

ESTABLISHING REFERENCE CONDITIONS AND SETTING CLASS BOUNDARIES

Richard K. Johnson, Mats Lindegarth, Jacob Carstensen

WATERS Report no. 2013:2

WATERS Report no. 2013:2
Deliverable 2.1-2

Establishing reference conditions and setting class boundaries

Richard K. Johnson, Swedish University of Agricultural Sciences

Mats Lindegarth, Gothenburg University

Jacob Carstensen, Aarhus University

WATERS partners:

WATERS: Waterbody Assessment Tools for Ecological Reference conditions and status in Sweden

WATERS Report no. 2013:2. Deliverable 2.1-2
Title: Establishing reference conditions and setting class boundaries
Publisher: Havsmiljöinstitutet/Swedish Institute for the Marine Environment,
P.O. Box 260, SE-405 30 Göteborg, Sweden
Published: March 2013
ISBN 978-91-980646-4-3

Please cite document as:
Johnson, R.K., Lindegarth, M., Carstensen, J. Establishing reference conditions and setting class boundaries Deliverable 2.1-2, WATERS Report no. 2013:2. Havsmiljöinstitutet, Sweden.

http://www.waters.gu.se/rapporter

WATERS is a five-year research programme that started in spring 2011. The programme's objective is to develop and improve the assessment criteria used to classify the status of Swedish coastal and inland waters in accordance with the EC Water Framework Directive (WFD). WATERS research focuses on the biological quality elements used in WFD water quality assessments: i.e. macrophytes, benthic invertebrates, phytoplankton and fish; in streams, benthic diatoms are also considered. The research programme will also refine the criteria used for integrated assessments of ecological water status.

This report is a deliverable of one of the scientific sub-projects of WATERS focusing on establishing reference conditions of inland and coastal ecosystems. The report presents reviews of WFD requirements and current Swedish approaches for establishing reference conditions and for setting class boundaries. These results will be further elaborated in coming work, thus providing a framework for a more harmonised treatment of reference conditions and classification using biological quality elements in monitoring programmes.

WATERS is funded by the Swedish Environmental Protection Agency and coordinated by the Swedish Institute for the Marine Environment. WATERS stands for 'Waterbody Assessment Tools for Ecological Reference Conditions and Status in Sweden'. Programme details can be found at: http://www.waters.gu.se

Table of contents

Summary

A central feature of the European Water Framework Directive (WFD) is that deviations in ecological quality have to be established as the difference between expected (reference condition) and observed conditions (European Commission, 2000). This approach underpins the importance of reference conditions for defining a reference biological community, for establishing the upper anchor in setting class boundaries, and, ultimately, for identifying departures from expected that may be caused by anthropogenic stress. According to the WFD, the normative definition of good ecological status refers to a slight deviation from undisturbed conditions, i.e. "The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions". Good environmental status (GES) according to the Marine strategy framework directive (MSFD) refers to a condition associated with sustainable use, i.e. "… the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable…" (European Commission 2008). Hence, the underlying principles between the two environmental directives differ, with the WFD focusing on concepts related to "naturalness" of biological structure and function, while the MSFD is based on long-term sustainable use of marine areas. This report reviews methods currently used to establish reference conditions of inland and coastal waters in Sweden.

Although the concept of reference conditions has been widely developed, in particular for inland surface waters, much contention still exists concerning what defines a reference condition (e.g. use of pressure criteria) and how reference conditions can be established in areas (catchments) that are heavily modified by land use. Our review showed that spatial approaches and pressure criteria are commonly used for establishing reference conditions of inland surface waters both within Sweden and elsewhere. In an earlier revision of developing classification schemes for Swedish lakes and streams the working groups collaborated in harmonizing approaches for dealing with reference condition (SEPA 2007). Using a pressure filter approach, ecoregion delineations are used to partition large-scale natural (biogeographic) variability, thereafter more local scale variables are used to partition among- and within-site variance (aka typology-based). With few modifications, the three main ecoregions used coincided with the ecoregions described in the WFD (i.e. Illies 1978), namely, the Central Plains, the Fenno-Scandian Shield and the Borealic Uplands. Identification of reference conditions for fish assemblages used a modeling

approach, with minimally disturbed sites used in model calibration. In contrast to lakes and streams, finding minimally disturbed areas in marine systems is difficult due to their openness and connectivity, and the relative importance of diffuse pressures (e.g. excess nutrients). Not surprisingly, our review showed that approaches used for coastal/transitional waters differ markedly and are more heterogeneous compared to those used for inland surface waters. Current approaches include the use of minimally disturbed sites, historical data, modeling and expert judgment, with particular focus on spatial representativity and functional differences. WATERS will build on the work done in establishing reference conditions of Swedish waterbodies, as well as work done in intercalibration exercises, to harmonize future approaches. For example, typology- versus modeling-based approaches for establishing reference conditions and detecting ecological change will be studied.

Approaches for setting class boundaries differed both within (e.g. biological quality elements within lakes) and between systems (freshwater, marine). Where minimally disturbed conditions were available in adequate numbers, many groups used EQR (ecological quality rations; observed to expected) distributions of high quality sites to set High-Good boundaries. Good-Moderate boundaries were established using a suite of approaches such as breakpoints, sensitive/tolerant taxa or equidistance. Given the diversity of approaches used and general lack of knowledge concerning uncertainty in classifications, WATERS is focusing on uncertainties associated with environmental assessments in general, and classification specifically.

Svensk sammanfattning

Ekologisk kvalitet, fastställd som en skillnad mellan observerade och förväntade ”naturliga” referensförhållanden, är ett centralt inslag i det europeiska ramdirektivet för vatten (vattendirektivet) (Europeiska kommissionen, 2000). Bestämning av referensförhållanden för de biologiska kvalitetsfaktorerna bottenfauna, makrovegetation, bentiska kiselalger, växtplankton och fisk, är därför ett viktigt steg för att klassificera status och för att urskilja avvikelser orsakade av mänsklig påverkan i kust- och inlandsvatten. Enligt ramdirektivet för vatten avser ”god” ekologisk status en mindre avvikelse från referensförhållanden, dvs "*…värdena för de biologiska kvalitetsfaktorerna för typen av ytvattenförekomst visar låga nivåer av distorsion till följd av mänsklig verksamhet, men avviker endast i liten från dem som normalt förknippas med typ av ytvattenförekomst vid opåverkade förhållanden*". God miljöstatus enligt det marina direktivet definieras däremot i relation till ett tillstånd som inrymmer ett hållbart nyttjande, dvs *"... det miljötillstånd för marina vatten där dessa utgör ekologiskt variationsrika och dynamiska oceaner och hav som är rena, friska och produktiva inom sina inneboende förutsättningar och användningen av den marina miljön är på en nivå som är hållbar...* " (Europeiska kommissionen 2008). Detta innebär att de underliggande principerna för statusklassning skiljer sig mellan två viktiga miljödirektiv; medan vattendirektivet är relaterat till "naturlighet" bygger det marina direktivet på långsiktigt hållbar användning av havet. Denna rapport sammanfattar de metoder som anges i vattendirektivet och dess stödjande dokument samt hur metoderna används för att fastställa referensförhållanden och klassgränser för sjöar, vattendrag och kustvatten i Sverige. Vi diskuterar även hur metoderna förhåller sig till andra direktiv och miljömål.

Kunskapen om referensförhållanden har utvecklats mycket på senare år. För inlandsvatten pågår diskussioner om hur man definierar ett referenstillstånd i praktiken (t.ex. hur påverkanskriterier används), hur referensförhållanden skall fastställas i avrinningsområden som är starkt påverkade av markanvändning, eller i områden med starka naturliga gradienter. Typologi och påverkanskriterier används vanligen för att fastställa referensförhållanden av sjöar och vattendrag, både inom och utanför Sverige. I en tidigare revidering av klassificeringssystem för svenska sjöar och vattendrag samarbetade arbetsgrupperna för att harmonisera metoder för att fastställa referenstillstånd (SEPA 2007). Biologiska kvalitetsfaktorer varierar vanligen naturligt på grund av storskaliga (biogeografiska) faktorer, motsvarande ekoregioner, och småskaliga faktorer som skapar lokala skillnader. Med vissa små justeringar används de tre stora ekoregionerna som beskrivs i ramdirektivet (Illies 1978), d.v.s. Centralslätten, Fennoskandiska skölden och

Boreala höglandet som ett första steg att definiera referenstillstånd. I utveckling av referensförhållanden för fisk används modellering med relativt opåverkade system.

I jämförelse med inlandsvatten, är det mycket svårare att finna relativt opåverkade områden i marina system, t.ex. på grund av systemets öppenhet och betydelsen av diffus påverkan. En genomgång av de metoder som använts för kustvatten visar på en större diversitet i jämförelse med inlandsvatten. Detta beror sannolikt på tillgången på data och skillnad i bakgrundskunskap mellan kvalitetsfaktorer. Metoder som används för kustvatten involverar tillståndet i relativt opåverkade områden (nationella referensstationer), historiska data, modellering och expertbedömningar, med särskilt fokus på representativitet och funktionella skillnader. WATERS kommer att bygga vidare på det arbete som har gjorts för att fastställa referensförhållanden, t.ex. interkalibreringsarbetet, med särskilt fokus på att harmonisera framtida strategier. Till exempel kommer jämförelser mellan typologi och modellering studeras för att fastställa referensförhållanden och upptäcka ekologiska förändringar.

Metoder för att fastställa klassgränser skiljer sig både inom de olika systemen (t.ex. mellan biologiska kvalitetsfaktorer i sjöar) och mellan system (inlands- och kustvatten). För vissa kvalitetsfaktorer har vattenförekomster med lågt påverkanstryck använts för att sätta gränsen mellan ”hög” och ”god” ekologisk status, om tillräckligt många opåverkade system fanns tillgängliga. Gränsen mellan ”god” och ”måttlig” ekologisk status bestäms med olika metoder såsom ekologiska brytpunkter, kvoter mellan känsliga och toleranta arter eller med hjälp av indelning i likstora intervall. Ett viktigt syfte med WATERS är att undersöka möjligheten att harmonisera de metoder som används för att fastställa klassgränser för olika biologiska kvalitetsfaktorer, med speciellt fokus på osäkerheter i klassificeringen och bedömning av miljöstatus.

1 Introduction

Ecological assessment of aquatic ecosystems is a growing area of research, and in Europe, in particular, this area is experiencing a rapid expansion since the ratification of the European Water Framework Directive (European Commission 2000). In contrast to earlier legislation pertaining to aquatic ecosystems, the European Water Framework Directive (WFD) is probably the most significant piece of ordinance to be assembled in the interests of preserving and restoring the biodiversity of inland waters, wetlands and coastal areas. For instance, whereas previous statutes focused on curbing emissions and monitoring using chemical indicators, the Directive focuses on catchment planning and management, viewing aquatic ecosystems not as isolated entities, but holistically as larger interconnected ecosystems. Indeed, a key feature of the Directive is its focus on detecting ecological change (i.e. degradation and recovery) and determining what human-generated pressures (or stressors) are acting as drivers of change.

To accomplish this the WFD recognizes that present and future pressures may dictate different monitoring and assessment designs such as surveillance, operative and investigative monitoring of ecological quality. These three types of monitoring can be summarized as 1) assessment of regional pattern and trends (surveillance monitoring), 2) detection of ecosystem recovery (operational monitoring) and 3) assessment of putative stressor(s) (investigative monitoring). One caveat or challenging aspect of all three of the assessment approaches is the need to establish a benchmark, reference condition to anchor judgments of change (that needs to be used in the calculation of ecological quality ratios of observed and expected condition). In brief, reference conditions are needed for defining a reference biological community to be used in establishing the upper anchor for setting class boundaries and for identifying departures from the expected that may be caused by anthropogenic stress. However, since the landscape of much of Europe has been altered for centuries, even finding minimally disturbed sites, let alone true pristine sites, is difficult for the majority of ecosystem types.

2 Objective

The objectives of this report are 1) to give a short review of methods commonly used to establish reference conditions (RC) and class boundaries of European inland and coastal surface waters, 2) to provide examples of contemporary work with establishing reference conditions in Europe, with special focus on results from the EU Intercalibration exercise, 3) to summarize how reference conditions are currently being established for Swedish inland coastal surface waters, and 4) to provide examples of how classification boundaries can be established.

3 Definition of reference condition

A number of problems emerge when trying to define and use a reference condition approach in monitoring and assessment programs. Although seemingly trivial, one problem is that many definitions exist of what constitutes a reference condition and another is that definitions are not always interpreted in the same way, both of which may result in misunderstanding and contention. Common definitions range from a "natural condition", where humans have no influence on the environment (see e.g. Bishop et al. 2009), to the "best attainable within a region", which recognizes that humans are often an inherent part of the ecosystem (e.g. Nowicki 2003). Depending on the ecosystem/region of interest both definitions may be appropriate. According to the WFD (Annex 5, section 1.2), the reference condition (or high ecological status) is defined as having "*no, or only very minor, anthropogenic alterations to the values of the physico-chemical and hydromorphological quality elements for the surface water body type from those normally associated with that type under undisturbed conditions*". For biological quality elements (BQEs) "*The values…for the surface water body reflect those normally associated with that type under undisturbed conditions, and show no, or only very minor, evidence of distortion.*".

Consequently, the WFD does not sanction the use of best attainable sites within a region, unless these sites can be shown to reflect a natural state with no or only minor human influence. Key issues in implementing the reference condition approach are: 1) defining what is meant by no, or only minor, alterations, 2) bettering our understanding of the strengths and weaknesses of approaches used to establish a reference condition, 3) distinguishing between human-induced changes and changes that are a natural, innate part of the ecosystem being studied (i.e. those changes that are "normally associated with undisturbed conditions"), and 4) determining what is meant by "normal" for the three (non-independent) ecosystem descriptors: hydromorphological, physical-chemical and biological quality elements. In this report we will focus on methods commonly used to establish reference conditions for inland and coastal waters.

In an attempt to add clarity to the use of reference condition terminology used in assessment Stoddard et al. (2006) have proposed a number of terms to be used when referring to reference condition (RC). These authors propose that the term "reference condition" be reserved for "naturalness" or "biological integrity", and when referring to reference conditions that deviate from naturalness or integrity the authors propose the use of four terms.

