Status, potential and quality requirements for lakes, watercourses, coastal and transitional waters

A handbook on how quality requirements in bodies of surface water can be determined and monitored

Handbook for the application of Chapter 4, Sections 1-4 and 7 of the Swedish Water Quality Management Ordinance (2004:660)
and
the Swedish Environmental Protection Agency's Regulations (NFS 2008:1)
and
General Guidelines on the Classification of and Environ-mental Quality Standards for Surface Water

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Preface

The aim of this handbook is to provide guidance when determining quality requirements for bodies of surface water as part of the work with the European Water Framework Directive (WFD) (2000/60/EC) in accordance with the Swedish Water Quality Management Ordinance (2004:660) and the Swedish Environmental Protection Agency's Regulations (NFS 2008:1) and General Guidelines on the Classification of and Environmental Quality Standards for Surface Water. Furthermore, the handbook is intended as a “tool-box” for how quality requirements for surface water can be determined.

The handbook shall be used as an aid when performing the various assessments that must be made before the water authority establishes environmental quality standards. An important component of this is assessment criteria, which are a tool enabling the data collected on surface water to be scientifically interpreted and evaluated.

The handbook has been produced by an internal working group at the Swedish Environmental Protection Agency (Swedish EPA), with the help of scientists, experts at water authorities and county administrative boards, and in cooperation with other relevant agencies and organisations.

The handbook is one of several guidance tools, including other handbooks, fact sheets and online articles, published by the Swedish EPA concerning the application of the Water Quality Management Ordinance (WMO). The handbook is aimed first and foremost at water authorities and county administrative boards. Others, such as consultants etc., may also utilise the handbook when they wish to acquire knowledge about how quality requirements in bodies of surface water can be determined and monitored.

The Swedish EPA wishes to thank all the scientists and consultants who have been involved in the project and helped to develop the assessment criteria as well as those who have participated in advisory groups and helped in the design of the handbook.

Stockholm, 20 December 2007

Martin Eriksson
SWEDISH ENVIRONMENTAL PROTECTION AGENCY

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1 Introduction

1.1 Background

EU Member States have agreed to create a homogenous water management system by adopting Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy, aka the Water Framework Directive or WFD. All water bodies in Europe shall have achieved good ecological and chemical status by 2015. Water bodies with a less-than-acceptable status shall be remediated and programmes of measures and management plans shall be drawn up (Figure 1.1). To this end, binding quality requirements shall be developed that describe the quality the water must have. The Water Framework Directive therefore specifies the framework, the objective and the time-limit that applies to achieve the aim. It is then up to each Member State to adopt its own national laws and regulations needed to implement the provisions of the directive.

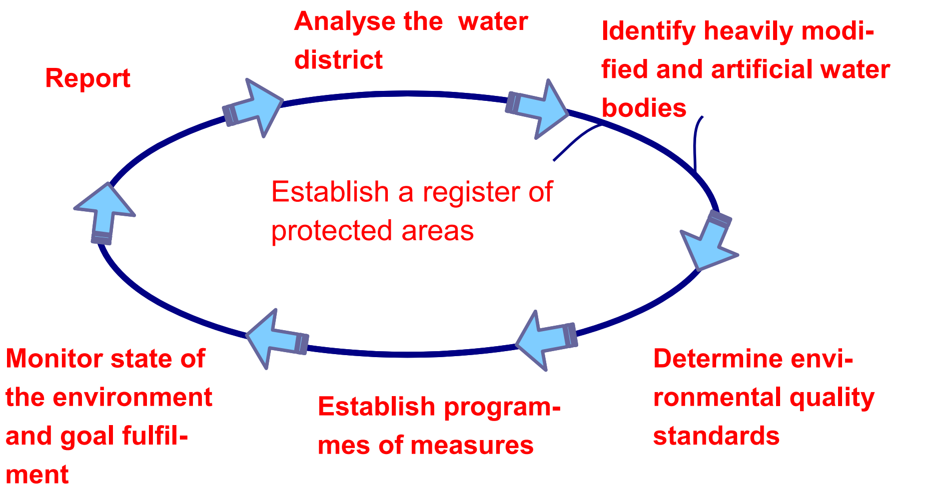

Figure 1.1 The water authority's planning cycle describes the procedure to be followed in water management work. A cycle normally takes six years to complete and contains e.g. analysis of the water district, establishment of environmental quality standards, the setting-up of programmes of measures, monitoring and reporting.

Sweden has incorporated the WFD into its national legislation, which means that Swedish water management is basically regulated by the following three legal statutes:

- The Swedish Environmental Code (1998:808)
- The Swedish Water Quality Management Ordinance (2004:660) *(sometimes referred to in short as the Water Management Ordinance or WMO)*
- The Swedish Ordinance (2002:864)containing Instructions to the County Administrative Boards (shortened to the *County Administrative Board instructions*)

Furthermore, the Swedish Environmental Protection Agency (Swedish EPA) and the Geological Survey of Sweden (SGU) have the power to issue additional regulations.

The Water Management Ordinance is hence the legislation that formally applies in Swedish law and the WFD only applies in cases where specific references are made to it in the ordinance. This handbook refers therefore first and foremost to the Water Management Ordinance, although sometimes to the WFD in cases where the ordinance contains such references.

The WFD is supplemented by two "daughter directives", one for groundwater[1] and one for prioritised substances[2] . The latter directive establishes limit values for 33 priority substances and 8 other pollutants. Priority substances are substances or groups of substances that are harmful and that shall be reduced or phased out.

Under Chapter 5 (Sections 10-11) of the Swedish Environmental Code, Sweden is divided into five water districts, each of which is to be coordinated by a water authority (Figure 1.2). A county administrative board (CAB) in each water district has been designated the water authority and is responsible for water quality management within the district. The water authority shall draw up a management plan and programmes of measures. The management plan shall, among other things, specify the conditions and the environmental quality standards that are to apply within the water district and this handbook is intended as a guide for some of the work involved. The programme of measures shall specify the measures needed to achieve or to maintain a certain environmental quality standard.

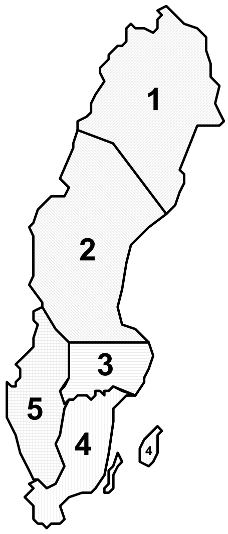

Figure 1.2. The five water districts in Sweden are: (1) The Bothnian Bay, (2) The Bothnian Sea, (3) Baltic North, (4) Baltic South and (5) Skagerrak and Kattegat.

[1] Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration

[2] DIRECTIVE 2008/105/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 16 December 2008 on environmental quality standards in the field of water policy, amending and subsequently repealing Council Directives 82/176/EEC, 83/513/EEC, 84/156/EEC, 84/491/EEC, 86/280/EEC and amending Directive 2000/60/EC of the European Parliament and of the Council

Based on the Water Management Ordinance, the Swedish EPA issued three sets of regulations at an early stage:

- Swedish EPA Regulations (2006:1) on the mapping and analysis of surface water in accordance with the Water Quality Management Ordinance (2004:660)
- Swedish EPA Regulations (2006:11) on the monitoring of surface water in accordance with the Water Quality Management Ordinance (2004:660)
- Swedish EPA Regulations (2007:1) and General Guidelines on programmes of measures for surface water in accordance with the Water Quality Management Ordinance (2004:660)

This handbook is intended as a support to the application of Swedish EPA Regulations (NFS 2008:1) and General Guidelines on classification and environmental quality standards regarding surface water.

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1.2 The aim of the handbook

This handbook is primarily intended for those who work with the classification of ecological status or potential and surface water chemical status, and with establishing environmental quality standards for surface water bodies.

The aim of the handbook is to clarify and interpret Swedish EPA Regulations NFS 2008:1, the Water Management Ordinance and the WFD. The intention is to give general guidance as to how quality requirements in surface water can be determined and monitored. The handbook focuses primarily on knowledge that currently exists or that can be obtained before it is time to draw up the next management plan. The idea has been to produce a "step-by-step" guide and to make it easier for the water authority in cases where an expert judgement needs to be made based on the limited supporting data that is currently available, including environmental data, models, etc.

The purpose of the handbook is also to help ensure that the assessment of water quality is as homogenous as possible throughout Sweden. It does not, however, contain detailed information on how the practical work can be done within a river basin district.

A central component of both how quality requirements are determined and how status or potential are monitored is the interpretation of the results obtained when applying the assessment criteria to the observed data. A major need for guidance has been identified here. There is also a considerable need for guidance when there is a lack of supporting data to the assessment criteria and an expert judgement must be made. Many of these issues are dealt with in this handbook, as well as in the handbook on mapping and analysis and the handbook on monitoring.

1.3 Delimitations

This handbook only deals briefly with how quality requirements for surface water bodies that are subject to exemption are to be determined. Such issues will instead be highlighted separately in forthcoming guidance material. This means that the handbook does not contain the entire process from determining quality requirements to actually establishing them.

Regarding artificial and heavily modified waters, this handbook provides guidance on how to determine quality requirements whilst the process for declaring these as artificial or heavily modified will be described in forthcoming guidance material. The term “determining quality requirements” refers to the process whereby e.g. county administrative boards produce supporting material prior to the water authorities taking a decision. The term “declaring quality requirements” refers to the water authorities actually taking the decision to consider the water body artificial or heavily modified.

Issues regarding environmental monitoring programmes are dealt with in the handbook on environmental monitoring and are not discussed in this handbook.

The surface water chemical status of a water body shall also be classified in accordance with the Water Management Ordinance. This is here a question of substances for which limit values have been specified at the Community level within the EU. Surface water chemical status partly includes the substances or groups of substances regulated in the EC fishing water directive[3] and shellfish directive[4], which have been implemented through Swedish Ordinance (2001:554) on environmental quality standards for fishing and bivalve waters. Mainly, however, it concerns the priority substances (pollutants) highlighted in the WFD. Limit values have been negotiated for the 33 priority and 8 additional substances of groups of substances and are included in a daughter directive to the WFD (see also Chapter 5).

The current assessment criteria do not cover the introduction of new non-native species. Because of a lack of scientific background data, the impact of non-native species has in principle not been included. Work is ongoing within the EU to draw up guidelines on how to deal with this as it is a widespread problem.

Regarding hydromorphology, the scientific background data has not been deemed sufficient to develop national assessment criteria for coastal and transitional waters. Regarding hydromorphological quality elements in coastal and transitional waters, Annex C gives only a short summary of feasible background data that can be used as an aid when classifying status and potential in coastal waters.

[3] Council Directive 78/659/EEC of 18 July 1978 on the quality of freshwaters needing protection or improvement in order to support fish life

[4] Council Directive 79/923/EEC of 30 October 1979 on the quality required of shellfish waters

1.4 How to read the handbook

To make it easier to read the handbook, two different types of boxes occur in the right-hand margin as a link to regulations (REG) or general guidelines (GG). The boxes tell the reader which chapter/section of the regulations/general guidelines they should refer to regarding the issue in question. A continuous line indicates a link to regulations (REG) and a dotted line to general guidelines (GG). An example is shown here to the right.

See REG
Chapter 1
Section 1

See GG to
Chapter 1
Section 1

1.4.1 Abbreviations

Abbreviations used in this handbook:

BDM	Boreal Dilution Model
BQI	Benthic Quality Index
CIS	Common Implementation Strategy
EC	European Community
EQR	Ecological Quality Ratio
EQS	Environmental Quality Standards
GEP	Good Ecological Potential
MAGIC	Model of Acidification of Groundwater in Catchments
MEP	Maximum Ecological Potential
WMO	Water Management Ordinance - Swedish Ordinance (2004:660) on Water Quality Management.
WISS	WaterInformationsSystemSweden

1.4.2 Concepts

Concepts	Definitions
Artificial water body	A surface water body created as a result of human activity in a location where there hasn't previously been a surface water body.
Checking procedure	A method for assessing whether the class boundary between good and moderate status, or between good and moderate potential has been correctly set for the physico-chemical quality elements. The objective of the checking procedure is to avoid the set class boundaries having a negative effect on the interpretation of the function of the water body's biology.
Class boundaries	The boundaries between the different classes in the assessment criteria, when classifying status or potential.
Classification	Assessment of the status of a water body. Regarding natural surface water bodies, this is an assessment of ecological status and surface water chemical status. Regarding artificial and heavily modified surface water bodies, however, it is an assessment of ecological potential and surface water chemical status. Parameters and quality elements are classified and then weighed together to give the ecological status or potential and given limit values are classified to be weighed together to give the surface water chemical status.
Ecological potential	The quality of the structure and function of aquatic ecosystems that are associated with the surface water of a heavily modified or artificial surface water body, classified in accordance with Annex V of the WFD and expressed as "maximum", "good", "moderate", "poor" and "bad".
Ecological Quality Ratio (EQR)	A scale between 1 and 0, where 1 is the highest reference value and is included in the "high status" class. Zero (0) is the greatest possible deviation from the reference value, i.e. "bad status". The interval between 1 and 0 is divided into the classes "high", "good", "moderate", "poor" and "bad" ecological status.
Ecological status	The quality of the structure and function of aquatic ecosystems that are associated with the surface water, classified in accordance with Annex V of the *WFD and* expressed as "maximum", "good", "moderate", "poor" and "bad".
Environmental criteria for environmental quality	Scientific criteria used when classifying the ecological structure and function of aquatic ecosystems in accordance with Annex V of the WFD (Directive 2000/60/EC). The assessment criteria contain reference values and class boundaries for all quality elements.
Expert judgement	A judgement made based on the best available knowledge in cases where the assessment criteria cannot be applied.
Heavily modified water body	A surface water body that as a result of human activity has significantly changed its physical character.
Less stringent quality requirements	A water body can be subject to less stringent quality requirements if it has been so heavily impacted by human activity that achieving good ecological status is completely out of the question. Another reason might be that, because of the water body's natural characteristics or the degree of human impact on it, it would be disproportionately expensive to implement the measures necessary to achieve the environmental objectives.
Limnic types	Classification criteria in accordance with the Regulations on mapping and analysis, NFS 2006:1, for characteristics that shall be applied when type-classifying lakes and watercourses. The determinant characteristics for lakes are "maximum depth", "surface area", "humic content" and "lime content". For watercourses, the determinant characteristics are "size of runoff area", "humic content" and "lime content". These should not, however, be confused with the type-classification that is performed when classifying water bodies using the assessment criteria, which are not as detailed. How the type-classification is to be performed is described in each assessment criterion.
Object-specific reference	Specific reference values can be developed for individual objects (water

Concepts	Definitions
values (see also type-specific reference values)	bodies) based on a method specified in the assessment criteria. The objects are then associated with a type for reporting purposes in accordance with the WFD. Object-specific reference values apply first and foremost to limnic water bodies.
"One out, all out"	Under Annex V of the WFD, the quality factor exhibiting the greatest anthropogenic disturbance shall decide the status classification. This does not normally apply in the parameter level apart from e.g. when weighing together pollutants and for biological and hydromorphological parameters indicating different degrees of impact.
Parameter	Sub-component of a biological, physico-chemical or hydromorphological quality element. A quality element consists of one or more parameters.
Quality elements	Biological, physico-chemical and hydromorphological elements listed in the annexes to NFS 2008:1. The status or potential of the different quality elements are weighed together into the ecological status or ecological potential according to the "one out, all out" principle.
Reference value	Value corresponding to a status that remains to all intents and purposes undisturbed by human activity. The reference values for a parameter or a quality element are given in the corresponding assessment criterion.
Significant impact	"Significant impact" is anthropogenic impact of such magnitude that, either on its own or combined with other impacts, it elevates the risk of a water body not achieving good status or potential by 2015.
Surface water body	The surface water body is the "sub-unit" within a river basin to which quality requirements in accordance with the Water Management Ordinance apply. A surface water body is characterised by the fact that it is homogenous in terms of its type and degree of impact. The surface water body is the smallest structural unit that can be classified in accordance with the WFD. The term "surface water body" is, according to the Water Management Ordinance, correct but to make this handbook easier to read, the simpler term "water body" (or "body of water") is used.
Surface water category	To be able to work efficiently with water bodies, they are divided into the different categories: lakes, watercourses, coastal waters and transitional waters. The term "surface water category" is, according to the Water Management Ordinance, correct but to make the handbook easier to read, the simpler term "water category" is used.
Surface water chemical status	The chemical quality of a water body, classified according to Article 4 and Annex V of *the WFD* and expressed as "good" or "failing to achieve good".
Surface water status	The status of a natural surface water body, determined by the water body's ecological status or surface water chemical status, depending on which is worse.
Type-group	The definition of a type-group is "a collection of water bodies that belong to the same type (according to the Regulations on mapping and analysis, NFS 2006:1) and that have the same degree and type of impact. Instead of describing the state of individual water bodies, a description can then be given of the state of a *type-group* of water bodies.
Type-specific reference values (see also object-specific reference values)	A reference value or ratio is given for a parameter within a quality element. The reference value applies within a specific type of water body and is given in the assessment criteria. All water bodies within the type have the same reference value.
Water authority	In this handbook, the term "water authority" is used as a collective name for the water authorities and other actors who do work on behalf of the water authorities (e.g. the county administrative board and other actors, since the division of responsibility for what needs to be done can differ from one district or county to the next).

2 About quality requirements

2.1 Good surface water status is the starting point

The Water Management Ordinance obliges the water authorities to establish binding quality requirements to protect, improve and restore all surface water bodies in their water districts.

This handbook is intended to give guidance for the work of the water authorities in determining, establishing and monitoring quality requirements for surface water bodies. The water authorities shall use Swedish EPA Regulations (NFS 2008:1) and the class boundaries specified therein for the various parameters or quality elements as a starting-point. Status and potential are classified on the basis of *"assessment criteria"* and chemical status is classified on the basis of specified limit values.

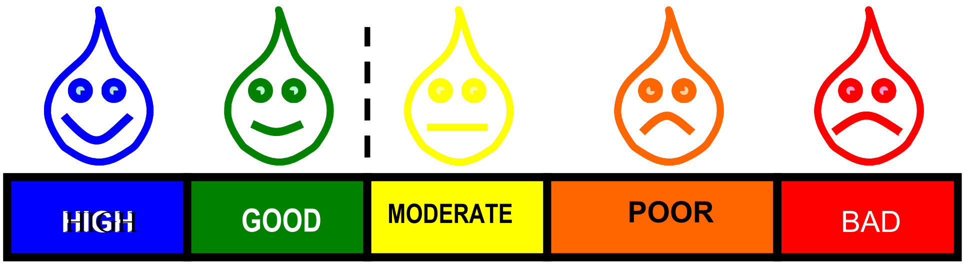

Figure 2.1 The five possible ecological status classes under the Water Management Ordinance as related to the Water Framework Directive (WFD). The boundary between good and moderate is important, since the starting-point is that water bodies that are below this boundary may be in need of remediation. For chemical status there are only two classes, "good" or "failing to achieve good".

In practice, "determining quality requirements" means that the water authority determines the level of environmental quality intended to be achieved by 2015 for each water body or group of water bodies. To assess the environmental qua-lity of water bodies, the water authorities shall use the assessment criteria scales for quality elements in NFS 2008:1 as their starting-point. These scales are divided into five status classes: high, good, moderate, poor and bad (Figure 2.1). The results of classifications for all quality elements are then weighed together to give the ecological status.

See REG Chapter 2 Section 2

In addition, surface water chemical status is assessed either as good surface water chemical status or failed to achieve good surface water chemical status (WMO, Chapter 4, Section 2 and Chapter 1, Section 4). The substances inclu-ded in the classification of surface water chemical status are those that have common EC limit values. This includes the substances and groups of substan-ces regulated by the EC Freshwater Fishing Directive and the Shellfish Waters Directive. These are implemented by Ordinance (2001:554) on environmental quality standards for freshwater fish and bivalve (mussel) waters. However, it is primarily a matter of the priority substances that have been identified in Com-mission Decision No. 2455/2001/EC amending the WFD. Limit values are un-der negotiation in the European Council and the European Parliament (not yet adopted as of December 2007) for the 33 priority and eight other substances or substance groups that will be included in a daughter directive to the WFD. When the daughter directive has been adopted, it shall be incorporated into Swedish legislation, and guidance on this will be included in this handbook (Chapter 5).

The starting-point for determining quality requirements for our water bodies is that they must achieve good surface water status by 2015 and that they must not deteriorate. For water bodies that are subject to exemption, or are declared to be heavily modified or artificial, other quality requirements shall be established, see Section 2.3. Surface water status is the condition of a water body deter-mined by that water body's ecological status or surface water chemical status, depending on which of the two is the worse. However, ecological status and surface water chemical status are not weighed together to produce surface water status, since it would be impossible thereafter to achieve high status. This is because the surface water chemical status can at best be classified as "good". Also, since the state of a surface water body may not deteriorate, weighing the statuses together in such a way may miss any deterioration from high to good. Quality requirements for water bodies are therefore determined separately for ecological status and for surface water chemical status. Regarding ecological status, the quality requirements are determined as good or high ecological status on the basis of the Swedish EPA's assessment criteria. Regarding surface water chemical status, the quality requirements are determined on the basis of the specified limit values only as good chemical surface water status. Regarding ecological status, a water body classified as high ecological status must continue to achieve high ecological status in the future because of the demand for "no deterioration". This means that when a water body with an environmental quality standard established as high ecological status is subsequently assessed, it is not sufficient that the water body shows good ecological status.

2.2 Environmental objectives, environmental quality standards or quality requirements

The Water Framework Directive states that environmental objectives are those objectives established in Article 4 thereof. These environmental objectives must not be confused with the Swedish environmental (quality) objectives, which are political aims and not legally binding. Sweden has chosen to implement the provisions about environmental objectives in the WFD by means of the legal instrument environmental quality standards[5] (Chapter 5 of the Environmental Code). The

[5] The concept "environmental quality standards" also appears in the WFD, where it refers to the concentration of a pollutant that should not be exceeded

environmental objectives under Article 4 of the WFD thus correspond in Swedish law with the environmental quality standards[6] laid down in Chapter 5. Section 2, first paragraph, Point 4, of the Environmental Code: “other environmental quality requirements arising from Sweden’s membership of the European Union”.

The Government may task a public authority to issue environmental quality stan-dards arising out of Sweden’s membership of the European Union (Environmental Code, Chapter 5, Section 1, second paragraph) The Government has done this by prescribing, under Chapter 4, Section 1 of the Water Management Ordinance (WMO), that each water authority shall define quality requirements for surface water bodies, ground water bodies and protected areas in its water district. The water authority must determine what standard of environmental quality has to be achieved via what are known as “water delegation decisions” (CABi 37, Section c)[7]. Water delegation decisions on quality requirements are therefore the same as environmental quality standards in this context. The Swedish EPA cannot at present say what form such a decision about quality requirements will take and the legal effects of the environmental quality standards are being discussed in several contexts. In Reports it has been emphasised that the environmental quality standards are directly binding on authorities, but only indirectly binding on individuals. In other words, individual enterprises and persons are affected in the same way and by the same instruments as they already are today, e.g. through permitting procedures and general regulations.

The Swedish EPA’s authority as regards environmental quality standards for water is to be found in Chapter 4, Section 8 of the Water Management Ordinance (WMO). The above shows that the water delegation’s decisions on quality requirements are synonymous with environmental quality standards. The water delegation’s decisions are, however, reached via a series of events and standpoints. Often, it is the county administrative boards that apply the assessment criteria and classify status or potential (including applying the provisions on expert judgement and further examination contained in NFS 2008:1). In addition, it may be the county administrative boards that make a first assessment of whether a surface water body is artificial or heavily modified or whether some other provision in Chapter 4 of the WMO is applicable (see Sections 2.3 and 2.4). Only after the water authority has considered the provisions of both Chapter 4 of the WMO and NFS 2008:1 can the water delegation decide on quality requirements, i.e. define what environmental quality standard is to apply to the water body in question. To describe the standpoints that lead to decisions by the water delegation, the Swedish EPA has chosen in this handbook to write “determine quality requirements” in accordance with the wording in the authorisation.

[6] In the Government Bill regarding amendments to the Environmental Code, it is stated that the concept "environmental quality standard" is not to be reserved for binding limit values and that "environmental quality standard" is synonymous with "provision on environmental quality" (Govt Bill 2003/04:2 p. 22)

[7] SOU 2005:59 The Environmental Code: environmental quality standards, environmental NGOs in the environmental decision-making process and charges and SOU 2005:113 Programme of measures for environmental quality standards

2.3 Artificial and heavily modified water bodies and exemptions

Of the thousands of water bodies found in Sweden, there will be a number where, for socioeconomic or other reasons, it is impossible to achieve good ecological status by 2015. These may be subject to exemption (Chapter 4 Sections 9-13 of the WMO).

If good ecological status cannot be achieved because of a hydromorphological impact (from e.g. a hydroelectric dam), and this impact cannot be remedied without having a negative effect on the purpose of the activity (hydropower production in this case), the water body may be declared heavily modified or artificial (Chapter 4, Section 3 of the WMO). Instead of good ecological status and good surface water chemical status this water body shall then achieve good ecological potential and good surface water chemical status (Chapter 4, Section 4 of the WMO). Good ecological potential is defined principally by the ecologically important measures that are technically feasible without having a significant negative effect on the activity. However, for the quality elements not affected by the hydromorphological impact in a heavily modified or artificial water body, the same boundaries apply as for good ecological status. For example, in certain cases it may be noted that the fish fauna in a regulated water body are disturbed by a dam while the number of diatoms remains unaffected.

2.4 The strictest requirement applies

According to Chapter 4, Section 6 of the WMO, quality requirements for protected areas must be established so that all standards and objectives are met by 2015, unless it is provided for otherwise in the legislation under which the protected areas have been established. The term “standards” here refers to the limit values, etc., in European Community legislative other than the WFD, whilst “objectives” are equivalent to the environmental quality standards under the WMO (i.e. the Swedish legislation incorporating the environmental objectives in the WFD). One example could be that requirements under both the Habitats Directive[8] and the WFD shall be established with the help of environmental quality standards and fulfilled for a particular water body.

If a water body is in a particular respect covered by quality requirements of varying stringency, the most stringent requirement applies (Chapter 4, Section 7 of the WMO). “Quality requirements” in this context refer to the quality elements or parameters of relevance in a specific water body. A quality requirement in accordance with e.g. the Habitats Directive is not, however, necessarily more stringent than those laid down in the WFD. It is the Swedish EPA’s interpretation that “the most stringent requirement” in this case refers to the protection value and not the

[8] Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora.

level of the requirement per se, i.e. the level of requirement established to protect something regarded as more worthy of protection could be regarded as being the most stringent requirement by law. This might lead to a situation where the level of requirement is less stringent, but where the object which the level of requirement aims to protect is regarded as more worthy of protection than the object which the more stringent requirement is intended to protect. In other words, there is a collision between, on the one hand, requirements on the basis of a general law and, on the other hand, requirements based on a specific law, a conflict that must be resolved by balancing interests. In certain exceptional cases, a situation might ensue whereby the quality requirements in accordance with two specific laws collide. It is the water authority's task when balancing such cases to make an assessment as to which of the protection interests weighs the heaviest. In this assessment, it is of course not only the state of the water body in question that should be taken into account, but also the state of the water bodies downstream. That may, for example, result in a less stringent quality requirement being determined under the Habitats Directive for phosphorus content in a bird lake than the quality requirement applied under the WFD.

A number of water bodies with particular protection values will neither be designated as protected areas in accordance with the WMO, nor come under Swedish EPA Regulations (NFS 2008:1) and General Guidelines on classification of and environmental quality standards for surface water. For these, it can instead be better for the water authority/county administrative board to protect the water body by means of other instruments, such as area protection, or special regional environmental objectives, as part of the work on environmental quality objectives at the national level.

For further information about protected areas, please refer to the Swedish EPA fact sheet (No. 8304) "Skyddade områden enligt vattenförvaltningsförordningen [Protected areas under the Water Management Ordnance]".

3 Procedure for classifying status and determining quality requirements

3.1 Data requirements and methods for classifying status and determining quality requirements

3.1.1 Mapping and analysis

During the work on characterisation in accordance with the Swedish EPA's Regulations (NFS 2006:1) on mapping and analysis of surface water under the Water Quality Management Ordinance (2004:660), existing data on the status of, and impact on, Swedish water bodies has been compiled (see also the Handbook on Mapping and Analysis, Handbook 2007:3). This constitutes good background material when the status is to be classified and quality objectives are to be determined. It is easier to assess the status of water bodies that have ongoing monitoring programmes and where quality elements have been examined in accordance with the requirements for assessment criteria. In water bodies where monitoring is not being carried out or is deficient, a carefully conducted impact analysis can be of great assistance when assessing their status.

Impact and status are often related to one another and therefore data on impact pressure can be a good indicator of environmental status. Furthermore, there are now a number of modelling tools available to perform extrapolated assessments by means of what is known as "source distribution analysis". Such an analysis can be of great assistance when there is no current data on the status of the areas to be assessed. Using the impact data, the status can be estimated by comparing similar water bodies within the same "type-group". A type-group is defined as a collection of water bodies that belong to the same water type (as defined in the Regulations on mapping and analysis) and have the same degree and type of impact. Instead of describing the state of individual water bodies, a description can then be given of the status for a type-group of water bodies (Section 4.5).

Impact data is also of importance when a status classification needs to be corroborated, for example when the classification is close to a class boundary. A simple description may suffice saying that, if the impact is major, it may be an indication that the status is worse, and if there is little or no impact, the status is probably better. More details on impact analysis can be found in the Handbook on Mapping and Analysis, and in Section 4.4.4.

3.1.2 Assessment criteria

See REG
Chapter 2
Section 3

The Swedish EPA's assessment criteria shall be used as the basis for both classifying status and determining quality requirements. However, assessment criteria have not been produced for all the normative definitions given in Annex V of the

Water Framework Directive (WFD). In some cases, this was due to lack of data, while in other cases they have not been regarded as relevant environmental indicators, for example because of excessive variation or because they quite simply have not been regarded as an indication of any environmental problem in Swedish conditions. Parameters can still be added, replaced or deleted as their value as environmental indicators is clarified or as more data is collected. The majority of the assessment criteria indicate primarily the effects of eutrophication or acidification. Some assessment grounds, however, show more general impact on the quality elements as compared with a normal state (for example, fish in freshwater and benthic macroinvertebrates in coastal waters). An assessment of each quality element indicated in Annex V of the WFD is nonetheless regarded as feasible on the basis of the assessment criteria developed.

The purpose of this handbook is to give guidance on the application of the assessment criteria. Annexes A, B and C provide detailed descriptions of each individual assessment criterion. In cases where the assessment criteria are for some reason not applicable (see Section 4.2) an expert judgement must be made (see Section 4.4).

Under the WFD, EU Member States' assessment criteria and class boundaries for the biological quality elements shall be intercalibrated. A number of types and parameters have been intercalibrated, i.e. have been compared with neighbouring countries that have comparable waters. In certain cases, the boundaries have been slightly adjusted, up or down, depending on the parameter. In the majority of cases, however, the Swedish assessment of high, good and moderate status has corresponded well with that of other countries. This work has influenced the assessment criteria and can be regarded as an assurance of quality. In Annexes A and B, the parameters that have been intercalibrated are marked.

