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Ecological Quality Ratios for Ecological Quality Assessment in Inland and Marine Waters

REBECCA Deliverable 10

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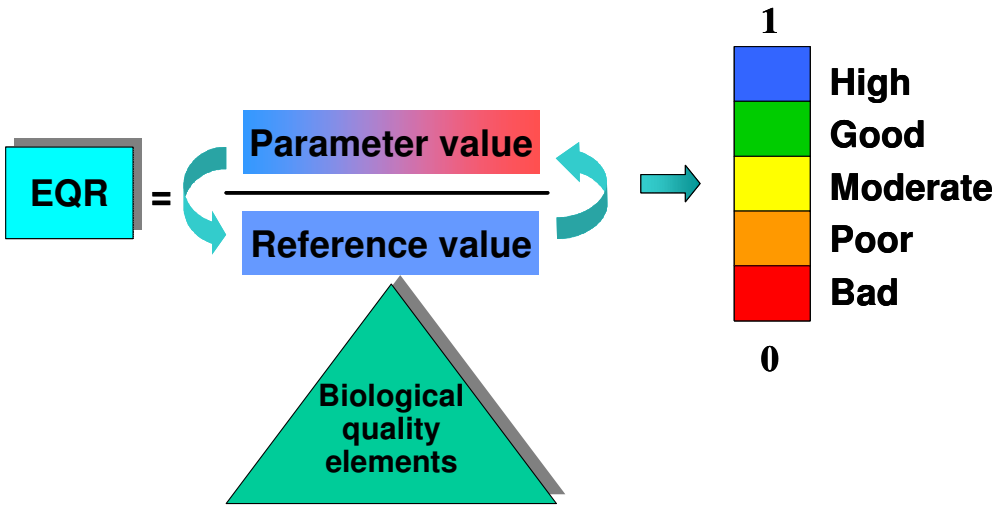

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1 Introduction

1.1 Background and outline

The REBECCA project focuses on establishing quantitative relationships between physicochemical and hydromorphological pressures and biological indicators. One of the major applications of such relationships is to support the establishment of WFD compliant assessment and classification methods. All EU Member States are required to establish such methods for the different biological quality elements (phytoplankton, benthic macroinvertebrates, fish, macrophytes, etc.).

This short Deliverable addresses the issue: What is a WFD compliant assessment method? This is done by focusing on the concept of the Ecological Quality Ratio (EQR). The EQR incorporates the key WFD requirements for ecological classification: typology, reference conditions, and class boundary setting. The Deliverable is targeted both to the policy makers and competent authorities implementing the Water Framework Directive and the scientists supporting them with their specific knowledge. The following issues are addressed:

- Ecological Quality Ratios in the Water Framework Directive (Chapter 1);
- Typology, reference conditions, and class boundary setting (Chapter 2);
- Uncertainty in EQR based assessments (Chapter 3);
- Discussion on the practical application of EQRs (Chapter 5)
- a checklist summing up criteria for WFD compliant classification systems (Chapter 5)

Most of this Deliverable was written by Wouter van de Bund (JRC-IES); Chapter 3 addressing uncertainty in EQR based assessments was written by Angelo Solimini JRC-IES). The authors thank Mike Dunbar (CEH) for providing valuable comments to Chapter 3. The views expressed are purely those of the authors and may not in any circumstances be regarded as stating an official position of the European Commission.

1.2 Ecological Quality Ratios in the Water Framework Directive

1.2.1 WFD requirements

The EU Water Framework Directive (EC 2000) requires the establishment of methods to quantify the ecological status of water bodies. Biological indicators play a key role in the assessment of ecological status. Biological assessment results need to be expressed using a numerical scale between zero and one, the 'Ecological Quality Ratio' (EQR). The EQR value one represents (type-specific) reference conditions and values close to zero bad ecological status (Figure 1.1). The use of EQRs is prescribed in Annex V, 1.4.1 of the WFD:

1.4.1. Comparability of biological monitoring results

(i) Member States shall establish monitoring systems for the purpose of estimating the values of the biological quality elements […]. Such systems may utilise particular species or groups of species which are representative of the quality element as a whole.

(ii) In order to ensure comparability of such monitoring systems, the results of the systems operated by each Member State shall be expressed as ecological quality ratios for the purposes of classification of ecological status. These ratios shall represent the relationship between the values of the biological parameters observed for a given body of surface water and the values for these parameters in the reference conditions applicable to that body. The ratio shall be expressed as a numerical value between zero and one, with high ecological status represented by values close to one and bad ecological status by values close to zero.

(iii) Each Member State shall divide the ecological quality ratio scale for their monitoring system for each surface water category into five classes ranging from high to bad ecological status, […] by assigning a numerical value to each of the boundaries between the classes. The value for the boundary between the classes of high and good status, and the value for the boundary between good and moderate status shall be established through the intercalibration exercise […].

In the CIS guidance on monitoring (Anonymous 2003d) the EQR is defined as follows - largely repeating the wording of the Annex V:

Ecological Quality Ratio (EQR) - The ration between the value of the observed biological parameter for a given surface water body and the expected value under reference conditions. The ration shall be expressed as a numerical value between 0 and 1, with high ecological status represented by values close to one and bad ecological status by values close to zero

The WFD explicitly states that the purpose of expressing results as an EQR is to ensure comparability between different assessment methods – in other words, to provide a common scale of ecological quality. Member States have the possibility to develop methods they see fit, tailored to their specific needs and taking into account differences in existing methods. Because of the common EQR scale it is possible to harmonise the outcome of the different methods through the intercalibration exercise.