1. **Minimally disturbed condition** (MDC) is used to describe the condition in the absence of significant human disturbance. The use of this term recognizes that finding truly undisturbed sites even in relatively undisturbed areas like the Nordic countries is not possible due to the influence of land use (e.g. forestry) and cross-boundary transport of pollutants by air or currents. An important aspect of the use of MDC in a spatial context is the recognition that indicator metrics/variables will vary naturally and that this natural variability (e.g. long term climatic and ecological fluctuations) needs to be considered when describing MDC.
2. **Historical condition** (HC) is used to refer to a condition of a lake, stream, estuary or coastal area at some point in time. The use of HC may reflect the true RC if the point in time chosen is before the influence of human disturbance. For example, the REFCOND guidance document recommended the use of pre-intensive land use (e.g. agriculture) as a past state corresponding to low anthropogenic pressure (Anonymous 2003). Accordingly, REFCOND recognized that since human-induced pressures vary across Europe a fixed-date HC was not possible. For example, the HC may represent ca 1850 in parts of the UK but may represent an even earlier epoch (e.g. 1600s) in other parts of Europe (e.g. Germany).
3. **Least disturbed condition** (LDC) is defined as the sites in the landscape having the best available physico-chemical and biological conditions. Often, explicit criteria are used to describe as is "best". For example, in some regions the best may be defined as having < 1% of the landscape classified as agriculture, whereas in another region a level of < 20% may be justified. As land use varies both spatially and temporally, LDC will vary accordingly.
4. **Best attainable condition** (BAC) is equivalent to the expected ecological condition (or LDC) if the best possible management practice were in use for some period of time. Since the upper and lower limits on BAC are sites by definitions of MDC and LDC, respectively, it is unlikely that the BAC will be neither better than MDC nor worse than LDC, but it may be similar to either of these depending on the prevailing level of human disturbance in a region.

4 Review of methods for establishing reference condition

A number of methods are currently used to establish reference condition and step-by-step protocols for selecting reference sites are readily available (e.g. Hughes et al. 1994; Hughes 1995). Reynoldson and Wright (2000) recommend, for example, a three-step approach for establishing reference conditions. Firstly, sites are spatially stratified to ensure that the full range of conditions is represented. In the second step, local knowledge is solicited as it may provide invaluable information on the degree of degradation not elucidated by the coarse screening criteria used in the first step. Lastly, in the third step, iterative data examination is used to select potential reference sites. The use of map information and screening criteria in the first step to identify (or screen for) "areas of interest", where pristine or minimally disturbed sites may be located, is strongly recommended as a cost-saving procedure. A similar approach is also implicit in the WFD (e.g. Anonymous 2003). For example, according to the Directive reference conditions are to be linked to stream typologies and the population of reference sites should represent, as well as possible, the full range of conditions that are expected to occur naturally within the stream type. In the final data examination step, caution should be exercised to avoid circularity. For instance, the use of the same biological element to establish and validate reference condition is not advocated.

A number of time or space approaches are presently being used to establish reference conditions (e.g. Stevenson et al. 2004). The most common methods can be grouped into four categories: 1) expert judgment, 2) temporally based approaches using historical data or paleoreconstruction, 3) spatially based approaches using for example survey data (Johnson 1999), and 4) modeling approaches such as hindcasting (e.g. Hughes 1995; Reynoldson et al. 1997; Anonymous 2003; Valinia et al. 2012). In areas where land use has not drastically altered the landscape the identification of reference conditions is rather straightforward, and spatially based (e.g. survey) approaches are frequently used as they include natural variability. In contrast, establishing a reference condition in areas where potential reference sites are few or lacking is more complex and may require a combination of approaches (e.g. data borrowing, modeling, paleoreconstruction and expert judgment) (e.g. Nijboer et al. 2004; Carstensen and Henriksen 2009). A brief description of methods commonly used to define reference conditions as well as some of their strengths and weaknesses is given below.

4.1 Expert judgment and/or the use of historical data

A reference can be thought of as what is perceived (e.g. using expert judgment) or known (e.g. using historical data) as being the former or original state of the environment in the absence of human influence. Although undisturbed conditions may be defined as the conditions existing before the onset of intensive agriculture or forestry and before large-scale industrial disturbances, the actual time period will obviously vary across Europe due to differences in anthropogenic stress. In many areas in northern Europe this time period would correspond to the mid-1800s, whereas in the southern parts of Europe a much earlier time period would be required to attain the same state of naturalness. However, considering that the landscape of much of Europe has been altered for centuries, identification of pristine or even minimally disturbed reference sites will be difficult for many ecosystem types (in particular large rivers and lakes).

Expert judgment or historical data, with the exception of paleoreconstruction, are seldom used as a single method to establish reference condition, although either one or both may be used to complement other approaches. For example, Brucet et al. (in press) showed that expert judgment, although not a method approved by the WFD, was frequently used, in combination with other methods, for setting reference conditions of European lakes. One of the strengths of using expert judgment in defining reference condition is that this approach may amalgamate historical data and/or opinion and present-day concepts. A caveat, however, is that expert judgment often consists of a narrative articulation of a perceived reference condition, and consequently this approach may introduce subjectivity (e.g. the common perception that it was always better in the past) and bias (e.g. experts may disregard sites having naturally low diversity). Even the use of historical data, albeit less prone to subjective bias, is not problem-free. For example, similar to the use of present-day or extant data, the interpretation of historical data can be complicated by a number of factors such as the timing and frequency of sampling and the use of different field (sampling) and laboratory (processing) methodologies. Another shortcoming in using historical data for defining reference condition is that data availability is often limited (e.g. only qualitative data are available). The use of the paleo record to reconstruct past conditions, either directly (by using the remains of taxa stored in the sediment to reconstruct an assemblage) or indirectly (by using taxon information to infer past water chemistry) overcomes many of the shortcomings associated with using past records or museum samples. This approach has been shown to be applicable for lakes (see below) and coastal ecosystems (Clarke et al. 2003; Andersen et al. 2003; Clarke et al. 2006; Andrén et al. 2007), although the validity of the approach is still debated. Finally, a weakness of using expert judgment, historical data (excluding the sediment record) or many other methods in defining reference condition is that the measure obtained is often a static measure that does not include the dynamic and inherent variability often associated with natural ecosystems.

4.2 The use of survey data and space-for-time substitution

In areas that are not heavily affected by anthropogenic stressors (e.g. the Nordic countries) reference conditions (or minimally disturbed condition, MDC) can be relatively easily established using a survey approach (e.g. Johnson 1999). One of the strengths of a survey approach is that it can either explicitly (sites are sampled to include among-year variability) or implicitly (space-for-time substitution) include natural variability. Another reason for the popularity of using a survey approach in areas where human influence is low is that this approach is relatively transparent and hence one of the least contested methods. In short, in a survey approach the reference condition is usually defined *a priori* using a set of clearly outlined criteria (e.g. catchment land use < x percent agriculture, deposition of airborne pollutants < x kg/ha/yr, no dispersal barriers). Thus, the sites included in the survey represent the natural variability of the response variable.

Survey data can be used to establish reference condition directly (e.g. using a typology based approach as described in the WFD) or indirectly (e.g. by calibrating predictive models, see below). The development of a typology-based approach for establishing reference condition assumes that the natural variability among sites can be partitioned with a parsimonious set of descriptor variables (e.g. ecoregions or landscape attributes, altitude, habitat types). Implicit in this approach is that if a substantial amount of ecological variance can be partitioned, then these features can be used to estimate the ecological potential of a new site with greater precision than if all sites were assumed to come from the same population. If a typology-based approach is not able to adequately partition the prevailing ecological variation (i.e. the approach has low statistical power to detect human-induced change), a predictive approach may be more appropriate. This may be true if the ecosystem attributes of interest (e.g. macroinvertebrate diversity) vary continuously and not discretely in response to environmental gradients. Variables used to partition natural variability or used in model calibration should be insensitive to human-related disturbance. Geographic variables such as latitude, longitude and altitude, as well as catchment land use and geology and habitat descriptors (substratum type) have been successfully used in both typology- and modeling-based approaches (Wright et al. 1996; Reynoldson et al. 1997; Simpson and Norris 2000; Johnson and Sandin 2001; Johnson 2003). In a review of methods used to establish reference conditions for lakes in Europe, Brucet et al. (in press) found that 17% of the 93 methods reviewed used near-natural reference sites to establish reference conditions, increasing to 48% when combined with other methods, such as historical data, modeling and expert judgment.

4.3 The use of predictive models and hindcasting

In many areas of Europe, humans have extensively altered the landscape over long time periods and the expected reference condition cannot be easily determined using survey or historical data. With this scenario, two modeling approaches may be used to establish a reference condition. If reliable stress-response relationships are known the reference condition can be predicted by modeling a stress-response relationship to a low (or target) level of stress. Ideally, the expected reference condition is obtained by interpolation (i.e. within the confines of the stress-response relationship) and extrapolation is done with caution. A second modeling approach uses knowledge of relationships between response- and predictor variables to predict the expected reference condition (e.g. community assemblage). Often an empirical model is calibrated using reference sites that allows the ecological attributes expected at a site (e.g. taxon richness or the probability of taxon occurrence) to be predicted from a suite of environmental variables (e.g. Wright et al. 1996; Hallstan et al. 2012).

So as not to confound predictor – response relationships, predictor variables should be insensitive to human-related disturbance. Sites included in the model calibration may consist of a subset minimally disturbed sites within the region of interest or sites "borrowed" from a nearby region. As in a survey approach, the use of inclusion/exclusion criteria is recommended as an efficient way of screening for a set of potential reference sites to be used in model calibration. Again, care should be exercised to determine that the sites included in the calibration dataset represent the expected natural variability of the population to be modeled. For example, to establish the reference condition of streams, the particular river types and the range of stream orders need be specified as well as the geographical limits of the system (e.g. Reynoldsen and Wright 2000). Both modeling approaches assume that the models used are truly representative of the relationships that exist in the undisturbed condition. This assumption might not be valid since model calibration is generally done using predictor-response relationships that lie outside the area of interest (i.e. minimally disturbed sites are few or lacking within the region of interest). Hence, care should be taken in calibration and prediction, and where possible validation of the model predictions should be attempted. Since the identification of even minimally disturbed sites is difficult due to large-scale alterations in landscape, the use of models to predict community assemblage structure and composition (e.g. the probability of taxon occurrence) has increased in popularity, in particular with the widespread use of RIVPACS-type models (e.g. Wright et al. 2000; Johnson 2000). Moreover, this is a promising direction of research as modeling approaches often de-emphasize the expertise of the individual investigator and allow for inferences regarding possible cause-and-effect relationships (Johnson et al. 1993).

5 Methods used by Member States to establish reference condition

This section summarizes early work to standardize the concept of establishing European reference conditions and intercalibration work that is currently being finalized by the EC Joint Research Centre (JRC).

5.1 The REFCOND project

The EU project REFCOND, CIS 2.3 work group, proposed 42 reference criteria to be used in the selection of reference sites (Annex 1).

The methods used to establish reference conditions (RC) for quality elements and parameters were reviewed by the EU-funded project REFCOND. The type of approach used by Member States (MS) to establish RC differed between lakes and streams and with quality element/parameter (Figure 5.1). Spatial approaches were commonly used in both lakes and streams (mean = 21% for lakes and 34% for streams). The second most common approach for establishing RC was the use of expert judgment (30% lakes, 27% streams), followed by historical (18% lakes, 21% streams), models (9% lakes, 15% streams) and paleoreconstruction (12% lakes, 3% streams). The order of methods used seemed to reflect the difficulty associated with establishing RC for the various quality elements and habitat types. For example, spatial approaches were commonly used to establish RC of phytobenthos in streams (54%), whereas expert judgment was more commonly used for macrophytes (43%).

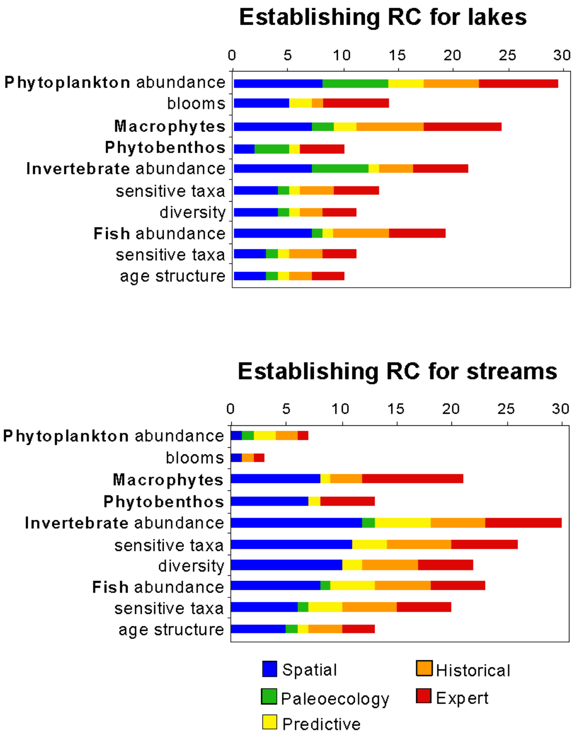

FIGURE 5.1
Methods used by Member States to establish reference condition (RC) of quality elements and parameters. Numbers show the number of Member States (REFCOND project.; Wallin et al. 2003 and CIS Working Group 2.3).

5.2 Intercalibration

In November 2002 the water directors endorsed the document "Towards a guidance on establishment of the Intercalibration network and on the process of the Intercalibration exercise" (EC 2003b). Intercalibration has since then been carried out within the Common Implementation Strategy (CIS) working group A - Ecological Status (ECOSTAT), responsible for evaluating its results and making recommendations to a strategic co-ordination group. The aim of the intercalibration process was to ensure consistency and comparability of classification results across MSs for the biological quality elements.

The first phase of the intercalibration process was carried out following CIS Guidance Document No. 14 "Guidance on the Intercalibration Process 2004- 2006" (EC 2005a). Among other things this document identified key principles of the intercalibration and a framework for deriving class boundaries consistent with the WFD normative definitions. For the second phase an update of the CIS Guidance Document No. 14 "Guidance on the Intercalibration Process 2004-2006" was developed, building on the first phase and providing further guidance for the intercalibration process during 2008-2011 (EC 2010).

The results of the first round of intercalibration were established in the Commission Decision of 30 October 2008 (European Commission 2008a) and were accompanied by a series of intercalibration technical reports (e.g. van de Bund 2009, Poikane 2009, Carletti and Heiskanen 2009). These reports describe the outcome of the intercalibration process and thus how the MSs have implemented concepts of reference conditions and class boundaries, and which principles have been applied in rivers, lakes and coastal waters. These reports were further analyzed by Pardo et al. (2011) with respect to screening criteria used by MSs to establishing reference conditions for European water bodies. Their approach was a questionnaire to establish the main pressures affecting the integrity of rivers with 17 MSs participating in the study.

