restricted to closely-related taxa this should have only a limited impact on any results. False negatives can be caused by over-looking particular taxa, while false positives can be caused by less experienced surveyors being tempted to record species as being present at a site, simply because they appear on a reference list or have been previously recorded at that site;
- differential survey effort. Surveyors specialising in particular taxonomic groups may spend extra time searching for, or identifying, these taxa;
- different assessments of the channel area to be surveyed. It can be difficult to determine how much of the river bed will be inundated for the required percentage of time. There was agreement at the project workshop that this is likely to be an important cause of operator variability. Surveyors who are operating on an unfamiliar stretch of river will not be able to draw upon prior experience of the area and so are likely to be less accurate in their estimations of the channel area;

- short-term (day-to-day) variation in flow. This could affect operators' ability to detect taxa and estimate cover, but there was little evidence from the 2008 monitoring data that this was the case. It could also affect the assessment of the channel area;
- differences in the equipment used by operators (e.g. waders vs. snorkel and mask). This had a marked effect on the results obtained, with underwater methods yielding markedly more taxa;
- there was some inconclusive evidence that pairs of operators produced less variable results than lone operators, but further studies will be required to test this hypothesis.

It is assumed that measurement error, unlike natural sources of variation, can be minimised. It could potentially be reduced by: more rigorous and frequent training and accreditation of operators; more consistent use of appropriate equipment (e.g. waders, snorkels); better adherence to current monitoring protocols; the development and use of clearer monitoring protocols; and employing pairs of operators instead of lone operators. Approaches to reducing operator variability are discussed in Section 4.2.

In summary, analysis of data from LEAFPACS surveys conducted in 2008 revealed that operator variability can make a significant contribution to variation in estimates of taxonomic richness and plant cover, and to a lesser extent to variability in community metrics such as EQR and RMNI. Natural spatial and temporal variability can be overcome by conducting replicate surveys, but implementing measures to reduce operator variability may be a more cost-effective way to improve the precision and confidence of survey results.

3.4 Can operator variability produce a biased assessment of conservation status?

Operator variability was a significant component of the variation in the 2008 LEAFPACS dataset used in Report SC070051/SR3 (Davey & Garrow, 2009). The relative importance of this component varied across the four parameters studied:

- operator variability was most important for NTAXA (38%), which indicates that missing and mis-identification of taxa can be an important source of error. Different assessments of river area are likely to have a significant impact on this parameter as taxa may be excluded by one operator for falling outside the survey area, while being included by another operator. If this finding is mirrored in the JNCC surveys then it could have implications for the survey results because the species composition sub-target is based upon a certain percentage of expected taxa being present.
- operator variability was quite high for %Cover (24%), which may reflect the fact that total cover is not routinely recorded in LEAFPACS surveys.
- operator variability was least important for EQR (14%) and RMNI (4%), which indicates that these calculated metrics are less sensitive to inaccuracies in the survey data.

Although analysed using LEAFPACS survey data, the high level of operator variability found for the NTAXA parameter is also of concern for JNCC surveys. An analysis of the only two reaches in the JNCC dataset, which were appropriate for studying inter-operator variability, found large differences between operators for this parameter in JNCC surveys (Davey & Garrow, 2009).

Mis-identifying, or over-looking, taxa may alter the pass/fail outcome of a JNCC survey. CCW has been investigating this issue in detail using the 2008 JNCC surveys performed on the River Dee in Wales (Rhian Thomas, CCW, *pers. comm*.). The initial findings are summarised below.

The CCW study suggests that there is no difference between operators in the overall results (pass/fail) for the surveyed reaches as all surveys were reported as unfavourable. However, there are differences observed in the sub-target results (for species composition, loss of species, abundant species and negative species). This variation in sub-target results can be significant. For example, at one reach the abundant species sub-target varied from 0% (of species recorded as dominant in baseline survey still recorded as dominant) to 60% between different surveyors in the same month.

A study of the Plant Community: Species Composition attribute showed that all but one of the surveys on the River Dee failed to record at least 60% of the species listed in the constancy table at Frequency IV. The lack of variation in the sub-target results might largely be due to a lack of Frequency IV species at all reaches.

It should be noted that the Negative Indicators: Negative Species attribute required an assumption to be made that a recorded cover value of 3 (>5% cover) was equivalent to a cover of >25%. This is due to an apparent discrepancy between the recording methodology and the sub-target requirements.

In addition, there were differences in the additional species lists recorded by surveyors. Some of these taxa are recorded at too coarse a taxonomic resolution to determine whether they are species listed in the constancy tables. Typically, this would not alter the sub-target pass/fail result. However, there are differences between operators in the recording of additional species.

In summary, these results indicate that operator variability is a significant source of variation in JNCC surveys. This can frequently produce a biased result at the sub-target level, but less so at the overall condition assessment level. The CCW study identifies inter-operator differences that could be reduced through improved training and methodologies (see Section 3.3).

3.5 What is the optimum spatial and temporal sampling strategy?

Natural variation in macrophyte communities gives rise to uncertainty in estimates of ecological status. This uncertainty can be reduced by conducting more surveys and by careful planning of the location and timing of surveys. If the relative magnitudes of various components of variation are known (Section 3.2), monitoring programmes can be designed to ensure that the limited sampling effort is targeted at the spatial or temporal scale where variation is greatest.

Clearly, the optimal sampling strategy in any given situation will depend on the aim and scope of the monitoring programme. This section assumes that the objective is to assess the condition of macrophytes throughout the whole water body over a three year reporting period.

WRc (Davey & Garrow, 2008) conducted a simulation study, which used the River Allen dataset, to compare the uncertainty in estimates of percentage cover that were produced by four different spatial sampling strategies:

1. random: five surveys conducted at random 100-m sites throughout a water body;

2. stratified random: five surveys, one conducted at a random 100-m site within each of five sections of the water body;
3. regular: five surveys conducted at equally spaced 100-m sites along the water body; and
4. continuous: five surveys conducted at five contiguous 100-m sites within the water body.

Figure 3.2 shows the 90% confidence intervals for each strategy around the simulated mean value.