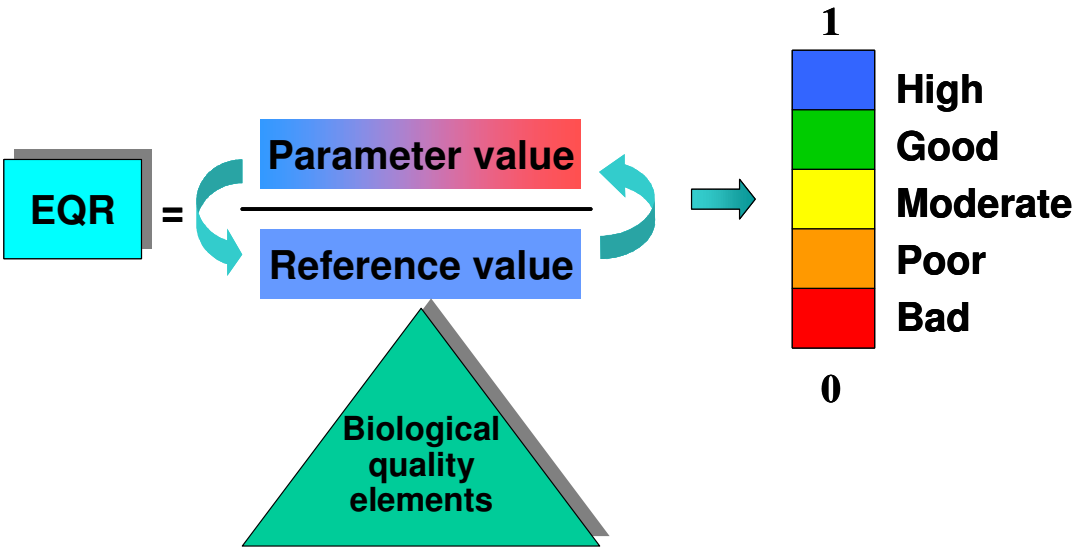

Figure 1.1 – Graphical representation of the concept of the Ecological Quality Ratio

Although simple enough in theory, the EQR concept is rather difficult to put into practice in the practical implementation of the WFD. It requires that several key issues are addressed, including the choice of appropriate indicators, typology, reference conditions, and agreement on common principles for setting quality class boundaries. Many of these issues have been addressed in several working groups of the WFD Common Implementation Strategy, as reported in numerous guidance documents (Anonymous 2003a, 2003b, 2003c, 2003d, 2003e, 2005a, 2005b).

For ensuring comparability the choice of appropriate indicators is extremely important – it is not possible to compare indicators that measure different things. The WFD concept of ecological status requires a assessment independent of pressure – calling for indicators of general ecosystem health to inform the surveillance monitoring programs (Anonymous 2003d). In practice this can be achieved by using multimetric indices combining the results of several pressure-specific indices (Barbour et al. 2000) Nevertheless, the EQR should be general enough to ensure comparability across different pressures. However, water managers have to know which problems occur in their water bodies to be able to solve them, and for this they are best served with indicators sensitive for specific pressures. Such more specific indicators are more appropriate for the operational monitoring programme (Anonymous 2003d). This issue, together with the issue of the right level of application of the EQR (e.g. parameter, quality element, combination of different quality elements) is further discussed in Chapter 1.2.2.

Type-specific reference conditions are the anchor point of EQR based classifications. Class boundaries are defined as a certain level of deviation from the reference conditions – and changing the anchor point directly affects those class boundaries. In other words, if there is no agreement on the principles and criteria for setting reference conditions, the value "1" does not represent the same of ecological quality, and the EQR scale is not comparable across countries and can not fulfil its main purpose of ensuring comparability across countries. The issue of typology and reference conditions is further discussed in Chapter 2.

1.2.2 Parameters, quality elements, and final classification: what is the right level for the EQR?

The WFD requires EQR values for the "values of the biological quality elements". It is therefore clear that the EQR does not apply to ecological status as a whole, but at a lower level. As explained before, the main aim of the EQR is to enable comparison between Member State's assessment methods through intercalibration. Since the WFD requires that the intercalibration takes place at the level of the biological quality element (Anonymous 2003b, 2005a), the logical level for the EQR is also the quality element.

It is a matter of interpretation what is meant by "values of the biological quality elements" – this can either apply to a quality element as a whole (e.g. benthic macroinvertebrates), or to single parameters within the quality element (e.g. the number of EPT taxa). This issue is addressed in the CIS guidance document on ecological classification (Anonymous 2005b); the conclusions are summarised in Figure 1.2.