5.2.1 Rivers

Pardo et al. (2011) reviewed the methods used by MSs to establish reference conditions for macroinvertebrates in rivers. Using the criteria given in Annex 1, direct measurement was the most common method used for assessing the impacts of pressure at putative reference sites, followed by field inspection and expert judgment. Somewhat encouraging was the finding that "measured" or "field inspections" accounted for almost 40% of the answers from the Central Baltic (CB) and Mediterranean (MED) Geographical Intercalibration Groups (GIGs) (Figure 5.2a). However, a large percentage also replied that information was missing: 38% (CB) and 27% (MED). Also somewhat disconcerting was the frequency of "okay" responses (e.g. 12% and 31% in the CB and MED GIGs, respectively). This response provided no information; since no information was given as to how reference conditions were established (Pardo et al. 2011). A breakdown by country showed that replies also varied markedly across member states (5.2b). For example, missing information ranged from ca 10% (Belgium) to 100% (Denmark).

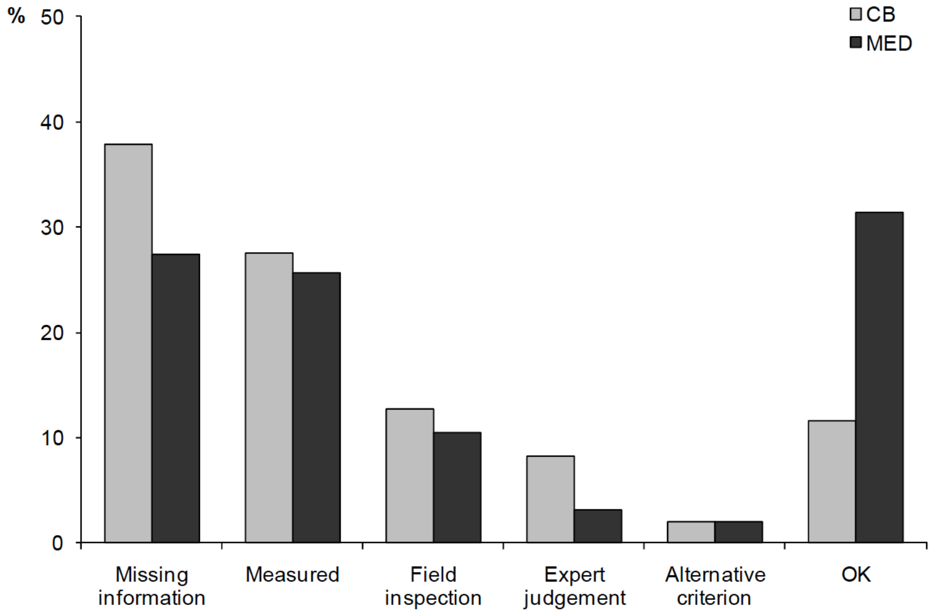

FIGURE 5.2a
Frequency (in %) of approaches used to establish reference condition. Taken from Pardo et al. (2011).

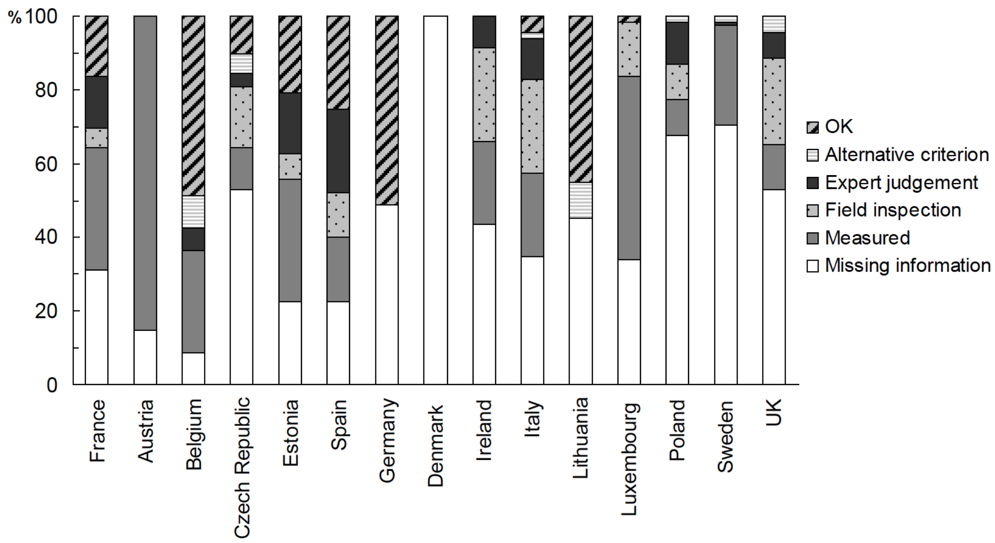

FIGURE 5.2b
Frequency (in %) of approaches used to establish reference condition by Member States. Taken from Pardo et al. (2011).

Summarizing work in the intercalibration process, Pardo et al. (2011) showed that spatial approaches were most frequently used by the MSs in classification of rivers (Table 5.1). Benthic macroinverebrates and phytobenthos were included in the first phase of intercalibration, whilst macrophytes and fish were included in phase II. For macrophytes and fish assemblages some MSs used expert judgment, and historical data was used in some cases to establish reference conditions of fish assemblages. Overall, these findings agree with those of the EU REFCOND project (see figure 5.2a and 5.2b).

TABLE 5.1
Approaches used to establish reference conditions for four biological quality elements (BQEs) according to Pardo et al. (2011). IC refers to Intercalibration phase.

BQE	IC phase	WFD approach to Reference condition
Invertebrates	I	Spatial network of minimally disturbed sites (reference sites).
Phytobenthos	I	Spatial network of minimally disturbed sites (reference sites).
Macrophytes	II	Spatial network of minimally disturbed sites (reference sites) and expert judgment.
Fish	II	Historical data and spatial network of minimally disturbed sites (not necessarily reference sites according to inverte-brate and phytobenthos criteria) (reference sites) and expert judgment/evaluation of pressures.

Screening criteria as recommended by the EU REFCOND project (see Annex 1) were commonly used for establishing reference sites for benthic macroinvertebrate and diatom assemblages (Pardo et al. 2011). However, although the approaches used were similar, differences in how the screening criteria were applied resulted in < 100% total overlap in reference sites established for benthic macroinvertebrates and phytobenthos. For example, although similar water quality variables were used for both macroinvertebrates and phytobenthos, threshold values for the latter were generally more stringent. The IC (intercalibration) group for phytobenthos recommended that there is a need to make available (e.g. published) and validate screening criteria in order for the IC process to be open and transparent. Focus should be on testing the validity of the IC typology and developing common protocols for key environmental variables used as screening criteria.

In phase II of the intercalibration work, screening criteria, like those used to select reference sites for macroinvertebrates and phytobenthos, were initially used in defining reference conditions for macrophyte assemblages (Pardo et al. 2011). However, this approach was dropped for two main reasons: 1) few potential sites were proposed by MSs, in particular for certain habitat types like sand brooks and lowland medium sized streams, and 2) some MSs were not able to identify any potential reference sites. Instead, most countries used geographical analogues, historical data or modeling to establish reference conditions for macrophyte assemblages. For fish assessments, some MSs used historical data for establishing reference sites (e.g. Germany and Austria), while the northernmost countries like Sweden and Finland implemented the concept of minimally disturbed sites in conjunction with expert opinion (Pardo et al. 2011). In brief, potential pressures and their intensity were summarized and a list of criteria was used to selected "undisturbed sites". In this work, consideration was given to the use of national criteria in establishing reference sites. In the final selection of screening criteria, the intercalibration working group decided upon the use of national criteria and the common criteria in screening for reference sites.

Using data from the European STAR and AQEM projects, Pardo et al. (2011) did preliminary analyses on the efficacy of screening criteria to establish stream reference conditions. Regression of EQR values against pressure gradients showed that slopes were not significant when analyses were restricted to putative reference sites, supporting the conjecture of no major anthropogenic gradients. Conversely, including sites that failed as reference using screening criteria (i.e. sites deemed as non-reference) resulted in significant relationships. Accordingly, the authors argued that screening criteria were effective at isolating high quality sites when applying both land use and water quality thresholds. However, in two cases regression slopes of reference sites alone resulted in significant slopes; indicating that the threshold values used for "intensive agriculture" and "mean dissolved oxygen" need to be re-evaluated.

5.2.2 Lakes

Responses of how MSs applied REFCOND criteria to establish reference conditions of lakes was summarized by Poikane (2009). The main criteria used by the Nordic GIG were agriculture (< 10% of the total catchment) and no major point sources (judged from visual observation of GIS land use and population data) (Table 5.2a). Impact criteria consisted of total phosphorus and chorophyll *a* or biovolume (Table 5.2b). Analysis of compliance to six classes of pressure types (point source pollution [criterion 1], diffuse pollution [2], morphological alterations [3], water abstraction [4], biological pressures [5], other, e.g. recreation, [6] showed that the Nordic GIG "partly" fulfilled criterion 1, "diverse" fulfillment of criterion 2, "partly, incomplete" for criteria 3-5 and did "not consider" criterion 6.

TABLE 5.2a
Pressure criteria used to establish reference condition of Nordic lakes, Northern GIG (Poikane 2009). Agriculture was mainly quantified using GIS.

Pressure criteria	Finland	Sweden	Norway	UK	Ireland
Agriculture	in data sets at present mainly ≤ 10%	< 10% of catchment	< 5%	< 10% arable or intensive grazing	no information or not used
Point sources	no major point sources	no major point sources	no major point sources	no information or not used	no major point sources
Urbanized area	no information or not used	< 0.1% of catchment	no information or not used	no information or not used	no urbanization, i.e. villages/towns < 1%
Population density	no information or not used	no information or not used	< 5 persons/km^2	< 10 persons/km^2	no information or not used
Other pressures	no significant water level regulation or morphological changes	annual mean ≥ 6 pH	no information or not used	no fish farms	no intensive use of lake, i.e. abstractions

TABLE 5.2b
Impact criteria used to establish reference condition of Nordic lakes, Northern GIG (Poikane 2009). Agriculture was mainly quantified using GIS.

Impact criteria	Finland	Sweden	Norway	UK	Ireland
Total P	no information or not used	< 10 µg/L or higher if high color	< 11 µg/L or higher if high color	no information or not used	< 10 µg/L
Chlorophyll	no information or not used	no information or not used	< 4 µg/L (low alk. Clear types), < 6 µg/L for other types	no information or not used	< 4 µg/L
Paleodata	no information or not used	no information or not used	no information or not used	if available	some sites
Expert judgment	yes, partly	no major point sources	yes	yes	yes

In summary, Pardo et al. (2011) recommended a three tiered approach for establishing reference conditions (screening) of inland and coastal surface waters.

Tier 1 - “true” reference sites – sites with no or minimal anthropogenic pressure that fulfill all criteria proposed in REFCOND Guidance for all pressures (i.e. Annex 1)
Tier 2 - “reference condition” sites or “partial” reference sites – impacted by some level of anthropogenic pressure but (some) biological communities corresponding to the reference conditions (e.g. “phytoplankton reference sites” with no or minimal eutrophication pressure but significant hydromorphological pressure which still is not affecting phytoplankton community in a significant manner);
Tier 3 - “alternative benchmark” sites – sites with some pressure and some level of impairment to biology (can be used for setting benchmark, see EC, 2010).

5.2.3 Coastal waters

Approaches used to establish reference conditions for coastal waters differ markedly from those used for inland surface waters (Table 5.3). A study of typology, reference conditions and classification of transitional and coastal waters (EC 2003b) concluded that data are often lacking to describe the chemical and biological status of high quality sites. One reason is that most studies have focused on monitoring pollution. Another reason is that across Europe few sites are at high status due to the widespread human pressures and impacts. Consequently, efforts to derive complete descriptions of reference conditions were deemed not possible in phase I of the intercalibration work.

Work for setting reference conditions of coastal waters deviated in many ways from the approaches used for inland waters. In contrast to the widespread approach of using abiotic screening criteria for inland waters, the reference conditions of coastal sites were biologically derived using modeling or expert opinion. Also, unlike the freshwater approaches, no concerted effort was made at quantifying the intensity of pressures. Finally, the use of reference condition often referred to best available or the highest quality along an ecological gradient. For example, the Baltic Sea GIG was not able to identify reference sites. Instead a relationship between Secchi depth transparency and chlorophyll *a* or nutrient concentration was used. Denmark and Finland defined the alternative benchmark as a Secchi depth of the early 1900s, Estonia used 8 m for high status, Sweden used 10 m and Poland used 6 m as the boundary between high and good status.

TABLE 5.3
Methods used by the Baltic countries to derive reference (high status) or the H/G boundary (in the case of Poland) in coastal waters. Taken from Pardo et al. (2011).

Member State	**Historical abiotic data** Secchi depth	TN	**Hind-casting of abiotic data**	**Recent abiotic/biotic relationship**
Finland	1925 - 1934	no information	no information	Chlorophyll *a* and Secchi depth, depth limit of *Fucus vesiculosus*, occurrence of cyanobacteria
Sweden	1900 - 1950 10 m	15.3 µM	no information	Secchi depth related to TN and TN related to chlorophyll *a*
Denmark	1903 - 1959	no information	nutrient loading and inputs related to TN	TN related to chlorophyll a with recent data (May-September)
Germany	no information	10 µM	TN loading	TN related to chlorophyll *a* with recent data (March-October, 1978-2004)
Poland	no information	no information	no information	Secchi depth and TN related to chlorophyll *a* with recent data (May-September, 1999-2005). For Secchi depth a summer value of 6 m is used for the H-G boundary.
Lithuania	no information	no information	no information	no information
Latvia	no information	no information	no information	no information
Estonia	no information	10.6 µM	1.1	TN related to chlorophyll *a* with recent data (June-September, 1993-2005) and max Secchi depth (8 m) during same time period

Other approaches involved the use phytoplankton, macroalgae, angiosperms, benthic invertebrates. For phytoplankton the frequency of blooms (e.g. *Phaeocystis* counts) and/or classification using chlorophyll a values were used. Intertidal assemblages of macroalgae in the North-East Atlantic GIG focused on diversity and composition (e.g. sensitive and opportunistic species). In the Baltic Sea GIG, the depth limit of eelgrass was intercalibrated between Denmark and Germany. Denmark used two approaches: 1) percent deviation from reference conditions based on historical data and 2) modeling the relationship between TN and depth limits. For benthic invertebrates, sensitive/tolerant species were used to define reference condition (Figure 5.3).