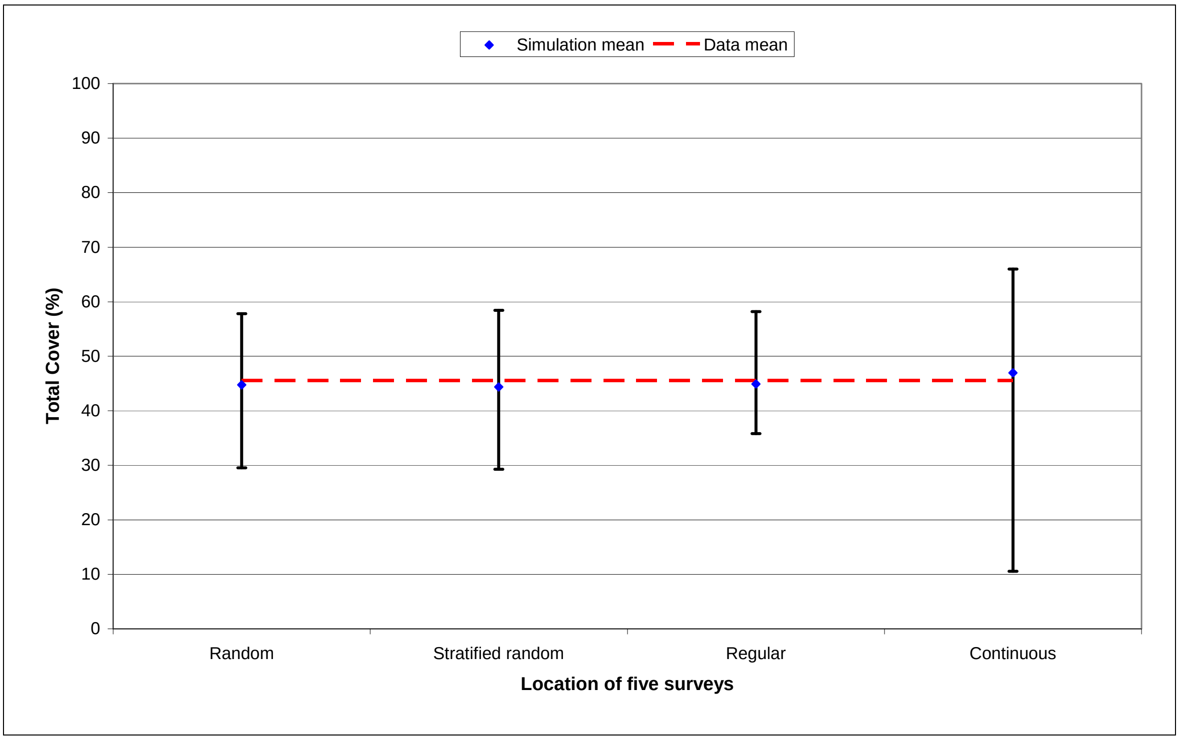

Figure 3.2 Mean and 90% confidence intervals for total cover in the River Allen, in 2000, from 100 simulations of each of four sampling strategies.

The regular spacing of surveys along the water body gave better precision than other sampling strategies. However, there was little difference between this strategy and either the random or stratified random strategies. Conducting a series of contiguous surveys gave the least precision. Regular sampling is the most efficient method of estimating ecological status within a water body, particularly when there are large-scale spatial patterns in the structure of the macrophyte community. Although the simulation was based on a limited set of data from the River Allen, the results are intuitive and illustrate the benefit of ‘spacing out’ surveys to average out natural spatial variation.

Understanding the relative magnitudes of various components of variation (Section 3.2) leads to a number of general recommendations for efficient monitoring:

1. low among-site variation means that a single 100-m survey will often be representative of the conditions within a 500-m reach;
2. significant among-reach variation means that replicate surveys in different reaches will be required to characterise the whole water body;
3. low among-month variation means that surveying in a single month each year should be sufficient to represent conditions within the river in any one

summer. However, if the goal is to assess community status throughout the whole growing season (May-October) then surveys in more than one month may be necessary;

4. significant year-to-year variation means that surveys should be undertaken in successive years to get a reliable measure of status over a three year reporting period.

The benefit of optimising a sampling strategy can be illustrated by considering a situation in which an organisation is tasked with assessing the status of a water body over a three year period, but has resources to conduct just three surveys in total. There are at least four sampling strategies that could be employed:

A. conduct a survey at the same site in each of three years;

B. conduct surveys at three sites in one reach in one year;

C. conduct surveys at one site in each of three reaches in one year; and

D. conduct surveys at three different sites, one in each of the three years.

These different strategies are illustrated in Figure 3.3, which shows the hypothetical spread of surveys along a horizontal spatial axis (showing the water body divided into three reaches) and a vertical temporal axis (showing time divided into three years). (Note: for simplicity, all four strategies assume that all surveys are conducted in the same month; in reality surveys could be conducted in different, randomly-chosen months in each year.)

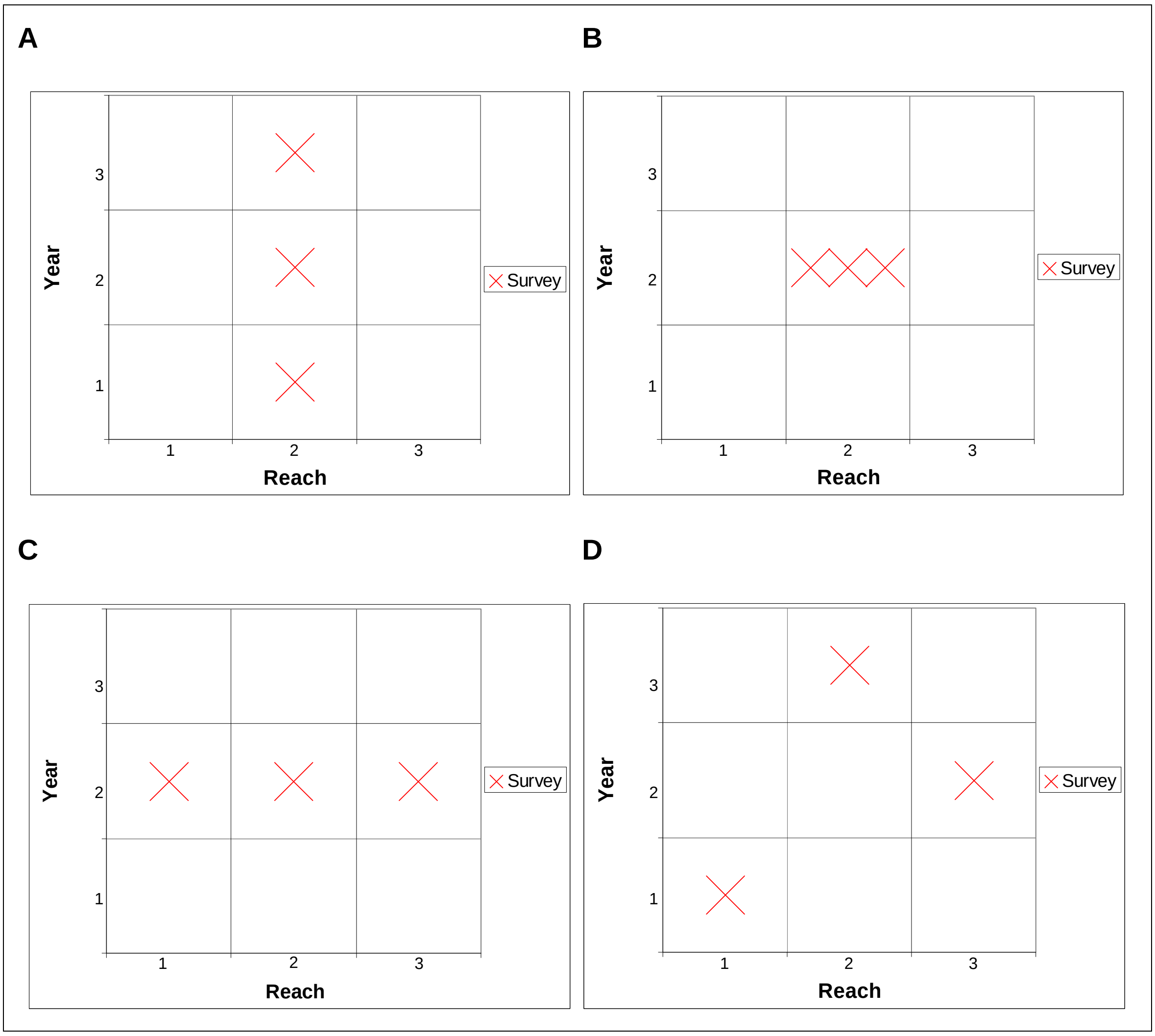

Figure 3.3 Four sampling strategies to estimate mean EQR of a water body over a three year period.

Ellis and Adriaenssens (2006) developed a simple Excel spreadsheet tool, called CAVE (Combines Appropriate Variance Estimates), which converts variance components into an estimate of the uncertainty in a reported result such as mean EQR. Ellis and Adriaenssens (2006) used CAVE to illustrate how the level of uncertainty is affected by how ecological status is defined, but CAVE can also be used to compare the uncertainty of different sampling strategies.