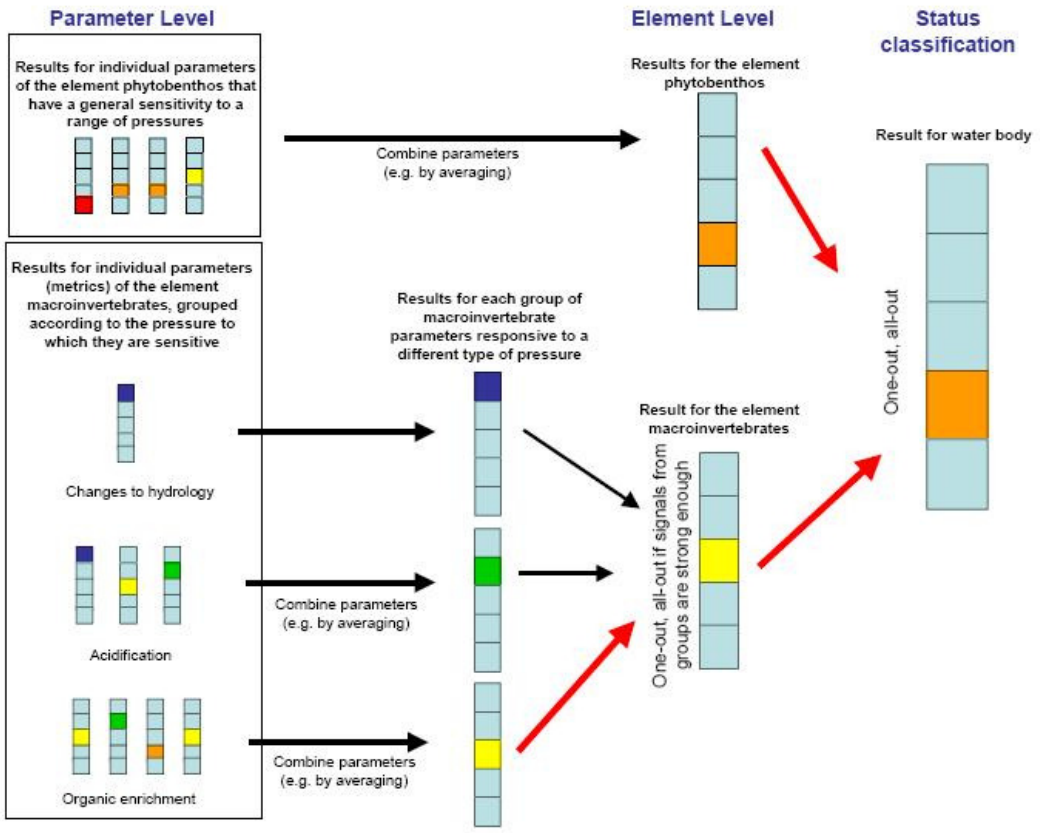

Figure 1.2 - Examples of how indicative parameters may be combined to estimate the condition of the biological quality elements. The one-out all-out principle has to be used on the quality element level as indicated with the phytobenthos example – from the CIS guidance document on ecological classification (Anonymous 2005b).

The Classification Guidance separates three levels in the biological assessment: the parameter level, the quality element level, and the status classification. The main conclusion is that the WFD requires classification of water bodies at the quality element level, and that the worst of the relevant quality elements determines the final classification (the "one out, all out" principle). How the different parameters within a quality element are combined is not prescribed; this can either be done by combining them in a multimetric index, or in any other way.

Up to date there are few studies that have brought the "one out, all out" principle in practice, and it is still unclear how this will work in practice. The more elements that feed into a final classification, the greater the probability, by chance alone, that the water body is misclassified (Irvine 2004, Anonymous 2005b). Using data from shallow lakes, the ECOFRAME project (Moss et al. 2003) has shown that combining may parameters using the "one out, all out" principle may very well result in a classification of virtually all water bodies as "moderate" or worse, even where expert judgment indicates that the quality is not so bad. In practice it may be a good idea to reconsider that "one out, all out" principle; a possible solution would be to allow multimetric indices across quality elements. It is also of high importance to take into account the reliability of the results; monitoring the entire spectrum of biological elements without regard of reliability is a waste of time and money (Irvine 2004).

2 Typology, reference conditions, and class boundary setting

Because the EQR uses type-specific reference conditions as 'anchor' for the classification, both typology and reference conditions need to be agreed upon. Typology and reference conditions are addressed in two CIS guidance documents focusing on freshwaters and coastal and transitional waters, respectively (Anonymous 2003a, 2003c). A more detailed discussion is also found in (Heiskanen et al. 2004).

The main purpose of typology is to enable type specific reference conditions to be defined which in turn are used as the anchor of the classification system (Anonymous 2003c). Water body types should be characterised based on geographical, geological, morphological and physical factors. The typology should group sites where the biology is similar in the natural baseline conditions, to enable the detection of the effects of human disturbance. This is only meaningful when the variability of the biological parameters is smaller within types than between types, depending not only on the typology, but also on the biological parameters chosen. The typology should therefore identify physically and morphologically distinct water body groups enabling comparison of 'like with like'(Anonymous 2003a, 2003c). This means, for instance, that naturally eutrophic lakes have different reference conditions than oligotrophic lakes, resulting in different scales and requirements for good ecological quality for these different lake types. The WFD allows two different approaches for typology – 'System A' and 'System B'. The difference is that System A prescribes how water bodies shall be characterised spatially (ecoregions) and with respect to specific altitude, size and depth intervals, and that System B, besides lacking this prescription, permits the use of additional factors (Anonymous 2003c).

'A priori' and 'a posteriori' typologies

Two main approaches can be taken in the determination of the surface water body types: 1) types are defined from knowledge of how physical drivers determine biological communities ('a priori' approach), and 2) types are distinguished by analysing survey data from reference sites ('a posteriori' approach) (Table 2.1). System A of the WFD is an example of an 'a priori' example; system B typologies can be defined using both approaches.

Table 2.1 Features of '*a priori*' and '*a posteriori*' typology systems

***'a priori'* typologies**	***'a posteriori'* typologies**
Should be based on knowledge of how biology is determined by geography/physical conditions	Based on physical and biological monitoring data from reference sites
Few data needed to define typology	Typology depends on available data (what quality elements, what parameters, from which region), and on the quality of the data
Types not necessarily biologically meaningful because of incomplete knowledge - need for validation using targeted field sampling	Types biologically meaningful
Reference conditions can be determined by different approaches (expert judgement, spatial, historical/ paleoreconstruction, modelling)	Reference conditions implicit

It is possible that 'a priori' typologies will not be biologically meaningful due to an incomplete understanding what drives the biology. An advantage of a verified 'a priori' typology is that it is likely to be relatively robust, because it is based on knowledge of the biology rather than purely on statistical correlation. The 'a posteriori' approach requires a sufficiently large number of sites in natural baseline conditions (reference sites) and good quality biological data. An advantage of the 'a posteriori' approach is that it has a high degree of objectivity. On the down side, 'a posteriori' typologies depend on the data available and therefore those are usually specific for a specific quality element.

Only very few countries have established advanced 'a posteriori' systems for classification and typology. One of the main reasons preventing the development of such systems is that it requires the availability of high quality data from many water bodies, sampled in a standardised way. The UK RIVPACS approach (Wright 2000), developed to predict reference macroinvertebrate communities in rivers, is a very good example. The potential of this approach is demonstrated in Swedish studies, where RIVPACS-type models (SWEPACS) have been successfully developed for both lake (littoral) and stream (riffle) macroinvertebrate communities (Johnson 1995). The European research projects STAR and FAME are extending such an approach over a larger geographical area, including wider range of river types, and more biological quality elements collected using harmonised methods. Furthermore, another European research project, CHARM is developing harmonised typology for the coastal Baltic Sea first starting from 'a priori' typology that will validated using existing biological monitoring data from the countries around the Baltic Sea.