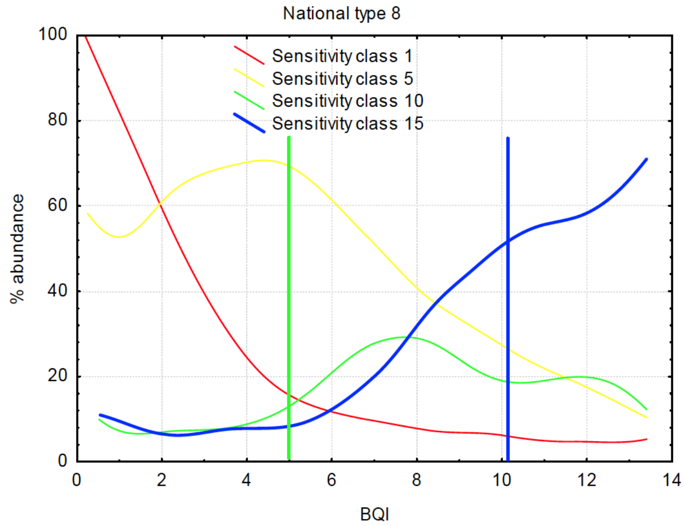

FIGURE 5.3
Example of how reference or alternative benchmark states were established for benthic macroinvertebrates. Distribution of relative abundance of four groups of macroinvertebrates with different sensitivity values. Class 1 and 15 are the least and most sensitive groups respectively. The G-M boundary is indicated by the vertical green line and the blue line represents the H-G boundary (from figure 48 in Pardo et al. (2011)).

5.3 Methods used in Sweden to establish reference condition

5.3.1 Inland surface waters

Table 5.4 a and b summarizes the methods currently used to establish reference conditions for lakes and watercourses in Sweden. With the exception of fish, reference conditions for all other BQEs are established using typology-based approaches. Reference conditions for fish metrics (EQR8 and VIX) are established using regression models using minimally disturbed sites in model calibration.

In the former revision of developing biological classification schemes an inland waters group collaborated resulting in relatively good harmonization of methods (SEPA 2007). As a result of this harmonization, the same typologies are often used to estimate reference condition of the BQEs. With the exception of fish, all other BQEs used ecoregion delineations as the first stratum to partition natural variability of BQEs in minimally disturbed sites (MDCs were established using pressure criteria, see below). The main ecoregions used in classification are: the Central Plains, the Fenno-Scandian Shield and the Borealic Uplands. However, some differences exist in ecoregion delineation. For phytoplankton the three ecoregions are 1) mountainous regions above the treeline, 2) north of the Limes norrlandicus (LN) ecotone (~ Fenno-Scandian Shield) and 3) south of the LN (~ Central Plains).

The working group decided on a number of criteria and threshold values to screen for potential reference sites (aka a pressure filter approach) (Table 5.5). Screening criteria consisted of both catchment land use/cover and physico-chemical variables. Both single and multimetric indices (fish and benthic invertebrates) were developed for BQEs. In metric calibration, the main pressures were eutrophication, acidity and general degradation. Estimates of uncertainty in classification consisted of the use of temporal variability (fish, phytoplankton, macrophytes), spatially-based estimates of variance (benthic macroinvertebrates; i.e. analogous to the use of mean values of BQE measures in an ecoregion to estimate expected conditions in the absence of stress) and method-based measures of variance (benthic diatoms).

TABLE 5.4a

Biological quality elements (BQEs) and indices used in the classification of lakes. Methods or determining reference condition: S= spatial, M=modeling/predictive.

BQE	Index	Index type	Pressure	Reference condition	
				Method	**Short description**
Fish	*EQR8*	multi-metric	general degradation, acidity, eutrophication	M	Reference condition: Site-specific reference conditions are modeled using multiple regressions, with model calibration using minimally disturbed sites passing a pressure filter. Uncertainty: The default procedure is to use a median SD of EQR8 (between years) = 0.077, i.e. the median SD observed using samples from 3-5 years within each of 113 lakes.
Benthic invertebrates	*ASPT*	single	general degradation	S	Reference condition: Mean value of minimally disturbed reference sites, using a pressure filter, based on three ecoregions (Central Plains, Fenno-Scandian Shield, Borealic Uplands). Uncertainty: mean SD of minimally disturbed reference sites, using a pressure filter, based on three ecoregions.
	BQI (profundal)	single	eutrophication	S	as above
	MILA	multi-metric	acidity	S	as above
Phytoplankton	*Total biomass*	single	eutrophication	S	Reference condition: five lake types based on ecoregions (3 regions: mountains above tree line, north of LN below tree line, south of LN) and water color. Uncertainty: mean values from at least three years of data must be used. If too few data are available, recommend using mean values of EQR SDs given for each of the five types.
	Proportion of cyanobacteria	single	eutrophication	S	as above
	TPI	single	eutrophication	S	as above
	Number of species	single	acidity	S	as above
Macrophytes	*TMI*	single	general degradation, nutrients	S	Reference condition: three lake types based on ecoregions (Central Plains, Fenno-Scandian Shield, Borealic Uplands). Uncertainty: recommend several repeated measures; if not available uncertainty is estimated.

TABLE 5.4b
Biological quality elements (BQEs) and indices used in the classification of streams. Methods or determining reference condition: S= spatial, M=modeling/predictive. NA=not applicable.

BQE	Index	Index type	Pressure	Reference condition	
				Method	Short description
Fish	*VIX*	multi-metric	general degradation, acidity, eutrophication	M	Reference condition: Site-specific reference conditions are modeled using multiple regressions, with model calibration using minimally disturbed sites passing a pressure filter. Uncertainty: Site-specific estimate of SD (between years) using multiple regression model, with model calibration using minimally disturbed sites sampled at least three times.
Benthic inverte-brates	*ASPT*	single	general degradation	S	Reference condition: Mean value of minimally disturbed reference sites, using a pressure filter, based on three ecoregions (Central Plains, Fenno-Scandian Shield, Borealic Uplands). Uncertainty: mean SD of minimally disturbed reference sites, using a pressure filter, based on three ecoregions.
	DJ-index	multi-metric	eutrophication	S	as above
	MISA	multi-metric	acidity	S	as above
Phyto-benthos	*IPS*	single	eutrophication and organic pollution	S	Reference condition: typology based; median of minimally disturbed sites using a pressure filter. Uncertainty: method-based measure of uncertainty.
	TDI	single	eutrophication	S	as above
	%PT	single	organic pollution	S	as above
	ACID	single	acidity	NA	no reference conditions are used as index is not used for class acidification

TABLE 5.5
Threshold criteria used in screening for potential minimally disturbed conditions in streams and rivers.

Pressure	Concentration	Land use
N and P from agriculture	TP < 10 µg/l If TP > 10µg/l, then use relationship between TP (flow-weighted annual mean) and water color (slightly modified from Swedish Environmental Quality Criteria*.	<10% agriculture in the catchment.
N and P from forestry	TP < 10 µg/l If TP > 10 µg/l, then use relationship between TP (flow-weighted annual mean) and water color (slightly modified from Swedish Environmental Quality Criteria*.	< 10% clear cuttings (not older than 5 years in Southern Sweden, not older than 10 years in Northern Sweden). NB This quantifies effects of N only.
Acidification	pH ≥ 6.0 If pH < 6.0, then use F-factor according to Swedish Environmental Quality Criteria*.	not applicable
Urbanization		< 0.1% population centers according to digital maps ("red maps").
Metals	Status class 1 or 2 according to Swedish Environmental Quality Criteria*.	not applicable
Alterations of hydro-morphology	No criteria available.	
Introduced species	No criteria available. Information can be obtained from local/regional authorities	

*Swedish EPA report 4913 (Anonymous 1999).

5.3.2 Coastal waters

A number of different approaches are used to define reference conditions in coastal waters (Table 5.6). The prevailing method in inland waters, the use of pressure filters to identify minimally disturbed areas, is not generally used to establish reference conditions in coastal BQEs. This is likely an effect of the openness and connectivity of marine systems and the relative importance of diffuse pressures, in particular excess nutrients, which makes it difficult to identify minimally disturbed areas. Instead, differences in availability of historical data, differences in spatial representativity of data, functional differences and perhaps to some degree lack of coordination among BQEs have meant that approaches for developing reference conditions are more heterogeneous and sometimes complex in coastal areas, compared to inland waters.

The status of benthic invertebrate fauna in coastal areas is measured using the benthic quality index, BQI (Rosenberg et al. 2004, Blomqvist et al. 2006, Leonardsson et al. 2009).

The BQI is developed to detect effects associated with eutrophication, organic enrichment and oxygen deficiencies. It is involves components of tolerance or sensitivity to these disturbances (relative abundances of tolerant and sensitive taxa), species richness and abundance, all combined into one index. The foundation for the assessment criteria using this index is the determination of the G-M boundary. This boundary was determined for each of the coastal water-types using data from the national monitoring program (located in supposedly undisturbed, "reference sites") and from other sources of data not affected by local pressures or sources of point-pollution. These data included samples from the 1950's, 1960's and 1970's, but the majority of data was from the 1980's to the early 2000's (Blomqvist et al. 2006). The type-specific G-M boundaries were determined using a combination of approaches. One important approach has been based on the basic assumption that the status of benthic fauna in national reference sites is "high" or "good". Acknowledging natural temporal fluctuations and spatial variability, the G-M boundary was then defined as the lower 20th percentile of the distribution of samples, i.e. BQI values lower than this limit were considered below "Good" status while values above were considered not to deviate from natural fluctuations. The G-M boundary was also compared and validated against observations of breakpoints in pressure-response relationships and in some instances when data were absent, it was determined by expert opinion. Note that this procedure does not involve definitions of any "pristine" reference condition, but rather a range of conditions which includes likely values in a fluctuation environment. Furthermore, the Swedish assessment criterion (NFS 2008:1) is in fact formulated in terms of absolute BQI-values and not as environmental quality ratios (EQRs). Nevertheless, for comparative and intercalibration purposes, boundaries are also given as EQRs where the type-specific maximum values of BQI are used as "reference values". These reference values also serve the purpose to indicate the direction of any future restoration efforts.

The approach for determining reference values of phytoplankton biomass and chlorophyll a (Chl a) was based on historical data on Secchi depth and nutrients in combination with various contemporary, empirical relationships among Secchi depth, nutrients, salinity and phytoplankton parameters (Larsson et al. 2006). Because of varying availability of data and because of varying ecological conditions in different coastal areas, methods differed among the Gulf of Bothnia, the Baltic Proper and the Kattegat-Skagerrak. In the Gulf of Bothnia, historical data on Secchi depth in offshore areas from the beginning of the 1900's and empirical relationships between coastal and offshore Secchi depths were used to model reference values of Secchi depth in coastal areas. These were in turn used to define reference values for Chl a using empirical relationships between Secchi and Chl a. Reference values for biovolume were, however, defined on the basis of a combination of relationships between Chl a and biovolume and reference values in the adjacent northern Baltic Proper. Reference values for different coastal areas in the Baltic Proper are based on a "salinity gradient approach". This approach uses reference values in offshore areas, observations of salinity and a simple mixing model to calculate reference values in coastal waterbodies. In the Kattegat-Skagerrak, reference values for Secchi depth and nutrients have been defined in offshore areas, accounting for differences in salinity, reference values for coastal types have been calculated for Secchi depth and nutrients and finally reference values for Chl a and biovolume was estimated from empirical relationships. In summary, methods for calculating reference conditions of coastal phytoplankton are necessarily very complex and diverse.

The current Swedish indicator for macrophytes (macroalgae and angiosperms) in coastal areas is the Multi Species Maximum Depth Index (MSMDI; Kautsky et al. 2006, Blomqvist et al. 2012). No formal reference values for MSMDI are given, but "reference depths" representing observed maximum depth in data from the 1940's until the early 2000's or in some cases expert opinions thereof are given for individual species and water types. Depths close to these "reference depths" for an individual species are given the score 5 (the difference between the "reference depth" and the depth for which the score 5 is given varies among species). If the species is found at shallower depths, observations are given scores between 4 and 2. The score 1 is given if the species is absent due to anthropogenic causes. The limits defining these scores are species- and type-specific and have been defined using expert judgment combined with estimates of Secchi depth. The ratio between the average score of at least three species and the maximum possible score (i.e. 5) constitutes the MSMDI and has the unit EQR (note that the same result is achieved by dividing all scores by 5 and then taking the average [Blomqvist et al. 2012]). Class boundaries are equidistant, i.e. H-G=0.8, G-M=0.6, M-P=0.4 and P-B=0.2, but note that these are not linearly related to changes in the depth range of individual species. Also note that maximum MSMDI=1, but this does not necessarily mean that all the species are found at their respective "reference depths". Furthermore, recent compilations of existing data suggest that the values used as "reference depths" are in many cases underestimates of historical maximum depths (Blomqvist pers. comm.).

TABLE 5.6
Biological quality elements (BQEs) and indices used in the classification of coastal and transitional waters. Methods or determining reference condition: S= spatial, M=modeling/predictive, H=historical, P=paleoecology and E=expert judgment.

BQE	Index	Index type	Pressure	Reference condition	
				Method	**Short description**
Benthic invertebrates	*Benthic Quality Index, BQI*	multi-metric	general degradation, e.g. oxygen deficiency	S, E	Data from (1) national trend monitoring sites (=long-term 20 percentile) and (2) pressure-response studies (breakpoints) used to define type specific G-M boundaries. Maximum observed BQI used as reference condition but because boundaries are defined in absolute terms rather than as deviations from the reference, the latter indicates direction rather than indication of pristine condition.
Phytoplankton	*Chlorophyll a (Chl a)*	single	eutrophication	H, M, E	Methods vary among coastal seas. In summary, contemporary relationships among Secchi depth, nutrients, salinity and chlorophyll a have been used with historical data on Secchi depth in offshore areas to model reference levels of chlorophyll. Type-specific reference values are given but corrections for salinity are made in four types based on observed salinities
	Biovolume	single	eutrophication	H, M, E	As above but occasionally reference conditions for biovolume have been based on empirical relations to chlorophyll a.
Macroalgae & Angiosperms	*Multi Species Maximum Depth Index, MSMDI*	single	eutrophication	H, E	No formal reference value is given to the MSMDI, but historical estimates and expert judgment on maximum depth limits of selected species represent “species and type specific reference depths”. For each species this depth (or close to) is given the score 5, shallower depths are given scores of 4-2 and absence (due to anthropogenic causes) is given a score of 1. The MSMDI is calculated as the ratio between the average score of 3-9 species and the maximum scores (=5).

5.4 Compatibility to the other quality elements and directives

The biological quality elements (BQEs) of the WFD are central to the assessment of ecological status in all Swedish and European surface waters. As such, their assessment criteria and classification schemes also provide the foundation for many status assessments and environmental targets internationally, nationally and locally (e.g. assessments according to HELCOM, OSPAR and the Swedish National Environmental Objectives). In these broad contexts the BQEs, their associated reference conditions and class boundaries are used in combination with other assessment criteria, which may or may not be entirely consistent with those of the BQEs. In order to develop more consistency among status assessments, it is important to document differences and similarities of principles behind setting reference conditions and class boundaries. It is also worth noting that any developments in the criteria for the coastal BQEs will have direct consequences for the assessment of achieving good environmental status according to the Marine Strategy Framework Directive (MSFD) (European Commission 2008b). This is because the Swedish implementation of the MSFD uses the G-M boundary as the boundary for good environmental status when applying the same BQEs (GES; see below).