To compare strategies A to D above, a modified version of CAVE was created that incorporated the following components of variation (Davey et al., 2008, Davey & Garrow, 2009):

- operator variability (variance of EQR = 0.0052);
- randomly monthly variation (0.0014);
- random annual variation (0.0004);
- random variation among 100-m sites within a 500-m reach (0.0028); and
- random variation among 500-m reaches within a water body (0.0083).

The values for these components are taken from Table 3.1, with the exception of random annual variation, which comes from the analysis undertaken in Davey et al., 2008.

Figure 3.4 shows, for each strategy, the uncertainty in the average EQR (expressed as a standard error) computed from the three surveys. Strategy A has the highest uncertainty because it does not average out any of the spatial variation, which is a relatively large component of variation. Likewise, Strategy B also performs poorly because it surveys replicate sites, but not replicate reaches. Strategy C produces considerably less uncertainty because surveys are spread out so as to measure conditions in replicate reaches. Finally, Strategy D is the best because it provides the best spread of surveys through space and over time (see Figure 3.3), but it is only a slight improvement over Strategy C because annual variation is relatively low.

Figure 3.4 Standard error in average EQR for each of four sampling strategies.

The best strategy to chose will also depend on how the data is to be interpreted. If the goal is to obtain the best possible estimate of absolute status within a water body, then Strategy D (or similar) will be the best approach because it covers all spatial and temporal variation. On the other hand, if the goal is to detect changes in the status of a particular water body over time, then it may be preferable to re-survey the same sites over time to make the results more comparable (i.e. to control for spatial variation). Similarly, if the goal is to compare the status of different water bodies, then it would be sensible to survey the water bodies in the same months/years to control for temporal variation.

There are different costs associated with implementing these four strategies. For example, Strategy D would be more expensive to implement than Strategy C because it requires three separate visits to the water body over the three year period, rather than just one. Strategy C would be more expensive than Strategy B because it is more time-consuming to visit distant reaches than adjacent ones. In situations where monitoring resources are the limiting factor, a cost-benefit analysis could be used to select the most cost-effective strategy.

The foregoing discussion assumes that the water body to be characterised is essentially homogeneous – i.e. that the water body has no marked spatial discontinuities in ecological conditions. This is a helpful assumption to make because it means that all surveys, wherever they are conducted, will provide a representative measure of conditions within the water body. In reality water bodies are often heterogeneous. Conditions can change markedly downstream as a result point-source inputs, abstractions and inputs from tributaries and result in a series of discrete sections of river, each with its own unique set of ecological conditions.

Heterogeneity is a serious challenge when attempting to characterise or classify a water body because surveys will not be representative of the water body as a whole. One solution is to ignore this structured spatial variation and to conduct surveys randomly, or regularly, throughout the water body. This will still yield an unbiased result in the long run, but will reduce the precision and confidence of the results. A better solution is to undertake a stratified sampling programme, where one or more surveys are undertaken in each distinct section of the water body. The results from each survey can then be weighted in proportion to the length of river that it represents and the survey results combined to give an overall assessment at the water body level. Ideally the number of surveys should be proportional to the length of each section of the water body, but in practice resources are often limited and a maximum of one survey per section will be all that is feasible.

A stratified sampling strategy requires *a priori* knowledge about the spatial variation in conditions within the water body, but has the advantage that surveys can be located in the right places and so deliver an appreciable gain in precision for the same amount of sampling effort (i.e. it is more efficient). A second advantage is that it permits the status of macrophyte communities in different parts of the water body to be assessed separately, which can help to guide further investigative monitoring or mitigation measures. This approach has been implemented by the Environment Agency for the first River Basin Management Plans using a tool called VISCOUS (full details in Davey 2009).

In summary, the most efficient sampling strategy for measuring the ecological status of a water body is to conduct surveys at replicate sites and reaches in replicate months and years, so that all sources of spatial and temporal variation are averaged out. Nonetheless, practical and financial constraints may necessitate using a different strategy.

3.6 What factors should be considered in site selection?

Having decided upon a sampling strategy (Section 3.5) it is then necessary to select appropriate survey sites. Ideally sites should be located randomly, or regularly, but this is often not possible because practical and logistical constraints (e.g. access restrictions, deep water) limit where surveys can be undertaken. Therefore, surveyors have to make a decision about where to locate their surveys and need to be aware of the effect that their decision can have on the overall outcome of the classification.

Site selection should be influenced by the purpose of the survey programme and the level of confidence that is required in any outputs derived from the survey data. The strategies outlined in Section 3.5 show how variability can be addressed by using the appropriate spatial and temporal pattern to the location of survey sites. Other factors that should be considered include:

- representativeness. In determining the ecological status of a water body, it is preferable to select sites that are as representative of the surrounding stretch of river as possible. Selecting sites that are representative of the zone in which they are found will increase the level of confidence that can be placed in the estimates of water body status. This requires a good knowledge of the spatial characteristics of the water body being monitored;

- specific water body characteristics. A good knowledge of the water body is necessary to recognise the existence of different zones within the water body. These should be taken into account when locating survey sites. If possible, all the different zones of the water body should be covered, but greater weighting should be given to dominant zones in order to give the best estimate of the ecological status of the water body as a whole (see Section 3.5). In addition, both impacted and unimpacted sites should be surveyed if they exist within the same water body;
- inundation level. At some sites it can be difficult to estimate the size of the channel to be surveyed because of problems identifying the area inundated at normal flow levels. These sites are likely to produce higher levels of operator variability and so it may be worth avoiding such sites, particularly if the goal is to assess change in condition over time through repeated surveys of the same site;
- point source pressures. Survey sites should not be located immediately within, or downstream of, significant point-source inputs. A better picture of the response of macrophytes to an input will be gained by surveying downstream of the mixing zone;
- site specific constraints. Some sites have constraints, such as deep hollows in the river bed, which can physically restrict the ability of surveyors to accurately or safely carry out a survey. Operator variability at these sites can increase if some surveyors use equipment that allows access to these sites while others do not. If deep water habitats account for a large proportion of the water body then efforts should be made to survey these habitats as accurately as possible.

In summary, care should be taken during site selection not to introduce conscious or sub-conscious bias to the survey programme. As far as possible, sites should represent conditions in the wider water body and surveyors should be mindful of the implications of site selection for both the precision and accuracy of survey results.

3.7 How many surveys are adequate to characterise the status of macrophyte communities in a water body?

The number of surveys considered adequate to characterise the status of macrophyte communities in a water body will depend on the monitoring objectives and on what level of precision and confidence is required in the results.

As a general rule, increasing the number of surveys will improve the precision with which status is assessed, but at a diminishing rate. For example, Figure 3.5 shows how the uncertainty, in estimates of mean percentage total cover within a water body, decreases with increasing number of surveys (Report SC070051/SR2A; Davey & Garrow, 2008). For each number of surveys, 100 simulated datasets were derived from the River Allen monitoring data and the 90% confidence interval around the mean result plotted. These findings suggest that the confidence interval around the result appears to level off after five to seven surveys and that any further increase in the number of surveys will deliver only minor benefits.