3.1.3 Limit values for surface water chemical status

In the summer of 2006, the Commission put forward a proposal for a daughter directive to the WFD, on the regulation of a number of priority substances. When this handbook was being drafted (December 2007), negotiations on this daughter directive were still ongoing in the European Council and the European Parliament. The Directive will establish limit values (EQS) for 33 priority pollutants designated in Decision no. 2455/2001/EC, and for 8 other pollutants. The water authority shall use these limit values when classifying and determining quality requirements for surface water chemical status.

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Other substances that have common EU limit values shall also be used in classifying and determining quality requirements for surface water chemical status. This includes the substances and substance groups that are regulated in the EC's Freshwater Fishing Directive and the Shellfish Waters Directive which have been incorporated into Swedish legislation through Ordinance (2001:554) on Environmental Quality Standards for fish and shellfish waters.

3.1.4 All available information shall be used

See REG Chapter 2 Section 8

In a well-examined water body, where all quality elements have been analysed, a status classification can be made fairly easily. In many cases, however, there will be a lack of data, which means that expert judgements need to be made. In an expert judgement, all available information must be used, such as impact data and other measurements. For example, the hydromorphological class boundaries in the assessment criteria can in this context constitute one of the tools for such an expert judgement. This is particularly important when major hydromorphological impact is suspected, since the majority of the assessment criteria have been developed on the basis of acidification and eutrophication impact, and the hydromorphological impact on the biology of the water body may not be registered in the classifications using the assessment criteria. When a status classification gives the result "good ecological status", despite significant hydromorphological impact, it may be necessary to carry out further investigations. The impact on migrant fauna and the presence of suitable spawning and breeding grounds are declared in Annex V, 1.2.5 of the WFD as being the type of conditions that are especially relevant to assess and mitigate. The hydromorphological parameters regarded as best reflecting this impact may therefore have a particular value when the hydromorphological assessment criteria are used to support status classification.

Unless otherwise stated in the assessment criterion, a rule of thumb can be not to use data on impact or condition that is older than one water planning cycle, i.e. six years. When making an expert assessment, for example, older data can only be regarded as representative when there are clear signs that the condition of, and impact on, the water body have not materially changed over time.

See GG to Chapter 2 Section 8

Improving the available assessment data, by e.g. additional sampling, may be important in order to ascertain whether or not a water body really does need to be remediated. It is, however, always sensible to have a strategy for the additional sampling. On a basic level, a good strategy is to follow the guidelines that have already been established e.g. for the developed assessment criteria. If surface water status is to be monitored, it is appropriate to first check whether the parameters and quality elements, on which the assessment criteria are based, respond to the effect one is trying to measure. Guidance on setting up a monitoring programme is to be found in the handbook on surface water monitoring.

3.2 Classification of status

3.2.1 Classification principles

See REG Chapter 2 Section 2

A certain pattern should be followed when classifying ecological status or potential (Figure 3.1). Under the WFD, biological quality elements weigh heaviest, followed by physico-chemical elements and finally hydromorphological quality elements.

The biological quality elements shall be classified to begin with. Physico-chemical elements (general conditions in Annex V of the WFD) need only be classified when the status or potential for biological quality elements has been classified either as good or high status or as good or maximum potential. Hydromorphological quality elements need only be classified when the status or potential for both biological and physico-chemical quality elements have been classified as high status or maximum potential. For substances included in surface water chemical status, those must be classified that are discharged into the river basin or sub-basin (see Chapter 5).

The reason for this order is that the aim of water management is healthy biology. If the biology is moderate or worse, it is of no importance what the physico-chemical or hydromorphological quality elements show, because a programme of measures must in any event be established to achieve good status. To classify a water body as good or high status, on the other hand, the supporting physico-chemical and hydromorphological quality elements must also show the same status.

3.2.1.1 ECOLOGICAL STATUS

The ecological status shall be classified for natural water bodies (those that have not been declared heavily modified or artificial) (see also Section 3.2.1). Figure 3.1 shows, in diagram form, a feasible process for classifying a water body. A more detailed working procedure is described in the checklist for classification of ecological status in Section 3.2.3.

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If none of the quality elements in a water body has an environmental status worse than high, the ecological status is classified as high. If the hydromorphological quality elements in a water body have an environmental status worse than high, when the status for other quality elements is assessed as high, the ecological status is instead classified as good. If the physico-chemical quality elements are classified as worse than good status, when the status for biological quality elements is classified as high or good, the ecological status is classified as moderate. This follows the “one out – all out” principle, which is explained in Section 4.2.4.

In principle, therefore, a water body can be only be assigned a status worse than moderate on the basis of biological quality elements. A water body can, however, be classified as worse than moderate on the basis of physico-chemical and hydromorphological quality elements, as a result of expert judgements. However, this must be justified and documented.

3.2.1.2 ECOLOGICAL POTENTIAL

For water bodies that have been declared heavily modified or artificial, it is not the ecological status but the ecological potential that shall be classified. Figure 3.2 shows, in diagram form, a feasible process for classifying the potential of a water body. The working procedure is broadly speaking the same as for the classification of status, but the names of the various classes are different. Instead of high potential, the term “maximum ecological potential” (MEP) is used. Otherwise, the names are the same as for status, but with “status” replaced by “potential”. A more detailed working procedure is described in Section 3.4.

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For a heavily modified or artificial water, the boundary between maximum and good potential (GEP) is assessed by determining what conditions prevail when the biological quality elements are only affected by the modified or artificial character of the water body. This modified or artificial character must be caused by the activity that justified the water body being declared heavily modified or artificial. The abovementioned conditions are further determined only after account has been taken of the improvements that can be achieved if all mitigation measures have been implemented. These measures must, in turn, consist only of those that have no significant negative effect on the purpose of the modification.

For water bodies that can be subject to exemption, or designated as protected areas, see Chapter 4, Sections 9-13 of the Water Quality Management Ordinance (WMO), forthcoming guidance material on heavily modified water bodies and exemptions, and the Swedish EPA's fact sheet on protected areas.

3.2.1.3 SURFACE WATER CHEMICAL STATUS

Classification of surface water chemical status is carried out for substances that have common European Community EQSs and are discharged into the water body. Surface water chemical status is classified either as good or failing to achieve good status, on the basis of the limit values given in Chapter 5.

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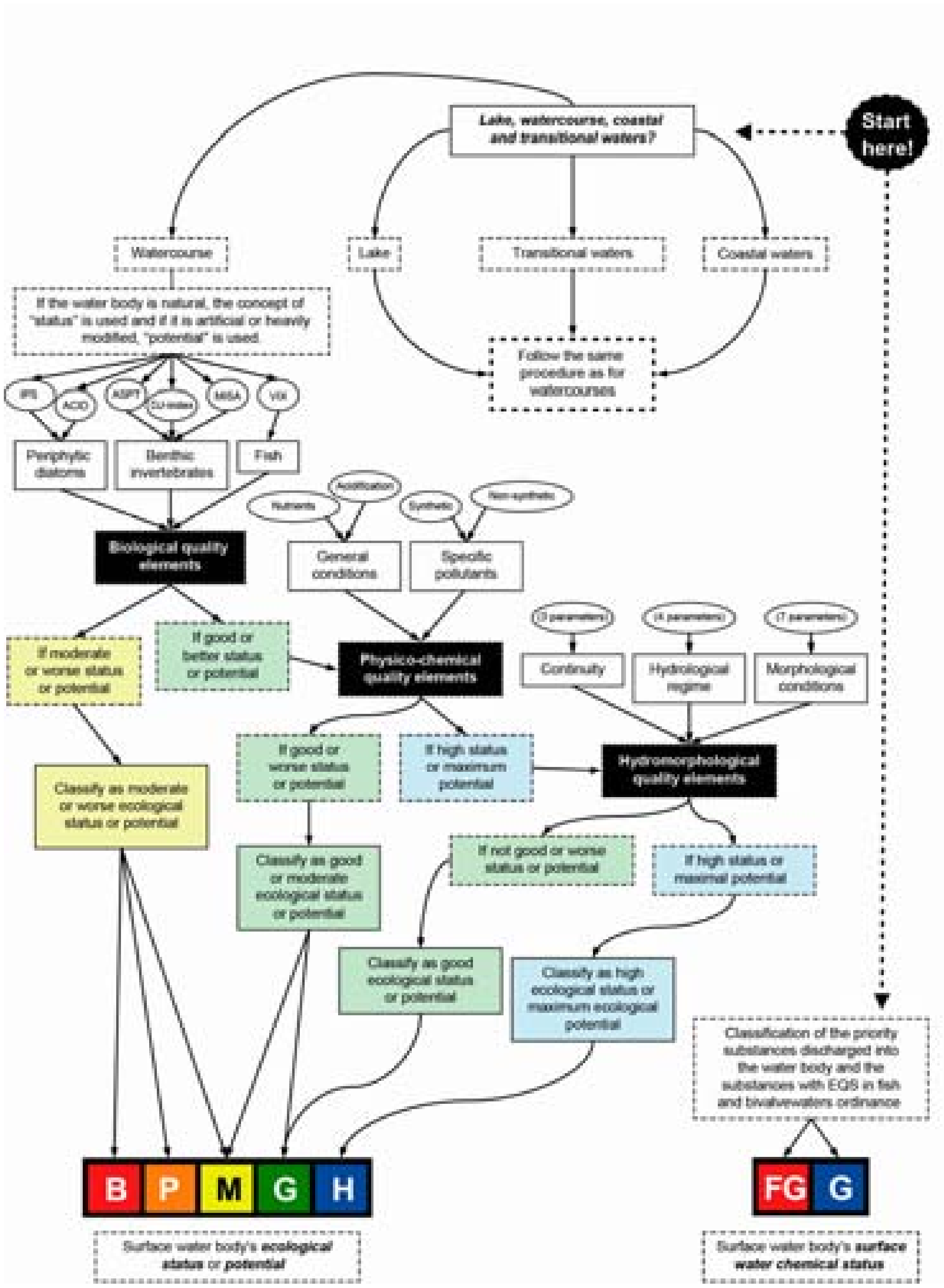

Figure 3.1 A simple flow chart describing the procedure for classifying the ecological status or potential and surface water chemical status of a surface water body. In certain cases, it may be practical to work in parallel on several stages at once. H, G, M, P and B stand for high, good, moderate, poor and bad ecological status respectively. G and FG stand for "good" / "failing to achieve good" surface water chemical status.

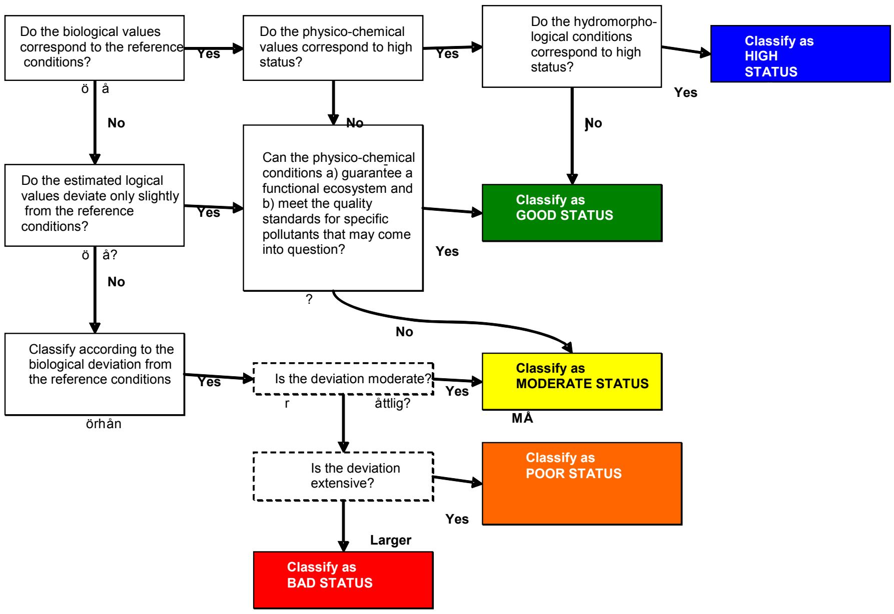

Figure 3.2 The relative role of the biological, hydromorphological and physico-chemical quality elements in the working procedure for classifying ecological status. The principle is the same as in the working procedure for ecological potential.

3.2.2 Ecological Quality Ratio (EQR)

As a common denominator for the results of different classifications in ecological status, the WFD uses the concept of Ecological Quality Ratio (EQR). The EQR is a value between 0 and 1 representing status or potential. The calculation of the EQR varies, depending on how the parameter responds to changes in water quality. If the absolute value increases with better water quality (e.g. depth dispersion of macroalgae in a marine environment), the EQR is calculated by dividing the observed value by the reference value. If the absolute value for the parameter diminishes with better water quality, the EQR is calculated by dividing the reference value by the observed value. The latter is the case with, for example, chlorophyll and biovolume in a marine environment. This ensures that the EQR value 1 always represents the reference ratio and values close to 0 represent poor status.

When the parameter value increases with improved water quality:

$$EQR = \frac{\text{Observed value}}{\text{Reference value}}$$

When the parameter value diminishes with improved water quality:

$$EQR = \frac{\text{Reference value}}{\text{Observed value}}$$

The EQR is calculated for a quality element or parameter in a specific water body. The interval between 1 and 0 is then divided into the classes: "high", "good", "moderate", "poor" and "bad" status. Each parameter has its own basis of division for what is moderate, good and high status as it represents deviation from the reference value. This means that the boundary between e.g. high and good is not always set at e.g. 0.8 but varies between the different parameters and quality elements. This division of classes can also differ between types for the same quality element (Figure 3.3).

(For surface water chemical status, there is a term very similar to EQR, namely Environmental Quality Standards (EQS). EQS are the boundary values, developed as a result of effect studies, for the priority substances).

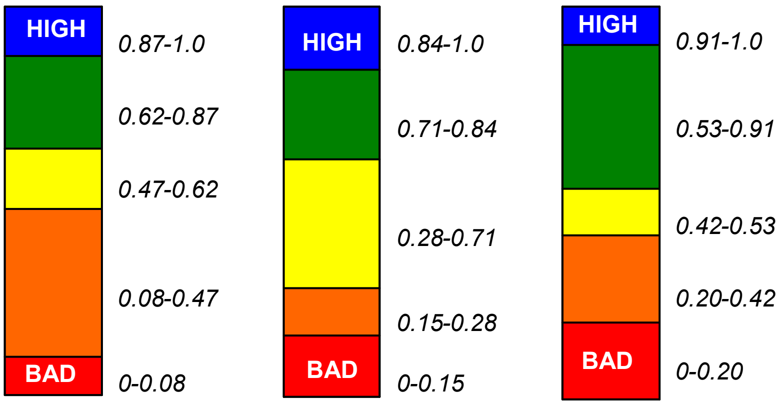

Figure 3.3 Three different examples of EQR classification. The EQR classes can vary in size, depending on what type and parameter it represents, since it shows the relation to the reference value.

3.2.3 Checklist for classification of ecological status

The step-by-step process, described below in the form of a checklist, may be followed when classifying ecological status (see also Figure 3.4). In certain cases, it may be practical to work in parallel on several stages at once. This can, for example, be relevant when a decision has to be taken on which quality elements ought to be classified first. In order to assess whether a water body should be declared heavily modified or artificial, the hydromorphological quality elements, for example, must be assessed. At the same time, the question as to which status class the water body will be classified in must also be answered. More detailed discussion of the various elements is contained in Chapter 4 and in the assessment criteria dealt with in Annexes A, B and C.

1. To which category does the water body belong?

To simplify work on water bodies, they are divided into the different categories "lakes", "watercourses", "coastal water" or "transitional waters" (see Regulations on Mapping and Analysis, NFS 2006:1). For each category, there are a number of quality elements to be used in the rest of the classification process (Table 3.1). A heavily modified surface water body shall be treated on a par with the surface water category it most resembles. See further Section 3.4.

Guidance on how to divide water bodies into categories is given in the Handbook on mapping and analysis[9].

2. To which type does the water body belong?

It is important to determine which type each water body belongs to, since the reference values and class boundaries for the majority of the quality elements are set in relation to type-specific criteria. Criteria for the division of limnic types are shown in the Regulations on mapping and analysis, NFS 2006:1. For the classification of lakes and watercourses, a less detailed mapping has been used, which to a certain degree varies between the different quality elements. For certain quality elements, object-specific, modelled reference values or national reference values apply. This is described for each quality element respectively in Annex A.

Regulations (NFS 2006:1) specify the classification criteria for Sweden's coastal and transitional waters, and include maps showing the distribution of these types.

[9] [Mapping and Analysis of Surface Water, Hand-book 2007:3]

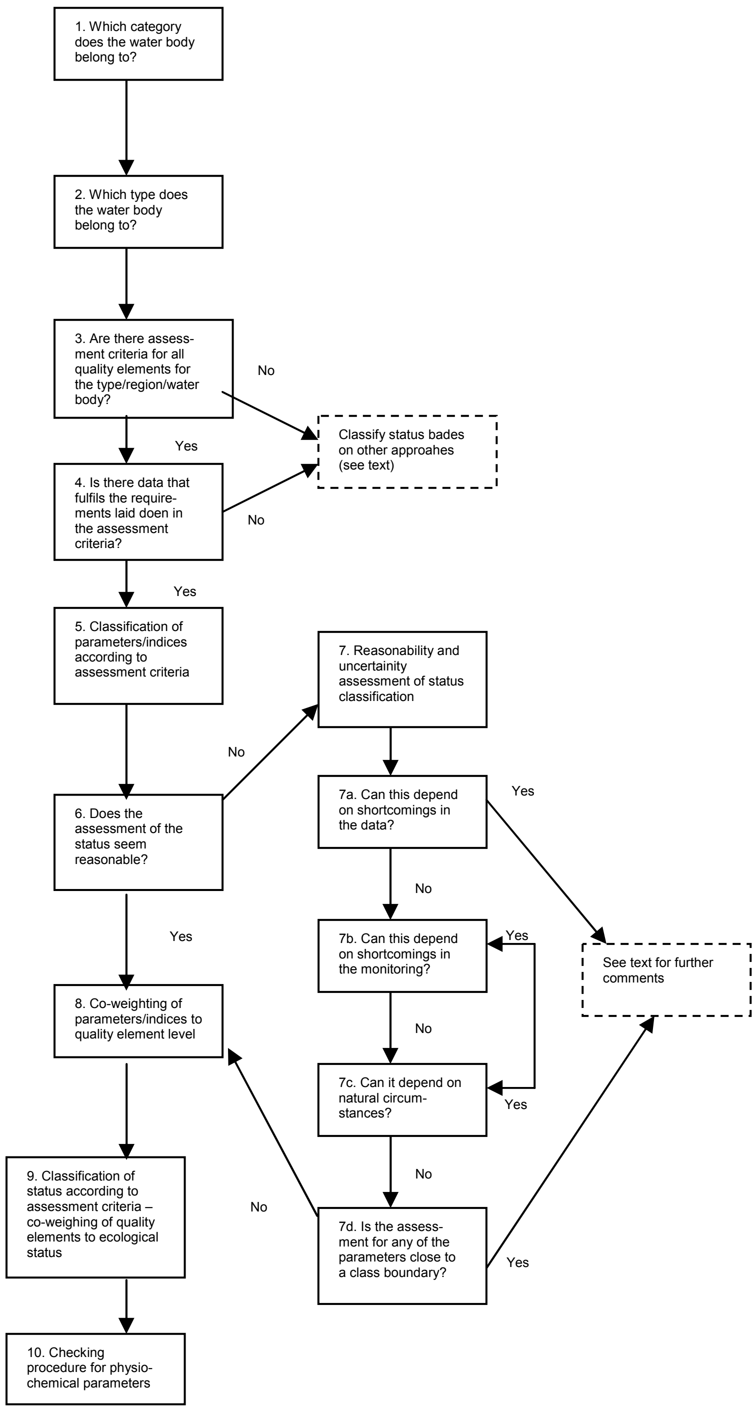

Figure 3.4. Outline of the checklist for status classification

Table 3.1 Summary of parameters or indices for all quality elements for ecological status where assessment criteria have been developed. Parameters in italics cannot be found in the regulations but can be used as an aid to classification.

Lakes	Quality elements		Parameter/ index
Biological elements	Phytoplankton		Total biomass
			TPI (trophic plankton index)
			Proportion of cyanobacteria
			Number of species
			Chlorophyll
	Macrophytes		Trophic macrophyte index (TMI)
	Benthic macroinvertebrates		ASPT
			MILA
			BQI
	Fish		EQR8
Physico-chemical elements[10]	General conditions	Nutrients	Tot-P
		Transparency	Transparency
		Oxygen	Oxygen
		Acidification	MAGIC library
	Specific pollutants		Substances discharged in significant quantities
Hydromorphological elements	Continuity		Presence of artificial migration barriers
	Hydrological regime		Prescribed regulation amplitude
			Impact on water-level changes
	Morphological conditions		Land-use in vicinity
			Land-use in sub-drainage basin
			Dead wood (number of pieces of wood)
			Modified littoral zone
			Number of ditches per km
Watercourses			
Biological elements	Diatoms		IPS
			ACID
			%PT (support parameter)

[10] Annex V of the WFD also lists priority substances discharged into the water body as a quality element under ecological status. However, under EU Guidance no. 13, the priority substances shall only be dealt with under surface water chemical status once common EU limit values have been developed. In these regulations, general guidelines and handbook, the priority substances are only dealt with under surface water chemical status

Lakes	Quality elements		Parameter/ index
			TDI (support parameter)
	Benthic macroinvertebrates		ASPT
			DJ index
			MISA
	Fish		VIX
			VIXsm (collateral index)
			VIXh (collateral index)
Physico-chemical elements[11]	General conditions	Nutrients	Tot-P
		Acidification	MAGIC library
			BDM/pBDM
	Specific pollutants		Substances discharged in significant quantities
Hydromorphological elements	Continuity		Presence of artificial migration barriers
			Degree of fragmentation
			Barrier effect
	Hydrological regime		Impact of flow regulation on watercourse
			Number of flow peaks per year
			Variation coefficient for daily flows
	Morphological conditions		Degree of straightening/canalisation
			Proportion of length cleared
			Number of road-crossings per km watercourse
			Land-use in vicinity
			Land-use in sub-drainage basin
			Number of ditches per km
			Dead wood (number of pieces of wood)
Coastal and transitional waters			
Biological elements	Phytoplankton		Biovolume
			Chlorophyll a
	Macroalgae and angiosperms		epth dispersion (only coastal waters)
	Benthic macroinvertebrates		BQI_m

[11] See footnote 4

Lakes	Quality elements		Parameter/ index
Physico-chemical elements[12]	General conditions	Transparency	
		Nutrients	total nitrogen, total phosphorus, DIN, DIP
		Oxygen	
	Specific pollutants		Non-synthetic
			Synthetic
Hydromorphological elements	Not available		

3. Are there assessment criteria for all quality elements for the type or water body?
In some cases it has not been possible to produce assessment criteria for a type due to the lack of knowledge or data. In certain cases there may also be water bodies that are not representative of the type, which in turn can entail that a specific assessment criterion is not applicable in that particular case. This is partly due to the fact that the available data is currently inadequate, and partly to the fact that the types must be relatively general to obtain a manageable number of types. This means that within each type, there will be individual water bodies that differ somewhat from the general type. This may affect the biological classifications.

One example of this can be water bodies that constitute extremely large or small lakes. The assessment criteria for fish in such lakes are not altogether reliable, because these lake sizes have not been included in the basic data used for the development of the assessment criterion. Another example can be a coastal water body that is situated by the mouth of a river and therefore has a lower salt content than other water bodies within the type, which makes the reference values not applicable here. For more information please see Annexes A-C for each assessment criterion respectively.

If Yes: Go to point 4.

If No: In cases where no assessment criteria are applicable, an expert judgement must be made on the basis of existing knowledge (Chapter 4). If further field samplings can increase the possibility of making correct judgements, they should be considered.

[12] See footnote 4

4. Is there data that fulfil the requirements laid down in the assessment criteria?

Different assessment criteria require different kinds of data. In addition to the fact that data from relevant quality elements is required, such data may need to have been collected using the correct method, in a specific environment, at the right time of the year, or according to other prerequisites specific to the quality element. More information about the kind of data required for the respective assessment criteria can be found in Annexes A, B and C.

If Yes: Go to point 5.

If No: In the first instance, it is appropriate to consider increasing the amount of available data by taking more field samples. If nevertheless there is still is a lack of measurement data needed to apply the assessment criteria, it may be necessary to classify the status of a water body using other approaches.

If the environmental data for a surface water body does not meet the requirements of the assessment criteria, type-grouping might be an alternative way of obtaining better supporting data. A type-group is defined as a collection of surface water bodies in the same category that belong to the same type and with the same impact pressure (Section 4.5). Instead of describing the status of an individual surface water body, it is thus possible to describe the status of a type-group of water bodies. Estimating quality elements on the basis of similar water bodies is not an exact method, but will suffice as a status classification when insufficient data about the water body is available.

In cases where water bodies cannot be type-grouped, a special expert judgement must be made on the basis of existing knowledge. Expert judgements can be made in different ways, as exemplified in Section 4.4. If there is reliable impact data, models can be used (e.g. the FYRIS model or the SMHI Coastal Zone Model) in order to assess the impact pressure on the surface water body. On the basis of the assessed impact pressure or other expert assessment, a status classification of the water body can then be made. Then go to point 8.

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5. Classification of parameters or indices in accordance with the assessment criteria Step 1. Assess each parameter individually. In certain cases, several parameters are embedded in an index, on which the classification is based instead. This applies e.g. to benthic macroinvertebrates in coastal waters, lakes and watercourses.

Step 2. Analyse the degree of variance. If the variation, after an investigation, is shown to be too large, the result based on this data for the parameter or index can be ignored in the status classification. The definition of what constitutes large or small variation differs, depending on which parameter is in focus. Certain parame-

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ters can e.g. have a large variation if the results from sampling during a whole year are considered, while the variation within a specific month is small. The concept of variation can also be interpreted in different ways depending on what knowledge one has about statistical methods. See Chapter 4 for more information about how to analyse the variation in order to make a correct classification.

6. Does the classification of status seem reasonable?

An evaluation of whether the result from the assessment criteria of the status classification seems reasonable shall be made on the basis of results at the parameter level and on the basis of knowledge about impact in the area in question.

See GG to Chapter 2 Section 9

Some examples of situations (not ranked in order) that should lead to further review of the status classification:

- The result deviates from the water authority's perception of the status in the water body.
- One or more values in the included data set deviate significantly from the others. A deviation that can be seen with the naked eye, in for example a time series, should be sufficient cause to make to a more in-depth analysis in order to find any explanations for this. Chapter 4 deals in more detail with what one should think about when deviations are found in the supporting data.
- The result of a status classification lies close to a class boundary which can lead to a requirement for measures (the boundary between high and good or between good and moderate status).
- There is a lack of available data for the status classification. The classification rests on only one or a small number of samples.
- An analysis of impact data gives a result opposite to that of the status classification.

If Yes: Go to point 8.

If No: If the result is not regarded as reasonable, an investigation must be performed in accordance with the procedure given under point 7, with the aim of improving the available data set.

7. Reasonability and uncertainty assessment of status classification

See REG Chapter 2 Section 9

For a more detailed description of reasonability assessments, please see Section 4.1.1. If the result of the status classification at the parameter level does not seem reasonable:

<u>7 a. Can it be due to deficiencies in the available data set?</u>

If Yes: When the classification is based on data from only one site, one sample or parameter, it probably contains a large degree of uncertainty and caution is therefore needed in drawing hasty conclusions from it. If moreover the classification differs from the result perhaps expected, it should be further investigated. An explanation should always be sought for a devia-

See GG to Chapter 2 Section 9

tion, for example an overflow from a wastewater treatment plant or high water flux, before ignoring a value, but if no event can be found that explains the deviation, it could in certain cases still be appropriate to ignore it in the assessment. Irrespective of the reason for ignoring the value, this must be justified and documented. This is part of the expert judgement that is described in greater detail in Chapter 4.

If No: Go to point 7 b.

7 b. Can it be because of monitoring deficiencies?

If Yes: When there are deficiencies in the monitoring, such as a failure to follow routines, e.g. samples taken using an inadequate method, or sampling stations that have been placed at inappropriate sites, the results obtained can be ignored when classifying the status. This must, however, be justified and documented. If it is only a matter of temporary deficiencies, during a single or small number of sampling events or parameters, it is sufficient to ignore the results from these samples or parameters. This is part of the expert judgement that is described in greater detail in Chapter 4.

If No: Go to point 7 c.

7 c. Can it be explained by natural causes?

When a deviant value can be explained as a result of unusual natural circumstances, it may be appropriate to give it less importance in the classification. In order to be able to judge what is normal and what is unusual, these values can be compared with meteorological data and the times series data that is available for environmental monitoring trend stations. Both meteorological (e.g. temperature) and hydrological elements (e.g. water levels or high water flows) are examples of natural circumstances that can influence a sampling value. There is more information in Section 4.1.1.2 about what can affect the value of a measured parameter.

If Yes:

- If a lot of data is available and it is only from one year, or a limited period that deviates, the mean values for e.g. three-year periods may be used to reduce the importance of the deviating values.
- If data is available from several parameters or quality elements, priority can be given to what are known as "robust" parameters, i.e. parameters or quality elements that react slowly to environmental changes. These ought to give a more long-term picture of the environmental condition.
- If status or data for other water bodies within the same type-group are available, classifications it may be appropriate to compare the values or assessments with these in order to evaluate reasonability. A prerequisite for the comparison is that the other data has not also been affected by the same deviating natural events.

- When all available data originates from a naturally deviating time-period, consideration can be given to ignoring results for the parameters in question. This must, however, be justified and documented. If it is only one or a few sampling events for which the parameter in question deviates, these values may simply be ignored.

If No: Go to point 7 d.

7 d. Does the classification for any of the parameters lie close to a class boundary?
When there are parameter values that lie close to a boundary between two classes, it may be necessary to investigate further whether the classification is correct. The difference between falling into one class, rather than the other, may possibly depend on natural events, such as heavy rain in the period when the field sampling was carried out. Since the difference between being assigned to one class or the other can be of crucial importance for whether or not measures are to be implemented, it is important to take this into account.

If the result of status classification lies close to one of the class boundaries between high and good, or good and moderate, it is appropriate to investigate the following:

- Have the samplings been adequately carried out?
- Have the samples been correctly handled?
- How can natural variation have affected the result?
- What effect can anthropogenic impact have had on the result?
- Is there anything else that may have affected the sample result?

In Section 4.1.2 there is more information about uncertainty assessments.

If Yes: The following should be considered to improve the above assessment result

- Measure other parameters, or measure several times to verify the classification. The Handbook on surface water monitoring contains more information about monitoring strategies.
- Assess previous and current impact in the sampling area. For more information about impact analysis, please consult the Handbook on mapping and analysis of surface water.
- If possible, assess the trends for the relevant parameters.

See GG to Chapter 2 Section 8

If No: If, after this procedure, the result is assessed as reasonable, the classification stands and we can proceed to point 8.

If the further investigation confirms that the assessment is not reasonable, the result of the status classification of the parameters in question may be ignored (see Section 4.1.1). This must be justified and documented.

See REG Chapter 2 Section 9

If a satisfactory status classification has been achieved by means of one of the procedures in points 7a-d, for example on the basis of additional sampling etc., proceed to point 8.

8. Co-weighting the parameters or indices at the quality element level
For each assessment criterion, there are already explicit methods for co-weighting parameters (see Annexes A, B and C). The general procedure is however:

Step 1. Compile the available parameters. Sort them on the basis of which impact pressure each respective parameter responds to (e.g. acidification, eutrophication, other).