5.4.1 Non-biological quality elements of the WFD

The WFD lists several non-biological quality elements which are to be used as support for status classifications in coastal and inland waters. The Swedish implementation of the WFD, with respect to these factors, include definitions of reference conditions and class-boundaries of water transparency, nutrients (various forms of N and P), acidity, oxygen, hydromorphological elements and pollutants. A comprehensive review of the principles behind the setting of reference conditions and class boundaries for non-biological quality elements is beyond the scope of this report, but perusal of the methods used show a great diversity in approaches (SEPA 2007). For many of the quality elements, reference values are not mentioned and in some instances (e.g. oxygen conditions in coastal areas) the term “reference condition” appears to be used to denote the boundary between “good” and “high” status.

Despite the fact that the main aim of assessments of non-biological quality elements is to provide support for assessments using BQEs, classification of these elements is often very important. In particular, during the first reporting phase of the WFD many assessments of ecological status of coastal and inland water bodies were made using expert judgment, albeit often in conjunction with BQE indicator quantification. Moreover, in some instances classification of ecological status was made using only non-biological quality elements (e.g. Rohlf 2010). Thus, the impacts of these quality elements may in fact have a greater influence on the implementation of the WFD and its classifications than expected from the formulation of the directive. Any discrepancies in relation to the BQEs in terms of principles for setting of reference values and class boundaries are therefore unfortunate.

5.4.2 Marine strategy framework directive (MSFD)

The overall goal of the MSFD is to achieve 'good environmental status' (GES) in the marine waters of EU Member States (MS). Both the WFD and MSFD focus on integrated catchment management and ecosystem-based approaches (Borja et al. 2010). The directives share a number of basic elements but there are also significant differences in terms of how class boundaries and reference conditions are defined, the types of indicators used, and the integrated assessment.

Firstly, the normative definition of good ecological status in the WFD refers to a slight deviation from undisturbed conditions, i.e. '*The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions*'. The definition of GES in the MSFD, on the other hand, refers to a condition associated with sustainable use, i.e. '*good environmental status means the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable…*'. Thus, the underlying principles behind definitions of "good ecological status" in the WFD and GES in the MSFD are different. The former relates to concepts of "naturalness" while the latter is based on long-term sustainable use of marine areas.

Second, there are differences in the number and types of indicators among directives. Similar to the WFD, the MSFD assessment of status should be based on a set of indicators, but while the WFD assessment of ecological status focuses on the quality of a limited number of biological quality elements (BQEs) which all have to achieve at least good status, the MSFD uses a suite of 11 descriptors (see Box 1) that covers the state of as well as impacts and pressures on the ecosystem. The MSFD descriptors express GES at an overarching level and are associated with a set of 29 criteria and 54 proposed indicators (2010/477/EU). The specific indicators to be used to assess whether GES is achieved are to be defined by the MS and the first set of indicators was reported to the EU commission in October 2012.

Twenty-two of the MSFD criteria are related to the state of or impact on biological components and the indicators represent a much wider array of biological characteristics and functional groups than the WFD e.g. health of mammals, distribution of birds, and the extent of habitats. Since the MSFD demands cooperation between MSs sharing the same marine waters, cooperation in the development of indicators takes place within the Regional Seas Conventions, for Swedish waters that means HELCOM and OSPAR.

Third, the MSFD requires that MSs define a set of environmental targets so that progress towards achieving GES can be assessed. However, while the WFD defines five status classes, with "good ecological status" as the minimum level of achievement, the MSFD only requires the distinction between GES and sub-GES. Furthermore, the MSFD guidelines for setting environmental targets refers to using 'where appropriate, specification of reference points (target and limit reference points)' (MSFD; Annex IV) but there is no requirement in the directive to define reference conditions. However, guidelines from HELCOM and OSPAR propose the use of reference conditions as the starting point for defining targets when historical data or reference sites are available (HELCOM 2012, OSPAR 2012). It should also be noted that the proposed MSFD indicators include aspects of the health of populations. Targets for indicators related e.g. reproductive capacity are likely to be based on limit values associated with critical deterioration of the status of a population. In addition, the MSFD does not rule out the use of trend-based targets, i.e. to express GES as a direction of change. Thus, the MSFD accepts new approaches to define the desirable state of the environment.

The first status assessment of the marine environment based on indicators and targets should be carried out by 2018 and the development of both indicators and targets is an ongoing process. In Sweden it has been decided that the WFD BQEs for the coastal environment should also be applied in the assessments of the marine environment. When this is the case the boundary between good and moderate status of the BQE will represent the boundary between GES and sub-GES.

Finally, another difference between the two directives is the process where the different descriptors of the MSFD or the biological quality elements of the WFD are combined to achieve an integrated assessment. To provide an integrated status assessment the WFD uses a rule (e.g. one-out all-out) that combines BQEs into a whole-system assessment. The MSFD does not prescribe how the descriptors, assessed by various indicators, should be combined when assessing the state of the marine environment. The development of integrated assessment rules can be expected to take place within working groups of the EU commission and the Regional Seas Conventions in the upcoming years.

BOX 1 The 11 descriptors of the Marine Strategy Framework Directive (2008/56/EC)

(1) Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions.

(2) Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems.

(3) Populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock.

(4) All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.

(5) Human-induced eutrophication is minimised, especially adverse effects thereof, such as losses in biodiversity, ecosystem degradation, harmful algae blooms and oxygen deficiency in bottom waters.

(6) Sea-floor integrity is at a level that ensures that the structure and functions of the ecosystems are safeguarded and benthic ecosystems, in particular, are not adversely affected.

(7) Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems.

(8) Concentrations of contaminants are at levels not giving rise to pollution effects.

(9) Contaminants in fish and other seafood for human consumption do not exceed levels established by Community legislation or other relevant standards.

(10) Properties and quantities of marine litter do not cause harm to the coastal and marine environment.

(11) Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment."

5.4.3 Habitats directive (HD)

The aim of council directive on the conservation of natural habitats and of wild fauna and flora (1992, "the Habitats directive") is to ensure that "favourable conservation status" is maintained for species and habitats in European countries. To achieve this Member States are to define "favourable reference values" for range (habitats/species), area (habitats) and populations (species). For example the favorable reference range is defined as:

> "Range within which all significant ecological variations of the habitat/species are included for a given biogeographical region and which is sufficiently large to allow the long term survival of the habitat/species; favourable reference value must be at least the range (in size and configuration) when the Directive came into force; if the range was insufficient to support a favourable status the reference for favourable

> range should take account of that and should be larger (in such a case information on historic distribution may be found useful when defining the favourable reference range); 'best expert judgement' may be used to define it in absence of other data." (EC 2005b).

Thus the "favourable reference value" of the HD does not refer to a reference value *sensu* WFD but rather a boundary of what is required to achieve favorable conservation status. This concept is thus more related to the G-M boundary in the WFD. However, the HD also defines a second concept, "natural range":

> "The natural range describes roughly the spatial limits within which the habitat or species occurs. It is not identical to the precise localities or territory where a habitat, species or sub-species permanently occurs. Such actual localities or territories might for many habitats and species be patchy or disjointed (i.e. habitats and species might not occur evenly spread) within their natural range. ... Natural range as defined here is not static but dynamic: it can decrease and expand. Natural range can also be in an unfavourable condition for a habitat or a species i.e. it might be insufficient to allow for the long-term existence of that habitat or species. … "

This concept is clearly more related to the "reference value" according to the WFD. As such it has the same shortcomings, often requiring historical records and/or expert judgment to be defined in a quantitative way.

In the practical application of the HD into Swedish status assessment, monitoring and action plans, focus has been on developing indicators and routines for monitoring and testing against "favourable reference values" at biogeographic and local levels (Haglund 2010). One notable feature here is the development of common routines for handling uncertainty in relation to conservation targets. Furthermore, specific manuals have been developed for defined Natura 2000-habitats in aquatic environments (Bergengren 2010a, Bergengren 2010b, Dahlgren et al. 2011).

Despite the slight mismatch of the reference concept between the WFD and the HD, there are important links. One particularly important aspect is that G-M boundaries defined in the WFD are sometimes recommended as conservation targets in assessment of status in protected areas, such as Natura 2000- areas (i.e. Dahlgren et al. 2011). Thus, development of new assessment criteria for the WFD will very likely also influence the implementation of the HD in Sweden.

6 Methods used to set class boundaries

6.1 Definitions

Approaches commonly used for setting class boundaries were recently summarized by Schmedtje et al. (2009) and the ISO (International Organization for Standardization)/TC (Technical Committee) 147/SC (Subcommittee) 5 workgroup (ISO/CD 8689-1). Here, the reference community was defined as a "*biological community present at a site when only natural conditions are present and man-made impacts are absent or not sufficient to influence the biology*". Evaluation of human-induced impacts on biological assemblages was made using data from impacted sites (observed data) and pre-defined data from an undisturbed community (the reference community). The difference is expressed as an Ecological Quality Ratio (EQR), or the observed value divided by the reference value; values range from 1 (high status) to 0 (bad status). If reference sites are not available, the ISO document recommends using other approaches such as modeling, historic data or paleolimnological studies to provide the range of variability for the reference biological data. Considering the range of natural variability (in each habitat type if typology is used), the lower end of the range of reference sites provides the anchor for the boundary of EQR between High and Good status. Once the reference condition has been obtained, the next step is to describe the biological response due to specific pressures, e.g. elevated nutrients or hydromorphological alteration, based on a conceptual model. Here, conceptual models should include normative definitions or an ecological description of the biological community of at least high, good and moderate status (Table 6.1).

TABLE 6.1
Conceptual model describing ecological status for High, Good and Moderate status using benthic macroinvertebrates in rivers. Taken from Annex 5, European Commission, (2000).

Classes	Ecological descriptors
High status	The taxonomic composition and abundance correspond totally or nearly totally to the undisturbed conditions. The ratio of disturbance sensitive taxa to insensitive taxa shows no signs of alteration from undisturbed levels. The level of diversity of invertebrate taxa shows no sign of alteration from undisturbed levels.
Good status	There are slight changes in the composition and abundance of invertebrate taxa compared to the type-specific communities. The ratio of disturbance sensitive taxa to insensitive taxa shows slight signs of alteration from type-specific levels. The level of diversity of invertebrate taxa shows slight signs of alteration from type-specific levels.
Moderate status	The composition and abundance of invertebrate taxa differ moderately from the type-specific conditions. Major taxonomic groups of the type-specific community are absent. The ratio of disturbance sensitive to insensitive taxa, and the level of diversity, are substantially lower than the type-specific level and significantly lower than for good status.

Scatter plots are used to study the relationship between the response and putative pressure variable. If none of the response variables shows a relationship with impact, then setting class boundaries is not possible. In this case, use of another response variable should be considered, as well as collection of more data for the response and pressure variables. Other factors to consider are the importance of other pressures affecting the response (multiple stressor situations) and partitioning natural variability, e.g. using typology-based approaches.

If the response variable shows a relationship with the pressure variable, the boundary between high and good status is established using the distribution of high status sites (e.g. the lower 10th-percentile). If distinct discontinuities are evident in the response-pressure plot (Figure 6.1 left panel) decide if the breakpoint(s) correspond to class boundaries (or centers) and to the biological functioning as described in the conceptual model, and specify how errors in the estimate of class boundaries or class centers are considered in setting class boundaries. Figure 6.2 shows how the use of a nonlinear relationship can be used to establish boundary between good and moderate status.

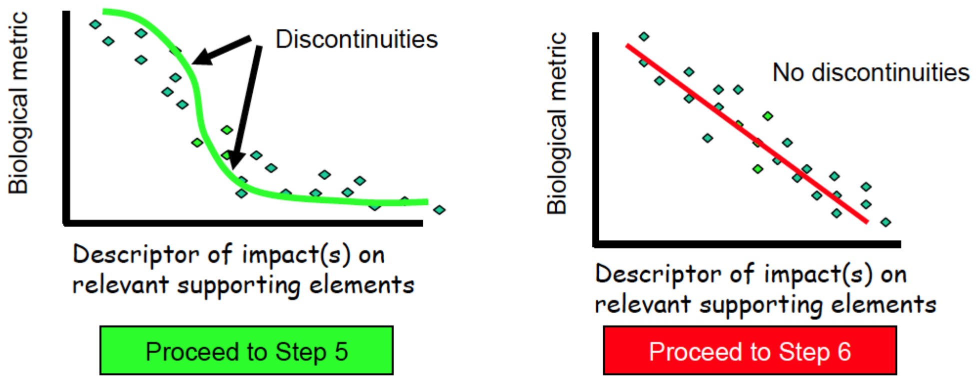

FIGURE 6.1
Non linear (left panel) and linear (right panel) plots of biological response versus pressure variables. Taken from Schmedtje et al. (2009).

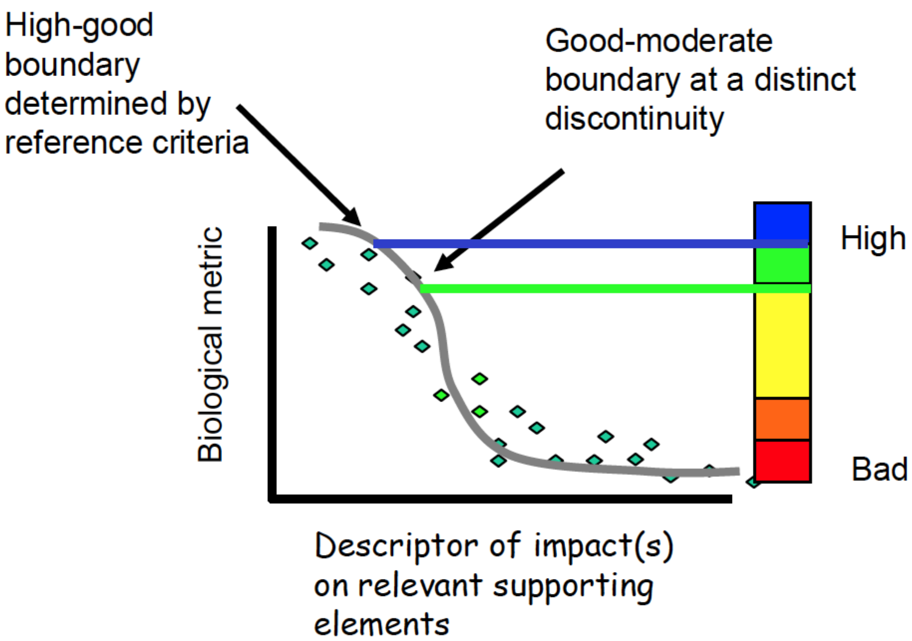

FIGURE 6.2
Example of using a breakpoint to establish the boundary between good and moderate status. Taken from Schmedtje et al. (2009).