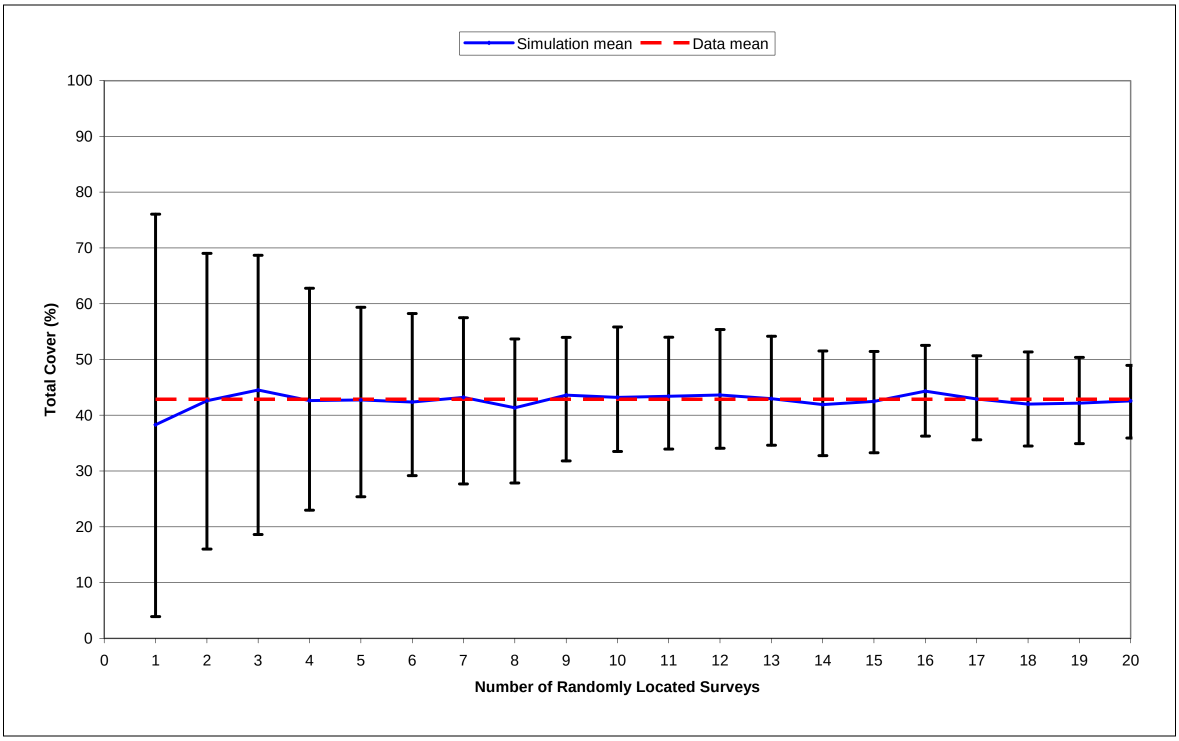

Figure 3.5 Mean and 90% confidence intervals for 100 simulations of different numbers of random surveys for total cover in the River Allen in 2008.

To reinforce this point, the CAVE tool (see Section 3.5) was used to illustrate how the level of uncertainty is affected by the number of surveys conducted. The surveying strategy with the lowest uncertainty (from Figure 3.4) was used. This is strategy D and is carried out by conducting surveys at different sites spread over different years. Figure 3.6 shows that the level of uncertainty in the results, as measured by the standard error in the mean EQR, decreases with increasing numbers of surveys. As with the River Allen simulation results (Figure 3.5), conducting more surveys will reduce the uncertainty in the estimate of water body status, but with a diminishing level of return.

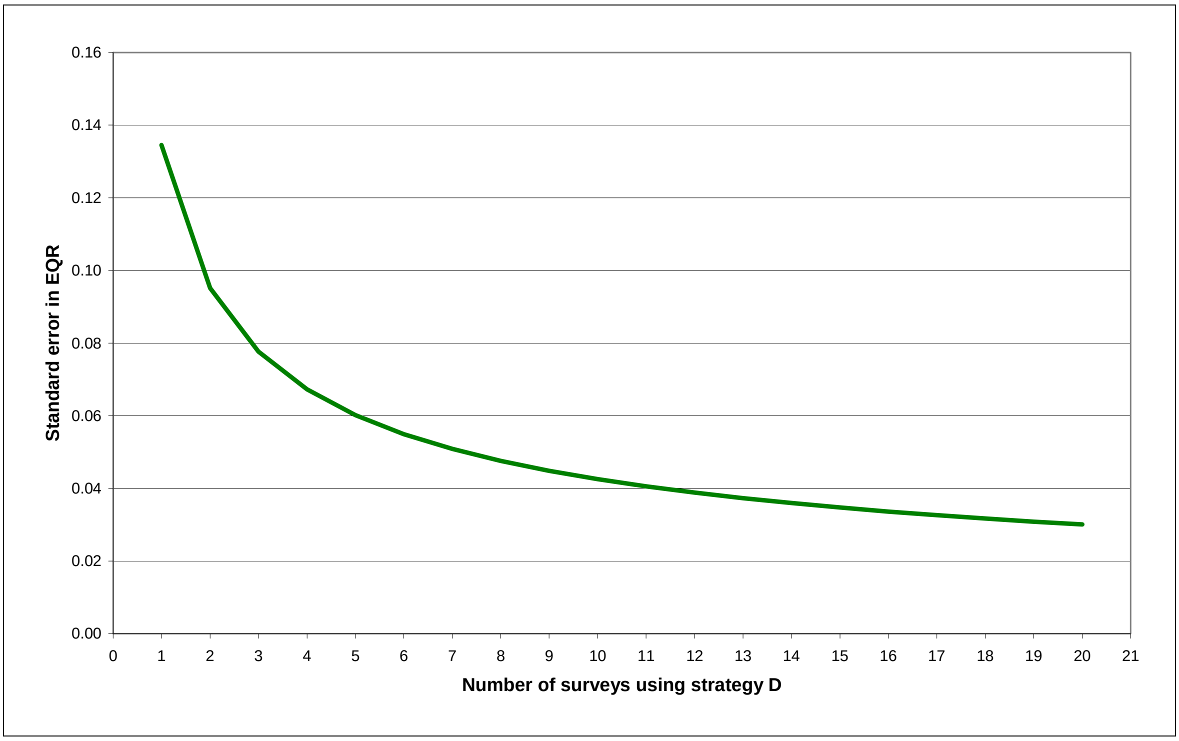

Figure 3.6 Standard error in mean EQR for different numbers of surveys using surveying strategy D.

To be really useful uncertainty should be expressed, not as a standard error, but as a risk of misclassification. Due to spatial and temporal variability in survey results there will always be some uncertainty in the estimated mean EQR, which in turn means that there is an inherent risk that a water body will be mistakenly placed in the wrong class.

To help manage this risk, WRc have developed a simple spreadsheet tool for the Environment Agency, called ROMANSE (Risk Of Misclassification And Number of Samples Evaluated), which relates the risk of misclassification to the number of surveys undertaken. ROMANSE allows users to make an informed judgement about the minimum level of survey effort required.

The calculations in ROMANSE are based on those in VISCOUS (Davey 2009), and it therefore makes the same assumptions. The main assumptions are that: (i) surveyed sites are representative of the water body, and (ii) the EQR for each site provides a full, integrated measure of the conditions at that site throughout the reporting period. ROMANSE, unlike the VISCOUS default setting, assumes that each water body is homogeneous (i.e. there are no marked discontinuities in ecological conditions).

Using the variance components presented in Table 3.1, ROMANSE estimates the risk of misclassifying a water body based on data from a given number of samples. For example, Figure 3.7 shows the risk of misclassifying a water body based on a mean EQR estimated from three replicate surveys. There are three important things to note: (i) the risk of misclassifying a water body is always 50% when the true mean EQR equals one of the class boundaries, (ii) the risk is lower for water bodies in the middle of a status class, and (iii) the risk is lowest for very High and very Bad status water bodies. A similar chart could be produced for any number of surveys; the risk would still be 50% at the class boundaries, but the troughs would get deeper as the number of surveys increased.