Step 2. Co-weight the parameters within a quality element that responds to the same impact, in accordance with the instruction in the assessment criteria in Annexes A, B and C, to obtain a common value. For e.g. phytoplankton in coastal waters, both chlorophyll and biovolume respond to eutrophication, which means that these shall be co-weighted into a phytoplankton class for the impact of eutrophication (see Figure 4.9).

Step 3. Thereafter, for each quality element, parameters that relate to different impacts are co-weighted, in accordance with the "one out – all out" principle (Section 4.2.4). Thus, parameters that respond to e.g. acidification or eutrophication shall be co-weighted into an overall assessment for each quality element.

9. Status classification according to assessment criteria - co-weighting quality elements into ecological status

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After the parameters within each quality element have been co-weighted, all quality elements are then co-weighted for each water body, in accordance with the "one out – all out" principle. The way this is done is illustrated in Figure 4.9. First co-weight the biological quality elements. If they indicate moderate status, or worse, that also becomes the result for the ecological status, since it is then of less importance what the physico-chemical or hydromorphological quality elements indicate. A programme of measures must in any case be established. If the biological quality elements indicate high or good status, the physico-chemical quality elements are classified. If the physico-chemical quality elements then show moderate or worse status, the ecological status will be classed as moderate. If both the physico-chemical and the biological quality elements indicate high status, the hydromorphological quality elements are also classified. If they indicate good or worse status, the ecological status will be good. If the hydromorphology also indicates high status, however, the water body must be classified as high ecological status.

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If the various quality elements give different results, a reasonability assessment may be carried out in accordance with the procedure described in point 7.

10. Checking routine for physico-chemical parameters

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Section 12

As a final step in this checklist, it should be considered whether the physico-chemical parameters relate accurately to the biology. The biological quality elements rank above the physico-chemical and hydromorphological quality elements. Under EU guidance no. 13, it is therefore possible to revise the limit values for the physico-chemical quality elements in a specific water body if it is obvious that the biology has not been disturbed, despite the fact that the physico-chemical quality elements indicate disturbance or vice-versa (see also Section 4.3.2 and Figures 4.10 and 4.11).

See GG to Chapter 2 Section 12

However, it is only the boundary between the classes moderate and good for the physico-chemical quality element that can be subject to adjustment in accordance with this model. The water authorities may make this adjustment for only a small number of individual water bodies within one type.

3.3 Determining quality requirements

3.3.1 Quality requirements for ecological status

Environmental quality standards for ecological status shall be established in accordance with the Water Management Ordinance (WMO). What follows below does not, however, apply to water bodies that are subject to exemption (see Chapter 4, Sections 9-13 of the WMO) or are characterised as heavily modified or artificial water bodies (see further in Section 3.3.2).

In order to take a decision about environmental quality standards, the quality requirements for the water body must first be determined. Quality requirements for natural water bodies are determined on the basis of the class boundaries between high and good, or between good and moderate, ecological status for the quality elements. Boundaries for these are given for each assessment criterion respectively in Annexes A, B and C. Support and guidance for correctly determining quality requirements can be found in the checklist for determining quality requirements for ecological status in Section 3.2.3.

In accordance with the Water Management Ordinance, quality requirements for surface water must be determined so that the status in surface water bodies does not deteriorate and so that all surface water bodies will by 22 December 2015 achieve good surface water status in accordance with the provisions in Annex V of the WFD. This means that if the ecological status of a surface water body has been classified as high, the quality requirement shall be determined as high ecological status on the basis of the “no deterioration requirement”. If the ecological status has instead been classified as good, moderate, poor or bad, the quality requirement shall be determined as good ecological status. If the status is worse than high, the quality requirement cannot, however, be determined as high pursuant to the Water Management Ordnance. If the water authority/county administrative board wishes to raise the level of ambition, this can be done by means of other instruments, such as area protection, or special regional environmental objectives, as part of the work on environmental quality objectives at the national level.

Decisions on environment quality standards for a specific surface water body mean that the water authorities take decisions on what environmental quality (high or good ecological status) is to be achieved by 2015 for each water body respectively. Environmental quality standards shall be reported in the WISS (Water Information System Sweden) database or equivalent.

See REG Chapter 3 Section 5

3.3.2 Quality requirements for ecological potential

If the hydromorphology of a water body has been so disturbed that the water body is declared heavily modified or artificial, the quality requirements are set determined for ecological potential. Quality requirements are determined as either maximum or good ecological potential. No uniform national assessment criteria have, however, been developed for quality elements that have been disturbed by changes in hydromorphology in a water body that has been declared heavily modified or artificial. This is because each water body will require a specific, tailor-made assessment criterion.

See REG
Chapter 3
Section 3

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Section 4

Quality requirements for good surface water chemical status shall be determined for heavily modified or artificial water bodies in the same way as for natural water bodies (see Section 3.3.3 and Annex C).

The classification of potential and the determining of quality requirements for heavily modified waters are largely based on the same fundamental principles as for natural water bodies. What distinguishes them is the possibility of permitting some degree of hydromorphological impact. Support and guidance for correctly determining potential and quality requirements are given in Section 3.4.

If the ecological potential in a surface water body has been classified as maximum, the quality requirement must be set as maximum ecological potential. If the ecological potential of a surface water body has instead been classified as good, moderate, poor or bad, the quality requirement must be determined as good ecological potential.

The above does not, however, apply to water bodies that are subject to exemption (see Chapter 4, Sections 9-13 of the WMO and forthcoming guidance material on exemptions).

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Section 5

The environmental quality standards determined shall be reported in the WISS (Water Information System Sweden) database or equivalent.

3.3.2.1 WHEN HYDROMORPHOLOGICAL IMPACT MUST BE TAKEN INTOACCOUNT

Impact from a physical modification of a water body may be taken into account when determining quality requirements for biology and hydromorphology. As regards physico-chemical elements, account may be taken of the hydromorphology only in cases where a parameter is regarded as disturbed by it[13]. This can apply e.g. for parameters under the general conditions such as oxygen, temperature and turbidity.

[13] European Commission (2003). Identification and Designation of Heavily Modified and Artificial Water Bodies. Guidance document no 4. Common Implementation Strategy for the Water Framework Directive (2000/60/EC).

As regards the specific synthetic pollutants, however, hydromorphology may not be taken into account and when classifying potential, the same requirements always apply to them as when classifying status.

3.3.2.2 HOW HYDROMORPHOLOGICAL IMPACT IS TAKEN INTO ACCOUNT

The impact which may be permitted when determining maximum ecological potential is the impact which remains after all mitigation measures have been implemented. Such hydromorphological measures shall not include measures that have a significantly negative impact on the purpose of the modification. This means that, for example, a harbour must still be capable of use as a harbour even after the mitigation measures have been implemented (see also Section 3.4.2).

Even after mitigation measures have been implemented, the status shall in practice be ecologically comparable to a natural water body within the same type, particularly as regards migration and spawning grounds for fish. Here there are at present two methods that can be used. One is based on EU guidance no.4[14] and the other on Annex 2 of *CIS ECOSTAT: Alternative methodology for defining Good Ecological Potential (GEP) for Heavily Modified Water Bodies and Artificial Water Bodies*[15] (see Section 3.4).

See GG to Chapter 2 Section 5

In this context, EU guidance no. 4 may be considered as the first official guidance, while the method to be found in CIS ECOSTAT is proposed as an alternative approach which was developed when the first was found to be very complicated to use. It is regarded as a matter of indifference which of these two alternatives is chosen.

3.3.3 Quality requirements for surface water chemical status

Good surface water chemical status means that a water body must not have higher levels of toxic substances than the levels stipulated in the quality requirements existing at the Community level. Substances included in the classification of surface water chemical status are those that have common EC limit values, i.e. substances and substance groups regulated by the EU Freshwater Fish Directive and the Shellfish Directive. These have been implemented by Ordinance (2001:554) on environmental quality standards for fish and bivalve waters, and the prioritised substances designated within the WFD and regulated in the forthcoming daughter directive (see Section 3.1.3). The water authority must set quality requirements for all water bodies at good surface water chemical status according to the limit values in Chapter 5. This cannot be done until the daughter directive for priority substances has been adopted and the Water Management Ordinance has been updated.

See REG Chapter 3 Section 4

[14] Ibid.

[15] CIS ECOSTAT (2006): Alternative methodology for defining Good Ecological Potential (GEP) for Heavily Modified Water Bodies and Artificial Water Bodies.

3.3.4 Checklist for determining quality requirements for ecological status

After carrying out all steps in the checklist for classification of ecological status, the classification of ecological status is complete. If quality requirements for a water body have not previously been determined, or if the water authority intends to revise them on the grounds of the result of the above status classification, this can be done in accordance with the checklist below.

1. Determining or revising quality requirements

Under Chapter 4, Section 2 of the WMO, the status may not deteriorate for any water body. For water bodies that have been classified as high ecological status, after completion of the checklist for status classification, the quality requirement is therefore determined at the boundary between high and good status. For water bodies that have been classified as good ecological status or worse, the quality requirements are determined at the boundary between good and moderate status (Figure 3.5). The starting-point for the above is the class boundaries for the assessment criteria.

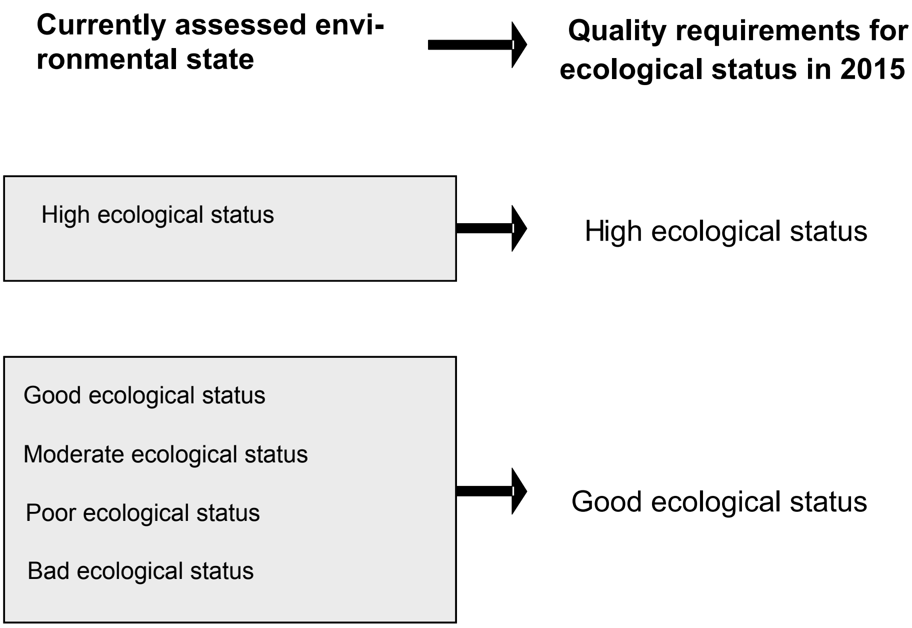

Figure 3.5. Starting-point for quality requirements. What is currently classified as high ecological status must not deteriorate to a lower status. What is today classified as good or lower ecological status must be retained as good ecological status or improved in order to achieve good status by 2015. However, possible exemptions may affect the final standard-setting.

2. Can the water body be subject to exemption?

If good ecological status cannot be achieved, and this is not due to hydromorphological impact, the water body can instead be subject to exemptions such as an extended *time-limit* or *less stringent quality objectives*. More information about determining quality requirements and establishing environmental quality standards regarding exemptions can be found in Chapter 4, Sections 9-13 of the WMO and in forthcoming guidance material about exemptions.

3.4 Classification and determination of quality requirements for ecological potential

No specific national assessment criteria have been developed for classifying potential. With a view nonetheless to giving some guidance, and in order to ensure the classification and determination of quality requirements for potential are implemented as uniformly as possible, a possible approach is described in this chapter.

3.4.1 Similarities and differences in the classification of natural and heavily modified and artificial water respectively

Exactly as for natural water bodies, the water authority must classify the present status of heavily modified and artificial water, what status must be achieved in a specific water body, whether this has been reached or not and whether the water body is in need of measures. Similar to natural water bodies, these water bodies can also be subject to exemptions if good ecological potential cannot be reached and the cause does not depend on hydromorphological impact (Chapter 4, Sections 9-13 of the WMO). The difference in this respect is that the status of these water bodies is termed "potential" instead of "status".

For those quality elements that are impacted by the hydromorphological modification, the water authority needs to:

- establish a specific assessment criterion for the relevant quality elements for each individual water body, in which it is made clear what value must be reached in order for a quality element to achieve maximum and good potential.
- measure (or assess) the present state of each relevant quality element.

The quality elements not affected by altered hydromorphology are classified with the aid of the usual assessment criteria for the category that best corresponds to the heavily modified or artificial water body. The status class thereby obtained is transposed to the corresponding potential class.

That shall then result in:

- the determination of quality requirements for the specific water body (see Figure 3.6).
- information about how the potential for the relevant quality elements stands in relation to the environmental quality standard.

Under Chapter 4, Section 2 of the WMO, the status may not deteriorate for any water body. For water bodies that have been classified as maximum ecological potential, the quality requirement is therefore determined at the boundary between maximum and good potential. For water bodies that have been classified as good ecological potential or worse, the quality requirement is determined at the boundary between good and moderate potential (Figure 3.6).

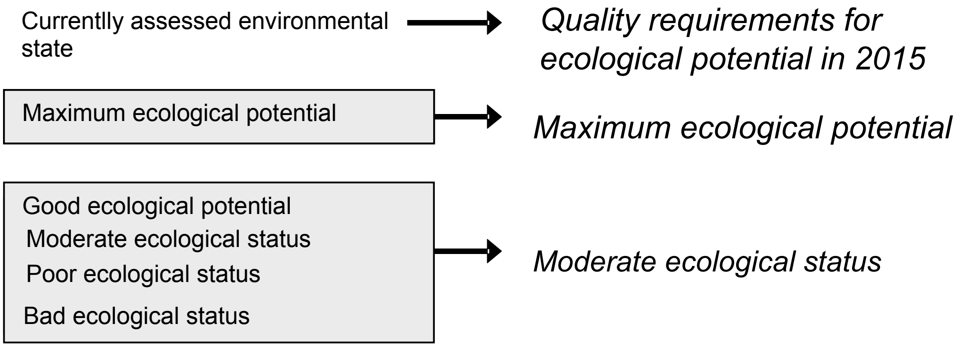

Figure 3.6. Starting-point for quality requirements. Water bodies that are currently classified as maximum ecological potential may not deteriorate to a lower potential. What is currently classified as good or lower ecological potential must be maintained as good ecological potential or improved to reach good ecological potential by 2015. However, possible exemptions may affect the final standard-setting.

See REG Chapter 2 Sections 4-5

For heavily modified or artificial water bodies, the boundary between maximum and good potential is assessed by determining the biological conditions that prevail when the only impact is the impact originating from altered hydromorphology (brought about by the activity that justified the classification). Before the boundary is determined, the conditions to be expected, after implementation of all mitigation measures, must be identified. Mitigation measures include measures that do not have a significantly negative effect on the purpose of the modification. The boundary between good and moderate potential is then determined by assessing the effect of a small deviation of the biological quality elements from maximum potential. Thereafter the biological conditions are determined that are deemed to prevail after this deviation has been made.

3.4.2 Guidance from the EU

A possible method of classification of potential has been described in two guidance papers from the EU (see Section 3.3.2.2 for the full reference). In the first guidance paper, here called the CIS method, this is carried out by comparing the heavily modified or artificial water body with the nearest comparable natural water body that is undisturbed and that belongs to the same category and type. If no such water body is available, a comparable modified water body that has been classified as having maximum ecological potential may be used as a comparison. In this way, reference values are developed for the parameters concerned that are subsequently applied to the modified water body when classifying ecological potential.

An alternative to the CIS method is the ECOSTAT method, which was produced because the CIS method was regarded by a number of countries as difficult to use. Here the boundary between maximum and good potential is assessed by assessing the ecological conditions that would prevail should "all mitigation measures" be implemented. These mitigation measures should lead to an ecological improvement without entailing any significantly negative effect on the activity that gave rise to the modification, or on the environment as a whole. The forthcoming guidance on artificial and heavily modified water bodies will deal with questions concerning the concept "significantly negative impact" and mitigation measures in more detail. In the meantime, there is only the text given in Chapter 4, Sections 9-13 of the WMO and a technical report with case studies from the EU[16]. The ECOSTAT method assesses the boundary between good and moderate ecological potential by estimating the conditions that would prevail if only mitigation measures with a significant ecological effect were to be implemented. In other words, the method excludes measures considered to have only slight ecological effect. The assessment as to which measures are to be regarded as ecologically significant must be made on a case-by-case basis by means of an expert judgement. Please see Section 3.4.3.3 for more information about ecologically significant measures.

3.4.3 Guidance for classifying and determining quality requirements for ecological potential

The guidance described in this handbook aimed at assessing potential is based on those parts of the CIS and ECOSTAT methods that the Swedish EPA deems to be currently feasible. Support in this work is provided in additional EU guidance material, entitled 'Toolbox on identification and designation of artificial and heavily modified water bodies'[17].The guidance paper, which has been produced with a view to making EU guidance no. 4 more concrete, proposes different methods for determining maximum and good potential based on both the CIS and ECOSTAT methods, as well as for different situations and different categories of water bodies. The guidance paper also gives examples of mitigation measures that have already been implemented, in the form of case studies discussing their advantages and disadvantages.

3.4.3.1 ALL RELEVANT PARAMETERS SHALL BE ASSESSED

Irrespective of whether the water body has only been disturbed by the hydromorphological modification or whether it has also been exposed to other impacts, the water authority shall always classify its ecological potential (Figure 3.7). This means that it is also important to classify quality elements that are not disturbed by the modifications, with a view to establishing whether there is any other impact, and also to determine which quality elements indicate the worst status, in accordance with the "one out – all out" principle. Only thereafter can the quality requirements that are to apply to a particular water body be indicated. As regards the

[16] improvement of ecological status/ potential by restoration/ mitigation measures. Separate Document of the Technical Report, November 2006

[17] European Commission (2003). Toolbox on identification and designation of artificial and heavily modified water bodies

hydromorphological quality elements, the "one out – all out" principle will in many cases also apply at the parameter level, because the parameters indicate different impact pressures. What applies to the respective hydromorphological quality elements is described in more detail in the various sections about them in Annex C.

3.4.3.2 THE RIGHT CATEGORY

For heavily modified water bodies, an assessment must also be made of whether or not the modification has resulted in the water body changing category. An example of a water body that, despite modification, has not changed category may, for example, be a watercourse that can afterwards still be regarded as resembling a watercourse. An analogous example of a water body that on the contrary has changed category can of course be a watercourse that after modification is more reminiscent of a lake. Here an assessment must also be made as to whether the quality elements have attained a new equilibrium in accordance with the new category. If the water body cannot be assigned to a category or type in accordance with the assessment criteria, no parameters can be status-classified based on the assessment criteria. The question of whether a new equilibrium has been adopted or not is of course difficult to assess and this must be done in the form of an expert judgement. A prerequisite is that the ecology is over time assessed to have adapted itself, so that the quality elements in principle correspond to those that could be expected in a natural water body in the same category. However, this does not always happen, even if much time has elapsed since the water body changed category. On the contrary, it seems that e.g. flow regulation in power station and regulation reservoirs continues even in the longer term to cause a constant disturbance to shoreline biodiversity. For example, in a major study of river-bank vegetation along regulated rivers in northern Sweden, no signs could be found of long-term recovery of species abundance[18].

[18] Nilsson, C., R. Jansson & U. Zinko. 1997. Long-Term Responses of River-Margin Vegetation to Water-Level Regulation. Science 276:798-800

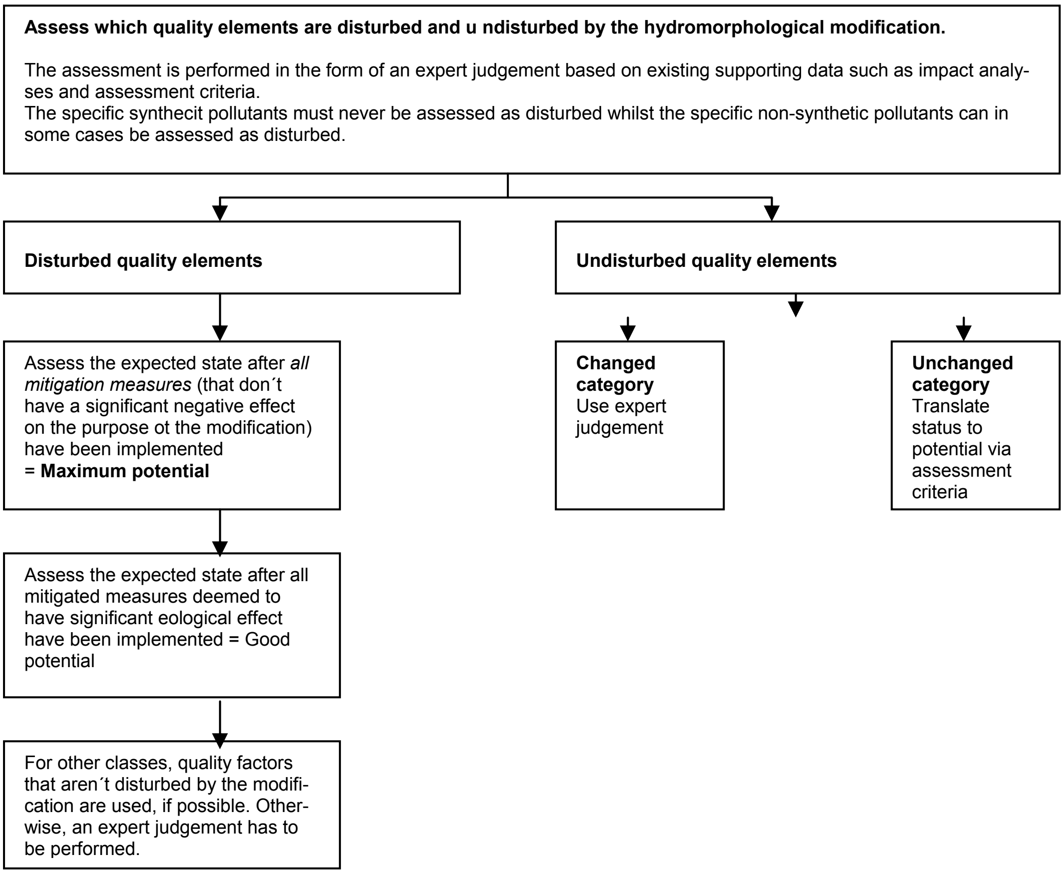

Figure 3.7. General timetable for classification of heavily modified and artificial water bodies.

3.4.3.3 METHODOLOGY FOR CLASSIFYING POTENTIAL AND DETERMINING QUALITY REQUIREMENTS

A working method for classifying potential and determining quality objectives for heavily modified and artificial water bodies is proposed below.

Step 1: Assess which quality elements are, or are not, affected by the hydromorphological modification.

Classification of quality elements that are affected by the hydromorphological modification

When classifying potential for the quality elements that are affected by the hydromorphological modification, no national assessment criteria have been developed. This is because each individual water body would require a specific and tailor-made assessment criterion. A proposal for how such an assessment criterion might be developed is given below.

Step 2: Establish an assessment criterion for the relevant quality elements

The assessments below should of course be made on quality elements that are regarded as relevant and possible to follow up. The quality elements included in the assessment criteria for status (for the category and type that most closely resemble the heavily modified water body) can for this purpose be used as a first recommendation for conceivable quality elements. In many cases, however, the changes in hydromorphology cannot be linked to any of these quality elements and it can therefore also be relevant to identify other quality elements. If the modification consists of a migration barrier, the number of fish species may, for example, be an alternative instead of the existing assessment criterion for fish, which consist of an index in which the effect of the modification is not always identified.

Even if each assessment is unique for each water body, it can be a good idea for the water authorities to enter all their classifications in a common database, in order to be able to benefit later on from each other's experiences, since in many cases they will be faced with similar assessment situations.

Step 3: Maximum potential

The boundary between maximum and good potential consists of a state in which all mitigation measures that do not have a significantly negative effect on the purpose of the modification have been implemented. This applies particularly to the migration of aquatic organisms and suitable spawning and breeding grounds. The next step is consequently to make an overall survey of such measures. In this context it is only hydromorphological measures that have to be taken into account. On the basis of this survey, an assessment is then made of the combined effect that "all mitigation measures" would have on the ecology if they were implemented. This assessment can be conducted in the form of an expert judgement with the support of existing knowledge, assessment criteria, modelling or other relevant supporting data. In the assessment, for example, other water bodies in the same category, and with a hydromorphology that has not been disturbed (or which show maximum ecological potential) can be a support and give a perception of what status can be expected for the quality elements that may be regarded as relevant.

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Section 5

It is not considered possible to give a standard definition of what a significantly negative effect on the purpose of the modification would entail, because what is significant will vary from case to case. Consequently, it is a matter of assessing in each case what measures can be required without the risk of the purpose of the modification not being able to be fulfilled. The basic idea is, for example, that even after implementation of mitigation measures, a harbour can still be used as a harbour, an embankment can still give protection against flooding and a protected area can still accommodate the biological values for which it was protected.

As a support for which measures that could be included in the concept of mitigation measures, the report "WFD and hydromorphological pressure[19]" , which

[19] WFD and hydromorphological pressure, Technical report, Case studies, Potentially relevant to the improvement of ecological status/ potential by restoration/mitigation measures. Separate Document of the Technical Report, November 2006.

was produced by an EU Working Group, can be used. The report lists case studies and potentially relevant measures. It also indicates their cost, and their significance as regards the purpose of the modification and as regards ecology. The report will be available on the Swedish EPA's website.

If the measures in reality have a negative effect on the purpose of the modification or not needs to be assessed from a holistic perspective, where the combined effects of all measures which do not have a significantly negative effect on the purpose in a specific water body are assessed

Step 4: Good potential

For the purpose of determining the boundary between good and moderate potential, all measures that have "significant ecological effect" can be combined.

See REG Chapter 2 Section 5

On the basis of this combination, an assessment is then made of what effect "all ecologically significant measures" would have on the ecology if they were implemented. This assessment may be conducted in the form of an expert judgement supported by existing knowledge such as assessment criteria, modelling or other relevant background data. Here too the report mentioned in Step 3 (WFD and hydromorphological pressure) can serve as good background material. In this context it is important to keep in mind that one and the same measure can have a significant ecological effect in one water body but not in another. What is really significant must therefore always be determined from case to case.

The above assessment is made on quality elements that are regarded as relevant and possible to follow up. The assessment criteria for status (for the category and type that most closely resemble the heavily modified water body) can be used as a first recommendation for conceivable quality elements. Exactly as in Step 3, it can in many cases be difficult to link the altered hydromorphology with one of these quality elements and it can therefore also be relevant to identify other quality elements.

The conditions that are assessed to be a result of the above measures provide the boundary between good and moderate for the quality elements that are affected by the altered hydromorphology.

Step 5: Moderate, poor and bad potential

The boundaries between these classes can in this proposed working method correspond to the same class boundaries that apply in the national assessment criteria, which are used in status classifications if they are appropriate (see also Step 9).

Step 6: Compare the observed or estimated status with the assessment criteria drawn up in accordance with Steps 3, 4 and 5.

To classify a water body as maximum potential, the measured, or in some other way estimated, status must be compared with the status that would be achieved implementing "all mitigation measures" (Step 3).

To classify a water body as good potential, the measured, or in some other way estimated, status must be compared with the status that would be achieved by implementing "all ecologically significant measures" (Step 4).

To classify a water body as one of the lower classes, moderate, poor and bad potential, appropriate assessment criteria can be used in the comparison of the measured, or in another way estimated, status (Step 5). Alternatively, an expert judgement can be carried out.

Classification of quality elements that are not affected by the hydromorphological modification

Step 7: Water bodies that despite modification have not changed category or have changed category but adopted a new equilibrium.

The quality elements that are deemed not to have been affected by a change in hydromorphology are classified with the aid of assessment criteria for the category that best corresponds to the heavily modified or artificial water body. The status class thereby obtained is substituted by the corresponding potential class. Otherwise the same classification procedure is used as described in Section 3.2.

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Step 8: Water bodies that have changed category because of a modification but have not yet had time to adopt a new equilibrium.

For quality elements that are deemed unaffected by changes in hydromorphology, an expert judgement is made of the potential, based on available knowledge about the status and impact (impact analysis), in order to determine the potential.

Determining quality requirements for ecological potential

Step 9: Determining quality requirements

The worst class that has been identified with the aid either of measures or the unaffected quality elements is then used to determine the quality requirement for the water body (according to the "one out – all out" principle) as maximum or good ecological potential.

If all quality elements have been classified as maximum potential, the quality requirement is determined as maximum ecological potential.

If the worst quality element has been classified as good potential or worse, the quality requirement is determined as good ecological potential.

Good ecological potential can be regarded as achieved when:

- all quality elements that are not affected by the modification have achieved good potential
  and
- ecological conditions, for quality elements that are affected by the modification, show only minor changes as compared with the values that lie at the boundary between maximum and good potential.

3.4.4 Artificial water bodies

The potential for an artificial water body can in principle be classified in the same way as the potential for a heavily modified water body. Since there was previously no water body at all in these cases, there aren't any reference values for the artificial water body. It can, for example, be a matter of an excavated reservoir or canal at a site where previously there was no water body of importance. That makes it difficult to identify a suitable category and type for which assessment criteria have been developed, which in turn means that assessment criteria must be used with great caution and that in many cases it will prove necessary to carry out an expert judgement directly.

3.4.5 When the water body is subject to exemption

If good ecological potential cannot be achieved in accordance with Chapter 4, Sections 9–11 of the WMO, the water body can on the same grounds as for a natural water body instead be the subject to exemptions, such as an extended time-limit, or be designated as a water body with "less stringent quality requirements". In the present situation there is no written guidance as regards exemptions when determining environmental quality standards and until further notice reference is therefore made to Chapter 4, Sections 9-13 of the WMO and forthcoming guidance on exemptions.

4 Classification of status - a more detailed description

4.1 Status classification in accordance with assessment criteria on the parameter level

4.1.1 Reasonability assessment

4.1.1.1 REASONABILITY ASSESSMENT – A WORKING PROCEDURE

Classifications of ecological status should normally be done preferably on the basis of data for several years (mean value, the median or similar) to reduce the importance of individual years. However, this varies between different assessment criteria and Annexes A-C show suitable intervals between measurements for each assessment criterion respectively. Despite this, it is important in an evaluation to examine and assess the reasonability not only of the final result but also of the results for individual years. This is done in order to ensure that no extreme events render the classification incorrect. Very divergent hydrological periods and individual point discharges are examples of such extreme events. Meteorological fluctuations in the climate are another example of a slower "extreme" event, which because of their impact over a longer period can affect both the physico-chemical and biological composition. The intention is to adjust the latter by revising reference values. When a classification deviates from what is regarded as reasonable, a reasonability assessment should be carried out.

See REG Chapter 2 Section 9

A draft working procedure for reasonability assessments is given in Figure 4.1. It is even more important to carry out reasonability assessments of the classifications obtained when they are close to a class boundary which requires mitigation measures and the result means that a class change may be indicated for the object under assessment. A reasonability assessment includes both an investigation of whether it is individual observations or years that deviate from the expected result and a comparison of the result with e.g. local trend indicators in the national environmental monitoring programme. It is hoped that comparison will throw up individual deviating observations resulting from analysis and input errors, misclassifications and other non-representative values. These deviating observations can then be excluded if there is good reason.