Reference conditions can either be spatially based, i.e. defined by collecting biological information from water bodies, which are (almost) in natural base-line conditions (sites with minor anthropogenic impacts), or derived by modelling, or by combination of those. If reference conditions are to be defined using modelling, either predictive models or hind-casting using historical, palaeolimnological, and other available data can be applied (Anonymous 2003c). In many countries there may be no reference sites available or data are insufficient to carry out statistical analysis or validate models. In that case, expert opinion may be the only possibility to define reference conditions. Also the establishment of common networks of reference sites could help in setting type specific reference conditions in a comparable way between different countries.

Stepwise approach for establishing reference conditions

A stepwise procedure for establishing reference conditions, based on availability of data, is suggested (Fig. 3). The most unimpacted sites for different types can be selected using both available monitoring data and/ or pressure criteria (Anonymous 2003c). This approach would also allow establishment of a reference site network, where data for biological quality indicators in reference conditions can be obtained. In combination to that also predictive models can be validated and used to establish reference values for the parameters that represent the different biological quality elements, and apply these models to sites where biological data may be scarce or not available for all quality elements. In some cases collaboration across national borders is required since natural baseline sites for a given types may be found in other countries. If there are no sites with minor anthropogenic impacts, historical monitoring data or palaeoecological methods should be used to reconstruct reference conditions before the

onset of significant human impact. Expert judgement may be needed to evaluate when the human impact started to increase, and which period would represent conditions with a minor impact. Finally, if neither a site nor any data is available for a given type, expert judgement remains the only alternative.

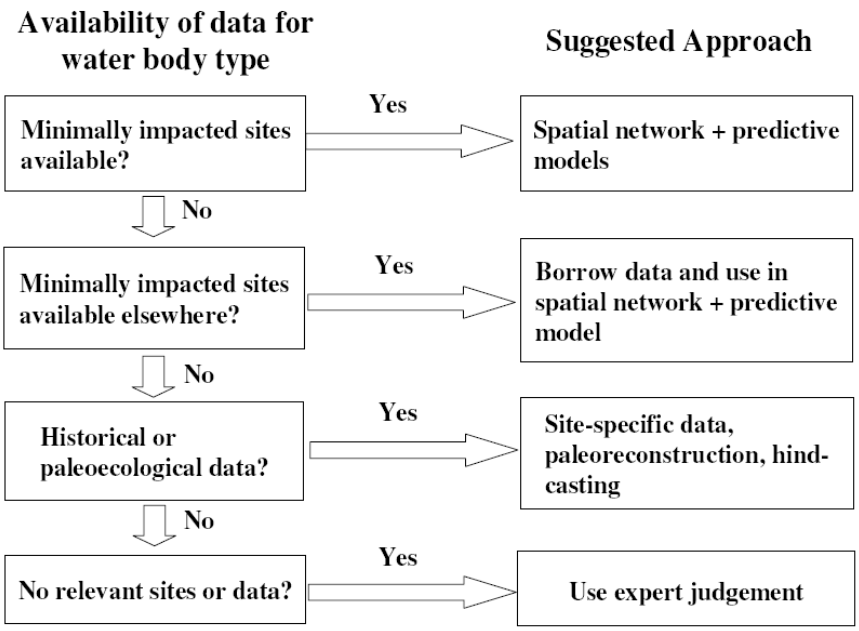

Figure 2.1 step-by-step approach for selection of the method for determination of reference conditions for surface water bodies depending on available information and data (from REFCOND guidance, Anonymous 2003c)

Reference conditions and classification

In the WFD, high ecological status is defined as 'slight' or 'minor' deviation from the reference conditions of a surface water body type, while the good status is defined as 'small' deviation. The CIS guidance documents suggest that due to the variability of type specific reference conditions, it will be more practical to consider that high status is equal to reference conditions (Anonymous 2003a, 2003c). In order to be able to set the quality classes and their borders, more detailed criteria are needed. There should be also an agreement of how the quality borders are set statistically (Anonymous 2003c). There is a need for a sound scientific basis for setting the class boundaries. In 'good' status the biological quality elements should indicate only 'slight' deviation from reference conditions, and the hydromorphological, physico-chemical, and chemical quality elements should ensure ecosystem functioning (Anonymous 2003c). However, it is not clear how the ecosystem functioning in good status should be defined. The functional diversity of the ecosystem's trophic structure may display high variability of response (Chase and Leibold 2002) when subjected to human impacts such as nutrient loading (Worm *et al.* 2002). More specific definitions and functional relationships between biological and chemical status needs to be established in order to develop operational tools for setting the quality targets in the practical management of water bodies. This requires expertise and availability of comparable biological monitoring data where functional relationships can be established across pressure gradients.

Reference conditions and class boundary setting in practice: the WFD intercalibration exercise

In the WFD intercalibration exercise the EU Member States are developing common views on the principles for setting reference conditions and setting class boundaries. A common "class boundary setting procedure" (Anonymous 2005a) has been agreed upon, a common framework that should ensure comparable approaches for rivers, lakes, and coastal and transitional waters, in different parts of Europe. At the time this Deliverable was written, the results of the intercalibration exercise were not published yet – a first technical report is expected in the autumn of 2006. However, intermediate reports (the "Milestone reports") are publicly available on the internet through the address http://forum.europa.eu.int/Public/irc/jrc/jrc_eewai/library . From these reports it can be concluded that for most of the "geographical intercalibration groups" (GIGs) have come a long way agreeing on common criteria for identifying reference sites based on the CIS "REFCOND" guidance (Anonymous 2003c) and that agreed high-good and good-moderate class boundaries can be expected for the quality elements benthic macroinvertebrates in rivers, phytoplankton (mainly chlorophyll) for lakes and coastal/transitional waters. Progress for other quality elements is mostly limited to specific GIGs. Especially in the intercalibration carried out in the lake GIGs the REBECCA project has played a very important role in the intercalibration process, by providing data and assisting with analysing relationships between pressures and biological parameters. An overview of the REBECCA contributions to the intercalibration process will be compiled after the first technical reports will be completed in the autumn of 2006.