When no distinct discontinuities are evident when the response variable is plotted against the pressure variable, plots of paired response variables (e.g. variables showing different functions of the biological community) can be used to assess the pressure-response relationship. If ecologically relevant interactions exist between the paired metrics, this relationship can be used to establish the class boundary as described above (Figure 6.3).

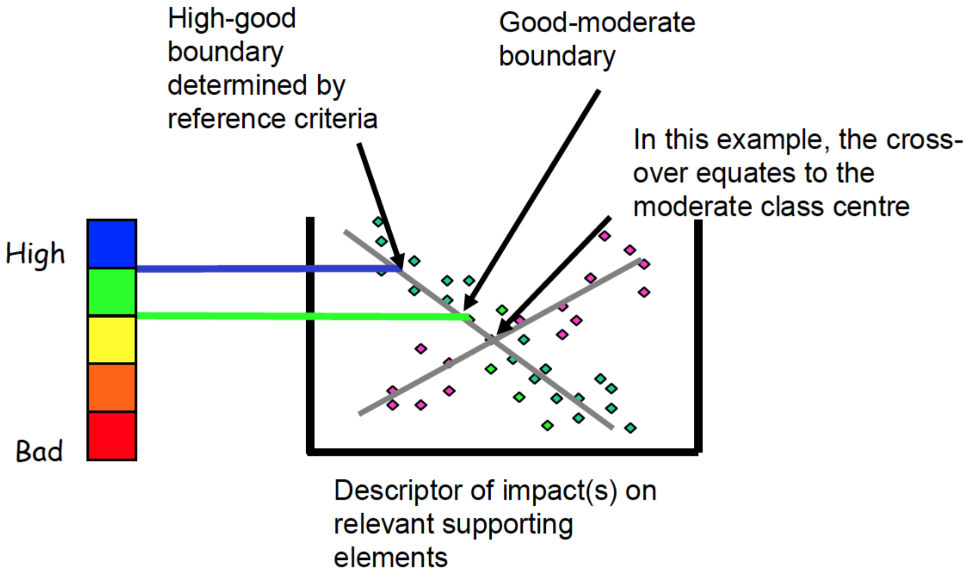

FIGURE 6.3
Example of setting a class boundary using paired biological response variables. Taken from Schmedtje et al. (2009).

If no distinct discontinuities are evident in the response-pressure plot then class boundaries can be ascertained by dividing the distance from the high – good boundary and lower anchor (minimum value possible) into four equidistant (width) classes (Figure 6.4). Determine if the classes established correspond to the conceptual model (e.g. changes in taxonomic composition) and, if they do not, revise the class boundaries accordingly.

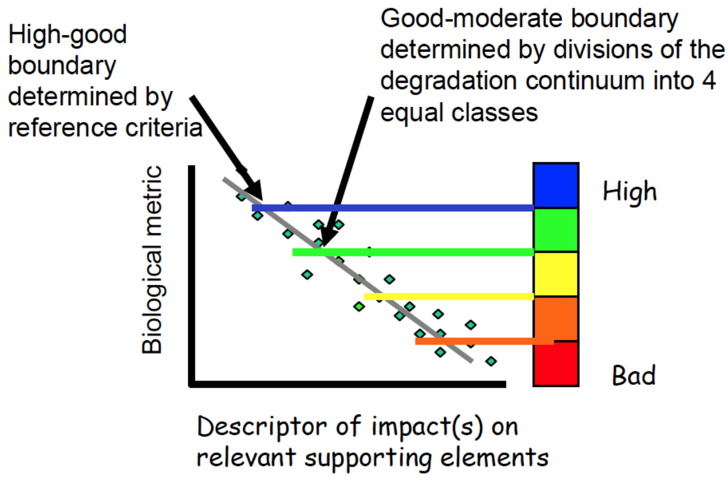

FIGURE 6.4
Example of setting class boundaries using equidistance. Taken from Schmedtje et al. (2009).

6.2 Methods used in Sweden to establish class boundaries

6.2.1 Inland surface waters

A number of approaches were used to establish class boundaries of BQEs in lakes and streams. Minimal disturbed sites were commonly used to set H-G boundaries; however, other methods such the use of stress sensitive/tolerant taxa (macrophytes) and modeling (fish) have also been used. Regardless of the approach, class boundaries were checked against the normative definitions in the WFD, in particular G-M boundaries.

Fish. The development of the current multi-metric fish indices VIX and EQR8 followed procedures used for developing a European Fish Index (EFI, Pont et al. 2004). Fish community metrics were assumed to be functions of continuous natural environmental factors, and to respond to a gradient of mixed anthropogenic pressures. Both high and good status sites were used for modeling site-specific reference-values, in order to increase the number of sites in the calibration data sets. Observed metric values were first expressed as standardized residuals (Z-values) of multiple regression models, and in the next step transformed to probabilities (P-values). The multi-metric indices were taken as the mean of the P-values of individual metrics. In this way both VIX and EQR8 got theoretical values between 0 and 1, and they were therefore not further transformed into EQR values, *sensu stricto* (e.g. Van de Bund & Solimni 2007). Class boundaries of VIX and EQR8, respectively, were also set according to Pont et al. (2004). The good-moderate boundary was set at the index value with equal probability of misclassification of sites predefined as ‘reference’ (high + good) or ‘disturbed’ (moderate or worse). The high-good boundary corresponds to less than 5 % probability of classifying a ‘reference’ site as ‘disturbed’, and the poor-bad boundary was set at less than 10 % probability for misclassifying a ‘disturbed’ site as ‘reference’ (Holmgren et al. 2007, Beier et al. 2007). Finally, the moderate-poor boundary was set in the middle of good-moderate and poor-bad boundaries.

Benthic Invertebrates. Scatter plots of EQR values for benthic invertebrate metrics of lakes and streams and pressure gradients revealed no clear breakpoints. Therefore, statistical approaches were used in setting class boundaries. EQR values for the reference population were calculated as the observed value divided by the reference (median value for the respective metric established by typology) value. The boundary between high and good ecological quality was set as the 25th-percentile of the reference distribution of EQRs (i.e. putative perturbed sites were not used in the step). The boundary between H-G ecological quality was set for each of the individual metrics in each of the three ecoregions (Central Plains, the Fenno-Scandian Shield and the Borealic Uplands). The remaining three class boundaries (G-M, M-P, P-B) were set by dividing the interval between the H-G value and the minimum value of each metric into equidistance groups. For acidification, many studies have shown a marked decrease in macroivertebrate diversity at pH of 5.6 (e.g. Johnson et al. 2007) Therefore, the intercept between pH 5.6 and the regression line of EQR values of MILA and MISA was used to set the G-M boundary. The remaining boundaries were set using the equidistance method.

Phytoplankton. EQR values for the reference population (screened using a pressure filter) were calculated as the observed value divided by the reference value. The boundary between high and good ecological quality was set as a percentile of the reference distribution of EQRs for each of the five ecosystem types (based on three ecoregions and water color, see above). The remaining boundaries were set using expert judgment and WFD normative definitions describing ecological classes.

Benthic diatoms. The reference value was established as the median of all IPS values of the streams passing the national filter for reference streams (Tot-P < 10 μg/l or no eutrophication (areal specific loss of Tot-P = class 1; in case of missing data for calculation of areal specific loss: Tot-P < 20 μg/l and color > 100 mg Pt/l), no acidification, land use (< 20 % farming, < 0.1 % urban area). Most focus was then placed on the good/moderate boundary, which was set as the ‘crossover’ between sensitive and tolerant taxa (Pollard and van de Bund, 2005), i.e. the IPS value where the nutrient tolerant and pollution tolerant species exceed a relative abundance of ca. 30 % (and the amount of sensitive species falls below ca. 30 %).

Diatom metrics tend to show gradual changes as the level of nutrient/organic pressure increases, with no distinct thresholds that can be used to set class boundaries. Ecologically, the point of ‘crossover’ means the point at which the taxa which form the ‘association’ characteristic of a site in the absence of pressure become subordinate to taxa which are favored by a pressure (nutrients and/or pollution, in this case). The high/good boundary was then set at the point where the sensitive and indifferent taxa still were in the majority, and the moderate/poor boundary was set at the point where the indifferent taxa still dominated over the tolerant ones. The poor/bad boundary was set at the point where the stress tolerant taxa predominate. For lakes, the stream classification system was used after assessing if the calculated IPS values fell into the same level of background nutrient values as for streams, which was the case. Regarding acidification, there is currently no biological index which can used to separate natural acid from anthropogenic acidified waters, therefore, acidity indices should not be used as a measure of acidification. Instead, we propose the use of chemical methods when biological indices indicate low pH. For diatoms, the ACID index is linearly correlated with pH. Considering future work, boundaries between acidity groups could be using relationships between different taxa. For example, the *Achnanthidium minutissimum* group is only dominating at pH > 5.9, whereas the genus *Eunotia* dominates only at pH < 5.5. Other clear thresholds with changing diatom groups are found between pH 6.5 and 7.3 (Kahlert 2005).

Macrophytes. Reference values and class boundaries for macrophytes using three ecosystems types: 1) north of Limes Norrlandicus, above the highest coastline, 2) north of Limes Norrlandicus, below the highest coastline, and south of Limes Norrlandicus. Each species was given an indicator values based on species-specific preference for total phosphorus concentration and indicator weights (niche breath). Class boundaries for H-G and G-M boundaries were set using the sensitivity/tolerance of individual taxa to nutrient concentration; other class boundaries were established by equidistance. In addition, for sites classified as being close to a boundary, indicator species are used in the final classification.

6.2.2 Coastal waters

As discussed above, approaches used for defining reference conditions vary considerably among coastal BQEs (section 5.3.2). These differences are to some extent also reflected in the way class boundaries are defined for the different BQEs. One common feature, however, is the focus on the boundary between “good” and “moderate”, the G-M boundary. This is of course due to the important implications for mitigation measures when the status is characterized as below “good”. A summary of how the G-M boundary in terms of ecological quality ratios (EQR) varies among coastal areas and quality elements reveal that there are substantial differences (Figure 6.5). For benthic fauna the difference is quite large among coastal areas with smaller EQRs tolerated in the Baltic Proper and Gulf of Bothnia. For macrophytes, the G-M boundary is set to 0.4 in all areas, while the phytoplankton indicators vary between 0.35-0.55.

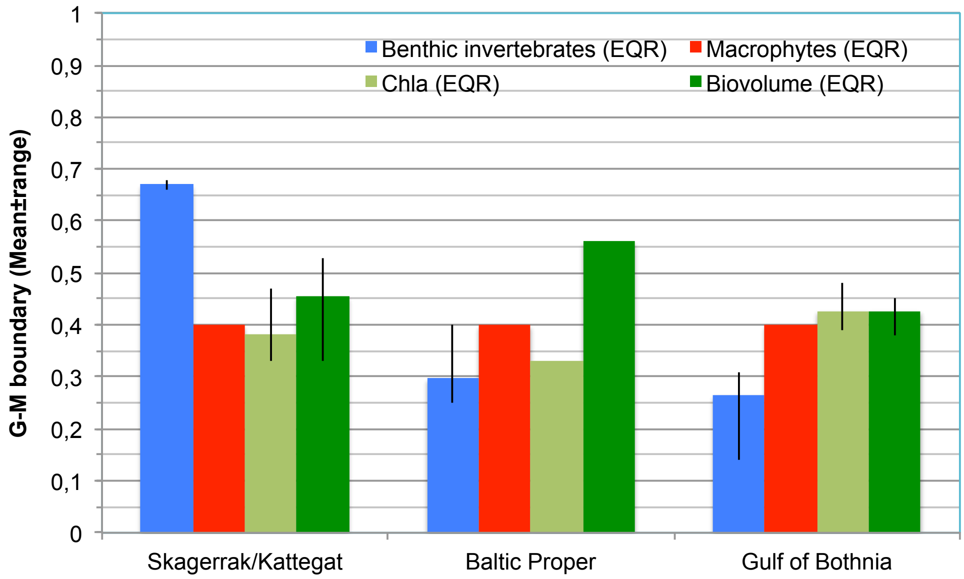

FIGURE 6.5
Mean values for G-M boundaries in terms of ecological quality ratios (EQR) in different coastal areas. Error bars represent maximum and minimum values of different types in coastal areas. Note that assessments of benthic invertebrates are made based on BQI (and not EQR values) but that EQR values are given for comparison (SEPA 2007).

Because the EQR scales are not linearly related to the units in which these indicators are measured, it might be useful to assess these boundaries in terms of relative change in the ecological units measured (Figure 6.6). As described earlier, the starting-point for definitions of class boundaries for benthic fauna was to define the G-M boundary as the 20th percentile of observations from minimally disturbed sites. In practice this means that in the Skagerrak-Kattegat area the G-M boundary is on average set at a 50% decrease compared to the maximum value ($\left[100\times\left(\frac{BQI_{G-M}-BQI_{max}}{BQI_{max}}\right)\right]$, note that the maximum value is not strictly equivalent to a reference value). In the Baltic proper and the Gulf of Bothnia the decrease was approximately 75% (Figure 6.6). By comparing this boundary to breakpoints in pressure gradients, this approach was generally considered appropriate. The remaining boundaries were set using generic rules. The H-G boundary was defined in a way that 2/3 of the interval between the G-M boundary and the maximum BQI-value for a particular type. Similarly, the Moderate - Poor boundary was defined in a way that "moderate" included 2/3 of the interval below the G-M boundary.

For macroalgae and angiosperms, the class boundaries were set using expert judgment partly based on analyses of observed relationships between depth distributions of individual species and Secchi depth. The approach to assign scores of 1-5 to species depending on the observed depth range (or absence due to human impacts) has the effect of standardizing among species and types, and in accordance with this the class boundaries on an EQR scale is arbitrary. In terms of changes in observed maximum depth for individual species, which is the basis for scores in the MSMDI, the G-M boundary differs slightly among species and coastal areas. In the Skagerrak-Kattegat area, the G-M boundary represents a 43% decrease in depth range while for the Baltic Proper and the Gulf of Bothnia the decrease is nearly 50% compared to the deepest observed specimens, the reference value. Thus, if the reference depth for a particular species in the Baltic is 10 m, it would not be considered below "good" until the deepest plant was found shallower than 5 m.

Finally, class boundaries for phytoplankton Chl a and biovolume are mainly set using relationships between phytoplankton, Secchi depth and nutrients. In the Baltic Proper the class boundaries are based on the generally accepted notion (e.g. HELCOM and national reports) that the Baltic Proper is eutrophied. Therefore, the G-M boundary is set at levels corresponding to levels of nutrients lower than those at the time when the criteria were developed. Similarly, in the Skagerrak-Kattegat area, class boundaries were set assuming that nutrient levels were increased due to human impacts and the G-M boundary is set using relationships between phytoplankton and nutrients. Remaining class boundaries were set at 50-75% increments, also accounting for the prevailing variability of plankton data. In summary, it can be concluded that the G-M boundaries differ slightly among coastal seas and that the average increase needed to cross the G-M boundary is 50-75% for Chl a and 75-130% for biovolume compared to the reference (Figure 6.6).