See GG to Chapter 2 Section 9

When it is established that the deviations are correct, the next step is to find out whether other comparable objects in the area show similar deviations. Comparable objects can for example be lakes, watercourses or coastal water in adjacent land areas, which are basically of the same type as the object in question. If no other objects in adjacent areas show similar deviations, it is appropriate to investigate whether there is any local impact, for example a point discharge, which can have caused the deviation. When other objects in the area show the same tendency, it is on the other hand reasonable to assume that some more extensive impact is the cause and then, based on whether it is a natural or an anthropogenic change, decide whether the observation, or that year's result, can be excluded from the classification on those grounds. A major impact of anthropogenic origin can for example be atmospheric deposition of acidifying substances or environmental toxins. A large scale impact originating in natural events can be e.g. abnormally large or small precipitation, which has a major impact on inland waters. It should, however, be noted that complex links often govern how material is transported from land to water and among other things the circumstances prior to the extreme weather situation may be a crucial factor. A long period of dry weather can, for example, cause organic material to accumulate in the catchment area which is subsequently washed out into lakes and watercourses when it begins to rain. Conversely, a long period of high water results in less organic material being washed out, which means that the levels in the water may be lower than normal.

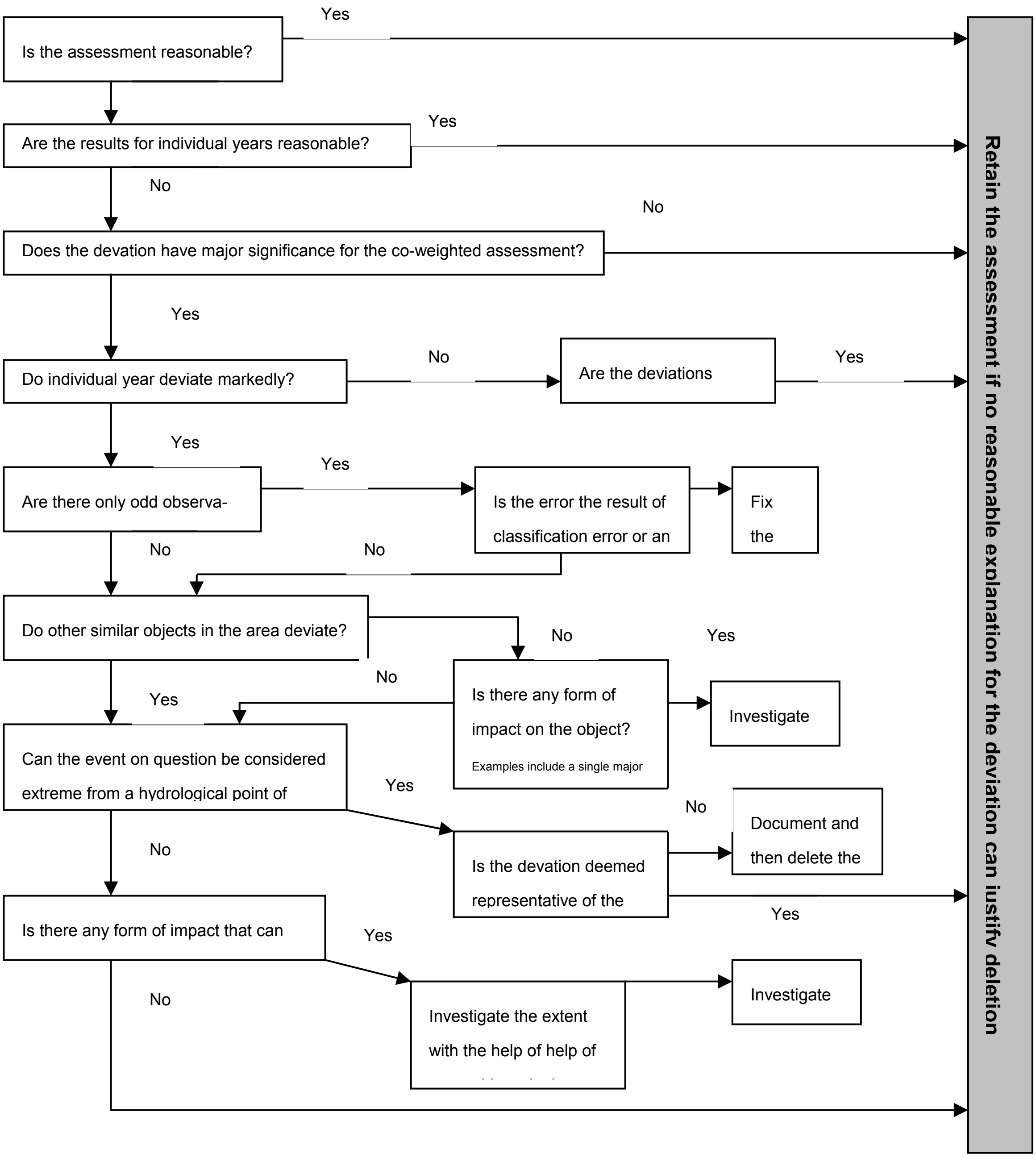

Figure 4.1. A flow diagram describing a reasonability assessment in connection with status classification.

4.1.1.2 REASONABILITY ASSESSMENT – CAUSES OF DEVIATIONS

When a reasonability assessment is to be made, it is appropriate to take into account both biotic and abiotic circumstances. Often it can be simplest to begin with abiotic circumstances, such as meteorological or hydrological factors. In some years, local or regional extreme events can lead to deviations from the expected values for a parameter. There are various causes for such events and some examples are given below.

Examples of meteorological or hydrological factors that can be the cause of deviations:

- Temperature – can have an impact in many different ways; extreme temperatures can, for example, wipe out species.
- The timing of the onset of a season can, for example, affect the extent and composition of spring-flowering.
- Precipitation – heavy rain with consequent changes in water flux, transport of sediment, sedimentation and changes in salinity can have an effect on the species composition, by favouring or disfavouring different species. Moreover nutrients are often affected by increased or diminished run-off.
- Wind - the frequency of storms and gales can cause turbidity or displacement of water masses, which in turn can bring about both improvement (through improved oxygen levels) and deterioration (through increased sedimentation and sediment accumulation).
- Currents – may be combined with wind and, for example, cause large-scale saltwater intrusion through seawater exchange. These water exchanges are often positive since they improve oxygen levels in deep water; but they can also cause increased sedimentation or sediment accumulation, which in turn can have a negative effect on a number of plants and animals. Currents can also accumulate and transport algal blooms from the deep sea to the coastal areas.
- Water levels – extreme changes in water levels can lead to increased nutrient leaching, greater turbidity and the loss of shallow-water species.
- Heavy ice-cover or scraping by large ice-masses can lead to the loss of perennial rock-clinging plants.

When physico-chemical and biological quality elements are being classified, the response to the above changes may appear different, depending on which parameters are being classified. Different quality elements may have different reaction times to environmental changes. For example, physico-chemical quality elements often have a very short response time, while phytoplankton respond in a time-frame of weeks, and macrovegetation and benthic macroinvertebrates have a slower response, which integrates events over a longer period, even over several years. What a phytoplankton species experiences as an extreme event may have no impact at all on, say, a macrovegetation species. Different species have, moreover, different tolerances of extreme events and e.g. certain benthic macroinvertebrates can even survive relatively short periods of oxygen deficiency.

When biotic factors are to be taken into account, it is often a matter of secondary effects which many times are caused by human activities. Such factors are seldom identified by monitoring and can furthermore often be difficult to ascertain. An example of this can be the mass occurrence of algae, which may consist of for example toxic phytoplankton and drifting macroalgae, which can lead to temporary local oxygen deficiency. This can affect the species composition, among e.g. benthic macroinvertebrates or vegetation, in that certain species are lost. Another example, with a more direct anthropogenic cause, can be certain trophic changes. If all predator fish are fished out, this can result in large-scale changes in trophic structure and lead to changes in the species composition among phytoplankton or benthic macroinvertebrates. The Water Framework Directive (WFD) is nevertheless not intended to cover the impact of fishing, other than if there is physical damage to habitats etc. This means that if fishing has led to changes in the structure of the fish community, the reference value must be changed on the basis of the new conditions. This also applies analogously for anthropogenically caused climate change. Fishing and climate are dealt with in other policy sectors within EC legislation.

If a deviation or unreasonable value can be explained by any of the above abiotic or biotic factors, it may be excluded from the status classification, but it should nevertheless be retained and documented for possible use in future analysis. Values must hence not be deleted simply because they deviate. If a deviating value cannot be explained by any of the above, it must still be retained so that the reason can be investigated. Recovery from a deviation can be expected to take several years and the impact on a status classification can also be significant for several years. Even though it is believed that a cause has been found for deviation or a significant change in a parameter, it can be difficult to provide proof. The sharp decline in the normally dominant benthic amphipod Monoporeia affinis, which occurred along the coast of Norrland during the late 1990s and early 2000s, is an example (Figure 4.2). There are several hypotheses about the cause of the decline, most of them based on changes in temperature or precipitation. The recovery is slow and according to a model calculation will take at least seven years. By comparing several years' data from the Norrland coast with time-series from e.g. trend areas, it should nevertheless be possible to filter out such a large-scale change.

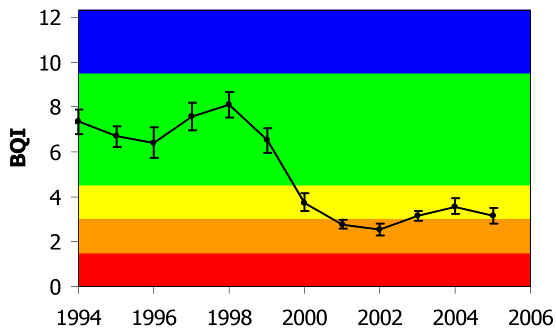

Figure 4.2. Changes in the benthic quality index (BQI) in Örefjärden. Spread measurements give a 60%-confidence interval according to the assessment criterion and classification is made with the lower boundary. Index calculated on data from 18 stations (1994: 11 stations). By 2000 the dominant species Monoporeia affinis has already greatly declined, which gives a major reduction in BQI because Monoporeia affinis is classified as a sensitive species. The recovery has still not yet really got going since then.

4.1.2 Uncertainty assessment

More measurements normally provide a more reliable classification of the parameter and its spread (standard deviation) and the uncertainty in the mean value (standard error) can be calculated for the water body concerned. In cases where only one year's data are available, the fixed value for method-bound uncertainty (standard deviation) for the respective parameters and types may be used, albeit with caution. In cases where this has been calculated, it is given for each assessment criterion respectively in Annexes A, B and C. The standard deviation gives a measure of how unreliable a classification is. In cases where an uncertainty interval around the ecological quality ratio (EQR) overlaps any of the class boundaries between high and good status or between good and moderate status, the calculated EQR-value lies very close to a class boundary. This indicates that a reasonability assessment should be made, as described in Section 4.1.1.2 above. See further under each assessment criterion respectively in Annexes A-C.

See GG to Chapter 2 Section 9

4.1.2.1 CERTAINTIES IN CLASSIFICATION OF ECOLOGICAL STATUS

Confidence and precision under the WFD

The classification of ecological status is an important element in the implementation of the WFD. Annex V, section 1.4.1.3 of the WFD provides that:

> "Each Member State shall divide the ecological quality ratio scale for their monitoring system for each surface water category into five classes ranging from high to bad ecological status, as defined in Section 1.2, by assigning a numerical value to each of the boundaries between the classes."

Furthermore Section 1.3 of the WFD provides that each Member State must provide information about the confidence and precision of the national monitoring programme:

> "Member States shall monitor parameters which are indicative of the status of each relevant quality element. In selecting parameters for biological quality elements Member States shall identify the appropriate taxonomic level required to achieve adequate confidence and precision in the classification of the quality elements. Estimates of the level of confidence and precision of the results provided by the monitoring programmes shall be given in the plan"

High uncertainty and low precision create a risk of classification error and consideration should be given to a number of steps to reduce it. Uncertainty in a classification can be revealed by estimating the uncertainty in the quality elements used in the status classification of the water body, stated as e.g. measured EQR ± x % uncertainty (standard error). If there is major uncertainty, producing an unsatisfactory classification, it is appropriate to take measures to reduce the uncertainty by:

- collecting more data (increased monitoring),
- improving the monitoring programme (e.g. stratified sampling) or modelling,

See GG to Chapter 2 Section 9

- using more quality elements that indicate similar response; and/or,
- reducing the number of quality elements that are used in the classification of the water body, i.e. by excluding quality elements that show high uncertainty and low precision.

Uncertainty in estimating a parameter

Each status estimate which is based on collected data is associated with a number of sources of uncertainty. It is well known that both field routines (e.g. sampling season, method, choice of habitat) and laboratory routines (e.g. sorting, identification of species, counting, data input) contain elements of uncertainty that can later affect a classification. In order to reduce their number, or at least to obtain better knowledge of the risk of classification error that can result from uncertainty, these sources of error should be quantified and taken into account when making a classification.

Examples of causes of uncertainty:

Sample collection

It can be important to take into account "person-bound variation", i.e. variation between different people. Different people can operate in different ways, despite the fact that they believe themselves to be following the same method. The precision of a sampling method depends on the number of samples, the number and type of habitats or the total area that has been sampled.

Sample handling and analysis

Several stages in sample handling or preparation can result in increased uncertainty. Examples of stages that can include uncertainty are sub-sampling in the sorting of samples, species identification, counting and data-input.

Natural spatial and temporal variation

Within each site there are variations that are both spatial (between samples taken at the same time, for example within one habitat) and temporal (between samples taken during different sampling events, e.g. annual variation) that are not linked with human impact. The spatial variation (spatial heterogeneity) affects the spread of species. This variation often depends on the scale, that is to say the distance between sampling points.

Impact variation

Environmental monitoring aims to detect environmental effects from human activity, known as “impact variation”. Impact variation can change rapidly if there is change in the human activity that causes it.

Confidence interval

Uncertainty in sample collection and handling, and also to some degree natural variation, should be reduced as far as possible, in order to increase the ability to distinguish anthropogenic impact from natural variation. Uncertainty caused by the collection and handling can be reduced by thorough standardisation of, and training in, the execution of field and laboratory methods. Spatial and temporal variation can, for example, be reduced by stratification in which the sampling is limited to a few types of environments or seasons.

A common method of expressing uncertainty in an estimated mean value is to calculate the "confidence interval". That means calculating an interval, within which the true mean value, μ, occurs with a given certainty (most frequently 95 %). To calculate it, estimated (or sometimes) existing information about the spread of the parameter and what is known as the "t-distribution" is used. The confidence interval is calculated as follows:

$$\mu = \overline{X} \pm tc_{crit} * \sqrt{\frac{s^2}{n}}$$

crit

where $\overline{X}$ = the estimated mean value,
t_{crit} = the critical value for t (determined by the uncertainty level and how many measurements the mean value is based on),
s^2 = the estimated variance and n is the number of readings the mean value is based on, ($\sqrt{\frac{s^2}{n}}$ = the "standard error). This expression means that the uncertainty in an estimated mean value diminishes with the increase in random-sampling (Figure 4.3).

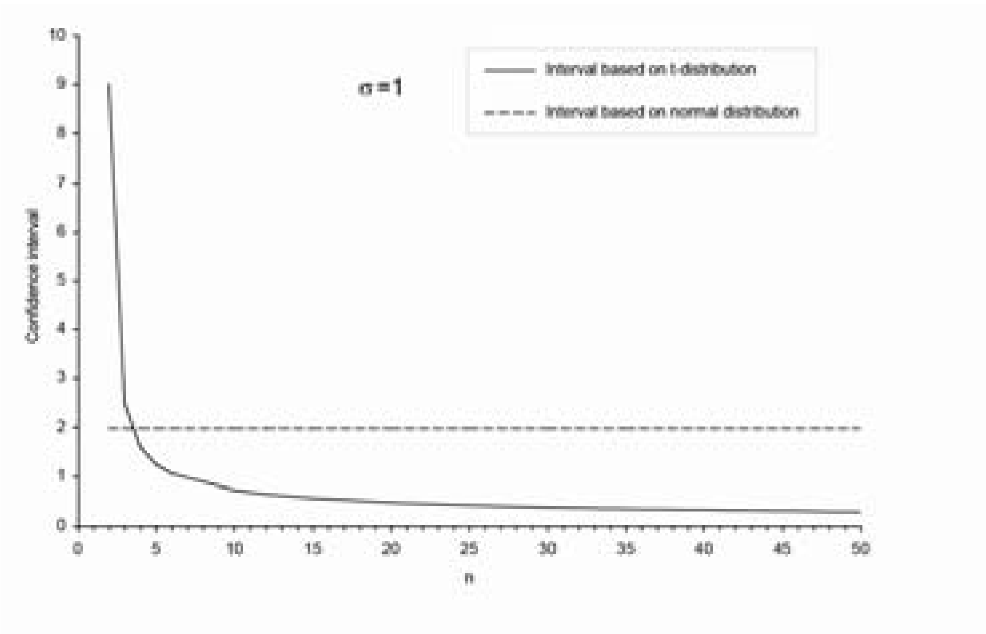

Figure 4.3. Uncertainty calculations of estimated mean values based on different numbers of random samples, with the aid of the t-distribution and assuming normal distribution. The example illustrates the uncertainty if the true standard deviation, σ, is 1.

For situations where only one measurement is available (no replication), it is impossible to calculate a confidence interval in accordance with the above principles. To obtain an idea of the population mean in such cases, it is necessary to have previous knowledge of the distribution of the parameter in question and what spread it may be expected to have. To avoid confusion, it may be mentioned here that the concept "parameter" in the statistical context is reserved for distribution characteristics such as mean value and standard deviation. When classifying status, "parameter" is used in a sense that corresponds to the concept "variable" in statistical literature. If, for example, it can be assumed that the parameter ('variable' is thus the more correct expression) has normal distribution and on the assumption that we have previous knowledge of the extent to which it can vary, a "population interval" can be calculated according to:

$$\mu = X \pm z_{crit} * \sigma$$

crit

where X = the measured value,
z_{crit} = the critical value for the normal distribution (for a 95% interval z_{crit}=1.96)
σ= the known standard deviation.

Thus in this expression no estimated information about spread is included (since none is available) and we can therefore not influence the precision of the mean value (Figure 4.3).

Note that for small random sample sizes (n<4), the confidence interval appears narrower for the latter method. This may seem contradictory, since two or three replications are, after all, more than one and therefore ought to give greater precision. It is however important to remember that this "improvement" can only come about if the assumption of a normal distribution holds true and if we have access to a good estimate of the population spread. Note also that only the first method offers opportunities to influence the precision. In random sample sizes of n≥4, the expected precision is thus greater using the method based on t-distribution, as compared with normal distribution (Figure 4.3).

In the strict sense normal distribution is only applicable if we have knowledge of the population's "true" spread, that is to say σ (compare the estimated variant *s* in the method with t-distribution). In step with the increase in the number of measurements (towards "infinity") we can, however, say that we approach this unattainable information and can in practice assume that we have complete information on the population spread. The question is however: How many random samples are necessary before we can consider ourselves close to the "true" spread? Through simulations it can be shown that if we take random samples from a normal distribution, the estimated standard deviation, *s*, deviates in a predictable way from the true, σ (Figure 4.4).

The observed relationship can be used to illustrate that approximately 70, 300 and 7 000 readings are required to attain 10, 5 and 1% deviation from the true value. Seen from a statistical perspective, it might perhaps be recommended that use of the formula for the population interval should be based on >300 measurements, since that gives an average error of 5% and can allow us to critically evaluate whether it is reasonable to assume normal distribution. The practical aspects and the costs may of course make it inexpedient, but it does at any rate indicate that classification is best based on a large number of measurements if it is to be reliable.

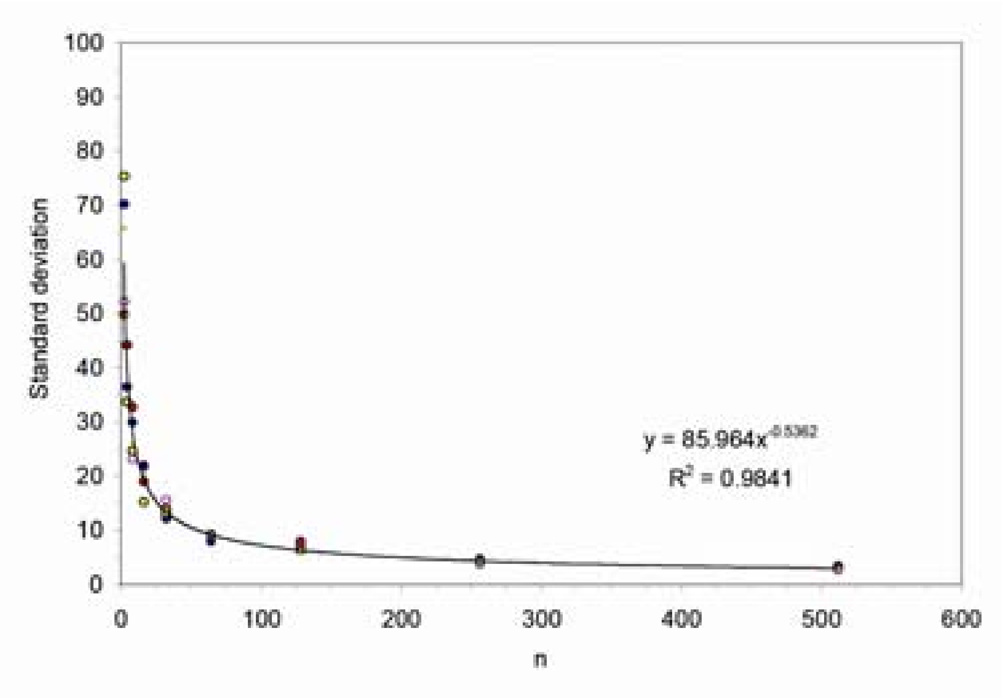

Figure 4.4. Mean deviation in an estimated standard deviation from a true standard deviation, as a function of random sample size, n. Different symbols represent simulations with different standard deviations ($1 < \sigma < 500$).

Risk of classification error - uncertainty interval

The uncertainty associated with an estimated value is always linked to a degree of risk that an estimated value falls into the wrong class. The nearer a class boundary the true value is situated, the greater the risk of classification error. Figure 4.5 illustrates the probability of classification error, depending on where an estimated value falls in a class. In this case, the estimated value has an uncertainty of 10% of the width of the whole class. In Figure 4.5a, the measured value lies in the middle of an ecological class, which in this case, despite added variation, implies a low risk of classification error. In Figure 4.5b, on the other hand, the measured value lies right on a boundary between two classes, implying a 50% risk of classification error.

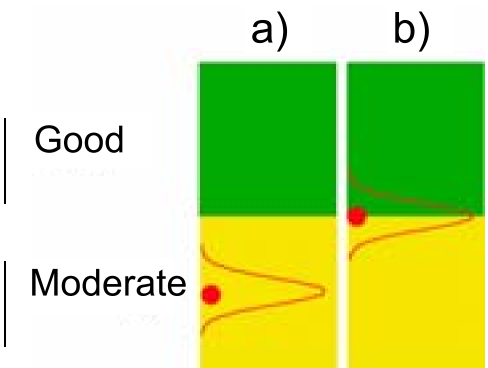

Figure 4.5. Examples of how uncertainty and the placement of measured values within two classes can affect the risk of classification error. a) The true value lies in the middle of a class, resulting in a low risk of classification error, b) The true value lies on the boundary between two classes, resulting in a 50% risk of classification error. In both examples the uncertainty is 10% of the band (the class) width.

Figure 4.6 and Table 4.1 show how the frequency of classification error increases with both uncertainty in the measured variable and the distance from class boundaries. Research[20] has shown that the probability of classification error increases markedly with the degree of uncertainty in measurements, and with diminished distance to a class boundary in a classification system. With a standard deviation of e.g. 10% of the class width, the classification error frequency for values in the middle of a class is between 0% and 8% if the measurement values are evenly distributed within a class. If the uncertainty increases to 50% of a class width, the classification error frequency increases 32% for values in the middle of a class, which means that approximately 40% of all assessments are misclassified, to a class either above or below the class boundary (Table 4.1).

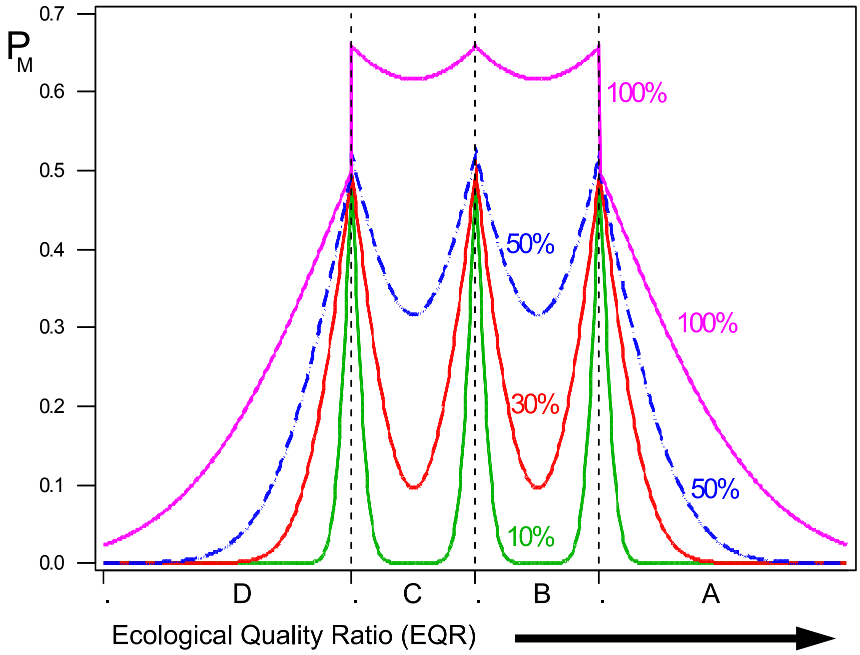

Figure 4.6. Simulated probability for classification error (PM) at varying sizes of distribution (SD). Classes A-D are purely hypothetical examples of EQR-classes (after Clarke 2000[21]).

[20] Clarke R. 2000. Uncertainty in estimates of biological quality based on RIVPACS. pp 39-54, In: J.F. Wright, D.W. Sutcliffe, and M.T. Furse (eds). Assessing the biological quality of freshwaters. RIVPACS and other techniques. Freshwater Biological Association, Ambleside, UK.

[21] Clarke R. 2000. Uncertainty in estimates of biological quality based on RIVPACS. pp 39-54, In: J.F. Wright, D.W. Sutcliffe, and M.T. Furse (eds). Assessing the biological quality of freshwaters. RIVPACS and other techniques. Freshwater Biological Association, Ambleside, UK

Table 4.1. The expected proportion of classification errors (PM) at 10%, 30%, 50% and 100% variation (% ESD = uncertainty SD as % of class width). According to Clarke (2000).

% ESD	P_M (mean value)	Range
10%	8%	0 - 50%
30%	24%	10 - 50%
50%	39%	32 - 52%
100%	63%	62 - 66%

Table 4.2 shows a hypothetical example of classification of a water body. In the example, the probability that the water body is classified as good or higher ecological status is 70% (60+10), at the same time as the probability that the same object can belong to a lower class, i.e. moderate or worse, is 30% (25+4+1).

Table 4.2. Hypothetical example of classification of a water body.

Class	Probability of classification (%)
High	10
Good	60
Moderate	25
Poor	4
Bad	1

Object-specific and method-bound uncertainty
Several stages in sampling, sample handling or preparation can result in increased uncertainty. This type of uncertainty is called method-bound uncertainty. In addition to this measure of uncertainty, another type of variation or uncertainty (natural and impact variation) can also be present which can affect the uncertainty in the classification of status. Brief descriptions of object-specific and method-bound, type-specific estimates of uncertainty and how these may be used to estimate uncertainty in the classification are given below. The first case is about how estimates of uncertainty can be used when we have replicated random samples, and the second case is the converse, i.e. when there are no replicated random samples. In both these cases, an expression of the probability of classification error is desired.

Estimating object-specific uncertainty (with random samples)
In accordance with the description given earlier in this chapter, the uncertainty in an estimated mean value can be expressed as a confidence interval. The certainty in the classification of a water body can then be assessed by investigating whether this confidence interval overlaps one or more class boundaries. If the distribution is skewed, it can on the other hand be more appropriate to estimate the confidence

interval and frequency of classification errors with the aid of randomisation (e.g. Clarke 2004[22]).

Examples of the calculation of uncertainty intervals for object-specific uncertainty

The status on soft bottoms in sea areas is evaluated using an index, BQI, which reflects the sensitivity to eutrophication and the species abundance among benthic macroinvertebrates. In measurements taken during 1997, an area in the Bothnian Sea had a mean value ($\overline{X}$) in the BQI of 7.36 and a standard variation (s) of 3.22. The size of the random sample (n) was 20. By using the formula in Section 4.1.2.1 and adopting a 95% confidence interval, it can be calculated that that the true mean value in 1997 (μ) was, with 95% probability,:

$$\mu = 7.36 \pm 2.093 * \sqrt{\frac{3.22^2}{20}} \text{, where } t_{n-1,0.025} = 2.093$$

That means that the lower boundary for the confidence interval is 5.85 and the upper boundary is 8.87. Since the maximum value of BQI in this sea area is 12.0, the mean value and boundaries for the confidence interval translate to 0.61, 0.49 and 0.73 on an EQR-scale (by dividing by 12). This procedure illustrates a general case for uncertainty in an estimated mean value in a parameter when the sampling includes replication.

Since the specific assessment criteria for benthic macroinvertebrates use the precautionary principle in such a way that the status classification does not consist of the mean value but of the starting-point from the lower boundary for an 80% one-sided confidence interval (i.e., we decide the value that with 80% probability includes the best conditions), it is appropriate to show how this can be calculated. In assessment criteria for benthic macroinvertebrates, two methods are given for managing uncertainty. In the first instance, it is recommended to use a method known as “bootstrapping”, which is based on repeated random sampling and replacement of the available data material. A second method, which can be used exceptionally, is based on the principles described here. On average, the two methods can be expected to give similar results. If one applies the latter method in the present case, the lower boundary, i.e. the status, is calculated, as:

$$Status = 7.36 - 0.86 * \sqrt{\frac{3.22^2}{20}} = 6.74 \text{, where } 0.86 = t_{n-1,0.4}$$

According to the assessment criteria for soft-bottom benthic macroinvertebrates, the status was thus 6.74 on the BQI scale and 0.56 on the EQR scale. In accordance with the assessment criterion, the quality should be classified as good.

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[22] Clarke R.T. 2004. 9th STAR Deliverable, Error/Uncertainty Module Software STARBUGS. STAR Bio Assessment Uncertainty Guidance Software. User Manual. www.eustar.at

Estimating method-bound uncertainty (with random samples)

A common way to assess method-bound uncertainty is to quantify uncertainty in the sample collection (e.g. the variation between various samplers) and sample handling (e.g. variation in species identification and counting between different people). For e.g. the diatom index IPS, the method-bound uncertainty is deemed to vary between 5 and 10%, of which 80% is judged as resulting from differences in species identification, 10% from the sample collection, 5% from the production of preparations and 5% from differences between various random samples (see assessment criteria for diatoms). It happens in the environmental monitoring programme that a number of random samples are taken in the same sampling area and at the same time. The variation between these samples includes uncertainty in the collection and handling of samples. However, we do not always have several random samples available, which means that there can be great benefit in having "static" measures of uncertainty, in cases where these have been developed (see e.g. Table 4.4). An example is given below of how knowledge about method-bound uncertainty can be used to give a measure of uncertainty in the classification status, when the variation cannot be estimated on the basis of the actual samples, i.e. when no replicated measurements are available.