3 UNCERTAINTY IN EQR BASED ASSESSMENT

3.1 Introduction

Predicting the causes and consequences of human activities on freshwater ecosystems requires the assessment of the level of uncertainty associated with our understanding of ecological processes. The EQR based assessment, as part of the WFD, ultimately results in the assignment of a given site to a certain ecological status class. However, to what extent are we confident that the resulting classification into a given ecological quality class is the true one for that site? The problem is to estimate the level of confidence of the class assignment for a given site or, in other words, the risk of misclassification. The uncertainty associated with ecological assessment reflects not only the inherent variability of ecosystem structure and processes but also technical, economical and political constrains of the assessment methods. Since the classification resulting from the assessment is relevant for decision makers, it is crucial to provide clue of the associated risk of misclassification.

For surface waters, the WFD refers to uncertainty as "level of confidence" and/or level of "precision" when dealing with reference conditions (Annex II) and when dealing with monitoring (Annex V). In the first case, the methods and the number of sites used to set spatial derived reference conditions must provide a sufficient level of confidence about the values for the reference conditions to ensure that the conditions so derived are consistent and valid for each surface water body type. In the case of monitoring, estimates of the level of confidence and precision of the results provided by the monitoring programmes must be given in the river management plan. The frequency of monitoring should be chosen so that an acceptable level of confidence and precision are achieved.

This chapter is not a comprehensive review of the aspects of WFD implementation that are related to uncertainty assessment (for this scope, see the outputs of research projects like HarmoniRiB, http://www.harmonirib.com or NeWater, http://www.newater.info). Here, we summarise the existing information on the sources of uncertainty in ecological data, with reference to the WFD-EQR based assessment. We also briefly describe some software packages, recently developed in research project supporting the implementation of the WFD, which can be used to assess uncertainty in ecological data. We conclude making some recommendations and possible future directions.

3.2 Sources of uncertainty in EQR based assessment

Uncertainty in ecological data

As stated earlier in this report, EQR measures the degree of departure of the value of a given ecological descriptor from an expected value. Often the calculation of metrics involve several intermediate steps, all with associated errors and all potentially adding to the resulting overall uncertainty. Therefore, the assessment of uncertainty requires specific information on the different sources of errors and is a laborious task. The nature of ecological descriptors (quantitative, semiquantitative, qualitative or binary) complicates the analytical solution. In general, uncertainty in data can be described in uncertainty categories (see Brown *et al.* 2005; table 3.1), for which different uncertainty models should be applied (Refsgaard *et al.* 2005). Ecological variables

belong at least to 8 of those types (table 3.2) and for some of them only a qualitative indication of uncertainty can be provided (Dunbar 2005).

Table 3.1. Attribute uncertainty categories (from van Loon *et al.* 2005) for some ecological variables taken as example (from Dunbar 2005). TC: taxonomic composition; AT: taxa abundances; DSDIT: ratio of sensitive/tolerant macroinvertebrate taxa; AS: age structure of fish population; DSS: number of disturbance sensitive fish species.

Space – time variability	Measurement scale		
	Continuous numerical	Discrete numerical	Categorical
Constant in space and time			
Varies in time, not space	DSDIT	DSS, AT	
Varies in space, not time	DSDIT	DSS, AT	TC
Varies in time and space	DSDIT	DSS, AT	TC, AS

Abiotic factors in freshwater systems are highly variable in time and space and often account for a significant proportion of the variation of community patterns in terms of species diversity, abundance, biomass, and production. This variability is not the only source of uncertainty. Dunbar (2005) summarised the various source of uncertainty in ecological data (table 3.2), with special reference to community type data. From this table it is apparent that the natural heterogeneity of communities in space and time are only two sources of error to account for. A variety of errors and biases can be introduced into the final results when considering also the uncertainty related to field and lab methodologies (sampling design and collection, sorting, taxonomy) and the metric used. It should emphasized that the size of the various sources of error change depending on the water bodies considered. For example, lowland, deep and turbid rivers require assessment techniques and have an associated variability that are different to wadeable piedmont rivers or streams.

Table 3.2 Sources of uncertainty in ecological data (from Dunbar, 2005 after Cao et al., 2003)

Source	Type	Comments
Unspecified sampling variability	Spatial variability	In practice, probably a combination of between and within site variability.
Choice of sampling sites within a region or catchment	Spatial variability	(where)
The ecological group or community being sampled	Assessment scheme variability	Commonly one group, e.g. fish, macro-invertebrates, macrophytes, phytoplankton, periphyton. Could be multiple groups..
Sampling effort used at the local scale	Spatial variability	Including habitats sampled, sampling equipment used (how)
Timing of sample	Time variability	Commonly time of year, but could be time of day (when)
Frequency of sampling	Time variability	Number of times over a day/season/year that a sample is taken
Personnel collected the samples	Methodological variability	(who)
Procedures for processing samples	Methodological variability	whether in-field or laboratory, including taxonomic resolution of identification, sub-sampling or compositing, whether abundance or presence-absence measured. QA/QC procedures, availability of specialist advice and up to date keys.
Similarity measure or metric used for summary	Assessment scheme variability	In order to assess the above sources of uncertainty, calculations on the raw data need to be made. The choice of method used for this assessment can affect the conclusions regarding the magnitude of uncertainty

Finally, human activities produce different pressures that interact with abiotic and biotic factors in shaping the structure and variation of communities. Dissecting the natural from the human induced sources of uncertainty can be extremely difficult (Clarke 2000). Quite often, all the above described sources of errors (spatial and temporal sampling variation, sample processing errors and human induced changes) are, to a certain (unknown) extent, reflected in the outcome of biological assessment.

Uncertainty in reference conditions

A degree of uncertainty is associated with both components of EQR, the expected and the observed value. Therefore, an assessment in the uncertainty associated with the

reference condition values should also be included in the overall assessment of EQRs uncertainty. The reference value of a given metric comes with an associated error because of the difficulties of defining the reference conditions for many water bodies. The WFD states that Reference conditions may be defined by collecting biological information from sites that are in natural baseline conditions, because of minor anthropogenic impacts. Alternatively, they may be derived by modelling or by a combination of modelling and actual biological data on minor impacted sites. For modelling purposes, historical, palaeolimnological and other available data might also be applied. However, in cases where human impact is so extensive that no reference sites available and data availability is insufficient for modelling, expert judgement may be the only possibility to establish reference conditions. Therefore, data uncertainty combines with model uncertainty and propagate to model prediction (e.g. the reference value; figure 3.1).