In conclusion, these analyses show that the G-M boundary is usually set in a way that allows for quite substantial deviations from reference values (note however that for some indicators these values are not strictly defined as reference values *sensu* WFD). Noting that the different indicators have different units and that there are differences among coastal areas, we can observe that the average difference between reference and the G-M boundary in different coastal areas on a relative scale varies among BQEs. For benthic invertebrates it is 50-75%, for macroalgae and angiosperms it is 40-50% and for phytoplankton 50-130%. These differences may be justified on ecological grounds or by differences in uncertainty among BQEs. Nevertheless, these results illustrate clear differences that may be important for the future development of coherent assessment criteria within WATERS. Note that this does not necessarily imply that there need to be a perfect match among BQEs and areas since there may be valid arguments for maintaining differences. However, for purposes of for example uncertainties and transparency of overall assessments these differences need to be addressed from an over-arching perspective.

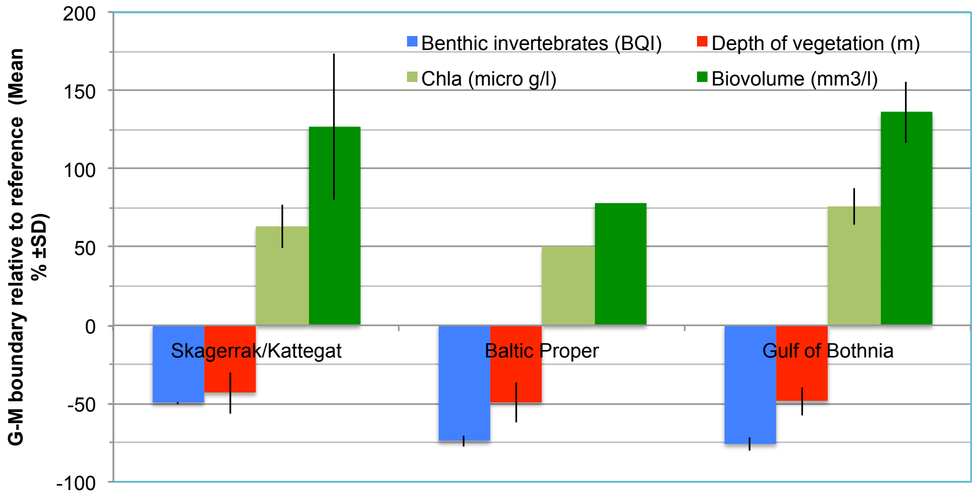

FIGURE 6.6
Mean percentage difference between reference and G-M boundary for individual BQEs in different coastal areas (see text for details about calculation). Note that deviations for phytoplankton, in contrast to other BQEs, are positive because poorer conditions are characterized by increased levels of biovolume and Chl a. Error bars represent standard deviation among water-types within areas. Data from Blomqvist et al. (2006), Kautsky et al. (2006) and Larsson et al. (2006).

7 Summary and conclusions

Effective management of aquatic resources requires knowledge of when a water body differs from the expected condition, and, for mitigation or rehabilitation, what has caused the deviation. Accordingly, accurate estimates of the reference condition and well defined class boundaries are pivotal aspects of many ecological assessments. From our review, it is clear that approaches used to define reference conditions differ between inland and coastal surface waters. Many of these differences are undoubtedly related to data availability, e.g. finding minimally disturbed sites is often easier for inland compared to marine systems, whilst other differences such as how class boundaries are established is likely more a result of poor coordination between working groups.

For inland surface waters, spatial approaches are commonly used to establish reference conditions in Sweden and elsewhere. Brucet et al. (in press) showed that 48% of 93 methods reviewed used near-natural reference sites combined with other methods (e.g. historical data, modeling, expert judgment) to establish reference conditions. Current classification schemes for lakes and streams are also to a large extent harmonized due to collaboration among working groups in developing WFD compliment methods (SEPA 2007). As a result of this harmonization, approaches for delineating reference conditions (typologies) are similar across most BQEs. Three main ecoregions are commonly used in partitioning natural variability and deriving reference conditions (Central Plains, the Fenno-Scandian Shield and the Borealic Uplands). The one exception being fish; reference conditions for EQR8 and VIX are established by regression models, with minimally disturbed sites used in model calibration. By contrast, the openness and connectivity of marine areas coupled with subtle changes in anthropogenic (nutrients) and natural (salinity) gradients result in difficulties in defining minimally disturbed areas that are representative of "ecosystem" types. Consequently, approaches for establishing reference conditions for coastal systems vary markedly among BQEs. Current approaches include the individual or combined use of minimally disturbed sites, historical data, modeling and expert judgment.

Similar to establishing reference conditions, methods for establishing H-G and G-M boundaries differed markedly both within and between inland and coastal systems. Related to the use of minimally disturbed sites in establishing reference conditions, classification schemes for inland waters often used distributions of high quality sites in setting H-G boundaries (fish, benthic macroinvertebrates, phytoplankton). For example, for benthic invertebrates the H-G boundary was set as the 25th-percentile of the reference distribution of EQRs. Other BQEs used modeling (fish) or relationships between sensitive and tolerant taxa (benthic diatoms, macrophytes) to set H-G boundaries. Given the need for mitigation and/or rehabilitation when sites are classified as being at or below moderate status, much focus has been placed on carefully defining the G-M boundary; also here methods varied among BQEs. If thresholds were evident, breakpoints were used in setting the G-M boundary. Other methods included modeling, use of sensitive/tolerant taxa or equidistant classes. Regardless of the method, for inland waters consideration was given to the normative definitions of the WFD. For setting class boundaries below the G-M boundary many approaches used generic methods such as using equidistance.

WATERS will address many of the issues related to defining reference conditions and setting class boundaries. As methods for inland waters are to large extent harmonized, future efforts will be placed on refining the pressure filter approach. This will be done by improving, where necessary, threshold values for some criteria (e.g. nutrient thresholds) and adding new criteria if data are readily available (e.g. for hydromorphological alteration, effects of invasive species). For coastal waters, effort will be on refining and harmonizing definitions of reference conditions, e.g. using historical data and modeling. In addition, there is a growing concern that typology-based systems are not as robust as site-specific approaches for establishing reference conditions and detecting ecological change. WATERS will test the usefulness of typology- versus modeling-based approaches. Finally, uncertainties are an inherent property of all environmental assessments. Hence, WATERS will identify how different forms of uncertainty (e.g. method based, natural variability) affect uncertainties in ecological classification. WATERS will attempt to harmonize methods regarding defining reference conditions and the setting of classification boundaries.

8 Acknowledgements

Many thanks to the work package leaders and members of WATERS for their contributions to this deliverable.

9 References

Andersen, J.H., Conley, D.J. and Hedal, S. (2003) Palaeoecology, reference conditions and classification of ecological status: the EU Water Framework Directive in practice. Marine Pollution Bulletin, 49, 283-290.

Andrén, E., Clarke, A., Telford, R.J., Weckström, K., Vilbaste, S., Aigars, J., Conley, D.J., Johnsen, T., Juggins, S. and Korhola, A. (2007) Defining reference conditions for coastal areas in the Baltic Sea for the Water Framework Directive (DEFINE). TemaNord 2007:583, available at www.norden.org/publications.

Anonymous, (1999) Bedömningsgrunder för miljökvalitet – Sjöar och vattendrag. Swedish Environmental Protection Agency, report 4913. (In Swedish)

Anonymous, (2003) Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance on establishing reference conditions and ecological status class boundaries for inland surface waters. Available at: http://forum.europa.eu.int/Public/irc/env/wfd/library

Beier, U., Degerman, E., Sers, B., Bergquist, B. and Dahlberg, M. (2007) Environmental quality criteria to determine the status of fish in running waters –development and application of VIX (In Swedish with English summary). Fiskeriverket Informerar (Finfo) 2007: 5, 59 p.

Bergengren, J. (2010) Manual för uppföljning av vattendrag i skyddade områden. http://www.naturvardsverket.se/upload/04_arbete_med_naturvard/Skydd_och_skotsel_vardefull_natur/Uppfoljning/5_UFmanual_Vattendrag_faststalld20100503.pdf

Bergengren, J. (2010) Manual för uppföljning av sjöar i skyddade områden. http://www.naturvardsverket.se/upload/04_arbete_med_naturvard/Skydd_och_skotsel_vardefull_natur/Uppfoljning/4_UFmanual_Sjoar_faststalld20100503.pdf

Bishop, K., Beven K., Destouni, G., Abrahamsson, K., Andersson, L., Johnson, R.K., Rodhe, J. and Hjerdt, N. (2009) Nature as the "natural" goal for water management: A conversation. Ambio, 38, 209-214.

Blomqvist, M., Cederwall, H., Leonardsson, K. and Rosenberg, R. (2006) Bedömningsgrunder för kust och hav – Bentiska evertebrater [Assessment criteria for coastal and marine waters - Benthic invertebrates].

Blomqvist, M., Krause-Jensen, D., Olsson, P., Qvarfordt, S. and Wikström, S. A. (2012) Potential eutrophication indicators based on Swedish coastal macrophytes. Deliverable 3.2-1, WATERS Report no. 2012:2. Havsmiljöinstitutet, Sweden.

Borja, A., Elliott, M., Carstensen, J., Heiskanen, A.-S. and Bund, W. van de (2010) Marine management – Towards an integrated implementation of the European Marine Strategy Framework and the Water Framework Directives. Marine Pollution Bulletin, 60, 2175-2186.

Brucet, S., Poikane, S., Lyche-Solheim, A., and Birk, S. (2013) Biological assessment of European lakes: ecological rationale and human impacts. Freshwater Biology, in press.

Carletti, A. and Heiskanen, A. (eds.) (2009) Water Framework Directive intercalibration technical report. Part 3: Coastal and Transitional waters. JRC Scientific and Technical Reports. EUR 23838 EN/3.

Carstensen, J. and Henriksen, P. (2009) Phytoplankton biomass response to nitrogen inputs: a method for WFD boundary setting applied to Danish coastal waters. Hydrobiologia, 633, 137-149.

CIS Working Group 2.3 (2003) Principles and methods for establishing reference conditions and ecological status class boundaries for inland surface waters. Policy Summary to the REFCOND Guidance Document, version 2003-03-21, 15 p.

Clarke, A., Juggins, S. and Conley, D.J. (2003) A 150-year reconstruction of the history of coastal eutrophication in Roskilde Fjord, Denmark. Marine Pollution Bulletin 46, 1615-1629.

Clarke, A., Weckström, K., Conley, D.J., Anderson, N.J., Adser, F., Andrén, E., de Jonge, V.N., Ellegaard, M., Juggins, S., Kauppila, P., Korhola, A., Reuss, N., Telford, R.J. and Vaalgamaa, S. (2006) Long-Term Trends in Eutrophication and Nutrients in the Coastal Zone. Limnology and Oceanography, 51, 385-397.

Clarke, R. T. (2000) Uncertainty in estimates of river quality based on RIVPACS. In: Assessing the biological quality of freshwaters: RIVPACS and similar techniques. Wright, J. F., D. W. Sutcliffe & M. T. Furse (eds), Freshwater Biological Association, Ambleside: 39–54.

Clarke, R. T. (2011) Deliverable D6.1-3: WISERBUGS (WISER Bioassessment Uncertainty Guidance Software) tool for assessing confidence of WFD ecological status class. http://www.wiser.eu/download/D6.1-3.pdf

Dahlgren, T., Lindegarth, M., Kilnäs, M. and Hammersland, J. (2011) Manual för uppföljning av marina miljöer i skyddade områden. http://www.havochvatten.se/download/18.4ac364c413727adcb8d8000372/Manual+uppf+marina+milj%C3%B6er+i+skyddade+omr%C3%A5den.pdf

Davidson, T., Bennion, H., Sayer, C., Rose, N., David, C. and Wade, M. (2005) A 250 year comparison of historical, macrofossil and pollen records of aquatic plants in a shallow lake. Freshwater Biology, 50, 1671-1686.

Dixit, S.S., Smol, J.P., Charles, D.F., Hughes, R.M., Paulsen, S.G. and Collins, G.B. (1999) Assessing water quality changes in the lakes of the northeastern United States using sediment diatoms. Canadian Journal of Fisheries and Aquatic Sciences, 56, 131-152.

Council of European Communities (1992) COUNCIL DIRECTIVE 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

European Commission (2000) Directive 2000/60/EC of the European Parliament and of the Council – Establishing a framework for Community action in the field of water policy. 23 October 2000. European Commission, Brussels.

European Commission (2003a). Common implementation strategy for the Water Framework Directive (2000/60/EC) Rivers and Lakes - Typology, Reference Conditions, and Classification Systems. Luxembourg, Office for Official publications of the European Communities. http://circa.europa.eu/Public/irc /env/wfd/library

European Commission (2003b) Common implementation strategy for the water framework directive (2000/60/EC) Towards a guidance on establishment of the intercalibration network and on the process of the intercalibration exercise. Luxembourg, Office for Official publications of the European Communities. http://circa.europa.eu/Public/irc/env/wfd/library

European Commission (2005a) WFD CIS Guidance Document No 14. Guidance on the Intercalibration Process 2004-2006.

European Commission (2005b) Note to the Habitats Committee Assessment, monitoring and reporting of conservation status – Preparing the 2001-2007 report under Article 17 of the Habitats Directive (DocHab-04-03/03 rev.3)

European Commission (2008a) Commission Decision of 30 October 2008, establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise (notified under document number C(2008) 6016) (2008/915/EC). Official Journal of the European Union L332, 20–44.

European Commission (2008b) Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive)

European Commission (2010) WFD CIS Guidance Document No 14. Guidance on the Intercalibration Process 2008-2011.

Haglund (2010) Uppföljning av skyddade områden i SverigeRiktlinjer för uppföljning av friluftsliv, naturtyper och arter på områdesnivå. RAPPORT 6379

Hallstan, S., Johnson, R.K., Willén, E. and Grandin, U. (2012) Comparison of classification-the-modelling and species-by-species modeling for predicting lake phytoplankton assemblages. Ecological Modelling, 231: 11-19.

HELCOM (2012) Development of a set of core indicators: Interim report of the HELCOM CORESET project. PART A. Description of the selection process. Balt. Sea Environ. Proc. No. 129A.

Henrikson, L. and Medin, M. (1986) Biologisk bedömning av försurningspåverkan på Lelången tillflöden och grundområden. Aquaekologerna, rapport till länsstyrelsen i Älvsborgs län. (In Swedish).