Examples of the use of uncertainty intervals for method-bound uncertainty

Within the environmental monitoring programme, five random samples of benthic macroinvertebrates are taken per site at each sampling event. The spread between these five samples can be regarded as a measure of uncertainty in the collection and handling of samples. Instead of estimating uncertainty at each sampling, we can, however, like the use of type-specific reference conditions, use a method-bound, type-specific estimation of uncertainty in the classification (see e.g. Johnson and Goedkoop 2007[23]). A similar procedure for estimating the uncertainty in the classification of ecological status has recently been proposed by Clarke et al. (2006)[24]. An example of this is shown in Table 4.3, which contains examples of the type-specific reference values (the median of the reference object) and uncertainties (the median standard deviation, SD, for five replicated samples taken in the reference object), and ecological class boundaries for the ASPT index used in the Illies Ecoregion 14 (Central Plain). For example, a measured value of 4.80 for the ASPT Index would result in an EQR of 0.82 (4.80/5.85). Taking account of the uncertainty in the method (in this example 5.7%), this can be expressed as 0.82 ± 0.057.

[23] Johnson, R.K. & Goedkoop, W. 2007. Bedömningsgrunder för bottenfauna i sjöar och vattendrag – Användarmanual och bakgrundsdokument [Assessment criteria for benthic macroinvertebrates in lakes and watercourses - User manual and background document]. Report 2007:4.

[24] Clarke R.T., Davy-Bowker J., Sandin L., Friberg N. & R.K. Johnson. 2006. Estimates and comparisons of the effects of sampling variation using 'national' macroinvertebrate sampling protocols on the precision of metrics used to assess ecological status. Hydrobiologia, 566: 477-503.

Table 4.3. Benthic macroinvertebrates in lakes in Illies Ecoregion 14 (Central Plain). The figures show type-specific reference values, the standard deviation (SD) for five replicated samples taken in the reference object, and ecological quality ratios (EQR) for the ASPT index.

Littoral ASPT	
Reference value	5.85
Uncertainty (SD of EQR)	0.057
High	≥0.95
Good	≥0.70 and <0.95
Moderate	≥0.50 and <0.70
Poor	> 0,25 and <0,50
Bad	< 0.25

Estimation of uncertainty interval

Annexes A, B and C provide descriptions of how uncertainties in an ecological classification should be managed for each assessment criterion respectively. Table 4.4 gives an example of how uncertainty is managed in the use of the quality element Phytoplankton in lakes (see Annex A, Section 3.10 for further information). For those parameters used in the assessment of surface water quality in freshwater, it is common to use a measure of the method uncertainty in ecological classification.

Table 4.4. Examples of how uncertainty is managed for the quality element Phytoplankton in lakes. The Table shows median values of the standard deviation in EQRs for reference lakes in the dataset.

Indicator	Mountains	Northern Sw. clear	Northern Sw. humic	Southern Sw. clear	Southern Sw. humic
Total biomass	0.05	0.09	0.13	0.19	0.12
Proportion of cyanobacteria	0	0.02	0.02	0.04	0
TPI	0.17	0.18	0.18	0.23	0.002
Number of species	0.14	0.05	0.03	0.07	0.07

Quick way of calculating the uncertainty interval

If the number of random samples is sufficiently large, with a normal distribution, 68.2% of all observations lie within the interval mean ± 1 standard deviation and 95.4 % within the interval mean ± 2 standard deviation (Figure 4.7). In environmental monitoring, the probability that an object is classified as worse than good ecological status is often of the utmost importance, which means that the greatest interest centres on the probability that the observed values lie in the “lower” or “upper” part of the normal distribution. In this case, the “lower” part of the normal distribution means that approximately 84 % (68.2+13.6+2.1+0.1) of observations lie higher than mean - 1 standard deviation and approximately 98 % lie higher than mean - 2 standard deviation. By applying this rule to object or type-specific estimations of uncertainty, an estimate of uncertainty in the classification can be obtained. This means that the risk of classification error is calculated with the aid of the already known uncertainty in the measurements, comprising a confidence interval around the measured value. To obtain a rapid indication of the probability that an object can be classified as worse than good ecological status, we can also calculate the probability of classification error as the measured EQR value minus 1 or 2 standard deviations.

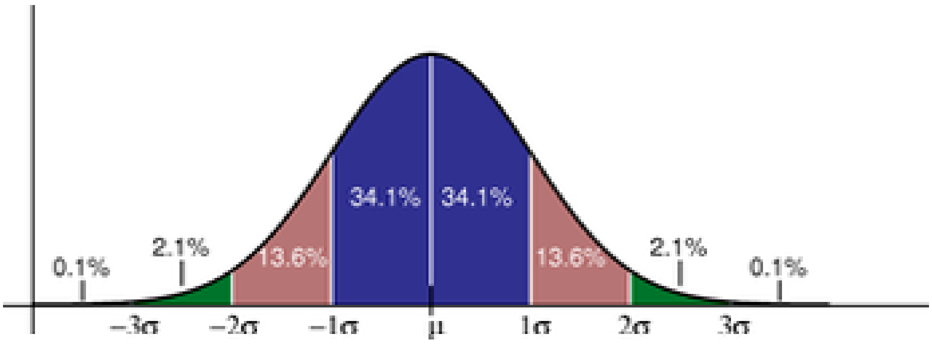

Figure 4.7. Mean value and standard deviation in a normal distribution.

Example of a quick calculation of the uncertainty interval
In Table 4.3, good status for benthic macroinvertebrates in lakes is defined as an ecological ratio that lies between 0.7 and 0.95. If the measured value is e.g. 0.82, the water body should be placed in the middle of the band (the class) 0.7 – 0.95 and thereby classified as good status. However, there is degree of uncertainty in every measurement, which means that we must determine the probability of a classification being correct. If we take into account the uncertainty in the measurement, the true value probably lies between 0.934 (0.82 + 2 * 0.057) and 0.706 (0. 82 – 2 * 0.057) (Table 4.5). Applying the normal distribution, 98% of observations therefore lie above the value 0.706, which in other words means that there is only a small risk that the object should be classified as worse than good status.

If, on the other hand, the measured value had been 0.750, which lies closer to a class boundary, the probability increases that the object should be classified in a lower status class (Figure 4.6). By applying the normal distribution, these calculated confidence intervals range between 0.864 (0.75 + 2 SD) and 0.636 (0.750 - 2 SD) (Table 4.5). In this example, when there is a high risk that the object should be classified as worse than good status, it is appropriate also to calculate the probability of the object being above the mean - 1 standard deviation (e.g. 0.750 – 1 * 0.057 = 0.693) in order to achieve a somewhat more precise breakdown of the confidence interval. According to the normal distribution, 84.2% lie above this boundary, which indicates that there is then a risk (probability = 15.7%) that the object should be classified as worse than good status.

Table 4.5. Examples of estimations of uncertainty in the classification of ASPT in Illies Ecoregion 14.

	% of distribution	EQR values	
+ 2 SD	13.6	0.934	0.864
+ 1 SD	34.1	0.877	0.807
Mean		0.82	0.750
- 1 SD	34.1	0.763	0.693
- 2 SD	13.6	0.706	0.636

Calculation of probability of classification error

If there is an evident risk that the object can be classified as worse than good status, the probability of classification error should also be calculated more carefully. With the aid of the four class boundaries (i) and the uncertainty (s, in the form of standard deviation) in the measurement, we can calculate the probability of the object being classified in each of the five ecological classes[25] respectively . Four calculations are required for this in order to respond to the probability (pi) of the observed index values of x, or the true mean quality (μ) of the class boundary (Li). This can be calculated as:

$$p_i = \Pr(X \geq x \text{ if } \mu=L_i) = 1 - \Phi\{(x - \mu)/s_i\},$$

in which Φ is the cumulative normal probability.

This probability expression asserts that $\Pr(X \geq \mu + u.s_i) = p_i$ (in which u is the normal standard deviation or $1 - p_i$). By inversion the probability is generated as:

Probability $(\mu \leq x - u.s_i) = 100p_i$.

Probability of class 5 = $100p_5$.
Probability of only class 4 = $100(p_4 - p_5)$.
Probability of only class 3 = $100(p_3 - p_4)$.
Probability of only class 2 = $100(p_2 - p_3)$.
Probability of only class 1 = $100(1 - p_2)$.

Examples of the calculation of the probability of classification error

Table 4.6 shows calculations of the probability for an object with a mean value of 0.82 for EQR for ASPT and a standard deviation of 0.057. Like the example above, the risk that the object should be classified as worse than good status is low (1.8%). According to a normal distribution of the variation, the probability that the object should be classified as higher than moderate status is 98.3% (96.5 +1.8). However, if the measured values lie closer to a class boundary (e.g. 0.75) there is a greater risk that the object should be classified as worse than good status. In example b) in the same table, there

[25] Ellis J. & Adriaenssens V. 2006. Uncertainty estimation for monitoring results by the WFD biological classification tools. Environment Agency, Rio House, Water-side Drive, Aztec West, Almondsbury, Bristol, UK. 32 p

is a 19% probability that the object should be classified as moderate status. This estimate tallies well with the calculation above (that is to say mean -1 standard deviation) = 0.750 – 1 * 0.057 = 0.693), which showed a 15.7% probability that the object should be classified as worse than good status.

Table 4.6. Example of a more thorough calculation of the probability of the object being classified in each of the five classes respectively. An explanation of how to calculate this in MS Excel is given below the table.

a)	**Input data:**	EQR ASPT = 0.82 SD = 0.057		
Status class	**Established class boundaries**	**Calculated P-values**	**Calculation of probability**	**Probability of classification (%)**
High		P1:	100 * (1 - P2)	1.8
Good	0.95	P2: 0.982	100 * (P2 - P3)	96.5
Moderate	0.7	P3: 0.018	100 * (P3 - P4)	1.8
Poor	0.5	P4: 0.000	100 * (P4 - P5)	0.0
Bad	0.25	P5: 0.000	100 * (P100)	0.0

b)	**Input data:**	EQR ASPT = 0.75 SD = 0.057		
Status class	**Established class boundaries**	**Calculated P-values**	**Calculation of probability**	**Probability of classification (%)**
High		P1:	100 * (1 - P2)	0.0
Good	0.95	P2: 1.000	100 * (P2 - P3)	80.9
Moderate	0.7	P3: 0.190	100 * (P3 - P4)	19.0
Poor	0.5	P4: 0.000	100 * (P4 - P5)	0.0
Bad	0.25	P5: 0.000	100 * (P100)	0.0

How to calculate this in MS Excel:
In Excel, P can be calculated by inserting the following command in any cell (with values for the different variables included): =NORMDISTR(μ;X;SD;TRUE)
To calculate P2 = NORMDISTR(0.94.0;82.0.057;TRUE) = 0.982;
To calculate P3 = NORMDISTR(0.70.0.82.0.057;TRUE) = 0.018;

To calculate the probability that the classification falls in the ecological class good status = 100 * (P2 – P3) or 100 * [0.982] – [0.018] = 96.5%

X = the value for which you wish to calculate the distribution.
Mean value (μ) = the arithmetical mean value of the distribution.
Standard deviation (SD) = the standard deviation for the distribution.
Cumulative = a logical value; TRUE for the cumulative distribution function and FALSE for the mass of the probability function.

Finally, to choose the class

Table 4.6 shows the probability of classification into five classes. In the first example (a) the probability that the object belongs to a specific ecological class worse than good ecological status is low (< 2% risk). In the second example (b), on the other hand, the probability is much greater, that is to say 19%. There is uncertainty in all measurements, and the question is how to apply and use this information in a status classification. The objects in the second example can be said with great probability (80%) to belong to the status class “good”, or the risk that they should be classified as worse than good status is smaller (19%). Since the most probable classification is good, the final status classification of the objects should be “good”. However, it is necessary to be aware that there is a 19% risk of classification error. This risk of classification error can be reduced by collecting several samples to increase the precision of the classification and by observing several quality elements either to confirm or to deny the classification. This is the advantage of having several indicators.

4.2 Classification of status according to assessment criteria at the quality element level

4.2.1 Limitations leading to a quality element not being applicable to the water body

For each assessment criterion respectively in Annexes A, B and C, there is information about the types in which an assessment criterion is applicable. In some cases, because of a lack of knowledge or data, it has not been possible to develop assessment criteria for a type. For coastal and transitional waters there are no assessment criteria for macroalgae in types 13, 24 and 25 or for fish in transitional waters. Moreover, assessment criteria are lacking for all the hydromorphological quality elements in coastal and transitional waters. However, there are no lake or watercourse types that lack assessment criteria. In certain cases, however, there can be water bodies that are not representative of the type, which in turn can mean that a specific assessment criterion is not applicable in that particular case. The types are relatively general, to keep them manageable in number. This means that within each type there will be individual water bodies that differ somewhat from the general type, which can affect the biological conditions. One example of this can be a coastal water body that lies close to a river mouth and which therefore has a lower salt content than other types, as a result of which the reference values do not apply here. Water bodies that consist of extremely large or small lakes comprise another example. The assessment criteria are not altogether applicable for fish in such lakes, because the size of lakes has not been included in the background data for the assessment criteria.

If the water body does not meet the objectives, it is necessary to make an expert judgement. This can be done in different ways on the basis of the available knowledge about the impact and condition of the water body in question and the water bodies in adjacent land areas (Section 4.4). Over time, it can also be of assistance in the expert judgment to construct a special index for the specific water body, but this is primarily relevant for water bodies that lack applicable assessment criteria.

4.2.2 Assessment of acidity and acidification

For many years, researchers have been trying to find indicators that can make a simple distinction between anthropogenic acidification and natural acidity. The result is everything from simple species-based indices to advanced acidification models. Acidification models, such as MAGIC and the episodic model, have now been sufficiently developed and tested that they can be said to be very accurate in indicating acidification. The species-based indices still have further to go before they can reliably be used to detect anthropogenic acidification. They are, however, sufficiently developed to be capable of indicating acidity and hence also potential acidification. To assess the acidification or acidity status, the following procedure can be used (examples in Figure 4.8):

A. In the first stage, acidic conditions must be investigated according to the assessment criteria for biological quality elements. To classify acidic conditions the scale used is not the ordinary one from high to bad, but instead an acidity scale with the *classes alkaline, almost neutral, moderately acidic, acidic, highly acidic and extremely acidic* (somewhat different classes for the each quality element respectively). If the assessment of the biological quality elements falls in the classes *acidic, highly acidic* or *extremely acidic* the water body may potentially be acidified. If at the first step it can already be established that the water should be classified as *alkaline, nearly neutral* or *moderately acidic*, the water body does not need to be further analysed with regard to acidification. For the final classification of its biological acidity status the acidity classes *alkaline* and *almost neutral* are translated to high status and *moderately acidic* to good status. The quality element fish does not have any special parameters that indicate solely acidity, and acidity/acidity impact is included in the index as part of the general impact. Therefore for these too, when they fall in the class moderate or worse, an analysis should be made of whether this results from acidity and, in that case, whether this is caused naturally or anthropogenically. Guidance on how this can be done is contained in the assessment criteria for fish in Annex A. Before a final classification is made, the human activities that can have a pH-raising effect on the water body, for example liming, should also be taken into account in the assessment. If, for example, the high pH value is due to liming, it should be investigated whether there is a need for further liming. Limed water bodies should be classified after the water chemistry has been corrected for liming impact using the ratio between non-marine Ca and Mg, or using other methods that give equivalent results. The ratio between non-marine Ca and Mg can be derived from measurements taken prior to liming or from a nearby unlimed reference lake.

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Section 10

B. If the assessment criteria after this first step indicate acidic conditions, one should go further and make use of the acidification models available to distinguish anthropogenic acidification from natural acidity. Models that should be used for this are MAGIC, MAGIC library or the episodic model BDM, described in Annex A. At this stage it may also be justified to carry out supple-

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10 §

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Sections 5-7

mentary field measurements if there is insufficient data to run the model. The analysis can be further improved by making an assessment of the acidification pressure on the ecosystem. Data on the local impact (e.g. of forestry) can provide important background material about this. If the acidification pressure over relatively large areas is being assessed, deposition data can also be of assistance (please see further Annex A).

C.

- If the outcome of the acidification analysis is that the water is to some extent naturally acidic, a revision of the reference value and class boundaries for the specific water body can be made on the basis of the water authority's expert judgement. The reference value for pH from MAGIC library is calculated on the basis of the ANC change in accordance with MAGIC library with the aid of the pH model given in Annex A (or possibly using the online application available on the Swedish EPA website). Alternatively, the reference value for pH can be calculated directly using the MAGIC model. With this calculated reference value for pH, a new reference value for the biological quality element or parameter can be generated through the diagram or table for assessment criteria for phytoplankton, diatoms and benthic macroinvertebrate assemblages, to be found in Annex A. For fish, it is not currently possible to generate new reference values and class boundaries easily, which means that in this situation an expert judgement has to be made. On the basis of this new reference value, the new class boundaries for status are calculated in accordance with the instruction for each assessment criterion respectively. Thereafter, the biological acidification status can be classified on the basis of the new status classes developed.

See GG to Chapter 2 10 §

- If the assessment is that the water is anthropogenically acidified, the final classification of the acidification status is made according to the classes in for each assessment criterion respectively.

Other quality elements, and impacts other than acidity and acidification, should also be assessed and considered in the final status classification in accordance with the ordinary procedure for status classification (see the checklist in Section 3.4). The "one out – all out" principle is applied between the different quality elements in all above cases (Section 4.2.4).

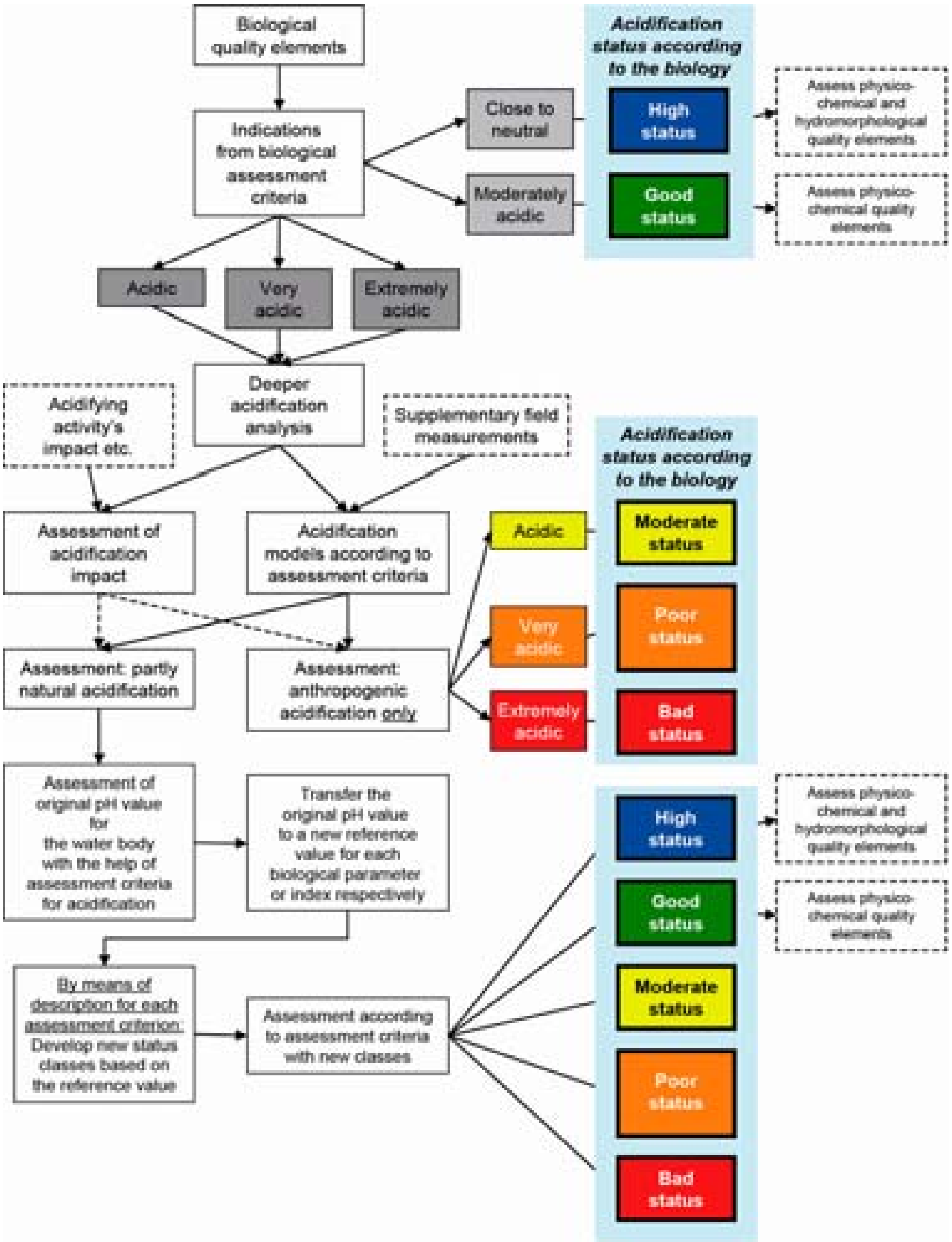

Figure 4.8. Examples of the working procedure when classifying acidification status. This example applies for benthic macroinvertebrate assemblages in lakes (MILA). For phytoplankton, benthic macroinvertebrate assemblages in watercourses and diatoms, the corresponding working procedure is followed, but with somewhat different classes (please see the description for each quality element in Annex A).

4.2.3 Assessment of nutrient richness and eutrophication in lakes and watercourses

When the status classification for lakes and watercourses results in moderate status, as indicated by the parameters showing nutrient richness or eutrophication, an assessment may be necessary as to whether this is a result of anthropogenic eutrophication or whether the watercourse is naturally nutrient-rich. However, naturally high nutrient content is not particularly common, especially not in lakes. The procedure may be compared with that for acidity and acidification.

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When nutrient status is to be assessed as regards whether the origin is anthropogenic or natural, the result of the biological assessment criteria may be compared with the result of the assessment criteria for phosphorus, which is the main regulating substance in freshwater. There are indications that nitrogen can be a limiting factor in certain nutrient-poor lakes and watercourses (in e.g. mountains) and in heavily eutrophied lakes and watercourses. If there are clear indications that the nitrogen content is controlling growth and having an impact on the species composition in a water body where there is significant anthropogenic nitrogen stress, the water authority can carry out an expert judgement to establish the appropriate nitrogen content for the boundary between good and moderate status for nitrogen (please see under Nutrients, in Annex A).

The assessment can further be improved by examining the impact and stress the water body may be exposed to. Important evidence for this includes source distribution data, historical data and floods. Supporting data is produced in connection with the characterisation (see the Handbook on mapping and analysis).

If the assessment is that the water is naturally nutrient-rich, a revision of the reference value and class boundaries for lakes and watercourses is made for the specific water body, on the basis of the water authority's expert judgement. A new reference value can then be generated for the respective biological assessment criterion on the basis of the assessment criterion for phosphorus.

See GG to Chapter 2 Section 11

If the assessment is that the water is anthropogenically eutrophied, the original biological classification is used to establish a final eutrophication status.

4.2.4 Co-weighting of parameters

Several different approaches may be used to co-weight (weigh together) the status of several parameters into the co-weighted quality element status. The "one out – all out" principle may be used at the parameter level when groups of parameters that indicate different impact pressure (e.g. acidification or eutrophication) are to be weighed together into the quality element level. Other methods of weighing together different parameters within one quality element may be to use the mean value of the classifications made. These principles are clarified in Figure 4.9. The details of how parameters should be weighed together for each quality element are given under each assessment criterion respectively in Annexes A, B and C. When weighing together different biological quality elements, the "one out – all out" principle must be observed. Any deviations from this must be justified and documented.

4.3 Status classification by assessment criteria – co-weighting quality elements into ecological status

4.3.1 Co-weighting quality elements

See REG Chapter 2 Section 2

When biological quality elements are to be co-weighted or weighed together in a status classification, the quality element indicating the greatest anthropogenic impact is the deciding factor. This principle, illustrated in Figure 4.9, is known as "one out – all out". If e.g. the quality element fish indicates poor status, while benthic macroinvertebrate assemblages indicate moderate status, the combined biological status classification must be "poor". First assess, therefore, the combined status for the biological quality elements. If they indicate moderate status, or worse, that also becomes the result for the ecological status, since it is then of no great importance what the physico-chemical or hydromorphological quality factors show. A programme of mitigation measures must in any case be established. If the biological quality elements indicate high or good status, the physico-chemical quality factors are assessed. If the physico-chemical quality elements then show moderate or worse status, the ecological status will be classed as moderate. If both the physico-chemical and biological quality elements indicate high status, the hydromorphological quality factors are also assessed. If they indicate good or worse status, the ecological status will be good. If the hydromorphology also indicates high status, however, the water body must be classified as high ecological status.

If a reasonability assessment of the classification of a quality element/parameter leads to the conclusion that the status or potential is different from the classification of other quality elements/parameters, and this is not assessed as being reasonable, the earlier classification can be disregarded. In the next step, an expert judgement (Section 4.4) must be made to assign a new classification of status or potential, either for the whole of the water body or for an individual quality element. It is in this context important to be attentive to whether any "inert parameters" (see point 7c in Section 3.2.3) have had sufficient time to react to the impacts in question.

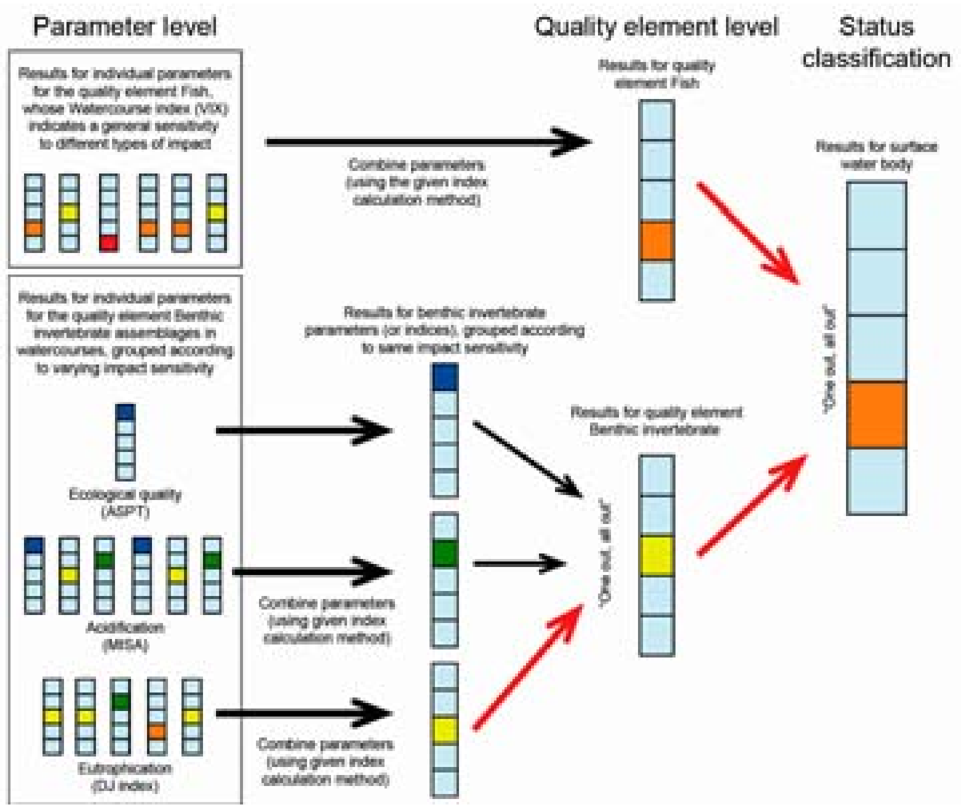

Figure 4.9. A schematic description of how to weigh together parameters and quality elements.

4.3.2 Checking procedure

There is an approach known as 'checking procedure', to ensure that a water body is not incorrectly downgraded to moderate status because the physico-chemical quality elements are more stringent than the requirements laid down in the WFD, or vice-versa. Checking procedure must only be used to assess whether the class boundary between good and moderate status or potential is correctly set for physico-chemical quality elements. This procedure is also explained in paragraphs 4.3-4.9 of EU Guidance Paper No. 13.

See REG Chapter 2 Section 12

It is normally the case that if results from both the biological and physico-chemical quality elements indicate good ecological status or potential for a water body, the co-weighted assessment of ecological status or potential is good. If, on the other hand, one or more physico-chemical quality elements do not attain the objectives for good ecological status or potential, despite the fact that the biological quality elements do so, the co-weighted assessment is moderate ecological status or potential.

Checking procedure can be used when the classification using the physico-chemical quality elements gives moderate status, while the classification using biological quality elements shows good status (Figure 4.10). To demonstrate that a class boundary is incorrect, a number of different questions must be answered showing that the difference does not rest on, for example, a response delay in the biology and that the result of the physicochemical quality elements is therefore correct.

See GG to Chapter 2 Section 12

In addition, checking procedure may be used when good status is achieved for the physico-chemical quality elements, even though the impact and condition data may support an assumption that (a) good status or potential for the biological quality elements is not reached in water bodies; or (b) there is proof of deterioration in the functioning of the ecosystem in water bodies within the type. Checking procedure as in Figure 4.11 can then be used to assess whether the established class boundaries for physico-chemical elements are insufficiently stringent to safeguard the functioning of the ecosystem and to attain good status or potential for biological quality elements. However, checking procedure cannot be employed when temporary deterioration in the physico-chemical state occurs because of unusual natural conditions, such as long periods of drought or floods, and we must then resort to the reasonability assessment described in Section 4.1.1.

See GG to Chapter 2 Section 12

When checking procedure is used, it should be kept in mind that the physico-chemical methods have been developed over a long period and can initially give a better and more reliable indication of ecological impact than many less proven biological methods. Nonetheless, the physico-chemical quality elements may only complement biological quality elements, not substitute for them. Both are required under the Water Management Ordinance.