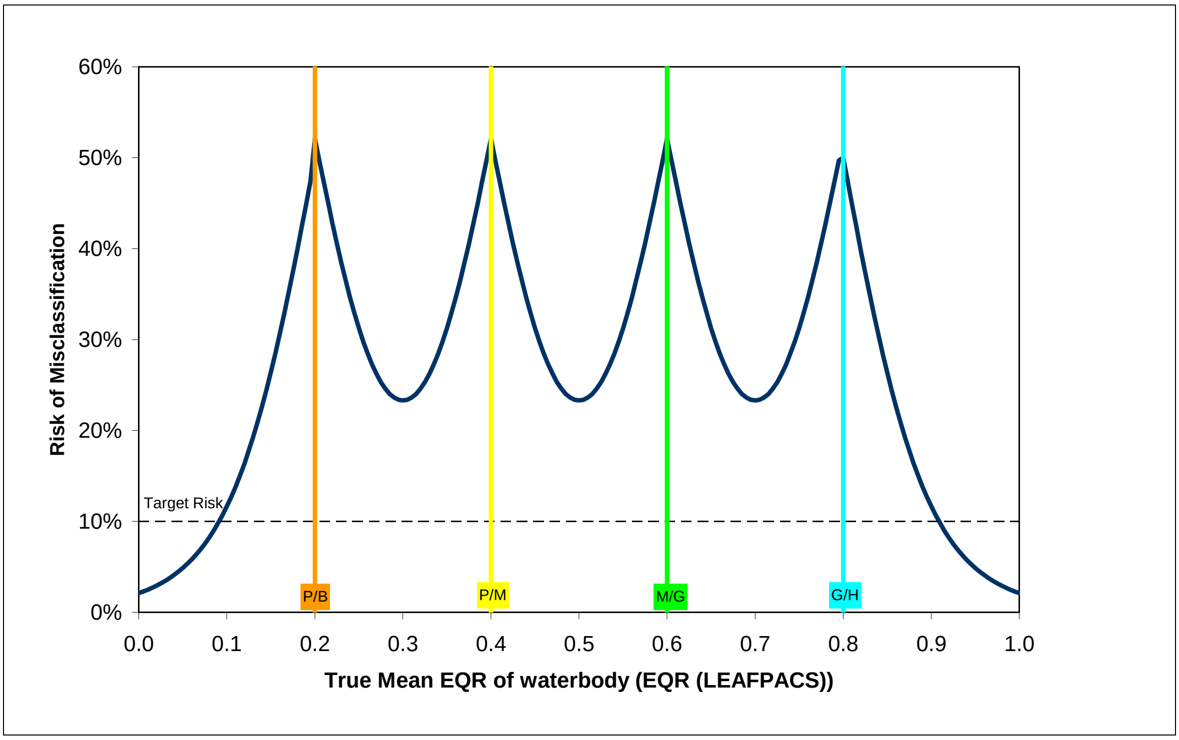

Figure 3.7 Risk of misclassifying a water body as High, Good, Moderate, Poor or Bad, based on 3 surveys per water body.

Figure 3.8 shows a similar pattern, except that here the water body is classified into one of two classes: Good or better, or Moderate or worse. As before, a water body that is exactly on the Good/Moderate boundary (i.e. of Good status) will have a 50% chance of being misclassified as Moderate and the risk of misclassification declines as the mean EQR tends towards 0 or 1.

When planning a monitoring programme, it is important to ensure that the level of sampling effort gives an acceptably low risk of misclassification. For example, in Figure 3.8, three surveys per water body would mean a 10% of risk of misclassifying a water body with a mean EQR of 0.5 or in other words a 90% chance of correctly detecting that that the water body is Moderate or worse. The person responsible for planning the monitoring programme must decide what constitutes an acceptable level of risk.

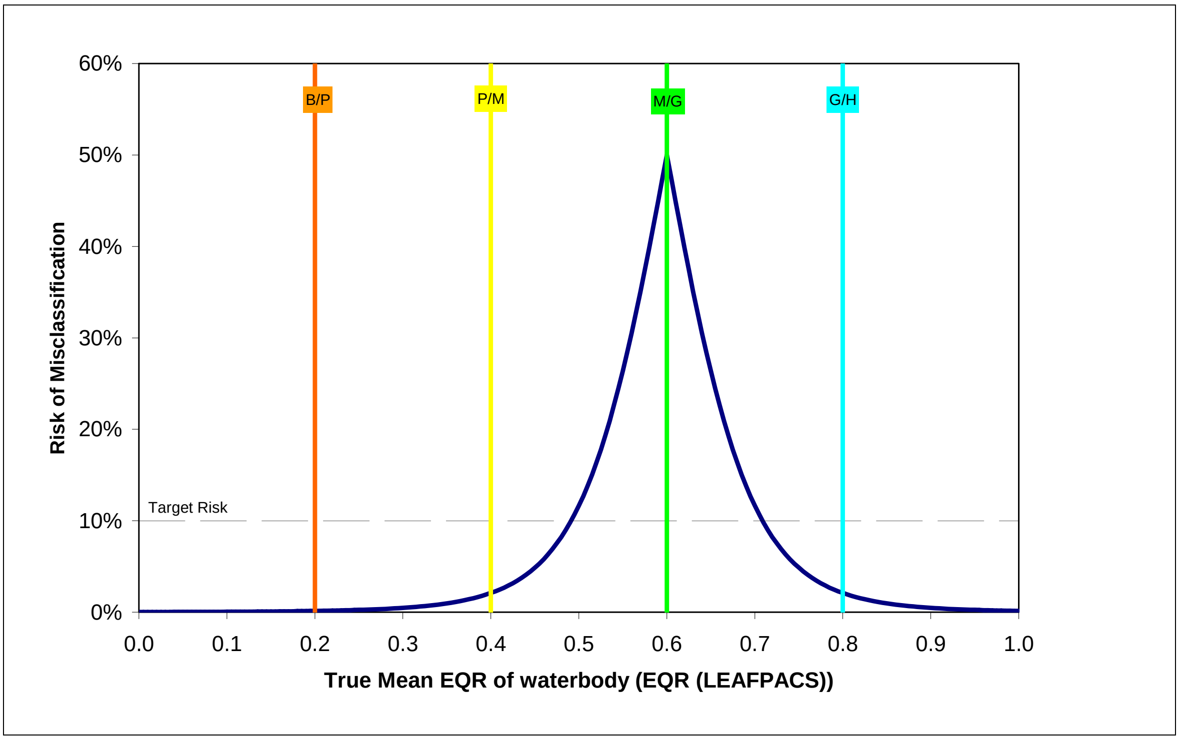

Figure 3.8 Risk of misclassifying a water body as Good or better or Moderate or worse, based on 3 surveys per water body.

ROMANSE can inform this decision by computing the number of surveys needed to achieve a given level of precision and confidence in the results. Figure 3.9 shows the predictions for LEAFPACS EQR, based upon the variance components in Table 3.1. This shows us that one survey per water body will be sufficient to classify a water body, with a mean EQR of 0.4, correctly 9 times out of 10 (i.e. with 90% confidence), but that to detect a Moderate water body, with a true mean EQR of 0.55, would require 15 surveys per water body. The lesson here is that conducting more surveys improves the ability of the monitoring programme to distinguish between water bodies that are of acceptable (Good) status and those that are not and require remedial action.