Holmgren, K, A. Kinnerbäck, S. Pakkasmaa, Bergquist, B. and Beier, U. (2007) Assessment criteria for ecological status of fish in Swedish lakes – development and application of EQR8 (In Swedish with English summary). Fiskeriverket Informerar (Finfo) 2007: 3, 54 p.

Illies, J. (1978) Limnofauna Europaea. Gustav Fischer Verlag, Stuttgart, Germany.

ISO Committee Draft ISO/CD 8689-1. Water quality – Biological classification of fresh waters – Part 1: Guidance on a procedure for setting boundaries between biological classes using data from surveys of aquatic fauna and flora.

Johnson R.K. (1999) Regional representativeness of Swedish reference lakes. Environmental Management, 23: 115–124.

Johnson, R.K. (2000) RIVPACS and alternative statistical modelling techniques - accuracy and soundness of principles. p 323-332. In Assessing the biological quality of freshwaters. RIVPACS and other techniques. (J.F. Wright, D.W. Sutcliffe, and M.T. Furse eds.). Freshwater Biological Association, Ambleside, UK.

Johnson, R.K. (2000) Spatial congruence between ecoregions and littoral macroinvertebrate communities. Journal of the North American Benthological Society, 19, 475–486.

Johnson, R.K. (2003) Development of a prediction system for lake stony-bottom littoral macroinvertebrate communities. Archiv für Hydrobiologie, 158, 517-540.

Johnson, R.K., Goedkoop, W., Fölster, J. and Wilander, A. (2007) Relationships between macroinvertebrate assemblages of stony littoral habitats and water chemistry variables indicative of acid-stress. Water, Air and Soil Pollution, 7: 323-330.

Johnson, R.K. and Battarbee, R. (2006) Establishing reference conditions for European inland waters, Eurolimpacs (GOCE-CT-2003-505540) Deliverable 72, 53p.

Johnson, R.K., Wiederholm, T. and Rosenberg, D.M. (1993) Freshwater biomonitoring using individual organisms, populations and species assemblages of benthic macroinvertebrates. p 40-158. In Freshwater biomonitoring and benthic macroinvertebrates. (D.M. Rosenberg and V.H. Resh eds.). Chapman and Hall, New York.

Johnson, R.K. and Sandin, L. (2001) Development of a prediction and classification system for lake (littoral, SWEPACLLI) and stream (riffle SWEPACSRI) macroinvertebrate communities. Department of Environmental Assessment, Swedish University of Agricultural Sciences, Report 2001:23.

Johnson, R.K., Goedkoop, W., Willén, E. and Larsson, D. (2003) Typanpassning av referenssjöar och vattendrag: Kritisk granskning av biologiska kvalitetsfaktorer med bedömningsgrunder. Department of Environmental Assessment, Swedish University of Agricultural Sciences, Report 2003:17, 48pp. (In Swedish).

Kautsky, L., Wibjörn, C. and Kautsky, H. (2006) Förslag till och vidareutveckling av bedömningsgrunder för kust och hav enligt krav i ramdirektivet vatten – Makroalger och några gömfröiga vattenväxter [Proposal for and further development of assessment criteria for coastal and marine waters in accordance with requirements in the WFD - Macroalgae and some angiospermous aquatic flora].

Larsson, U., Hajdu, S., Walve, J., Andersson, A., Larsson, P., and Edler, L. (2006) Bedömningsgrunder för kust och hav – Växtplankton, näringsämnen, klorofyll och siktdjup [Assessment criteria for coastal and marine waters - Phytoplankton, nutrients, chlorophyll and transparency].

Leonardsson, K., Blomqvist, M. and Rosenberg, R. (2009) Theoretical and practical aspects on benthic quality assessment according to the EU-Water Framework Directive – examples from Swedish waters. Mar. Poll. Bull. 58: 1286–1296

Nijboer, R.C., Johnson, R.K., Verdonschot, P.F.M., Sommerhäuser, M. and Buffagni, A. (2004) Establishing reference conditions for European streams. Hydrobiologia, 516: 91-105.

Nowicki, P. (2003) The ecosystem approach. p. 13 – 24. In Conflicts between human activities and the conservation of biodiversity in agricultural landscapes, grasslands, forests, wetlands and uplands in Europe. (Young, J., Nowicki, P., Alard, D., Henle, K., Johnson, R.K., Matouch, S., Niemelä, J., and A. Watt, eds.). A report of the BIOFORUM project, August 2003. Centre for Ecology and Hydrology, Banchory, Scottland, UK.

OSPAR 2012. MSFD Advice Manual and Background Document on Biodiversity. A living document - Version 3.2 of 5 March 2012.

Pardo, I., Poikan,e S. and Bonne, W. (2011) JRC scientific and technical reports. Revision of the consistency in Reference Criteria application in the phase I of the European Intercalibration exercise. Luxembourg: Office for Official publications of the European Communities; 2011.

Pardo, I. Gómez-Rodríguez, C., Wasson J-G., Owen R., Bund W., Kelly, M., Bennett C., Birk, S., Buffagni, A., Erba, S., Mengin, N., Murray-Bligh, J. and G. Ofenböeck. (2012) The European reference condition concept: A scientific and technical approach to identify minimally-impacted river ecosystems. Science of the Total Environment, 420, 33-42.

Poikane, S. (ed.), (2009) Water Framework Directive intercalibration technical report. Part 2: lakes. JRC Scientific and Technical Reports. EUR 23838 EN/2.

Pont, D.B. Hugueny, Roset, N., and Rogers, C. (2004). Development, evaluation & implementation of a standardised fish-based assessment method for the ecological status of European rivers (FAME). Final Report WP6-8. Analysing reference conditions and assessing degraded conditions. Definition of an European fish based index (EFI) using the reference condition modelling approach. 59 p.

Reynoldson, T.B., Norris, R.H., Resh, V.H., Day, K.E. and Rosenberg, D.M. (1997) The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. Journal of North American Benthological Society, 16, 833–852.

Reynoldson, T.B. and Wright, J. (2000) The reference condition: problems and solutions. p 293-303. In Assessing the biological quality of freshwaters. RIVPACS and other techniques. (J.F. Wright, D.W. Sutcliffe, and M.T. Furse eds.). Freshwater Biological Association, Ambleside, UK.

Rohlf, C. (2010) Hur har de marina bedömningsgrunderna utfallit i statusbedömningen? En beskrivande nationell överblick. SWECO. (Report in Swedish commissioned by the SEPA)

Rosenberg R., Blomqvist M., Nilsson H.C., Cederwall H. and Dimming, A. (2004) Marine quality assessment by use of benthic species-abundance distributions: a proposed new protocol within the European Union Water Framework Directive. Mar. Poll. Bull. 49: 728–739

Sandin, L. and Johnson, R.K. (2000) Ecoregions and benthic macroinvertebrates in Swedish streams. Journal of the North American Benthological Society, 19, 462–474.

SEPA (2007) Status, potential and quality requirements for lakes, water courses, coastal and transitional waters: A handbook on how quality requirements in bodies of surface water can be determined and monitored. Swedish Environmental Protection Agency. 2007:4, 421 p.

Schmedtje, U., Birk, S., Poikane, S., Jepsen, N. and Bonne, W. (2009) Guidance on the Intercalibration Process 2008.2011 (version 5.0). 62 p.

Skriver, J., Friberg, N. and Kirkegaard, J. (2000) Biological Assessment of water course quality in Danmark: Introduction of the Danisk Stream Fauna Index (DSFI) as the official monitoring method. Verhandlungen der Internationalen Vereinigung für Theoretischen und Angewandte Limnologi 27: 1822–1830.

Stoddard, J.L., Larsen, P., Hawkins, C.P., Johnson, R.K. and Norris, R.H. (2006) Setting expectations for the ecological condition of running waters: the concept of reference condition. Ecological Applications, 16: 1267-1276.

Valinia, S., Hansen, H.-P., Futter, M.N., Bishop, K., Sriskandarajah, N. and Fölster, J. (2012) Problems with the reconciliation of good ecological stastus and public participation in the Water Framework Directive. Science of the Total Environment, 433, 482-490.

van de Bund, W. J., (2009). Water Framework Directive intercalibration technical report. Part 1: Rivers, JRC Scientific and Technical Reports. EUR 23838 EN/1.

van de Bund, W., and Solimni, A.G. (2007). Ecological quality ratios for ecological status assessment in inland and marine waters. REBECCA Deliverable 10. EUR 22722 EN, 23 p.

Wallin, M., Wiederholm, T. and Johnson, R.K. (2003) Guidance on establishing reference conditions and ecological status class boundaries for inland surface waters. Final version 7.0, 2003-03-05, 93 p.

Wright, J.F., Blackburn, J.H., Gunn, R.J.M., Furse, M.T., Armitage, P.D., Winder, J.M., Symes, K.L., and Moss, D.(1996) Macroinvertebrate frequency data for the RIVPACS III sites in Great Britain and their use in conservation evaluation. Aquatic Conervation: Marine and Freshwtaer Ecosystems, 6: 141-167.

Wright, J.F., Sutcliffe, D.W. and Furse, M.T. (2000) Assessing the biological quality of fresh waters: RIVPACS and other techniques. Freshwater Biological Association, Ambleside, UK.

Annex 1

Criteria for determining suitable reference sites (“reference filter”) in streams and lakes developed within the REFCOND project (http://circa.europa.eu/Public/irc/env/wfd/library?l=/framework_directive/guidance_documents/guidancesnos10sreference/_EN_1.0_&a=d).

1. Point source pollution

"Reference" threshold: < 0.4% of artificial land use in the catchment area. "Rejection" threshold: 0.8% of artificial area in the catchment. Between 0.4 and 0.8%, a validation with physico-chemical parameters at the site scale is necessary.

2. Diffuse source pollution

Intensive agriculture: < 20% of the catchment area as reference threshold. Rejection threshold: > 50% of intensive agriculture in the catchment. Between 20% and 50% of intensive agriculture, a validation with physico-chemical parameters at the site scale is strongly recommended.

3. Riparian zone vegetation

In agricultural landscape (intensive agriculture between 20% and 50%), intensive agriculture land cover < 10% of the reach. Riparian corridor land use > 90% semi natural or low intensity agricultural areas.

In non agricultural landscape (intensive agriculture < 20%): valley floor and riparian corridor occupied by semi natural or low intensity agricultural areas.

Artificial areas: < 10% of the reach.

The riparian zone of the site is entirely bordered by the type specific natural vegetation or semi-natural land cover, with the possible exception of access to the river site. Riparian vegetation zone continuity: uninterrupted or with few interruptions (access to the site).

The lateral connectivity between river and riparian corridor is maintained along the site.

No direct impact of cattle trampling.

4. Morphological alterations

Sediment transport: No dams which significantly modify the sediment regime (sediment retention) leading to morphological alterations, evidenced by signs of incision of the river bed (e.g. incision > 0.2 m * stream order, bare bed rock appearing…).

"Continuity" for fish should be related to the maintenance of river and stream continuity to facilitate movement of type specific species that should be present in reference state.

If this condition is not fulfilled and some migratory species have disappeared, these species should be added to the type -specific list of fish species.

Flow impedance: < 10% of the reach is affected by flow impedance, due to hydraulic effects of weirs, sluices, etc…

Channelisation: < 10% of the reach is affected by "hard works" (like modification of longitudinal and/or transverse profiles, narrow embankment, loss of lateral connectivity…).

Stabilisation: < 20% of the reach is affected by "soft works" (like bank protection on one side, distant dikes, bank maintenance, not affecting the longitudinal and/or transverse profile, and lateral connectivity globally maintained…).

If both types of works are combined (Annex1, lines 134 and 135) < 10% of the reach must be affected.

Siltation: reaches with anomalous siltation suspected, due to agricultural soil erosion, should be avoided (expert judgment).

Connection to groundwater: Total lateral and vertical connection to groundwater.

Substrate conditions: Correspond to related typology.

River profile and variation in width and depth: Correspond to related typology.

River continuity: At the reach scale, the continuity of the river is not disturbed by anthropogenic barriers and allows undisturbed migration of aquatic organisms (including resident fish populations).

River continuity: At the reach scale, the continuity of the river is not disturbed by anthropogenic barriers and allows free sediment transport. The site is not situated in a zone directly or indirectly impacted by a nearby artificial structure upstream or downstream. Lacking any instream structural modifications (weirs or dams) that affect the longitudinal and lateral connectivity, and natural movement of river bed, sediment load, water and biota (except for natural waterfalls). Only very small artificial constructions with very minor local effects can be accepted.

5. Water abstraction

No dams or water storage significantly altering the low flow regime; low flow alteration < 20% of the monthly minimum flow. No significant water abstraction in the reach. The cumulative effect of water regulation and abstraction at the basin and reach scales is < 20% of low flow discharge.

6. River flow regulation

No dams which significantly modify the natural hydrological flow regime (flow regulation): e.g. suppression of frequent floods (< 5 years) with anomalous development of vegetation in the channel, or low flow alteration. The total storage capacity of the reservoirs in the catchment is < 5% of the mean annual discharge at the site. No change of the natural (type specific) annual flow characteristics (seasonality of high and low flow). No by-passed section with residual flow (legal minimum discharge). No significant hydropower peaking effect (ratio Q hydropeaking/Q baseflow < 2). Absence of flow regulation (dam) on the reach itself.

7. Biological pressures

At the site scale, no invasive species, but alien species which are not at the invasive stage are tolerated. No intensive (commercial) fishery. No or very limited direct pollution by aquaculture plants. No biomanipulation.

8. Other pressures

No intensive use of reference sites for recreation purposes (no intensive camping, swimming, boating, etc.). No nearby intensive recreational use at the site scale: No regular bathing activities or motor boating. Occasional recreational uses (such as camping, swimming, boating, etc.) should lead to no or very minor impairment of the ecosystem.

Establishing reference conditions and setting class boundaries

Systems for classification are being increasingly used to gauge the effects and magnitude of human disturbance on aquatic ecosystems. For inland and coastal surface waters, the European Water Framework Directive prescribes definition of a reference condition representing a state of "naturalness" or "no, or only very minor, anthropogenic alterations" and a set of boundaries are defined to delimit status classes. In contrast, the European Marine Strategy Framework Directive does not require definition of reference conditions, but focus on achieving "Good environmental status" based on criteria related to "sustainable use". Our review showed that methods used to establish reference conditions varied for coastal waters, whereas approaches for inland surface waters were more harmonized. Likewise, methods for establishing class boundaries, in particular boundaries between good and moderate ecological status, differed both within systems (e.g. BQEs within lakes) and between systems (inland and coastal waters). WATERS will propose general frameworks for establishing reference conditions and class boundaries, with special focus on uncertainties associated with ecological classification.

WATERS is coordinated by:

Havsmiljöinstitutet
Swedish Institute for the Marine Environment

WATERS is financed by:

NATUR VÅRDS VERKET
SWEDISH ENVIRONMENTAL PROTECTION AGENCY