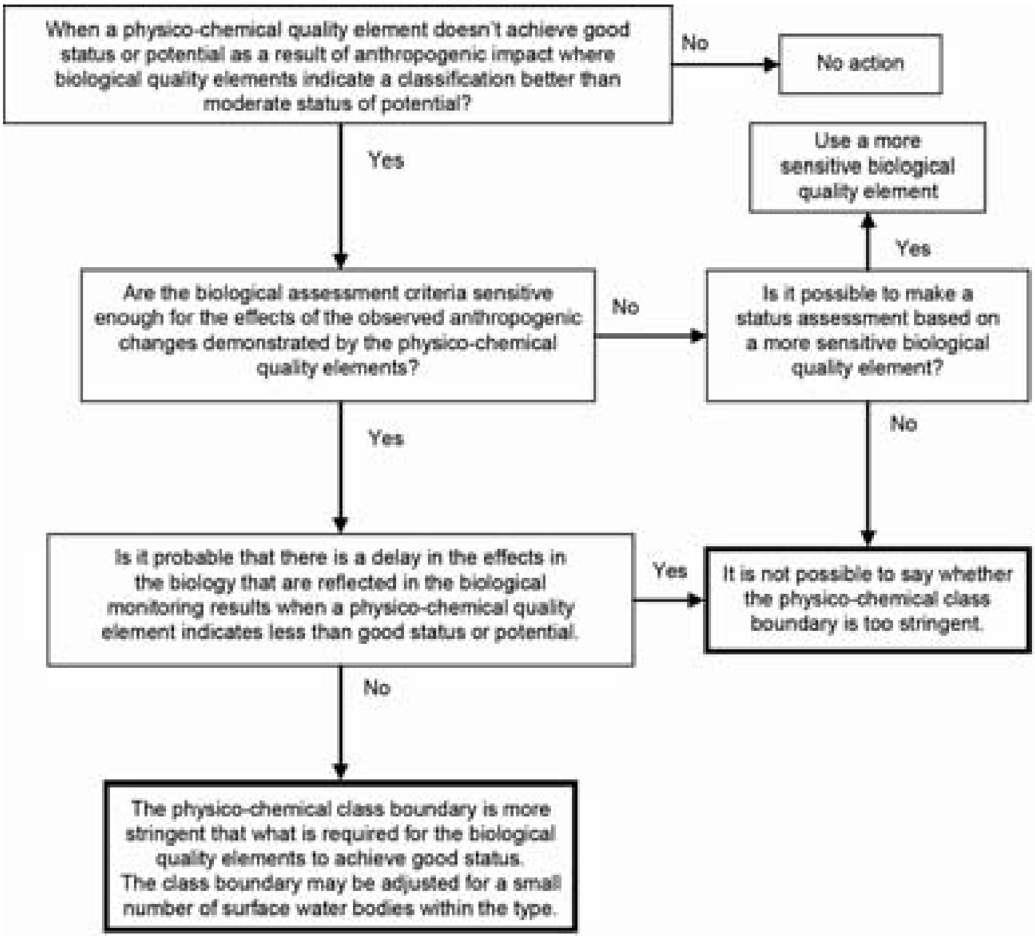

Figure 4.10. Checking procedure to assess whether an established class boundary for a physico-chemical quality element is more stringent than is necessary to enable the biological quality elements to achieve good status.

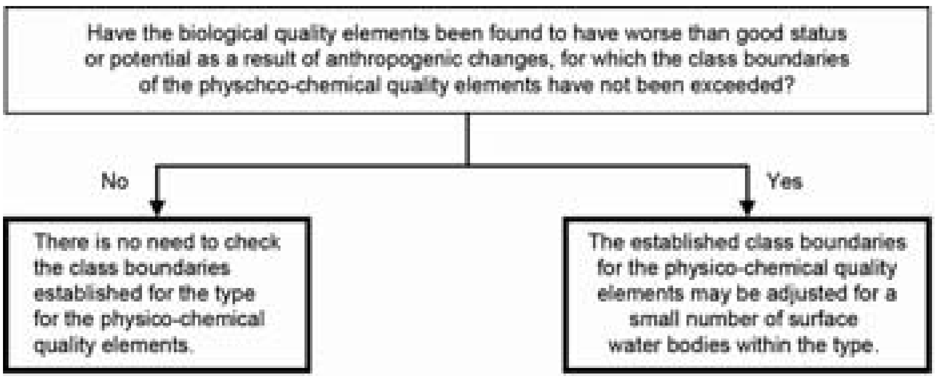

Figure 4.11. Checking procedure to assess whether an established class boundary for a physico-chemical quality element is insufficiently stringent in relation to the biological quality elements.

4.3.2.1 EXAMPLES OF WHEN THE BIOLOGY HAS GOOD OR HIGH STATUS OR POTENTIAL AND PHYSICOCHEMICAL QUALITY ELEMENTS INDICATE MODERATE OR WORSE STATUS OR POTENTIAL

In certain cases, the physico-chemical quality elements indicate moderate or worse status or potential, despite the fact that the biology has good or high status or potential. Some conceivable situations and conceivable remedial measures are exemplified below.

Site selection

- If a sample has been taken in a microcommunity, e.g. behind a rock in a watercourse, too close to reeds in a lake, etc, the biology can be misleading. A conceivable remedial measure is to investigate whether the sample point is representative for the measurement we wish to take and, if necessary, move it. Thus no class boundary is changed here.
- If there are different water chemistry and biology sampling sites in e.g. a watercourse that does not obviously lie in the same flux, it is possible that the samples have been taken in different fluxes by mistake, which can lead to different classifications of the water chemistry and biology. A conceivable remedy is to move the sampling sites closer to one another to minimise the risk of samples being taken in the wrong places. Thus no class boundary is changed here.
- If a coastal area with undisturbed biology shows high phosphorus and nitrogen concentrations at measured stations, it can for example be because, close to a river mouth, the area has high water exchange and good oxygenation, which means that the biology is not disturbed by high concentrations. A conceivable remedial measure is to move the sampling points, which may have been placed too far in. Thus no class boundary is changed here.

Temporal variation

- If there has been a recent discharge of chemical substances (e.g. as a result of overflow) the biology may not yet have had time to react to it. A conceivable measure here is to take new samples and to investigate whether the biology has been disturbed by the discharge. The class boundary is thus not changed until further investigations have been made.
- Abnormally low water can result in a concentration of physico-chemical substances. If the water has been low for only a relatively short period, the biology may not have had time to react to any physico-chemical substances. A conceivable remedial measure is to take samples when the water flow is normal, to check whether any high concentrations of physico-chemical substances persist. The class boundary is thus not changed without first making further investigations.

Biological response

- If a biological or physico-chemical quality element or parameter is used which is not sensitive to the type of impact which is of interest, it is difficult to interpret the results. A conceivable remedial measure is to change to a quality element or parameter that is more sensitive to the impact that is of interest. Thus no class boundary is changed here.
- If the indications are that the macroalgae are undisturbed despite high nutrient levels, it may be because they have a delayed response to nutrients. In this case, a checking procedure may be considered appropriate and could be used as an early warning system for what might happen. A conceivable remedial measure is to check nutrient level trends in the area. If they have been stable for a long time, this should be acceptable, but if there is an increasing trend, the physico-chemical quality elements should perhaps be the deciding factor.

Other interaction

- If a low pH value is measured in a lake, despite the fact that the biology is undisturbed and there are plenty of e.g. roach and salmon trout yearlings, it may be because high humic content offers protection. A conceivable remedial measure, when there is a high humic content and low inorganic aluminium content, is possibly to change the reference values for physico-chemical quality elements (i.e. change the quality requirement for e.g. good status for the relevant physico-chemical quality element to match the current measured values since these are considered to guarantee good status in this type of water body).

4.3.2.2 EXAMPLES OF WHEN THE BIOLOGY HAS MODERATE OR WORSE STATUS AND PHYSICO-CHEMICAL QUALITY ELEMENTS INDICATE GOOD OR HIGH STATUS

In certain cases the physico-chemical quality elements indicate good or high status or potential, despite the fact that the biology has moderate or worse status or potential. Some conceivable situations and conceivable remedial measures are exemplified below.

Site selection

- If there are different water chemistry and biology sampling points in e.g. a watercourse that does not obviously lie in the same flux, it is possible that the samples have been taken in different fluxes by mistake, which can lead to different classifications of the water chemistry and biology. A conceivable remedial measure is to move the sampling points closer to one another to minimise the risk of samples being taken in the wrong places. Thus no class boundary is changed here.

Temporal variation

- If the biology has moderate or worse status, despite recent mitigation measures that have improved the chemical stress, it may mean that the biology has not yet caught up. A conceivable remedial measure is to monitor the biology to ensure an improvement in its status. The assessment of the class boundary based on the biology should be retained.
- If there is high water when the physico-chemical quality elements are sampled, it may mean that any pollution has been diluted. That, together with the sample results, may be misinterpreted as showing that the water has become "purer". A conceivable remedial measure is to take additional samples when the flux is normal. Thus no class boundary is changed until further investigations have been carried out.

Biological response

- If the physico-chemical quality elements do not indicate the same sort of impact as the biology, the sample results may be misleading. A conceivable remedial measure is first to decide which impact to measure. Then it should be considered whether the right quality elements have been selected, and make an adjustment if that is not the case. Thus no class boundary is changed here.

Other interaction

- If the biology is locally disturbed by an extreme but natural event (e.g. severe predation, competition, etc), this may mean that the physico-chemical quality elements perhaps give a truer depiction of the general state. A conceivable remedial measure is to move the sample points for biology. Thus no class boundary is changed here.

4.3.2.3 WHEN THERE IS REASON TO CHANGE THE REFERENCE VALUE OR CLASS BOUNDARIES IN INDIVIDUAL WATER BODIES

Even if the quality element indicates a specific status, there may be reason not to follow this indication in individual water bodies within a specific type. If the water authority can demonstrate that a water body has been assigned the wrong status classification when a specific assessment criterion has been used, this may be a reason to change the reference value or class boundaries for physico-chemical quality factors in the specific water body.
Adjustments may be made only for a relatively small number of water bodies within each type. This deviating assessment must nevertheless be justified and documented by the water authority.

4.3.2.4 WHEN THERE IS REASON TO CHANGE A REFERENCE VALUE FOR A TYPE

When it is more or less proven that water bodies within a type are constantly assigned the wrong status classification when a specific assessment criterion is used, it may be a reason to change the class boundaries for the whole type. The expediency of so doing should then, if possible, be discussed with officials from other authorities that have carried out classifications of water bodies within the same type. The water authority can then convey this information to the Swedish EPA.

To facilitate communication, answers to the following questions should be prepared:

- Does the request relate to general misclassifications?
- Does it apply to all or only some objects within the type?
- Have others had the same experience (such as county administrative boards, water authorities, the EPA)?
- Is it requested that a sub-type be defined?

If, after consideration, it appears justified to change the reference values or class boundaries for a type, the EPA will make the amendment and communicate it to the water authority.

4.4 Expert judgement

When a water body is not representative of the type and there are no effective assessment criteria, the assessment criteria can nevertheless be used as a basis. It is often possible in different ways to use them in combination with e.g. expert judgement, extrapolation or modelling. This can apply when there is a lack of assessment criteria for a type or they do not function in a water body. Irrespective of which road we go down, a rule of thumb can be that the impact or condition data used in the assessment should not be older than one water planning cycle, i.e. six years. Older data can only be regarded as representative when there are clear indications that the condition of, and impact on, the water body have not changed much over time.

4.4.1 Expert judgement can be made in different ways

Expert judgement is necessary in many situations but it is difficult to provide written guidance on it. What procedure should be followed, e.g. when the biology indicates good status but the known anthropogenic impact indicates something quite different? A judgement can be made in many different ways, depending, among other things, on the knowledge and experience of the person making it, and what background information is available.

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Whatever the judgement made, it is important that the procedure is documented in some form. To track data or judgements where no information is available about their origin can be extremely time-consuming and at times impossible. If others are to work with the same data or judgements at a later stage, the origin of the information needs to be clear.

If, for example, models are used that automatically carry out many calculations and can therefore facilitate the work appreciably, there still remains a type of expert judgement to be made. Models are never perfect and interpretation of them requires knowledge and experience. Even if running a model produces a ready-made judgement, for example about a status, it may not be sensible to trust it blindly, and a reasonability assessment of the result should first be made. Models are not dealt with at length in this chapter, since it is difficult to identify a model that suits many contexts. Models are often constructed for a specific task and before running one, it is important to think about its real purpose and what it does.

If data is available for a parameter which is not part of an assessment criteria, it can in certain cases be convenient to use this data in an expert judgement. Nevertheless, it is a prerequisite that this data indicates the impact or effect which it is intended to assess. The genus Unionoida (large mussel) may be mentioned as an example of a parameter for which assessment criteria are currently lacking. In cases where reproducing Margaritifera margaritifera, which belong to the Unionoida genus, have been found, this is probably a sign that the watercourse has good or high status. In many watercourses, on the other hand, Margaritifera margaritifera are not a good indicator at all, because they may not, for example, have reproduced for many years, or because they are simply not present in the watercourse.

Another example is when available data for a few macroalgal species or angiosperms in coastal water is used together with the qualitative description in Annex B, Section 3.7.

It is complicated to draft guidance for every situation that can arise since the pre-conditions for each situation vary. For that reason, this handbook takes up only a few selected parts of the concept “expert judgment”.

4.4.2 Expert judgement when there is a lack of supporting data

Judgement and classification of the environmental quality in a water body should ideally be carried out on the basis of empirical data of high quality. In practice, however, many water bodies will be judged on the basis of more or less incomplete empirical material. In many cases data will be lacking for all parameters required for quality elements or assessment criteria, in other cases data will be available for only some of these parameters. Since the Water Management Ordinance requires that quality elements be weighed together, this means that in many water bodies it will be necessary to establish quality objectives and assess whether the environmental quality matches them, with the aid of a combination of environmental data and "expert judgements".

There may be anxiety that a doubtful expert judgement can have major adverse significance, perhaps resulting in very expensive mitigation measures. However, this need not be so. If an expert judgement concludes that a water body probably has moderate status, it automatically requires the water body to be monitored operationally. This then entails the collection of additional data that can verify the expert judgement. A dubious expert judgement need not therefore result in unnecessary programmes of measures.

Table 4.7 shows conceivable methods for expert judgement of environmental quality for specific parameters within a quality element. The list is incomplete, and certain methods overlap, or are used in combination. The grouping of methods is to some extent arbitrary but the purpose is to be able to show the advantages and disadvantages of different approaches and to propose criteria and methods to measure the certainty of a status classification made on the basis of expert judgements. An important objective of such a grouping is to create common terminology which can be used in, for example, WISS to describe and document the different types of currently used methods for expert judgement, and the criteria on which status classification is made.

The methods can be divided into two groups, informal and formal. Their internal order in the list is not obviously synonymous with a ranking, but it is appropriate as far as possible to strive after objectivity and maximum information content (Figure 4.12). The informal methods can in certain cases be the only ones possible, but there is a risk of their being subjective and subject to human error. Formal methods are less subjective but on the other hand they require more quantitative information and technical expertise. Quantitative information is also the strength of the available models.

See AR to Chapter 2 Section 8 §

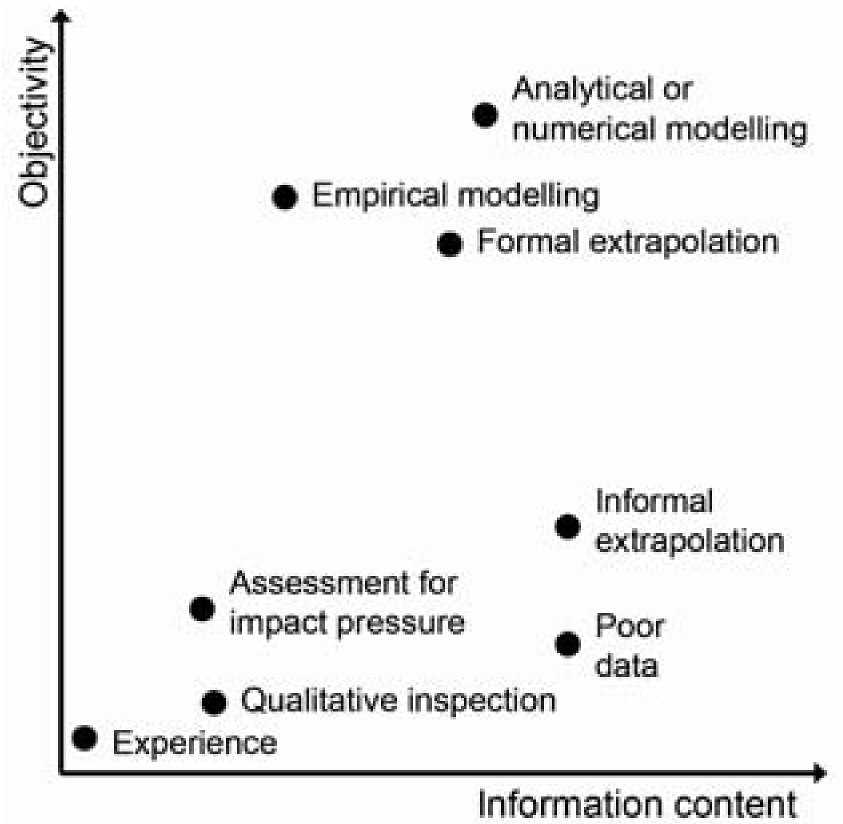

Figure 4.12. Picture of the relationship between objectivity and information content in an expert judgement. Note, however, that the diagram says nothing about the accuracy of results obtained with the different methods.

How should the quality of an expert judgment be measured and documented?

As regards empirical data, there are accepted methods and criteria for how information quality can be measured and managed. The precision of an estimated mean value for an individual parameter can, for example, be measured with the aid of a 'standard error' or confidence interval (Section 4.1.2.1). Using statistical theory, the number of random samples needed to achieve a desired precision can then be calculated. Using information about the uncertainty in specific input parameters, the probability of misclassification can then be calculated for an entire quality element. Assessment of the quality and uncertainty in an expert judgement at the parameter and quality element level is nevertheless more difficult. Since expert judgments can be made in a number of different ways, there is no general method for how their quality and uncertainty can be assessed. For the assessments to be comparable from case to case, the minimum requirement must be that the methods of expert judgement used, and the extent to which they are used, are always documented in the WISS database. In addition, expert judgements, like judgements based on data, should as far as possible be accompanied by some measure of uncertainty.

The method for determining uncertainty in an expert judgement of a specific parameter varies depending on what method is applied (conceivable approaches are described in Table 4.7). Even if there are general differences in uncertainty between different methods of expert judgement, it is not possible to give any simple order of priority. Judgements based on sophisticated analytical models are probably less vulnerable to human error, and have estimatable uncertainty, as compared with those based on subjective expert judgement by personnel with special qualifications. The aim must therefore be to develop objective, quantitative methods based on scientific facts rather than on personal "experience" that is difficult to describe. Nonetheless, it is possible that subjective judgments can occasionally give better quality judgements than more objective methods. Whatever the method used, it is of the utmost importance that the supporting data and criteria are clearly stated.

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Table 4.7. Examples of methods for expert judgement of individual parameters. The use of the term extrapolation relates to the fact that judgements from one or more water bodies are extrapolated to other water bodies. With this starting-point, extrapolation also includes situations where the condition of geographically intermediate water bodies is interpolated from water bodies with known conditions, using "krieging" or regression methods.

Method of expert judgement	Explanation	Examples of methods for estimating uncertainty
Informal methods		
Experience	Previous "impression" of the parameter in the water body	
Qualitative inspection	Current, visual or other unquantified inspection.	
Defective data (does not meet the recommendations in the assessment criteria)	For example, access to old or incomplete data about the parameter in question.	Independent information about variation in the parameter.
Assessment from impact pressure	Knowledge of specific impact pressures and their effects on the parameter are used to draw conclusions about the parameter.	
Informal extrapolation	Neighbouring or similar water bodies are used in an informal way to assess the condition or to set quality requirements.	Concordance between neighbouring water bodies.
Formal methods		
Analytical or numerical mod-	Quantitative, geographically	Uncertainty analysis with

Method of expert judgement	Explanation	Examples of methods for estimating uncertainty
elling	explicit models of processes and impact pressures that affect the parameter, e.g. the FYRIS model, HBV-PN, Watchman or SMHI's Coastal Zone Model.	the aid of Monte Carlo randomisation methods or independent validation.
Empirical modelling	Modelling in which statistical, empirical relations are used to extrapolate or interpolate to water bodies where the parameter has not been measured, e.g. the habitat modelling presented in SAKU.	Uncertainty analysis with the aid of Monte Carlo randomisation methods or independent validation.
Formal extrapolation	A group of water bodies is used to set quality requirements or assess the condition, by means of e.g. "type-grouping" in WISS.	Concordance between neighbouring bodies and clear criteria on what requirements must be met for type-grouping.

Since it will be impracticable, or financially impossible, to achieve complete, representative sampling in all water bodies in accordance with Figure 4.13a, different types of extrapolation will probably be a common type of expert judgment (Table 4.7). That means that results from the three sampled water bodies in Figure 4.13b will also be used for statements about the status in the four where no samples have been taken. This method can also be used in WISS, where it is possible to group several water bodies within the same type into a "type-group" and to jointly classify them The logic behind type-grouping is that the water bodies that are included in the group are all of the same type and affected by similar impact factors and impact pressures. In a limnic environment the tool System Aqua[26] (example in Section 4.5) can be used to assess whether these requirements are fulfilled. For the marine environment, there are no developed formal procedures, but one can imagine that, for example, the SMHI coastal waters model could function in a similar way. Irrespective of what tool is used, decisions about type-grouping of biological parameters will nevertheless be based on information about a small number of factors in adjacent areas, whose impact on biology in many cases will be uncertain. The basis for type-grouping will therefore be uncertain and this will in some cases lead to misclassification. Since this form of expert judgement will be common, there is reason to reflect on methods to measure and document the extent of this uncertainty. This is exemplified below by the benthic quality index (BQI) in sea areas.

[26] Bergengren, J. and Bergquist, B. 2004. System AQUA 2004 – Part 1. Hierarkisk modell för karaktärisering av sjöar och vattendrag.[Hierarchical model for characterising lakes and watercourses] Jönköping County Administrative Board Communication 2004:24.

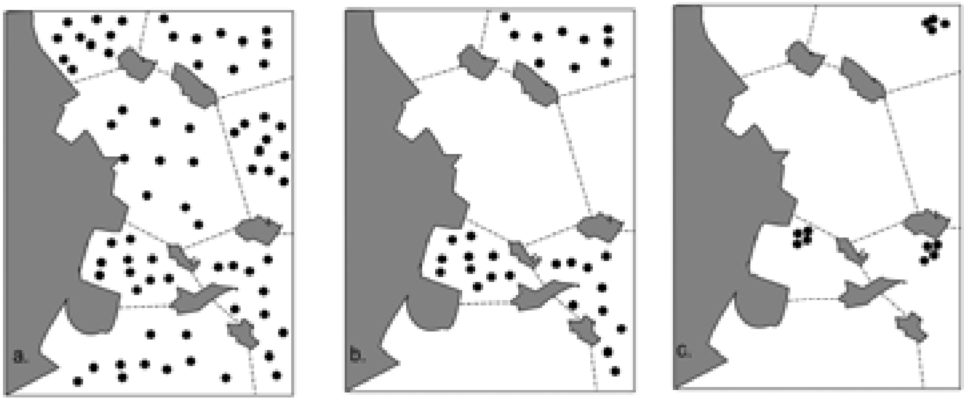

Figure 4.13. Examples of stationing sample points in seven marine water bodies: (a) geographically representative sampling in all water bodies, (b) geographically representative sampling in three water bodies, (c) non-representative sampling in three water bodies.

Uncertainty with BQI for benthic macroinvertebrate assemblages in type-grouping in coastal areas of the Gulf of Bothnia.

Even if there may be good reasons to assume that water bodies of the same type and with similar impact pressures have similar ecological status, there is reason to be careful in drawing such conclusions. There can be both known and unknown natural factors that mean that the assessment criteria function with varying success in different water bodies, but there may also be differences in the impact pressures that are not yet known, or where the effects on the biology are not known. Status classification by extrapolating from a small number of water bodies therefore leads to the introduction of further uncertainty.

One way of assessing the size of this uncertainty can be to consider sampled water bodies as a representative random sampling of water bodies within a certain type and impact class. This is furthermore a precondition without which the grouping cannot be justified. The degree of uncertainty will then be determined by the size of the difference between the water bodies for which data is available. Even if measurements have not been taken in all water bodies, the difference between the mean values from the three sampled water bodies in Figure 4.13b represents the size of the difference to be expected between water bodies in the group in question.

If the mean values vary a great deal between these three water bodies, it must be expected that there is in general a major difference between water bodies and the extrapolation must be regarded as unreliable. If there is little difference, the type-grouping and extrapolation are more reliable.

Table 4.8. Table showing the variance analysis for one year's sampling i with n tests in each of a water bodies (WB). 1For reasons of chance can sometimes be estimated as <0. This is an illogical result (but not improbable if is small). In such cases, can be estimated by calculating the mean values for each water body and the variance between them.

Source	Levels	df	Component,	MS estimates	[1]Calculation	Confidence of interval
WB	a	a-1	σ_{VF}^2	$\sigma_\varepsilon^2 + n\sigma_{VF}^2$	$\frac{(MSVF - MS\mathrm{Re}s)}{n}$	$\sqrt{\frac{\sigma_{VF}^2}{a}} * t_{krit,a-1}$
Res	n	a(n-1)	σ_ε^2	σ_ε^2	$MS\mathrm{Re}s$	$\sqrt{\frac{\sigma_\varepsilon^2}{n}} * t_{krit,n-1}$

A simple way of performing these calculations is to use the structure in a variance analysis (ANOVA) to estimate the mean squared sums (MS) and the contributions caused by different variation sources. If, for example, there is data which in a representative way estimates the condition in a number of water bodies (Figure 4.13b), a calculation can be made of the variation that represents differences between samples, σ_ε^2 , and that which represents differences between water bodies, (Table 4.8). With the aid of and t-distribution, we can subsequently calculate the limits within which, with a given probability, the true mean value within an area lies. For example, in the assessment criteria for soft-bottom fauna for coastal waters, we have chosen to classify the status according to the lower boundary for a one-sided confidence interval. Estimation of the difference between water bodies, , can be used to calculate the boundaries within which the mean values for other water bodies should lie. Assuming a normal distribution, this can be done by calculating the "population interval":

$$\mu = \overline{X} \pm 1.96 * \sigma_{wb},$$

The estimation of can also be used to calculate a confidence interval for mean values for the whole area (Table 4.8).

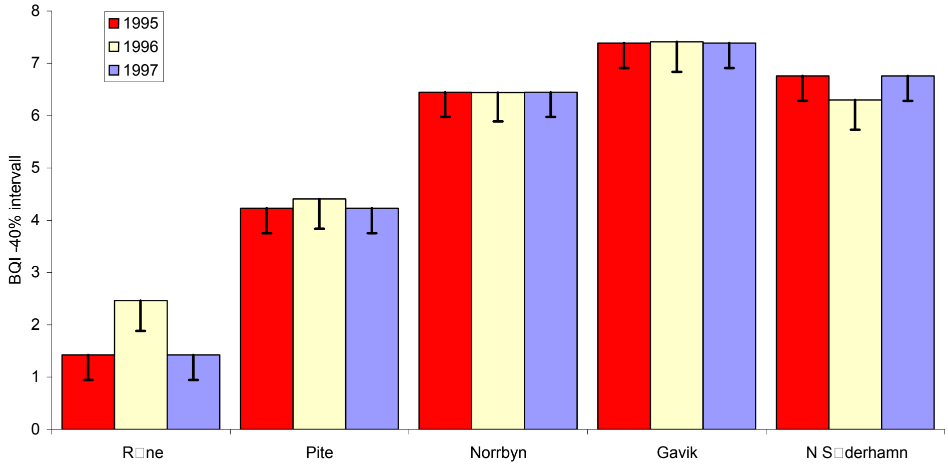

Figure 4.14. Mean values – 40% confidence interval of BQI for five areas in the Gulf of Bothnia.

Example – calculation using data from areas with different types
To exemplify the above, it is assumed that all water bodies in the coastal areas of the Gulf of Bothnia are to be grouped. Because the areas represent different types, this ought not to be done, but the intellectual experiment can nonetheless be made in order to illustrate the principles. Using three years' data from five areas in the Gulf of Bothnia, we can calculate the mean value and the spread within and between areas, which are assumed to correspond to individual water bodies. There are relatively large differences in the mean values for the benthic quality index, BQI, and in the estimated classification (lower boundary for a one-sided 20% confidence interval) as between the areas (Figure 4.14), but the internal order seems similar as between years. Above all, the two areas in the northern part of the Gulf of Bothnia (Råneå and Piteå) differ from those in the Bothnian Sea. As an example, analysis of the 1997 data shows that the variation between the areas is approximately 2.3 while that within the areas is about 10 (Table 4.9). The 95% confidence intervals for estimated mean values within the areas is 1.50, which means that the true mean value within an area with 95% probability lies within an interval is ±1.50 BQI from the estimated mean value (X=5.24). The confidence interval for the mean value in the whole area is ±1.88 from the total mean value.

With the aid of an estimate of σ^2_{wb} and the expression for the population interval, we can calculate the boundaries within which the mean values for other, unsampled, water bodies, should lie with 95% probability. In 1997, 95% of the water bodies thus lay within the interval 5.24±4.68. If similar calculations are made for all years, it transpires that 1997 was the year that had the greatest variation between water bodies, but that the interval generally is large (> 2.96) if one were to group the whole of the Gulf of Bothnia (Figure 4.15a). To group the benthic macroinvertebrates in the coastal areas on this scale would thus lead to very great uncertainty when extrapolating to other, unsampled, water bodies. That corresponds well with what can be expected since the example represents an attempt to group water bodies from different types.

Table 4.9. Table for the variance analysis of BQI for 1997 sampling of benthic macroinvertebrates in five coastal areas in the Gulf of Bothnia (for the calculation of σ^2 and confidence intervals, please see Table 3). Even if the formal test is not shown, the results reveal a significant variation between the areas (p<0.001).

Source		**Levels**	**df**	**MS**	σ^2	**SE**	**$t_{0.05, N-1}$**	**KI**
Area	5	4	56.11	2.29	0.68	2.78	1.88	
Residual	20	97	10.30	10.30	0.72	2.07	1.50	

Example – calculations for areas within the same type

If the same calculations are instead applied to areas which belong to the same type, i.e. only those areas that lie in the Bothnian Sea, the situation becomes a little different. For data collected in 1997, the mean value of BQI is 7.14. The population interval during the same period is calculated as ± 0.65, which means that in the Bothnian Sea, 95% of all areas should lie between 7.79 and 6.49. According to expectation bearing in mind the typology, the type-grouping in this case seems more reliable.

Since the assessment criteria for benthic macroinvertebrate assemblages are designed in such a way that the classification is made on the basis of the lower boundary for a one-sided 20% confidence interval, it may be of interest to investigate how much the classification may be expected to vary between different water bodies in the Bothnian Sea. To do this, calculate the boundaries for the individual areas (which in 1997 were 6.77, 6.70 and 6.14 for Norrbyn, Gavik and Söderhamn respectively) and estimate the average value and standard deviation (6.53 and 0.34, respectively). With the aid of these estimates it can be calculated that during 1997 the classifications in 95% of the water bodies in the Bothnian Sea appear to lie within the interval 6.53±0.66. Apart from small deviations, this pattern was observed in all three years (Figure 4.14). Furthermore, it can be noted that more than 95% of water bodies should be expected to have been of good status in all years. Once again, it seems as if there is some empirical support that grouping water bodies in the Bothnian Sea does not introduce any dramatic uncertainty.

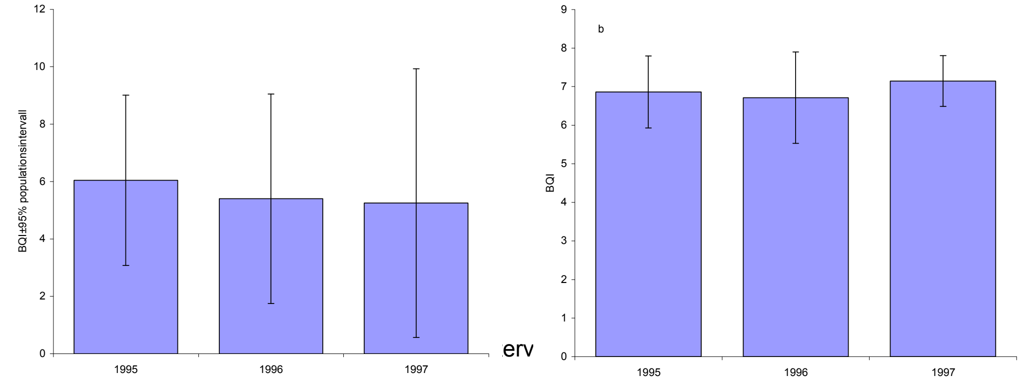

Figure 4.15. Mean values – 95% population interval of BQI for (a) the whole of the Gulf of Bothnia and (b) the Bothnian Sea.