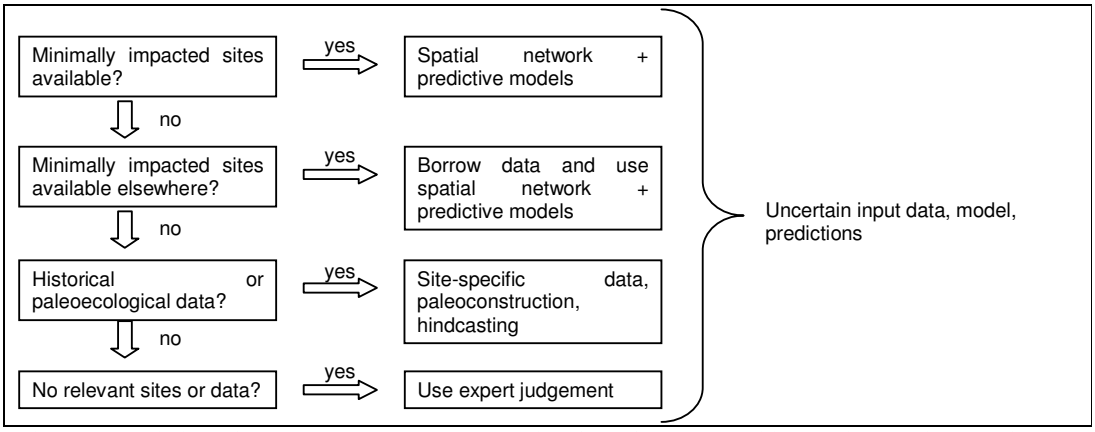

Figure 3.1 Relationship between suggested approaches for reference condition settings and uncertainty

The overall data uncertainty of a given biological metric at reference sites has the same sources of variation described in the paragraph above. It is clear that metrics should be selected after examining their variability... Metrics too variable, even at reference sites, are unlikely to be effective for the assessment. Although, relevant typology factors of water bodies can be included to diminish variability in reference values for ecological measures (see Lyche Solheim (2005); for a review on reference conditions in European lakes), variability may remain high. For example, a recent assessment of total phosphorus TP values in a large dataset of reference European lakes led to coefficient of variation ranging from 1% to over 69% depending on the lake type and sample size (Cardoso *et al.* Submitted).

Also the uncertainty of the model used for setting the reference values can be large in statistical terms. As an example consider the regression model to predict reference values for total phosphorus (TP) in European lakes from simple edaphic factors that was recently developed within the Rebecca WP3 (figure 3.2; (Cardoso *et al.* Submitted). The variance explained by this model (51%) is similar to other models to predict reference TP levels. Consider also a hypothetical eutrophic low altitude Nordic lake with an average alkalinity of 0.1 meq/l and 10 m average depth for what we want to predict "reference" TP. Note that this lake typological parameters are within the range well defined by the relationship of Cardoso et al. Using the model parameters the resulting predicted reference TP level (annual average) will be between 3.9 µg/l and

17.8 µg/l., with an average of 8.1 µg/l. Water managers may face the problem of deciding if the uncertainty associated with the predicted average value is small enough to detect changes with the actual lake conditions.

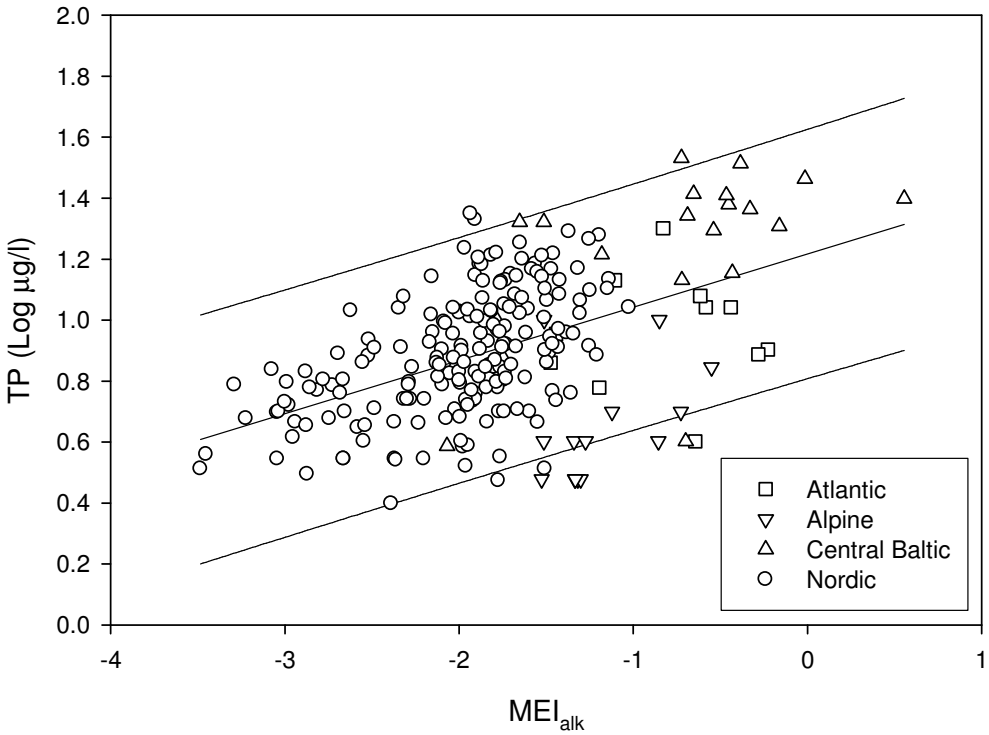

Fig. 3.2 Total phosphorus (TP) – Morphoedaphic index relationship for European lakes (from Cardoso *et al.* (Submitted) with 95% prediction interval. Lakes in different geographical intercalibration groups are identified. A= Atlantic, Al= Alpine, CB= Central-Baltic, N=Northern.