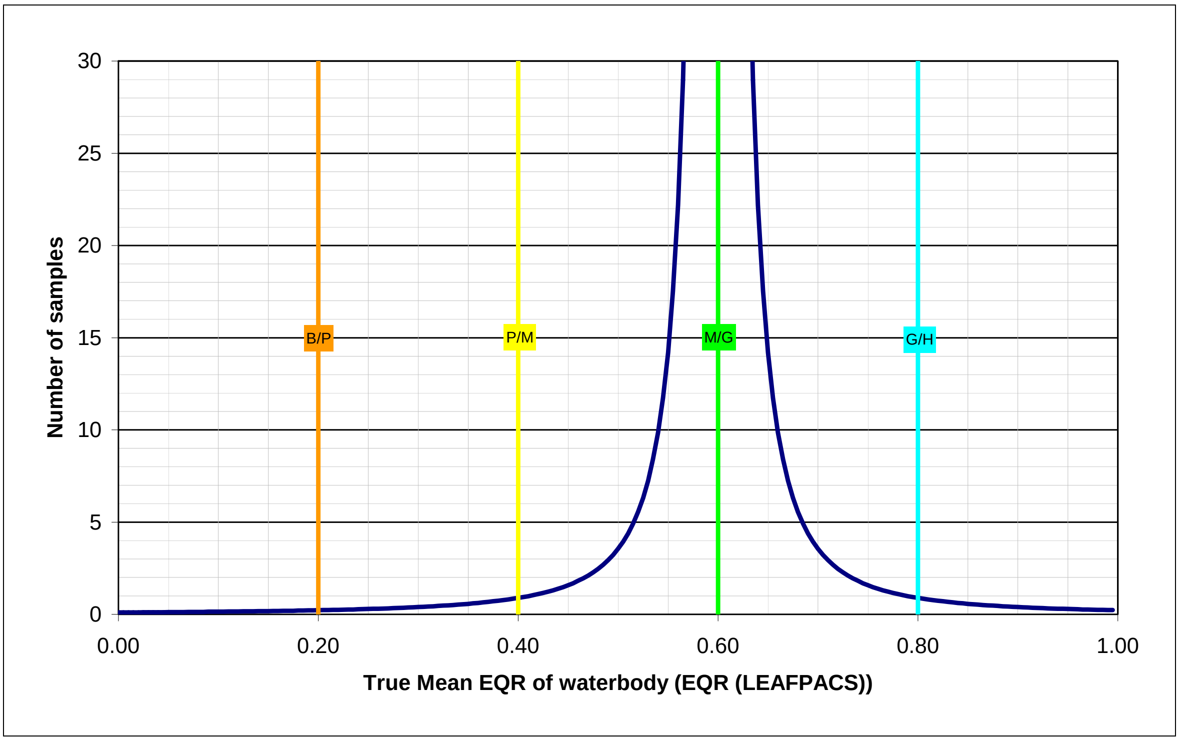

Figure 3.9 Number of samples required to be 90% confident that the status of a water body is Good or better or Moderate or worse.

In summary, there is no simple answer to the question: "How many surveys are adequate to characterise the status of macrophyte communities in a water body?" Critically, the number of surveys required will depend on the level of precision and confidence required in the results. For example, a monitoring programme whose goal is to have a 95% chance of detecting a water body of Moderate status or worse, will require far more surveys than a programme whose goal is to have just a 50% chance of detecting a water body of Moderate status or worse. However, variance components can be used to make an informed decision about the number of surveys per water body required to classify a water body on the basis of mean LEAFPACS EQR.

3.8 How long should each survey length be?

The length of a survey depends on the purpose of the survey, i.e. whether the survey is for WFD monitoring (using LEAFPACS) or for the assessment of conservation status (using the JNCC method).

For WFD monitoring, the LEAFPACS methodology surveys 100-m lengths of river. The relatively high level of spatial variation among reaches, compared with that among sites (see Section 3.2), means that greater precision would be achieved by surveying two 50-m sections in different reaches rather than a 100-m section in a single reach. However, this approach would be more time-consuming and therefore expensive and would not be comparable with historical data collected at a 100-m scale. Therefore we recommend that the LEAFPACS survey length remain at 100-m.

A study of RMNI scores derived from different lengths of LEAFPACS surveys found that a 100-m survey can be sufficient to judge the taxonomic composition in a 500-m reach, although there is some variability between 100-m surveys. Figure 3.10 shows that the median RMNI score of 100-m surveys is very similar to that of the 500-m reaches in which the surveys are found. Increasing the survey length from 100-m to

200 or 300-m does not appear to improve significantly the estimation of the RMNI metric (Davey & Garrow, 2009). This is in agreement with the relatively low between-site variation reported in Section 3.2.

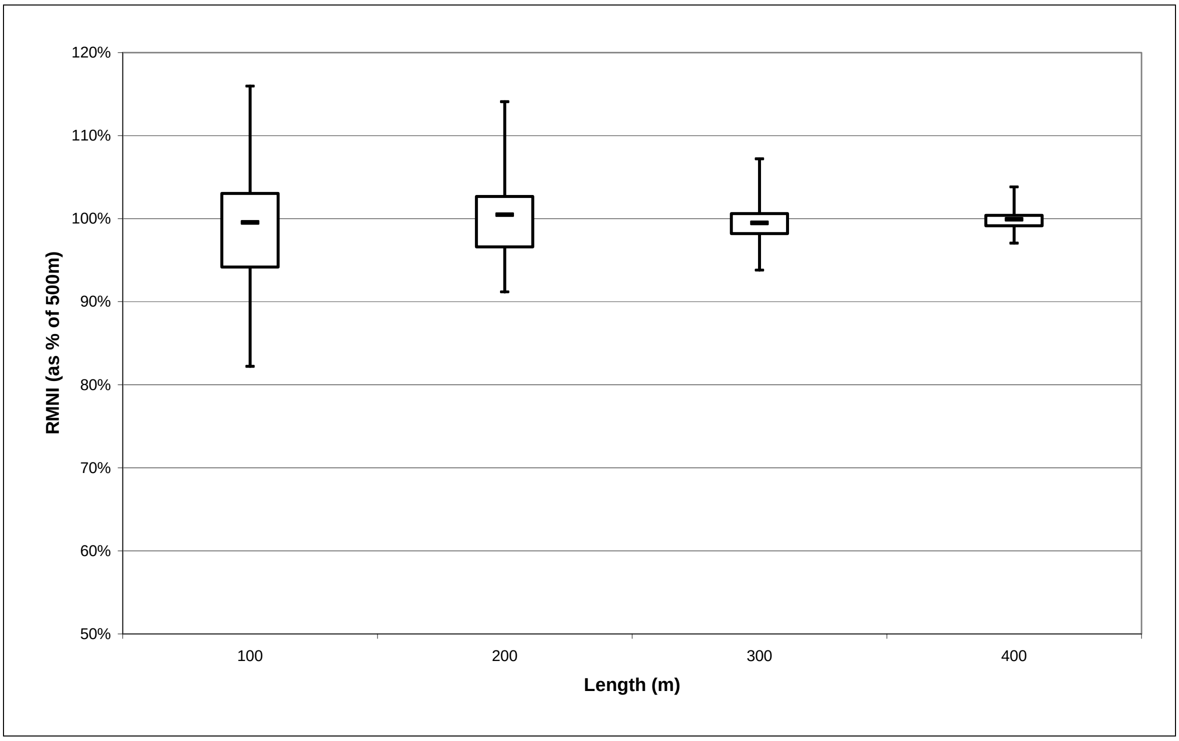

Figure 3.10 Box-plot of calculated RMNI scores for different survey lengths as percentage of 500m reach RMNI score.

The WFD requires an assessment of ecological status at the water body level, while the JNCC methodology is used by the conservation agencies to assess the conservation status of individual 500-m reaches, which are assumed to be representative of a designated SSSI or SAC area. The JNCC monitoring is more qualitative in that it focuses on determining the presence or absence of macrophyte taxa. In this situation, spatial and temporal variability in the macrophyte community are of less importance and the key question is: “how long should the survey be to reflect accurately the composition of the macrophyte community in the area of the river being studied?”