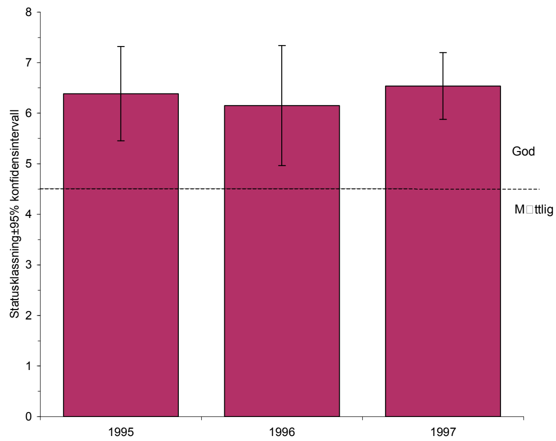

Figure 4.16. Mean values for status classification of BQI in the Bothnian Sea. The class boundary between good and moderate status has been set at 4.5.

On a general level, the above example illustrates a formal method that can be used to assess uncertainty when grouping. It is based on representative underlying sampling in relation to one water body (Figure 4.13b). In certain cases, for example for other quality elements, some sampling was conceivably designed otherwise. One possibility is that the sampling was planned in such a way that only a small part of the water body is monitored (for example, one "station", Figure 4.13c). If the current measurement variable shows a significant spatial structure within the water body (for example, gradients or local variations), this means that monitoring of one station per water body may overestimate the variation (between water bodies, that is). This will lead to considerable uncertainty being introduced when grouping water bodies. How great this problem will be in practice probably depends on the context. Irrespective of how the sampling has been designed, the method described can be used to give an approximate idea of the uncertainty.

4.4.3 Classifying status when water chemistry data is virtually the only data available

When status classification is to be made in a water body for which there is no biological data, data should if possible be gathered beforehand. If not, there is no alternative but to use whatever information is available, and to make an expert judgment instead. If there are physico-chemical parameter measurements (often termed water chemistry) they can be of great assistance in making an expert judgment. Together with the potential impact we already know of, they can be sufficient for making a simple status classification.

We could perhaps begin by, say, collecting available data about the area, in order to create an overall picture. The data can then be sorted according to what it might indicate. An assessment ought also to be made of how far the available samples are representative of the water body. Representativity can be affected by e.g. the point in time at which the samples are taken, the sampling locality, or the quality of the sampling and analysis. The working procedure might then be as follows:

1. **List the physico-chemical parameters that have been measured in the water body.**
   If there are assessment criteria for the respective parameters, these can be used to classify status. If there are no assessment criteria for the parameter, search for other information that can give a similar indication about status

2. **List potential impacts in the water body.**
   Major impact can be an indication of worse status. Worse status means in this case moderate or worse status. Minor impact can be an indication of better status. Better status means in this case good or high status.

3. **Assessment:**
   a) If physico-chemical parameters indicate better status and other impacts in the water body are minor, the status can be assumed to be good or high.
   b) If physico-chemical parameters indicate better status and other impacts in the water body are major, it may be difficult to assess the status. Further investigations should then be made in order to be able to classify the status of the water body.
   c) If the physico-chemical parameters indicate worse status and other impacts in the water body are minor, it may be difficult to classify the status. Further investigations should then be made in order to be able to classify the status of the water body.
   d) If the physico-chemical parameters indicate worse status and other impacts in the water body are major, the status can be assumed to be moderate or worse.

In cases a and d, the probability of making a correct judgement is high, while in cases b and c further investigations are necessary in the form of new sampling or the production of more information about impacts in the water body.

The above approach is intended as a simple status classification when there is a lack of biological data, but new data should be collected in order to verify the judgements made.

4.4.4 Classifying status on the basis of an impact assessment

In contrast with the example in the previous section, there will be areas where a status classification must be made, despite the fact that no condition data is available. This applies in large parts of the country (particularly in northern Sweden) where there is no or little data from field mapping and sampling, which means that it may initially be necessary to make a status classification wholly or partly on the basis of an impact assessment. The starting-point is then the impact analysis that has been carried out during the characterisation work. By sorting the water bodies on the basis of the extent of anthropogenic impact, a rough classification of a presumed status can be obtained. It is assumed that there is a strong link between anthropogenic impact and status, and also that no other factors are obviously causing or have caused changes in the water body. The normative definitions for classification of ecological status in Table 1.2 in Annex V of the WFD emphasise a strong link between anthropogenic impact and status.

High status according to the general definition of ecological quality is defined as:

> There are 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. The values of the biological quality elements 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. These are the type-specific conditions and communities.

Good status is defined according to the general definition:

> 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 status is defined according to the general definition:

> The values of the biological quality elements for the surface water body type deviate moderately from those normally associated with the surface water body type under undisturbed conditions. The values show moderate signs of distortion resulting from human activity and are significantly more disturbed than under conditions of good status .

When there is a lack of condition data for certain water bodies, their status can be classified based on condition data from water bodies that are similarly disturbed, that is to say based on other bodies in the same type-group. It is also appropriate that the focus should lie on one type of impact at a time (e.g. acidification, eutrophication or metal pollution.) A proposed five-step working method is illustrated in Figure 4.17. It begins with two steps that are merely an example of how the data can be grouped, before, in the third step, type-grouping the water bodies are type-grouped and the status is classified based on the condition data available in each type-group. In step four, a reasonability assessment is made of the intended status for each water body and in the final step the status that is to apply to each water body respectively is determined. This step-by-step working method is explained in greater detail below.

Step 1. Assessment of the level of impact in the water body resulting from human activity

One can begin with a rough breakdown of the water bodies into three groups, based on how disturbed the water body seems to be. In this case the groups have been termed “insignificantly disturbed”, “slightly disturbed” and “highly disturbed”, this terminology having been chosen to avoid confusion with other concepts (such as "significant distortion"). This assessment can be made both subjectively, on the basis of one’s own knowledge of the impact in the water bodies, and objectively, on the basis of stress calculations with the aid of source distribution models and appropriately set limit values for the impact in focus. In the first instance, objective methods should be used. It may be appropriate, as in the example, to choose three groups that correspond to the definitions of ecological status classification in Table 1.2 in Annex V of the WFD. If any water bodies should be put in the wrong group, it does not matter, since this should in any case emerge when type-grouping the water bodies in Step 3. The purpose of this step and the next is merely to create a certain order among the water bodies in order gain a better overview of them. If it appears that very good knowledge about impact in the water bodies is already available, we can progress immediately to Step 3 and perform a type-grouping.

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Step 2. Subjective status classification

Since the status classification partly builds on the level of impact in a water body resulting from anthropogenic activity, a subjective and very preliminary status classification can be made at this stage. In the example, the water bodies in the “insignificantly disturbed” group are assigned the status “presumed high status”, the “slightly disturbed” group are termed “assumed good status” and the “highly disturbed group” are termed “presumed worse than good status”.

Step 3. Verification by type-grouping and status classification according to assessment criteria.

In this step the subjective status classification is verified by type-grouping the water bodies and classifying the status of water bodies within the group for which condition data is available. The classification is then made using the assessment criteria. The intention is then to assign the same status classification to the water bodies for which there is no condition data as the status assigned to water bodies within the type-group that were classifiable using assessment criteria. If there is insufficient condition data within a type-group, as regards data on the condition of the water bodies, it is recommended to consider supplementary sampling of relevant quality elements in order to be able to classify status. If no supplementary measurements are made in this case, the expert judgement that will probably be required will be very unreliable. If the classification is moderate or worse, supplementary sampling of relevant quality elements is appropriate.

Step 4. Reasonability assessment

When the status classification of the type-groups has been carried out, a assessment is made about the reasonability of the status classification regarding each water body included. In this case, there are often no condition values, but it is nevertheless appropriate to try as far as possible to follow the flow diagram in Figure 4.1, to determine the reasonability.

Step 5. "Extrapolated" status classification

The status deemed to be the result after using the assessment criteria in Step 3 can then be classified within the type-group as the status for each water body respectively, on condition that the reasonability assessment has shown that this status is reasonable.

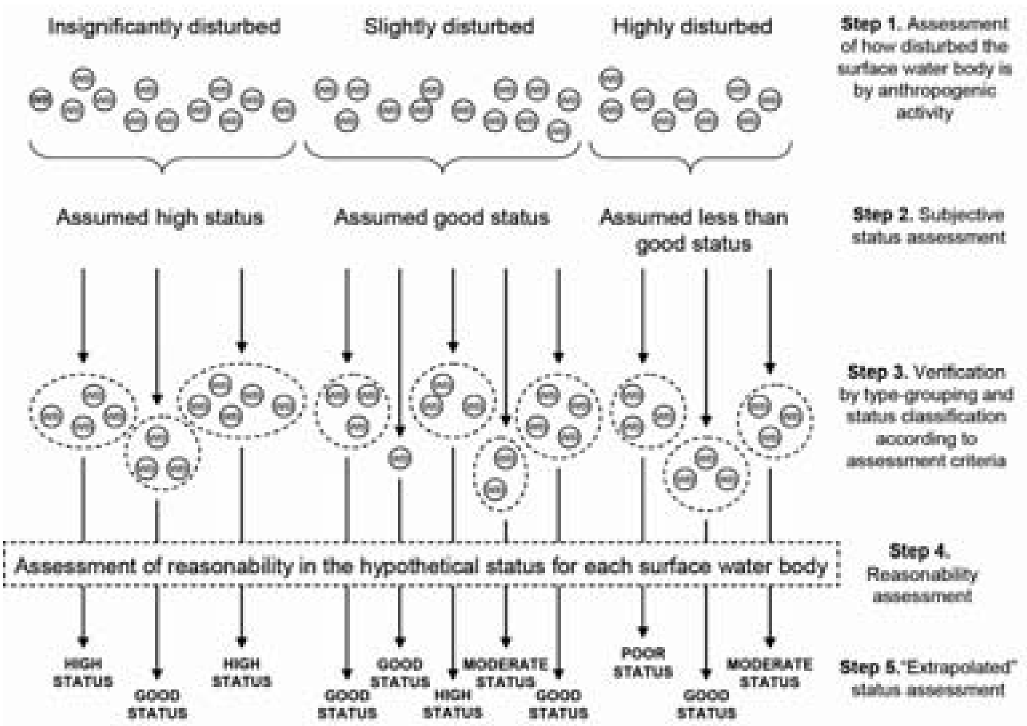

Figure 4.17. Example of working procedure in status classification based on how disturbed the water body is thought to be. WB is in this case an abbreviation for water body.

An expert judgement can be made based on the impact analysis and other available knowledge, in order to determine the status of water bodies that lack condition data and do not match any type-group; or the type-groups for which information is incomplete; or for which there is no supporting data at all in the form of sampled quality elements. It should nevertheless be kept in mind that expert judgements can prove very unreliable. It is therefore appropriate that a reasonability assessment is made for each water body in this case too. If after the above procedure it is proposed to classify status in a water body as moderate or worse, it is appropriate to carry out new sampling of the relevant quality elements and to classify the body using the assessment criteria, in order to reinforce this classification. This is done to avoid implementing mitigation measures on incorrect grounds. IN addition, the water body should not become subject to operative monitoring when it is not necessary.

4.4.5 Documentation

To make it easy to check how specific classifications have been carried out, it is important to document clearly how different types of information have been used in a classification. The water authority shall for each water body post a report in WISS, or other corresponding database, of how the status classification has been carried out and of the result for each classified quality element in it. This should also be reported in the form of ecological status and surface water chemical status or ecological potential and surface water chemical status. Moreover, information about the supporting data used in the classification should be documented for each classified quality element respectively.

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The documentation of the above should show whether the classification has been made for parameters based on condition data in accordance with the requirements in the assessment criteria or whether the classification is wholly based on modelling based on impact data, such as discharges, land use, stress etc., or whether the classification has been performed based both on condition data and on impact data. The documentation for all water bodies should be carried out in a uniform way, since that facilitates, for example, comparisons of different water bodies and gives a clearer indication of well how well supported a specific classification is. Documentation is required for each classified quality element.

4.5 Grouping of water bodies

In Sweden there is very large number of surface water bodies and it is impossible to take measurements in all of them. To simplify the management of water bodies, they can be divided into sub-populations such as type-groups. In a type-group, all water bodies are of the same category, i.e., lake, watercourse, coastal waters or transitional waters. The definition of a type-group is “a collection of water bodies that belong to the same type (according to the Regulations on mapping and analysis, NFS 2006:1) and that have the same level and type of impact”.

Instead of describing the condition of individual water bodies, a description can then be given of the condition of a type-group of water bodies. Estimating status on the basis of other water bodies within the same type-group is not an exact method, but it gives guidance for the classification. The alternative can be to simulate data for quality elements using models.

4.5.1 Division into type–groups – an overview

Before dividing the water bodies into type-groups several other steps are necessary; characterisation in water types under NFS 2006:1, division into types for assessment criteria as is described for each quality element in Annex A and assessment of potential impact (Figure 4.18). For more information about these elements, please refer to the Regulations on mapping and analysis (NFS 2006:1) and the associated handbook (2007:3).

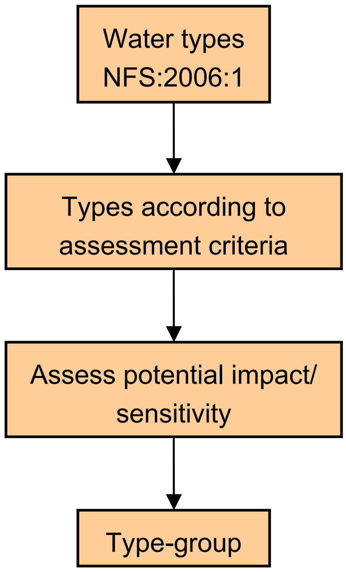

Figure 4.18. Different stages necessary to make a division into type-groups.

4.5.1.1 DIVISION INTO TYPES IN ACCORDANCE WITH NFS 2006:1

The characterisation into limnic types (according to NFS 2006:1) in freshwater is based on natural preconditions (geographical ecoregions). For lakes, the determinant characteristics are humic content, lime content, maximum depth and surface area. For watercourses it is length and the size of catchment area. The main purpose with the division into types is to develop comparable types so that assessments of deviations from the reference condition can be made in a uniform way. A water body can belong to only one type and different types may not overlap one another.

The Swedish coast is already divided into 23 coastal water types (under NFS 2006:1) and two types of transitional waters. A type of coastal or transitional water consists of a major sea area which is relatively similar as regards salt content, water exchange, layering and wave impact. Hydromorphological and physico-chemical conditions create preconditions for relatively similar biological conditions within the respective coastal water type. The water bodies that have been classed as transitional waters lie in the inner Stockholm archipelago and Hallsfjärden, as well as the estuaries of the Göta River and the Nordre River. The majority of coastal areas have an inner type near the coastline and an outer type towards the open sea and each individual type can contain a varying number of water bodies.

4.5.1.2 DIVISION INTO TYPES ACCORDING TO ASSESSMENT CRITERIA

After characterisation into limnic types in accordance with NFS 2006:1, the limnic water bodies shall be divided into the types used for the assessment criteria (Annex A). This is often a less detailed division into major types, which it has been possible to distinguish based on the background material used to develop the assessment criteria. Details about this for each quality element respectively are found in the text describing the assessment criteria. As an example, it may be mentioned that benthic macroinvertebrate assemblages in freshwater are divided into three types according to Illies Ecoregions and phytoplankton into five types according to ecoregions and humic content. A consequence of this is that two water bodies within a limnic type under NFS 2006:1 can fall into different types for the various quality elements. Typing water bodies under NFS 2006:1 also helps when arranging the environmental monitoring programme so that better supporting material is obtained to be able to develop reference conditions in the future for more, and more detailed, types.

4.5.1.3 PROPOSED APPROACH

A proposal for an approach to dividing water bodies into type-groups is presented below. Please note that this is not an exact method, but it gives an estimate of the condition and quality elements. The more water bodies with measurement data, the more homogenous type-groups can be constructed. A type-group can of course extend beyond a water district and several water authorities can therefore contribute data.

1) Compile information on the **impact** in the water bodies.
2) What is the level of the **potential impact** or how **sensitive** is each water body to any impact?
3) **Divide the water bodies into type-groups** according to how similar they are as regards impact. All water bodies in a type-group must be exposed to the same potential impact or have the same sensitivity to the impact. If the level of impact varies between different water bodies, these water bodies form different type-groups irrespective of similarities in other respects. Within one and the same type-group, the difference between the levels of impact in the water bodies must be as small as possible. Two lakes with catchment areas consisting of 70% and 20% agricultural land respectively cannot, for example, belong to the same type-group, even if in other respects they have similar impact. The proportion of agricultural land must be approximately the same and have similar distribution within the catchment area for the lakes to be able to belong to the same type-group.
4) Calculate for **how many water bodies measurement data exists for** each type-group. There should be measurement data for at least three water bodies in each type-group. If this is not the case, it may be appropriate to take additional samples.

5) **Classify status in the type-group** or **estimate biological and chemical quality elements** in water bodies without measurement data. Mean values, confidence interval and the median are calculated based on water bodies with measurement data. This requires measurement values from at least three water bodies per type-group. Use the confidence interval, for example, for status classification in high, good or moderate status. It is desirable for the whole confidence interval to fall within one status class. If not, the water bodies are classified according to the precautionary principle, i.e. in the lower status class. Based on knowledge of the water bodies, an expert can assess whether there is reason to exclude a certain extreme water body from a type-group if this water body is not representative of the group. This water body is then excluded when the mean value and confidence interval are calculated.

- *Classification of status in water bodies where data is available from trend stations and flux monitoring stations - three feasible scenarios:*
  1) If, within a type-group, there are only measurement values from three trend stations with annual measurements, the actual measurement values are used to assess the condition.
  2) If within the type-group there are measurements only from flux monitoring stations, the condition is assessed based on, say, the mean, the confidence interval and the median, in accordance with the description above.
  3) Within the type-grouping, there are measurements in both trend and flux monitoring stations. With the aid of trend stations, the condition is assessed on the basis of the measured values. With the help of the flux monitoring stations, the condition is, say, assessed based on the mean, confidence interval and median as described above.

- *Classification of status in water bodies where there is no data from trend stations and flux monitoring stations*:
  Use as a starting-point the water bodies within the type-group for which measurement data exists. Calculate the mean value, the confidence interval and the median for the type-group.

To illustrate how type-group division can be done, two simplified examples are given here. In reality it can be more complicated to type-group, more lakes and greater impact matrices. Use of System Aqua[27] indicator values can be useful to classify impact pressure and hence simplify the grouping.

[27] Bergengren J, and B Bergqvist. 2004. System Aqua 2004 – del 1 – Hierarkiska modell för karakterisering av sjöar och vattendrag [Hierarchical model for characterising lakes and watercourses]. Jönköping County Administrative Board Communication 2004:24.

Example 1: Simplified division into type-groups (freshwater)

Here a simulated example of the division into type-groups is presented, where no account has been taken of the degree of impact. This example only ascertains whether there is potential impact or not. Table 4.10 shows 50 lakes that make up a common type. There are measurement data for 16 of the 50 lakes. 32 of the lakes lie in a forest landscape (coniferous/mixed forest) and 18 in an agricultural landscape. Other potential impacts include clear-cut areas, hydro-electric dams, holiday homes and housing (built-up areas). The potential impact gives seven type-groups. There is measurement data for all but one type-group, between one and four lakes with measurement data in each type-group. Measurement data for these lakes is used to classify the whole group. In type-group 2, there is e.g. measurement data for three out of eight lakes. Using these three lakes, we can assess the condition in the whole group alternatively. In this example, additional sampling will need to be done in at least three of the type-groups (type-groups 1, 4 and 6) where the number of lakes with measurement data is fewer than three.

Supplementary sampling may not be necessary for all quality elements in all type-groups. If the potential impact from forestry and agriculture in the example above is of no significance for benthic macroinvertebrates, type-groups 1 and 2 could be weighed together for benthic macroinvertebrates. On other hand, they can perhaps not be combined for diatoms.

Table 4.10. Division of lakes into type-groups (simulated example). *"forest" means here coniferous/mixed forest.

Typ-grupp	Skog*	Jordbruk	Hygge	Kraftverk	Fritidshus	Bebyggelse	Antal sjöar per typgrupp	Antal sjöar med data	Typgrupp med kompletterings behov
1	X						6	1	X
2		X					8	3	
3	X		X				10	4	
4	X			X			4		X
5	X		X	X			8	4	
6	X		X		X		4	1	X
7		X		X		X	10	3	
Totalt							50	16	

Example 2: Division into type-groups based on degree of impact

System Aqua contains indicator values for different types of impact, varying between 0 and 5 (0 corresponds the highest and 5 the lowest level of impact; Table 4.11). These indicator values can be used to type-group the water bodies. All water bodies with the same indicator value are put in the same type-group.

For the 10 lakes in type-group 3 in example 1 which are potentially disturbed by forest (coniferous/mixed forests) and clear-cut areas, the division, by area, of forest and clear-cut areas in the catchment area is given in Table 4.12. The proportion, by area, of forest land and clear-cut areas in the catchment area corresponds to the potential level of impact. To divide the lakes into type-groups, an assessment must be made as to which lakes resemble one another most based potential impact from, respectively, forest and clear-cut areas. Lakes with identical indicator values for both forest and clear-cut areas (Table 4.13) are put in the same type-group. In this case, there are three type-groups with different types and levels of impact:

- Lakes with the indicator value 0 for forest and 5 for clear-cut areas
- Lakes with the indicator value 1 for forest and 4 for clear-cut areas
- Lakes with the indicator value 2 for forest and 2 for clear-cut areas

There is measurement data in one or two lakes for all type-groups. In this case, there are only three and four lakes in each type-group; therefore supplementary sampling to obtain measurement data from more lakes can be carried out for the division into type-groups.

Table 4.11. Land-use in the catchment area (arable land, clear-cut areas and built-up areas are counted as heavily disturbed). (Source: System Aqua)

Indicator value	Land-use intensity
5	<10% of catchment consists of heavily disturbed vegetation/land-use type
4	≥10 - <20% of catchment heavily disturbed
3	≥20 - <40% of catchment heavily disturbed
2	≥40 - <60% of catchment heavily disturbed
1	≥60 - <90% of catchment heavily disturbed
0	≥90% of catchment heavily disturbed

Table 4.12. Division into further type-groups of the lakes belonging to type-group 3 in Example 1.

Lakes in type-group 3 (Example 1)	Lakes with data	Proportion of forest* (%)	Indicator value forest	Proportion clear-cut areas (%)	Indicator value clear-cut areas
1		99	0	1	5
2		91	0	9	5
3	x	97	0	3	5
4	x	95	0	5	5
5		88	1	12	4
6		85	1	15	4
7	x	81	1	19	4
8		51	2	49	2
9		55	2	45	2
10	x	58	2	42	2

* coniferous/mixed forest

Table 4.13. Number of lakes with the same indicator value for forest/clear-cut areas respectively (the lakes comprise type-group 3 in Example 1). Lakes with the same indicator value belong to the same type-group.

Clear-cut area / 4.5.2 Forest	5	4	3	2	1	0
5						
4						
3						
2				3		
1		3				
0	4					

5 Surface water chemical status

Surface water chemical status shall be classified for substances which have common EC environmental quality standards. This applies particularly to the priority substances. Decision No. 2455/2001/EC to the WFD established 33 priority substances regulated in Annex X of the WFD. In December 2008, a daughter directive to the WFD was published . This directive establishes limit values for 33 priority substances and 8 other pollutants. The water authority shall use these limit values when classifying and determining quality requirements for surface water chemical status. Environmental monitoring and status classification only need to be done for substances that are discharged into the water body. In Guidance Document No. 3 within the EU (Analysis of Pressures and Impacts[28]) the concept of "discharge" is interpreted in a broad sense. It covers discharges from point sources in the river basin/catchment area, leakage from diffuse sources and e.g. atmospheric deposition from other areas. We should therefore take all the possible pathways by which the pollutant can reach the water body into consideration.

See REG
Chapter 3
Section 4

The 33 priority substances and the 8 other pollutants that will be regulated are as follows:

Priority substances

1. Alachlor
2. Anthracene
3. Atrazine
4. Benzene
5. Brominated diphenyl ethers
6. Cadmium and its compounds
7. C10-13-chloroalkanes
8. Chlorfenvinphos
9. Chlorpyrifos
10. 1,2-Dichloroethane
11. Dichloromethane
12. Di(2-ethylhexyl)phthalate (DEHP)
13. Diuron
14. Endosulfan
15. Fluoranthene
16. Hexachlorobenzene
17. Hexachlorobutadiene
18. Hexachlorocyclohexane

[28] Common Implementation Strategy for the Water Framework Directive (2000/60/EC) Guidance no 3 Analysis of pressures and impacts, produced by working group 2.1 – IPRESS, 2003

19. Isoproturon
20. Lead and its compounds
21. Mercury and its compounds
22. Naphthalene
23. Nickel and its compounds
24. Nonylphenol (4-nonylphenol)
25. Octylphenol (para-tert-octylphenol)
26. Pentachlorobenzene
27. Pentachlorophenol
28. Polyaromatic hydrocarbons
(Benzo(a)pyrene),
(Benzo(b)fluoranthene),
(Benzo(g,h,i)perylene),
(Benzo(k)fluoranthene),
(Indeno(1,2,3-cd)pyrene)
29. Simazine
30. Tributyltin compounds (Tributyltin-cation)
31. Trichlorobenzenes
(1,2,4-Trichlorobenzene)
32. Trichloromethane (Chloroform)
33. Trifluralin

Other pollutants
1. DDT total
2. Aldrin
3. Dieldrin
4. Endrin
5. Isodrin
6. Carbontetrachloride
7. Tetrachloroethylene
8. Trichloroethylene

Directive 2008/105/EC provides Member States with the opportunity to take natural background concentrations and bioavailability for metals into account. For the metals lead, mercury and cadmium, however, waters with no local discharges cannot be considered to be undisturbed. Concentrations of these metals in the environment are general and clearly affected by air emissions and long-distance dispersion in the atmosphere. Regarding mercury and cadmium, we have to go back to the pre-industrial era, about 150 years ago, to find natural concentrations, whilst the impact of lead has been going on for a very long time, about 3 000 years. More information on how background levels can be assessed and how to take bioavailability into account can be found in Annex A, Chapter 16 and Annex B, Chapter 8.

Other substances that have common EC limit values shall also be used when classifying and determining environmental objectives for surface water chemical status. This applies to substances and groups of substances regulated in the EC Freshwater Fish Directive and Shellfish Waters Directive which have been implemented in Sweden through Ordinance (2001:554) on environmental quality standards for fish and bivalve (mussel) waters. The chemicals regulated in these provisions are nitrites, phenol compounds, mineral-oil-based hydrocarbons, ammonia, ammonium, residual chlorine, organic halogen compounds and a number of metals. The levels for these environmental quality standards are, however, primarily set to prevent fish and bivalve from tasting bad and they have in most cases no numerical values.

More guidance on the classification of chemical status will be issued at a later date once the daughter directive has been adopted and it is clear how it is to be incorporated into Swedish legislation.

6 Literature

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Annex A - Assessment criteria for lakes and watercourses

(This annex contains the text for all assessment criteria for lakes and watercourses and can be downloaded as a separate document from the Swedish EPA's website at www.naturvardsverket.se. The reason for this is so that the user can avoid having to download files that are very big and hence difficult to handle).

Annex B - Assessment criteria for coastal and transitional waters

(This annex contains the text for all assessment criteria for coastal and transitional waters and can be downloaded as a separate document from the Swedish EPA's website at www.naturvardsverket.se. The reason for this is so that the user can avoid having to download files that are very big and hence difficult to handle).

Annex C - Assessment criteria for hydromorphological quality elements

(This annex contains the text for all assessment criteria for hydromorphological quality elements and can be downloaded as a separate document from the Swedish EPA's website at www.naturvardsverket.se. The reason for this is so that the user can avoid having to download files that are very big and hence difficult to handle).

Annex A - Assessment criteria for lakes and water-courses

(This annex contains the text for all assessment criteria for lakes and watercourses and can be downloaded as a separate document from the Swedish EPA's website at www.naturvardsverket.se. The reason for this is so that the user can avoid having to download files that are very big and hence difficult to handle).

Contents

1 Introduction

The assessment criteria for lakes and watercourses have been produced by the National Board of Fisheries, SLU (Swedish University of Agricultural Sciences), Luleå University of Technology, Stockholm University and the consultancy firm Jarlman HB on behalf of the Swedish Environmental Protection Agency (Swedish EPA).

Within the EU, intercalibration of the class boundaries between high and good, and also between good and moderate, has been carried out for the biological quality factors in accordance with the standards laid down in the Water Framework Directive (WFD). Intercalibration work has been carried out within the Common Implementation Strategy (CIS) and has been based on a comparison between the different Member States' class boundaries for the respective parameters or quality factors and, where necessary, adjustment of the boundaries in order to guarantee equal protection of the water environment. EU waters have been divided into different types to enable the comparison to be made between waters with the same conditions. This work has been carried out in a series of different working groups and has involved a considerable number of experts.

Because of the lack of comparable data and classification systems, it was not possible to calibrate all parameters within the different quality factors. As far as Sweden is concerned, the quality factors and parameters for lakes and watercourses that have been intercalibrated up to and including 2007 are as follows:

LAKES:
Phytoplankton – chlorophyll, cyanobacterial abundance (completed but not formally adopted)
Macrophytes – TMI (completed but not formally adopted)
Benthic (bottom-dwelling) fauna – MILA (work in progress)
Fish - EQR8 (work in progress)

WATERCOURSES:
Macrophytes – DJ-index (completed but not formally adopted)
Benthic fauna – MISA (work in progress)
Diatoms – IPS (completed but not formally determined)
Fish – VIX (work in progress)

Following intercalibration, certain boundaries have been adjusted slightly but in most cases the Swedish assessment of high, good and moderate status has corresponded well with assessments made by other Member States. Decisions on boundaries, both absolute values and Ecological Quality Ratios (EQR), will be taken in the course of 2008 for phytoplankton, macrophytes, diatoms and benthic fauna (DJ-index). The decision will be taken at the EU level.

In the WFD, it is stated that the results of the status classification are to be given in Ecological Quality Ratios (EQR) to guarantee comparability between the Member States. EQRs show the deviation from the reference value. In the course of the work on intercalibration, both nationally and internationally, it has become apparent that the extent of the acceptable deviation for the different status classes varies between different quality factors and parameters. In the cases where there are class boundaries based on values for the parameters themselves e.g. µg/l total biomass of phytoplankton or IPS value for diatoms, these class boundaries are also presented in this handbook. The purpose is to facilitate understanding of the class boundaries.

Many of the assessment criteria contain several parameters that show different kinds of impacts. The most common are the nutrient impact or the acidity impact. In cases where, on the basis of impact assessment or local knowledge, it is known what kind of impact the body of water is exposed to, it is appropriate to use primarily the parameters that correspond to the actual impact in order to make a status classification. According to regulations (NFS 2008:1) issued by the Swedish EPA, all parameters are to be calculated but since that can give misleading results, e.g. to look at an acidity indicator when there is no acidity in the area, it is possible with the aid of the reasonability assessment described in Section 4.1.1 of the main handbook to disregard those findings that do not appear reasonable in the light of the known impact picture.