3.3 Examples of tools for estimating uncertainty relevant for WFD ecological assessment

DUE- Data Uncertainty Engine

The Data Uncertainty Engine (DUE) software was developed within the HarmoniRiB project (Harmonised techniques and representative river basin data for assessment and use of uncertainty information in integrated water management; Contract EVK1-CT-2002-00109) and is freely available on the web (http://www.harmonirib.com). Although DUE is not targeting directly the uncertainty related to EQR based assessment, it has useful general features that apply to uncertainty analysis of ecological datasets. For example, it account for variables that vary continuously in space and time and supports continuous, discrete or binomial variables. It should be noted that techniques for handling the sorts of discrete data that occur in ecology (e.g. species lists) are limited. Also uncertainty associated with the spatial and temporal autocorrelations of data as well as uncertainties in geographic objects can be handled. As described in the manual (Brown and Heuvelink 2005), DUE allows uncertainties in model inputs to be described and their impacts propagated through model predictions. An uncertainty model is constructed from sample data and expert judgement and variability in the

model inputs are defined with a probability distribution, a confidence interval or possible scenarios. Uncertainty propagation is quantified by sampling from the uncertain inputs and implementing the model for each 'realisation' of the input values (see Brown and Heuvelink 2005).

Starbugs- Star Bioassessment Uncertainty Guidance Software
The Star Bioassessment Uncertainty Guidance Software (Starbugs) is a deliverable of project Star (Standardisation of river classification (http://www.eu-star.at/; Contract EVK1-CT-2001-00089). The aim of the package is to assess uncertainty in the classification of ecological status of rivers. Although designed for river macroinvertebrate data, Starbugs can be used with generic single metric, a combination of metrics and multi-metric indices. For each metric, users are required to provide estimates of variability (e.g. standard errors) including sampling variation, subsampling, sorting and identification biases. Starbugs uses standard errors to simulate the range of possible values that could have been produced by the same assessment protocol and calculate the probability of assigning a given site to a particular ecological quality class. Data on the variability of metrics in reference conditions is also required. For each metric entered, the program produces estimates of the observed status class and probabilities of belonging to each of the five possible classes, the EQR value, its standard deviation and the lower and upper 95% non-parametric confidence limits.

Rivpacs – River Invertebrate Prediction And Classification System
The River Invertebrate Prediction And Classification System, Rivpacs (http://www.ceh.ac.uk/sections/re/RIVPACS.html) is probably the oldest software dealing with ecological assessment data. Specific for river macroinvertebrates, Rivpacs III+ can generate statistical information on the risk of misclassification caused by uncertainty in the physical variables that are used to predict the reference type. The frequency distribution of simulated observed/expected values represents the degree of uncertainty in the true value of the metric at that site/time (Clarke 2000). Although errors due to the inadequacy of reference site selection, the choice of environmental predictors and the statistical methods are not considered in the software, comparisons of Rivpacs III+ results with alternative statistical methods produced similar results (Clarke 2000).

3.4 Concluding remarks and recommendations

Uncertainty is present at all levels of ecosystem management for conservation (Burgman *et al.* 2005). For decision maker, the importance of uncertainty of ecological data depends on the context and increases with larger spatial scales and longer time horizons (Burgman *et al.* 2005). However, many forms of uncertainty are not acknowledged in the models that support decisions or, in some cases, ignored altogether (Burgman *et al.* 2005). In the case of ecological data, the complexity of error budget and the magnitude of resulting uncertainty estimates may be very large and so the necessary effort (e.g. the cost) for its estimation. However, as ecological data are often used for comparisons, precision is probably more important than accuracy (Dunbar 2005). Ecologists have become more and more sophisticated in realizing suitable metrics that are stable and robust enough to assess ecological status in a reliable manner. Quantification of EQR uncertainty should be implemented in future

assessment programs. Software like starbugs (http://www.eu-star.at/) may help in the assessment of EQR uncertainty and provides a first attempt into this direction.
It should be remarked that the analysis of uncertainty of EQR classification of a given site resulting from the use of a specific assessment scheme does not reveal the (unknown) real quality class of that site. If the EQR assessment outcome can be incorporated into a modelling framework, uncertainties may be assessed through careful evaluation of model predictions. A possible approach is to account for as many forms of uncertainty as possible (such as error propagation and sensitivity analyses) and to use several modelling frames and examine if the choice makes a difference to management decisions (Burgman *et al.* 2005). Error propagation is the calculation of statistical error in quantities that comprises multiple components, each with associated error (Blukacz *et al.* 2005). Although Gaussian error propagation and Monte Carlo type error analysis are promising and powerful tools, those techniques have been rarely applied in ecological studies (see also Lo 2005).

4 The use of EQR values in classification

Common EQR values for class boundaries?
Because the main aim of ecological quality ratios is to enable comparability between different assessment, it would be tempting to agree on common EQR values for the class boundaries (e.g. 0.8 for the high-good boundary, and 0.6 for the good moderate boundary, etc.). Although this approach may be feasible in some cases, it should not be applied without considering the nature of the relationship between pressure and biological impact. This issue has been brought up in the guidance on the intercalibration process (Anonymous 2005a), that states that common EQR values only make sense, and are only possible, where very similar assessment methods are being used or where the results for different assessment methods are normalised using appropriate transformation factors. This is because different assessment methods (e.g. using different parameters indicative of a biological element) may show different response curves to pressures and therefore produce different EQRs when measuring the same degree of impact.