All JNCC surveys were 500-m in length, so this question was addressed using LEAFPACS surveys of five contiguous 100-m sites. Figure 3.11 shows the cumulative number of taxa found in 100-m, 200-m, 300-m, 400-m and 500-m lengths of river, expressed as a percentage of the average number of taxa found in a 100-m section (Davey & Garrow, 2009). The number of taxa recorded increases with increasing length, but at a decreasing rate. The number of taxa found in a 5 x 100-m LEAFPACS survey was, on average, 87% higher than the number of taxa found in a 1 x 100-m survey. Or, in other words, a 100-m survey yields, on average, 47% fewer taxa than a 500-m survey. These results indicate that reducing the JNCC survey length could have implications for the sub-target results in the JNCC assessment.

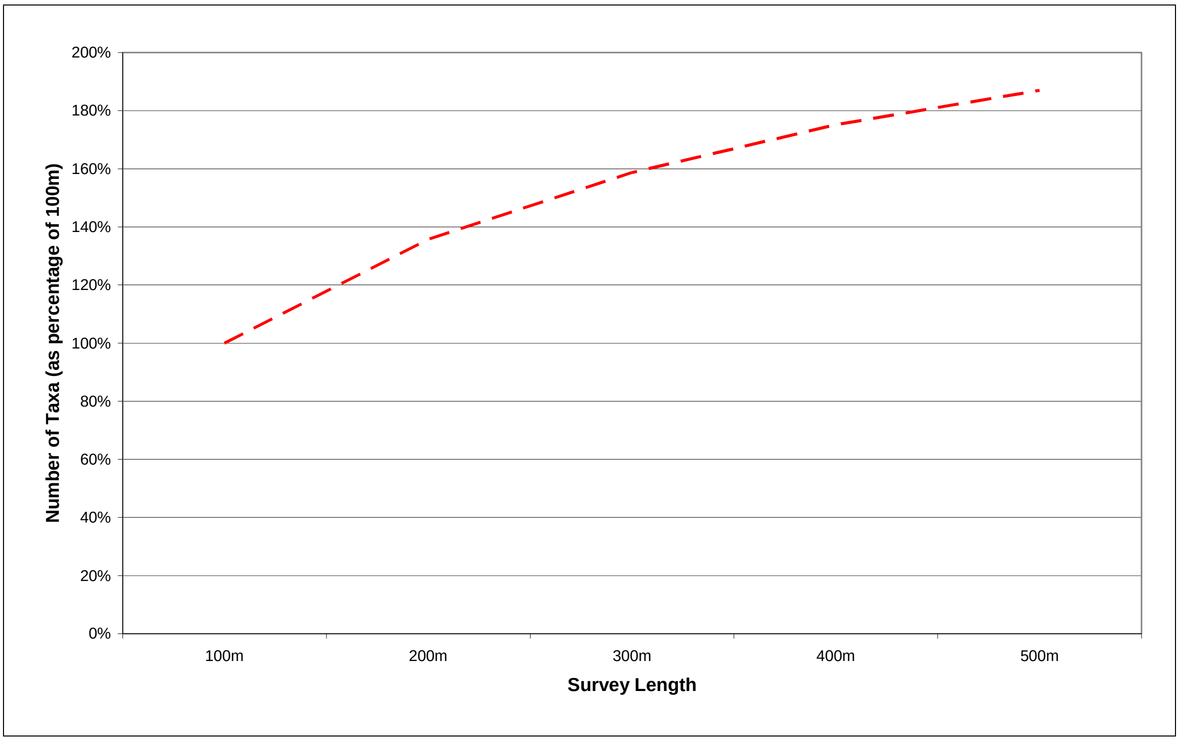

Figure 3.11 Average number of taxa found in LEAFPACS surveys of differing lengths.

Further analyses have been carried out by CCW on the River Dee survey data from 2008. These compared 18 x 500-m JNCC and 87 x 100-m LEAFPACS surveys performed at the same 500-m long reaches (Rhian Thomas, CCW, *pers. comm*.). The results are presented in Table 3.3, which shows that the 100-m long LEAFPACS surveys have a noticeably lower pass rate than the 500-m JNCC surveys. This analysis was conducted purely for comparison purposes as the condition assessment criteria are not really designed for use with LEAFPACS survey data.

Table 3.3 Number of surveys assessed as passes for the Plant Community: Species Composition attribute on River Dee.

Frequency of species	JNCC	LEAFPACS
V	10 of 18 (56%)	5 of 87 (6%)
IV	1 of 18 (6%)	0 of 87 (0%)
III	8 of 18 (44%)	4 of 87 (5%)

Unfortunately, it is not possible to directly compare the LEAFPACS surveys with the JNCC surveys, since a large proportion of the taxa listed in the JNCC constancy tables are not on the LEAFPACS species list and may not have been recorded, even if present. Many of these taxa are listed as additional species and so may not have been recorded by the surveyor. This makes it difficult to say whether the lower pass rate achieved by the LEAFPACS surveys is due to a shorter survey length or due to a more restricted taxa checklist. Furthermore, it is not possible to judge what survey length is sufficient to characterise accurately the macrophyte community in that location. A longer survey can never record fewer taxa than the shorter lengths that it is composed of, but a sufficient representation of the macrophyte community may be possible from these lengths. Further insight could be obtained by conducting five replicate 100-m surveys and recording all necessary taxa from the JNCC constancy tables using the JNCC methodology.

In summary, the JNCC method is more sensitive to survey length than the LEAFPACS method because of its dependence on specific taxa listed in the constancy tables. The taxa in the constancy tables are currently under review. The survey length used in a JNCC survey will depend on the level of error that is acceptable to the organisation performing the survey. Shorter length surveys will produce a higher rate of “false failures” as less taxa are likely to be observed.

4 Future Research Needs

The analysis of a range of datasets has significantly increased our understanding of the main components of variability in river macrophyte communities and provided valuable insight into the most efficient way of monitoring the ecological and conservation status of individual water bodies. The aim of this section is to highlight continuing gaps in understanding, to make recommendations for research to address these gaps and to discuss how these results could be translated into operational tools.

4.1 Understanding and quantifying natural variability

Understanding and quantifying variability in macrophyte communities, at appropriate spatial and temporal scales, is an essential prerequisite for developing cost-effective monitoring programmes and for measuring uncertainty in ecological and conservation status assessments. Analysis of a number of datasets has shown that spatial variation is the dominant source of variation when measuring a number of macrophyte parameters and that variation among reaches within a water body is considerably larger than variation among sites within a reach.

The magnitude of temporal variation remains poorly understood. The results of report SC070051/SR3 (Davey & Garrow, 2009) suggest that monthly variation is low relative to spatial variation. This analysis was based on a relatively narrow three month window and data collected over a wider range of months is likely to show greater variation. However, the period of study covered the standard surveying season and so seasonal variation is of little practical consequence. Report SC070051/SR3 considered data from only one calendar year, so it was not possible to determine whether the seasonal changes observed were systematic (i.e. repeatable from year to year) or random.

Annual variation was estimated using data from just seven sites that were surveyed by SEPA in both 2008 and 2009 (Garrow & Davey 2009) and a more comprehensive analysis is required to build a clearer picture of the importance of year to year variations in macrophyte communities. Understanding the level of variation in macrophyte communities from year to year is important for WFD monitoring because ecological status is assessed over a three year reporting period. If annual variation is significant it would be beneficial to spread monitoring efforts over the three years in order to measure and average out this variation. On the other hand, if annual variation is negligible surveys could be conducted in just one year, which would be cheaper and still provide a representative measure of conditions over the reporting period.