EQR scales in some special cases
For indicators that do not continuously decrease with anthropogenic pressure to minimum value of 0 it is necessary to make some kind of transformation in order to establishment of an EQR scale from 0 (bad quality) to 1 (reference conditions), as required by the WFD. Two of these cases are:

1. low values at reference conditions, increasing with pressure

A clear example of this situation is phytoplankton biomass expressed as chlorophyll-a or biovolume. A possible way to calculate the EQR in this case is simply to invert the relationship:

EQR = [reference value]/[measured value]

A doubling of the measured value compared to the reference value therefore results in the EQR of 0.5; this approach, proposed in the WFD intercalibration work on chlorophyll, results in rather low values for class boundaries. An alternative approach would be to use logarithmic transformations of the values:

EQR = [log(reference value)]/[log(measured value)]

Another approach is to take into account that in the real world the parameter values do not occur above a certain maximum value, as is clearly the case for chlorophyll-a. This maximum value can than be set as 0, the lower anchor for the classification.

2. Discontinuous classifications

Many existing and newly developed classification methods are not continuous, but have a limited number of categories. There are many examples of such methods for macroinvertebrates in rivers, such as the Danish Stream Fauna Index, the Italian IBI index, the Irish “Q” index, and many others. Although EQR values can be calculated easily for such methods, it is more difficult to establish pressure-impact relationships using such methods, especially in cases where they have only few possible values. This makes comparison with continuous indices difficult to make.

5 *Checklist for WFD compliant assessment methods*

The requirements of the WFD for the establishment of assessment methods for the biological quality elements can be summarised in a checklist (Figure 4.1).

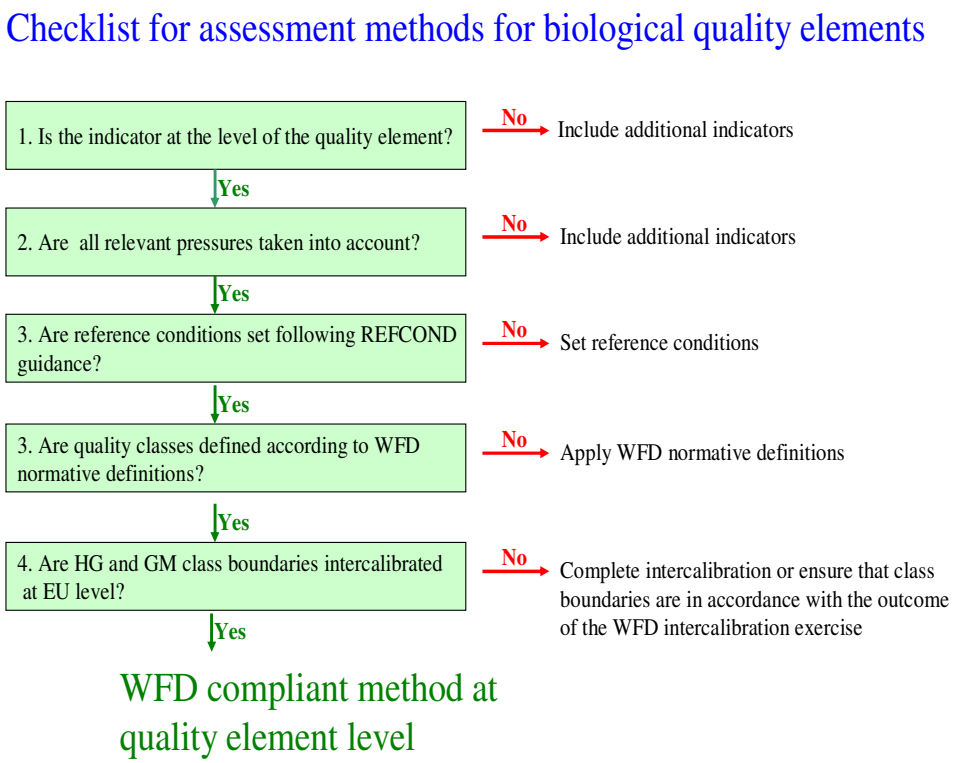

Fig. 4.1 Checklist for compliance of assessment methods for biological quality elements

Of the methods currently in use by the EU Member States it seems that very few will meet all these criteria at this moment. Especially the first two criteria are in most cases not met. Current methods are mostly addressing only part of a quality element, and are addressing only few pressures (mostly eutrophication). A possible exception is the situation with benthic macroinvertebrates in rivers, where most countries have developed multimetric indices that are sensitive for multiple pressures. The last three criteria – reference conditions setting, definition of quality classes, and intercalibration of those boundaries – are currently addressed in the WFD intercalibration exercise, with a strong focus on specific quality elements and pressures. It is expected that the remaining quality elements and pressures will be addressed in the next couple of years, and that there will be a tendency from very specific, single-parameter, pressure-specific methods towards more general multimetric approaches.

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7 Abstract

This report addresses the issue: What is a WFD compliant assessment method? This is done by focusing on the concept of the Ecological Quality Ratio (EQR). The EQR incorporates the key WFD requirements for ecological classification: typology, reference conditions, and class boundary setting. The Deliverable is targeted both to the policy makers and competent authorities implementing the Water Framework Directive and the scientists supporting them with their specific knowledge.

The Classification Guidance separates three levels in the biological assessment: the parameter level, the quality element level, and the status classification. The main conclusion is that the WFD requires classification of water bodies at the quality element level, and that the worst of the relevant quality elements determines the final classification (the “one out, all out” principle). How the different parameters within a quality element are combined is not prescribed; this can either be done by combining them in a multimetric index, or in any other way.

WFD- compliance criteria for assessment methods include reference conditions setting, definition of quality classes, and intercalibration of those boundaries. Those are currently addressed in the WFD intercalibration exercise, with a strong focus on specific quality elements and pressures. It is expected that the remaining quality elements and pressures will be addressed in the next couple of years, and that there will be a tendency from very specific, single-parameter, pressure-specific methods towards more general multimetric approaches.

Quantification of EQR uncertainty should be implemented in future assessment programs. Software like starbugs may help in the assessment of EQR uncertainty and provides a first attempt into this direction. It should be remarked that the analysis of uncertainty of EQR classification of a given site resulting from the use of a specific assessment scheme does not reveal the (unknown) real quality class of that site. If the EQR assessment outcome can be incorporated into a modelling framework, uncertainties may be assessed through careful evaluation of model predictions.

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