More generally, work to date has assumed that spatial and temporal variation is similar in all water bodies and has focused on deriving generic components of variation. It is difficult to evaluate how realistic this assumption is. Report SC070051/SR2 (Davey et al., 2008) attempted to derive variance estimates for different river types but the results were limited by a lack of data for many river types. It is possible that larger water bodies will show greater spatial variation because rivers characteristically show a continuum of physico-chemical conditions along their length. This is difficult to substantiate using the limited available data. Report SC070051/SR2A (Davey & Garrow, 2008) suggested that the amount of spatial variation reaches a plateau at a scale of 10-15 km, but this result was based on data from just one river that may or may not be typical of all UK rivers. It is also possible that the level of inter-annual variation may be affected by the ecology of the observed plant species; the macrophyte community will include taxa with differing life-cycles (annuals, perennials and biennials) and this should be borne in mind when analysing annual variation.

Potentially, large quantities of data are required to derive type-specific estimates of spatial and temporal variability and it is questionable whether existing WFD and conservation monitoring programmes will deliver data that is appropriate for such a task. More refined and targeted monitoring might be possible if there were a better understanding of how the sources and magnitude of variability differ between water bodies, but we recommend that generic estimates be used for now and that their performance is monitored during the next reporting period.

4.2 Survey design and practice

Report SC070051/SR3 (Davey & Garrow, 2009) identified operator variability as a major source of variation in LEAFPACS macrophyte surveys. Operator variability comprises both inter-operator variability, where different operators produce different results for the same survey, and within-operator variability, where the same operator may produce different results when repeating the same survey.

At present it is not possible to distinguish the relative importance of inter- and intra-operator variability, or to judge whether error arises mainly from over-looking or misidentifying taxa or from the misreporting of cover scores. Although a study could be devised to answer these questions, a more important question is: “how can operator variability be reduced?” Obvious options include more rigorous and frequent training and accreditation of operators, consistent use of equipment (e.g. waders, snorkels), the development and use of clearer monitoring protocols or better adherence to existing protocols if these are deemed sufficient, and employing pairs of operators instead of lone operators. These options will have different cost implications and it is important that their benefits are quantified so that a cost-effective approach can be adopted. We recommend that current surveying practices are reviewed and a research investigation undertaken to evaluate the effectiveness of a variety of measures in reducing operator variability. As an example, it would be useful to compare paired operators with lone operators, experienced operators with inexperienced operators and operators that have been recently tested with those that have not.

The JNCC conservation assessment appears to be particularly sensitive to misidentification or over-looking of taxa, which can produce a biased result at the sub-target level and, to a lesser extent, at the overall level. This is because the condition assessment is based on the observation of particular taxa. However, this conclusion is based on data from a limited number of surveys on a single river (the Dee) and a more comprehensive analysis would help to establish the severity of the problem.

Work to date has established an optimal sampling strategy for LEAFPACS surveys, but it was not possible to answer fully the question; “what length of survey is sufficient to accurately represent the status of a SSSI or SAC river using the JNCC method?” Further insight could be obtained by conducting five replicate 100-m surveys and recording all necessary taxa from the JNCC constancy tables using either the JNCC or LEAFPACS methodology.

Finally, there appear to be some limitations in the current JNCC methodology that need to be addressed. Firstly, there are inconsistencies between the taxa in the constancy tables and those in the methodology list; all taxa in the constancy tables should be recorded, even if they do not feature on the methodology list. The constancy tables are currently under review and it is expected that this issue will be resolved. Secondly, the condition assessment for the Negative Indicators:Negative Species attribute should be altered to be compatible with the cover score system used in surveys, instead of using an absolute percentage cover value for assessment.

4.3 Translating information into tools

The WFD requires the UK environmental protection agencies to classify all surface water bodies into one of five status classes (High, Good, Moderate, Poor or Bad) and to report the level of confidence associated with each water body classification – i.e. to state the probability of the water body being correctly allocated to its status class. In an ideal world of comprehensive monitoring data containing no errors, water bodies would always be assigned to their true class with 100% confidence, but in reality estimates based on monitoring are always subject to error. Understanding and managing the risk of misclassification, as a result of uncertainties in the results of monitoring, is important on two counts; firstly, because of the potential to fail to act in cases where a water body has been wrongly classified as being of better status than it is, and secondly, because of the risk of wasting resources on water bodies that have been wrongly classified as worse than they are.

The Environment Agency currently uses VISCOUS to compute the status and confidence of class of each water body. VISCOUS is a generic spreadsheet tool that converts variability in biological data (macrophytes, macro-invertebrates or diatoms) into a measure of uncertainty in the overall classification (Davey, 2009). Where water bodies are surveyed at multiple sites in multiple years, the confidence of class can be computed directly from the monitoring data, but for water bodies with less extensive monitoring coverage, confidence of class relies on external estimates of the components of variation of key macrophyte parameters (namely EQR).

VISCOUS currently incorporates an estimate of within-water body spatial variation in LEAFPACS EQR measurements derived from Report SC070051/SR3 (Davey & Garrow, 2009). However, VISCOUS could be extended to take account of both temporal and operator variability. This will make the tool more flexible and produce more reliable assessments of the confidence of class.

As well as supporting the assessment of water body status, an understanding of variability in macrophyte communities can also be used at the planning stage to estimate how many surveys are required to detect water bodies of less than Good status with the required level of precision. As described in Section 3.7, ROMANSE allows users to make an informed judgement about the level of survey effort that is required to achieve a given level of confidence in the survey results. For example, a user will be able to calculate how many surveys will be required to be 95% confident of detecting a water body of Moderate status or worse. ROMANSE is a useful planning tool, but has the potential to be made more powerful, for example by deriving water body type-specific variance estimates and taking account of the relationship between variability and status.

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List of abbreviations

CCW – Countryside Council for Wales

CSM – Common Standards Monitoring Guidance for Rivers

EA – Environment Agency

EQR – Ecological Quality Ratio

FCT – Favourable Condition Table

HD – Habitats Directive

JNCC – Joint Nature Conservation Committee

MTR – Mean Trophic Rank

NE – Natural England

NFG – Number of Functional Groups

NTAXA – Number of Aquatic Taxa

RMHI – River Macrophyte Hydraulic Index

RMNI – River Macrophyte Nutrient Index

SAC – Special Area of Conservation

SCD – Squared Chord Distance

SEPA – Scottish Environment Protection Agency

SNH – Scottish Natural Heritage

SSSI – Site of Special Scientific Interest

STR – Species Trophic Rank

UWWTD – Urban Waste Water Treatment Directive

WFD – Water Framework Directive

%Cover – Percentage Cover of Macrophytes

Statistical Glossary

Bias – Bias describes a systematic over- or under-estimation of a parameter. For example, if the number of species recorded in a survey reach is consistently fewer than the true total number, then the results would show a negative bias.

Confidence – A statistical term describing how certain we are about a result.

Confidence interval – A confidence interval quantifies uncertainty in the estimate of a parameter by giving a range of values that is likely to include the true (unknown) parameter. For example, a 90% confidence interval around a mean indicates that one can be 90% confident that the true mean lies within that range.

Parameter – A parameter is a number describing some aspect of (in this case) a macrophyte community.

Precision – A measure of the uncertainty in an estimated parameter, often expressed as a percentage. Precision varies with the level of confidence required. For example, an estimated EQR of 0.8 with a precision of ±10% at 90% confidence means that we can be 90% confident that the true (unknown) EQR lies between 0.72 and 0.88.

Variance - Variance is a measure of the spread of observations about a mean value. In this project it is used to summarise how much variability there is in a series of survey results.

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