Section numbering can be the same in different annexes to the handbook, but a reference to a given section in the annex always refers to the relevant section of this annex.

2 Input quality factors and parameters

Table 2.1 shows the quality factors and parameters for which assessment criteria have been developed and which are regulated in the Swedish EPA's Regulations (REG) and General Guidelines (GG) on Classification and Environmental Quality Standards for Surface Water (2008:1), Annexes 1 and 2.

Table 2.1 Tabulation of parameters or indices for all quality factors for lakes and watercourses for which assessment criteria have been developed. Parameters in italics cannot be found in the regulations but can be used as an aid to classification.

Lakes		**Quality factors**	**Parameter/index**
		Phytoplankton	Total biomass
			Cyanobacterial abundance
			TPI (trophic plankton index)
			Chlorophyll
		Macrophytes	Trophic macrophyte index (TMI)
		Benthic fauna	ASPT
			BQI
			MILA
		Fish	
Physico-chemical factors[1]	General conditions	Nutrients	
		Transparency	Transparency
		Oxygen	Oxygen
		Acidification	
		Specific pollutants	Pollutants released in significant amounts
Watercourses			
Biological factors		Diatoms	IPS
			ACID
			%PT (support parameters)
		Benthic fauna	ASPT
			DJ index
			MISA
		Fish	
			VIXsm (page index)

[1] Annex V of the WFD also contains priority pollutants released into water bodies but with a quality factor below ecological status. Under EU Guidance no. 13, the priority pollutants must only be dealt with under surface water chemical status when EU-wide limit values have been developed. In these regulations, general guidelines and handbook, priority pollutants are dealt with only under surface water chemical status

Physico-chemical factors[2]	General conditions	Nutrients	Tot-P
		Acidification	
			BDM/pBDM
	Specific pollutants		Pollutants released in significant amounts.

All background reports to assessment criteria are presented online at www.naturvardsverket.se. There may be differences between what is contained in the background reports and in the handbook. The handbook is the most up-to-date and represents the Swedish EPA's position on the material.

[2] See footnote 1.

3 Phytoplankton in lakes

Parameter	Shows primarily effects of	How often do measurements need to be taken?	At what time of year?
Total biomass	Nutrient impact	Once a year, but 3-year mean value	July - August
TPI (trophic plankton-index)	Nutrient impact	Once a year, but 3-year mean value	July - August
Proportion of cyano-bacteria	Nutrient impact	Once a year, but 3-year mean value	July - August
Number of species	Acidity	Once a year, but 3-year mean value	July - August
Chlorophyll	Nutrient impact	Once a year, but 3-year mean value	July - August

3.1 Introduction

Changes in the water's nutrient status are rapidly reflected in biomass and species composition of phytoplankton. Phytoplankton are therefore used as an indicator in order e.g. to monitor the recovery process after a nutrient reduction, to monitor an acidification process or as an early sign of increasing nutrient load. Phytoplankton respond rapidly to environmental changes and are a good "early warning signal" (Figure 3.1).

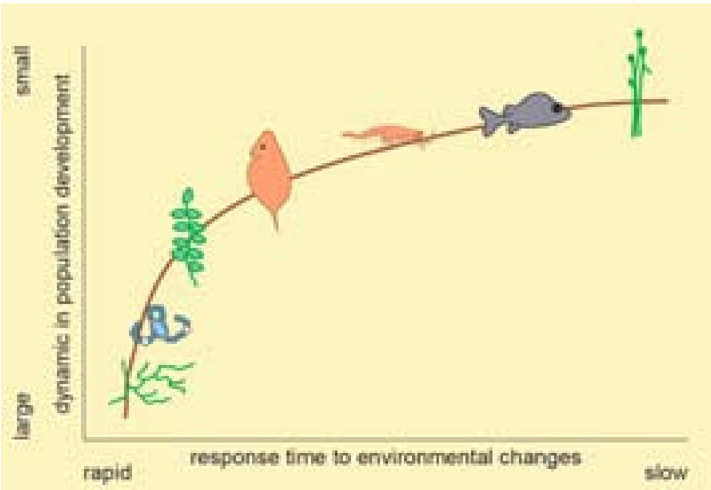

Figure 3.1. Relative reaction times to environmental changes for different organism groups in water.

Phytoplankton communities have a marked dynamic in their population development, in which weather and wind have overall importance. Despite this, the proportion of cyanobacteria is a good indicator of increasing nutrient levels (Figure 3.2). Certain individual species of other phytoplankton groups that can develop in nutrient-poor water are an exception. These species normally do not have gas vacuoles and hence do not rise to the surface. For example the clear link between the relative cyanobacterial biomass and increasing nutrients levels does not apply in lakes with the raphidophycean flagellate *Gonyostomum semen.* Lakes containing a lot of *Gonyostomum* are found mainly in southern Sweden and are of a humic nature. The *Gonyostomum* share of the total biomass in a lake must be at least 5% for it to be regarded as dominant.

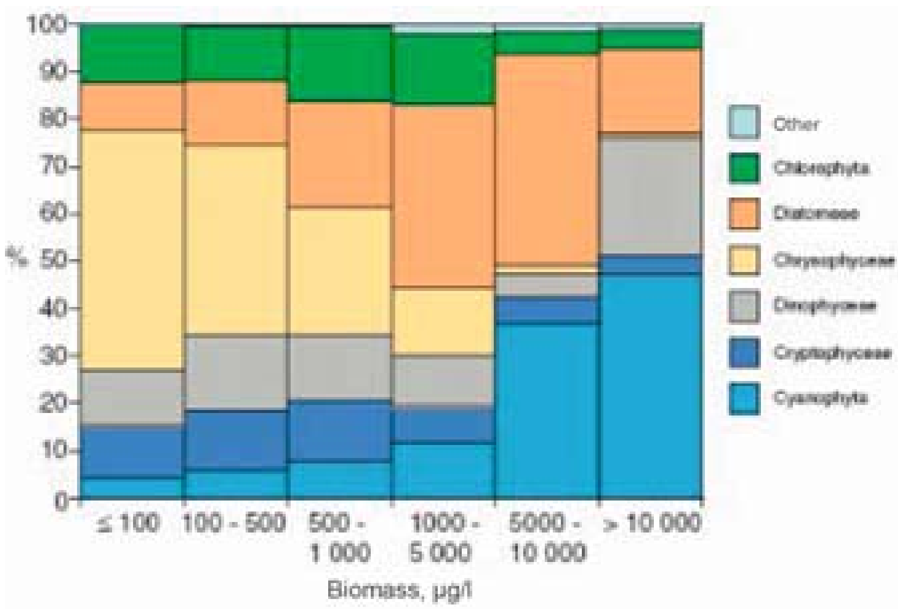

Figure 3.2. Percentage distribution of phytoplankton groups in July-August along a gradient of increasing biomass, which in turn follow increasing total phosphorus levels (number of lakes = 409). The proportion of cyanobacteria is increasing and the proportion of chrysophytes is decreasing.

Chlorophyll measurements are a comparatively quick and simple method to obtain an overview of the total phytoplankton biomass in a water body, but since the amount of chlorophyll a varies between different plankton groups, this method can be used only as an indication of the current situation. The method is applicable for screening, and to give indications of possible changes in the phytoplankton biomass in a water body. Where there are doubts, a complete phytoplankton analysis should always be carried out to verify results. Moreover, in certain situations a chlorophyll analysis does not give the whole truth about the current situation in a water body. For example, in mountain lakes where the water is clear, a relatively large proportion of the primary production is produced on the lake bottoms by benthic organisms like periphytic algae or higher vegetation. In such cases, reliance exclusively on chlorophyll a, or phytoplankton data, can lead to the false conclusion that the biomass of primary producers is less than is actually the case. Even in humic lakes, it is possible to be misled into the belief that the phytoplankton biomass is less than is the case if one relies solely on chlorophyll analyses. That is because in these systems phytoplankton biomass can in varying degrees consist of heterotrophic and/or mixotrophic plankton organisms, which can be poorly pigmented since these in varying degrees live on dead organic material.

As regards the reactions of phytoplankton to acidification, the results are less clear-cut at the species level, but it is evident that certain groups disappear almost completely in the most acidic environments. Such examples are cyanobacteria and diatoms, which both require somewhat more nutrients than is often to be found at, for example, pH <5.5. A drastic reduction in the number of species is an indication of an acidic water body (Figure 3.3).

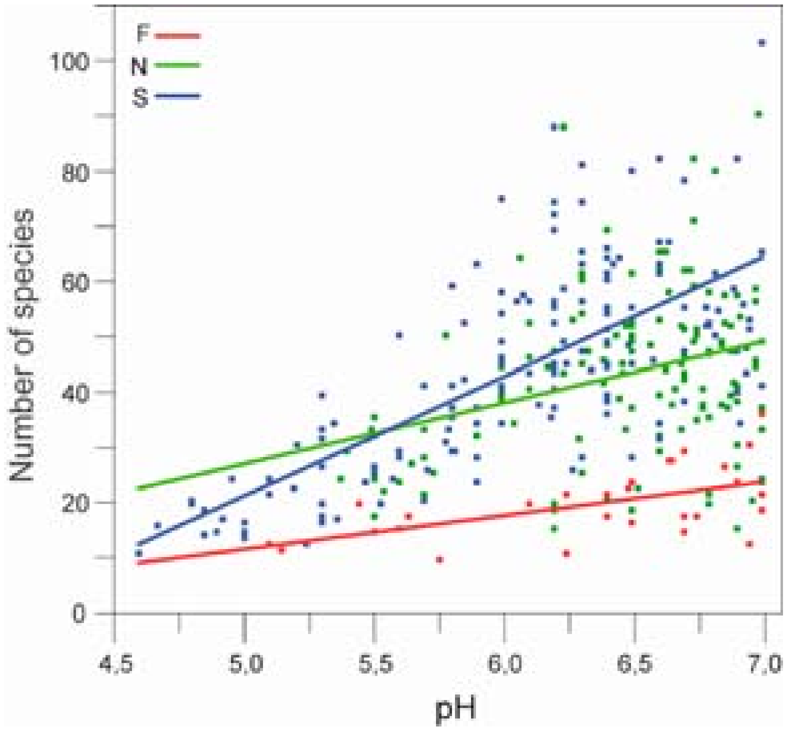

Figure 3.3. Number of species in an acidity gradient of pH 4.5-7 in three regions. The linear equations for the three regions illustrated in Figure 3 are:
Mountains (F): Number of species=-20.61+6.3•pH, n=28
Northern Sweden (N): Number of species=-28.98+11.1•pH, n=130
Southern Sweden (S): Number of species=-87.53+21.7•pH, n=151

The assessment criteria are intended for use in all types of lakes, but lakes with high metal pollution were not included in the supporting material and for that reason caution should be observed in the classification of such types of lakes.

3.2 Input parameters

For classification of phytoplankton as a quality factor in a trophic gradient, the following parameters must be used:

See REG Annex 1, Section 1.1

- **Total biomass of phytoplankton**. Total biomass can be expressed both as a volume unit or as a mass in which phytoplankton are assumed to have the same density as water i.e. 1 g cm^3. Total biomass can then be expressed as mg l^{-1} or µg l^{-1} and if the concept of 'total volume' is used, the corresponding units are mm^3 l^{-1}. The term 'total biomass' is used in these assessment criteria.
- **Proportion of cyanobacteria** (blue-green algae). I.e. the cyanobacterial biomass as a percentage of the total biomass.
- **Trophic plankton index** (TPI) based on indicator species on a scale from –3 to 3
- **Chlorophyll** (primarily as a screening method in the absence of phytoplankton analysis). The biomass of planktonic algae can be gauged in a general way by analysing the algae's chlorophyll *a* content. However, this analysis gives no detailed information about structures in the phytoplankton community.

For assessment of acidity/acidification:

- **Number of species**

3.3 Requirements for supporting data

If the assessment criteria for phytoplankton in lakes are to be applicable, the tests must be taken during the period July-August and the analysis must be conducted in accordance with standard SS-EN 15204:2006 or by another method that gives an equally good result. At least three years' data must be used for the classification. The sample should preferably represent the upper layer of the water above the thermocline (epilimnion). It is also possible to use the top metre(s) of this layer, particularly in humic lakes since parts of the supporting material has been derived from these levels. Since the plankton in humic water seek the surface, at least during daylight hours, the majority of organisms are to be found in the upper metres of these lakes. In clear lakes, on the other hand, the greatest biomass can be found a little way down in the water mass, because the plankton organisms can be damaged by excessive light radiation at the surface. To obtain the best possible comparison, it is therefore best if the sample represents approximately 75% of the epilimnion. The sample is analysed and the taxa counted in accordance with the Utermöhl method (Utermöhl 1958), preferably using the technical procedure described in the Swedish EPA's survey type 'Phytoplankton in lakes'. It is particularly important to use this method of analysis when classifying the number of species. In cases where only the most frequently occurring taxa have been counted, expert assessments may be made based on the index values, such as the total biomass and the proportion of cyanobacteria, even though that does not give the same precision as using a more detailed analytical method. As regards the use of the trophic plankton index for samples counting a limited number of species, a number of such tests from a survey of 1000 or so lakes in 1972 corresponded well with results from the material which constituted the basis for the construction of the TPI index. It is, however, important not to limit the count to only 4-5 taxa if there is no mass development, but to count at least 20 or more taxa, with exception made for lakes in the mountain region which are much more species-poor.

If fewer than four species with an indicator number (from Table 3.6 or 3.7) have been found in a lake, the TPI cannot be calculated and the classification of nutrient conditions must be based solely on total biomass and the proportion of cyanobacteria. Where there is a lack of supporting data even to make a classification of total biomass and proportion of cyanobacteria, a classification based solely on chlorophyll may be made. As regards chlorophyll, the Swedish standard methods that apply for tests and analysis are SS 02 81 46 and 02 81 70 or equivalent methods.

3.4 Typology

For the classification of phytoplankton, lakes in Sweden are divided into five types with different reference values (Table 3.2 and Figure 3.4). For the trophic plankton index, no distinction is made between clear and humic lakes in Norrland (northern Sweden). The types are based on the ecoregions given in the Swedish EPA's Regulations on Typology and Analysis (NFS 2006:1), and the humus content of the lakes (water colour). Under the regulations, the lakes are divided into low humus content (h) and high humus content (H) with a boundary of 50 mg Pt/l. For the classification of phytoplankton, however, the boundary has instead been set at 30 mg Pt/l, which corresponds with that used for intercalibration of classifications among the Nordic countries. In the regulations, there is also a more precise division into limnic types but the other factors for allocation have not been shown to affect the classification of phytoplankton with the supporting data currently available.

All the lakes which match one of the lake types established are given the same reference value for classification of phytoplankton.

Table 3.2. Typology of lakes for classification of phytoplankton. Ecoregions and humus class in accordance with the Swedish EPA's Regulations on Typology and Analysis (NFS:2006:1) are also shown.

Lake classifications for phytoplankton	Ecoregion in accordance with NFS 2006:1	Humus class in accordance with NFS 2006:1
Mountains above the tree-line	1	h, H
Norrland clear lakes[1]	2, 3	h[3]
Norrland humic lakes[2]	2, 3	H[3]
Southern Sweden clear lakes[1]	4, 5, 6	h
Southern Sweden humic lakes	4, 5, 6	H

[1]Water colour ≤30mg Pt/l or Abs420/5 ≤0,06 (filtered sample)
[2]Water colour >30mg Pt/l or Abs420/5 >0,06 (filtered sample)
[3]When classifying in accordance with TPI, no distinction is made between clear and humic lakes in Norrland

One type of humic lakes that have high and deviant biomasses (total biomass or chlorophyll) is those dominated by Gonyostomum semen. This is revealed only by analysis of the species composition of the phytoplankton community. Here, TPI in combination with proportion of cyanobacteria are the suitable indicators to use unless the lake is acidic, in which case it is instead the number of species that gives the status.

Figure 3.4 Typology of lakes in Sweden for phytoplankton is based on three ecoregions.

3.5 Total biomass

3.5.1 Classification of status

See REG Annex 1, section 1.3

For samples taken and analysed in accordance with the description in Section 3.3, the total biomass is determined. A mean value of at least three years' data must be used for the classification. The ecological quality ratio (EQR) for biomass is calculated as follows:

EQR = reference value/observed total biomass (mean value)

Reference values and class boundaries are given in Table 3.3.

3.5.2 Reference values and class boundaries

Table 3.3. Reference values and class boundaries for classification of the total biomass parameter(BM) in µgl-1 and as ecological quality ratio (EQR). Total biomass = total biovolume. If the total biomass ≤ the reference value, the EQR is set to 1.

Type	**Status**	**Total biomass ($\mu g\ l^{-1}$)**	**Ecological quality ratio (EQR)**
Mountains above tree-line	Reference value	120	
	High	BM≤200	EQR≥0.6
	Good	200<BM≤350	0.6>EQR≥0.34
	Moderate	350<BM≤500	0.34>EQR≥0.24
	Poor	500<BM≤650	0.24>EQR≥0.18
	Bad	BM>650	0.18>EQR≥0
Norrland, clear lakes, colour ≤30 mg Pt l^{-1} Southern boundary *Limes Norrlandicus*	Reference value	200	
	High	BM≤300	EQR≥0.67
	Good	300<BM≤650	0.67>EQR≥0.31
	Moderate	650<BM≤1000	0.31>EQR≥0.2
	Poor	1000<BM≤1350	0.2>EQR≥0.15
	Bad	BM>1350	0.15>EQR≥0
Norrland, humic lakes, colour >30 mg Pt l^{-1} Southern boundary *Limes Norrlandicus*	Reference value	300	
	High	BM≤400	EQR≥0.75
	Good	400<BM≤1000	0.75>EQR≥0.3
	Moderate	1000<BM≤1500	0.3>EQR≥0.2
	Poor	1500<BM≤2000	0.2>EQR≥0.15
	Bad	BM>2000	0.15>EQR≥0
Southern Sweden, clear lakes, colour ≤30 mg Pt l^{-1} Northern boundary *Limes Norrlandicus*	Reference value	400	
	High	BM≤600	EQR≥0.67
	Good	600<BM≤2500	0.67>EQR≥0.16
	Moderate	2500<BM≤5000	0.16>EQR≥0.08
	Poor	5000<BM≤10000	0.08>EQR≥0.04
	Bad	BM>10000	0.04>EQR≥0
Southern Sweden, humic lakes, colour >30 mg Pt l^{-1} Northern boundary *Limes Norrlandicus*	Reference value	400	
	High	BM≤600	EQR≥0.67
	Good	600<BM≤2500	0.67>EQR≥0.16
	Moderate	2500<BM≤5000	0.16>EQR≥0.08
	Poor	5000<BM≤10000	0.08>EQR≥0.04
	Bad	BM>10000	0.04>EQR≥0

3.6 Proportion of cyanobacteria

3.6.1 Classification of status

Proportion of cyanobacteria (blue-green algae) shall also be used as an indicator of increasing nutrient levels. The biomass of cyanobacteria is determined and divided by the total phytoplankton biomass in order to ascertain the cyanobacterial proportion. A mean value of at least three years' data must be used for the classification. The ecological quality ratio (EQR) for cyanobacterial abundance is calculated as follows:

See REG Annex 1, section 1.4

EQR = (100 – observed % cyanobacteria) / (100 – reference value)

Reference values and class boundaries are given in Table 3.4.

3.6.2 Reference values and class boundaries

Table 3.4. Reference values and class boundaries for classification of proportion of cyanobacteria (C) in % and as ecological quality ratio (EQR). If the proportion of cyanobacteria ≤ reference value, the EQR is set to 1.

Type	Status	Cyanobacterial abundance (C) in %	Ecological quality ratio (EQR)
Mountains above tree-line	Reference value	0	
	High	C≤1	EQR≥0.99
	Good	1<C≤5	0.99>EQR≥0.95
	Moderate	5<C≤10	0.95>EQR≥0.90
	Poor	10<C≤20	0.90>EQR≥0.80
	Bad	20<C≤100	0.80>EQR≥0
Norrland, clear lakes, colour ≤30 mg Pt l^{-1}. Southern boundary *Limes Norrlandicus*	Reference value	5	
	High	C≤10	EQR≥0.95
	Good	10<C≤24	0.95>EQR≥0.80
	Moderate	24<C≤43	0.80>EQR≥0.60
	Poor	43<C≤81	0.60>EQR≥0.20
	Bad	81<C≤100	0.20>EQR≥0
Norrland, humic lakes, colour >30 mg Pt l^{-1}. Southern boundary *Limes Norrlandicus*	Reference value	7	
	High	C≤14	EQR≥0.92
	Good	14<C≤30	0.92>EQR≥0.75
	Moderate	30<C≤46	0.75>EQR≥0.60
	Poor	46<C≤81	0.60>EQR≥0.20
	Bad	81<C≤100	0.20>EQR≥0
Southern Sweden, clear lakes, colour ≤30 mg Pt l^{-1}. Northern boundary *Limes Norrlandicus*	Reference value		
	High	C≤10	EQR≥0.95
	Good	10<C≤24	0.95>EQR≥0.80
	Moderate	24<C≤43	0.80>EQR≥0.60
	Poor	43<C≤81	0.60>EQR≥0.20
	Bad	81<C≤100	0.20>EQR≥0
Southern Sweden, humic lakes, colour >30 mg Pt l^{-1}. Northern boundary *Limes Norrlandicus*	Reference value	7	
	High	C≤14	EQR≥0.92
	Good	14<C≤30	0.92>EQR≥0.75
	Moderate	30<C≤46	0.75>EQR≥0.60
	Poor	46<C≤81	0.60>EQR≥0.20
	Bad	81<C≤100	0.20>EQR≥0

3.6.3 Comments

If one or more of the cyanobacterial taxa shown in Table 3.5 dominate, it may be a reason for particular attention as they can often give rise to nuisance or even be potentially toxic.

Table 3.5. Cyanobacterial taxa that are often associated with bad water quality as they often mass-develop or can form toxins. When developing en masse, all species can give off a bad odour or make the water taste like raw sewage.

Taxon	Comment
Anabaena	Produces nerve and liver poisons, as well as substances giving rise to bad odour and taste. Toxicity has been verified in samples from Sweden.
Aphanizomenon	Potentially toxic, not verified in Sweden with the species in cultivation, but present in cyanobacteria communities where toxicity has been registered.
Gloeotrichia	The species *echinulata*. Toxin production not verified in Sweden
Limnothrix	Potentially toxic, not verified in Sweden with the species in cultivation, but present in cyanobacteria communities where toxicity has been registered.
Microcystis	Producer of nerve and liver poisons, verified in Sweden. The species *wesenbergii* does not have the genes for toxin production.
Planktothrix	Primarily the species *agardhii* and *prolifica* both producers of liver poisons, verified in Sweden.
Pseudanabaena	Potentially toxic, not verified in Sweden with the species in cultivation.
Woronichinia	Primarily the species *naegeliana*. Gives rise to smell and taste in mass-development.

3.7 Trophic plankton index

3.7.1 Classification of status

The trophic plankton index (TPI) is calculated as follows:

See REG Annex 1, section 1.5

$$TPI_{lake} = \frac{\sum_{i=1}^{n} (I_{species\ i} \times B_{species\ i})}{\sum_{i=1}^{n} B_{species\ i}}$$

n = the number of species with indicator number in a lake
I = the indicator number for species i
B = biomass per litre for species i (the unit in which this is expressed may be µg/l, mg/l or mm^3/l the main point being that it is the same unit for included species and the total biomass of these species)

Tables 3.6 and 3.7 give the indicator numbers of the various species.

The EQR for TPIs that contain both negative and positive values is calculated using the following formula:

$$EQR= \frac{r_{75}-r_{50}}{x+r_{75}-(2\times r_{50})}$$

Where
r_{75}= the TPI value for high class status
r_{50}=the TPI value for the reference conditions
x=the TPI value of the object

In this procedure, the norm for the EQR for high status is 0.5. In this way, some consideration of the variation in the reference data material is taken.

Reference values and class boundaries are given in Table 3.8.

Table 3.6 Tolerant species with an indicator number on a scale 1-3, where 3 indicates "eutrophy indicators", or species that are specially tolerant and prevalent in the most nutrient-rich environments. They have been checked against Brettum and Andersen 2005[3] and some items have been grouped together in accordance with Binnengewässer 7:1[4]

Taxon	Indicator number	Remark
Actinastrum spp.	2	
Actinocyclus normanii f. subsalsa	3	
Anabaena lemmermannii	1	
Anabaena coiled	2	*circinalis, flos-aquae, mendotae*
Anabaena straight	2	*planctonica, solitaria, macrospora*
Anabaena spirals	3	*spiroides, crassa*
Aphanizomenon bundles	3	*flos-aquae, yezoense, klebahnii*
Aphanizomenon solitary	3	*issatschenkoi, gracile, flexuosum*
Aulacoseira ambigua	1	
Aulacoseira granulata	2	
Aulacoseira granulata v. angustissima	3	
Aulacoseira subarctica	1	
Ceratium furcoides	2	
Chodatella spp.	2	
Closterium acutum v. variabile	1	
Closterium limneticum	1	
Coelastrum spp.	3	
Cryptomonas large	2	length >40 µm.
Cyanodictyon spp.	3	
Dictyosphaerium pulchellum	1	this also includes *tetrachotomum*
Dimorphococcus lunatus	1	
Diplopsalis acuta	3	
Euglena spp.	3	all euglenophytes are classified as 3
Fragilaria berolinensis	3	

[3] Brettum, P. & Andersen, T. 2005. The use of phytoplankton as indicators of water quality. NIVA-report SNO 4818-2004. Norwegian Institute for Water Research, Oslo.

[4] Binnengewässer von Huber-Pestalozzi 1983. Chlorphyceae. Ordnung: Chlorococcales. Teil /:1. Schweizerbartsche Verlagsbuchhandlung. Stuttgart.

Taxon	**Indicator number**	**Remark**
Fragilaria crotonensis	2	
Fragilaria ulna	2	Brettum & Andersen 2005
Lagerheimia spp.	2	
Lepocinclis spp.	3	
Limnothrix planctonica	3	
Limnothrix redekei	3	
Micractinium pusillum	2	
Microcystis aeruginosa	3	*botrys* also included here
Microcystis flos-aquae	3	
Microcystis wesenbergii	3	
Microcystis viridis	3	
Monoraphididum minutum	2	
Pediastrum boryanum	3	Brettum & Andersen 2005
Pediastrum duplex	3	
Pediastrum privum	2	
Pediastrum tetras	2	
Phacus spp.	3	
Planktolyngbya spp.	3	*limnetica, contorta, bipunctata*
Planktothrix agardhii	2	
Planktothrix mougeotii	1	
Pseudanabaena limnetica	2	
Quadricoccus ellipticus	3	
Scendesmus gr. acutodesmus	3	includes *S. acutus, S. acuminatus,* and varieties of the *Scenedesmus gr.,* Cf. Binnengewässer 7:1
Scenedesmus gr. spinosi	2	Includes *S. spinosus* and varieties of it. Cf. Binnengewässer 7:1
Staurastrum chaetoceras	2	
Staurastrum smithii	2	
Staurastrum tetracerum	1	Brettum & Andersen 2005
Stephanodiscus spp.	2	
Tetraedriella spinigera	1	
Tetraedron incus	1	
Tetrastrum staurogeniaeforme	2	
Trachelomonas spp.	3	
Treubaria triappendiculata	3	

Table 3.7. Sensitive taxa "oligotrophic indicators" with indicator numbers on a scale from -1 to -3, where -3 indicates taxa that have been assessed as particularly competitive in low nutrient concentrations.

Taxon	Indicator number	Remark
Aulacoseira alpigena	-2	
Bitrichia chodatii	-2	
Bitrichia phaseolus	-3	also includes *ollula* and *longispina*
Chlamydocapsa spp.	-2	also includes *Gloeocystis* and *Coenocystis*
Chrysidiastrum catenatum	-2	
Chrysochromulina spp.	-2	
Chrysococcus spp.	-2	
Chrysolykos planctonicus	-2	
Chrysolykos skujae	-3	
Cyclotella spp. small	-2	diameter <10 µm
Dinobryon borgei	-2	
Dinobryon crenulataum	-2	
Dinobryon cylindricum	-3	especially *v. alpinum*. Varieties had not always been distinguished in the supporting material
Dinobryon njakajaurense	-3	
Dinobryon pediforme	-3	
Dinobryon sociale v. americanum	-3	
Gymnodinium spp. small	-3	length <10 µm
Gymnodinium uberrimum	-1	
Isthmochloron trispinatum	-3	
Kephyrion spp.	-3	all species have been given the same indicator number after tests of 7 separate species
Mallomonas akrokomos.	-2	
Mallomonas hamata	-3	
Mallomonas tonsurata	-1	
Merismopedia tenuissima	-2	
Monoraphidium griffithii	-2	
Oocystis submarina v.variabilis	-2	
Peridinium inconspicuum	-1	
Pseudokephyrion spp.	-3	all species have been given the same indicator number after tests of 7 separate species
Rhodomonas lacustris	-1	also includes *Rhodomonas minuta* and *Plagioselmis nannoplanctica*
Spiniferomonas spp.	-2	no species separation in the supporting material
Staurastrum lunatum	-2	also includes *v. planctonicum*
Staurodesmus sellatus	-2	
Stichogloea doederleinii	-2	also includes *olivacea*
Tabellaria flocculosa v. teilingii	-3	

3.7.2 Reference values and class boundaries

Table 3.8. Reference values and class boundaries for classification of the trophic plankton parameter (TPI) in index values and as ecological quality ratios (EQR). If TPI ≤ the reference value, the EQR is set to 1.

Type	Status	Trophic plankton index (TPI)	Ecological quality ratio (EQR)
Mountains above tree-line	Reference value	-2	1
	High	TPI≤-1.8	EQR≥0.5
	Good	-1.8<TPI≤-1.5	0.5>EQR≥0.29
	Moderate	-1.5<TPI≤-1.25	0.29>EQR≥0.21
	Poor	TPI>-1.25	0.21>EQR≥0
	Bad	-	-
Norrland, clear and humic lakes	Reference value	-1.5	1
	High	TPI≤-1	EQR≥0.5
	Good	-1<TPI≤-0.5	0.5>EQR≥0.33
	Moderate	-0.5<TPI≤0.5	0.33>EQR≥0.2
	Poor	TPI>0.5	0.2>EQR≥0
	Bad	-	-
Southern Sweden, clear lakes, colour ≤30 mg Pt^{-1}. Northern boundary *Limes Norrlandicus*	Reference value	-1.25	1
	High	TPI≤-0.9	EQR≥0.5
	Good	-0.9<TPI≤1	0.5>EQR≥0.13
	Moderate	1<TPI≤2	0.13>EQR≥0.1
	Poor	TPI>2	0.1>EQR≥0
	Bad	-	-
Southern Sweden, humic lakes, colour>30 mg Pt^{-1} Northern boundary *Limes Norrlandicus*	Reference value	-1	1
	High	TPI≤-0.5	EQR≥0.5
	Good	-0.5<TPI≤1	0.5>EQR≥0.2
	Moderate	1<TPI≤2	0.2>EQR≥0.14
	Poor	TPI>2	0.14>EQR≥0
	Bad	-	-

3.8 Number of species

3.8.1 Classification of status

See REG Annex 1, section 1.7

In order to assess the acidity of the water, the number of species is determined, i.e. the number of phytoplankton species in the sample. The number of species must above all be assessed if it is suspected that a lake is exposed to acidification, since the indicator is difficult to interpret and highly dependent on taxonomical effort. If a method other than that prescribed in the regulations has been used, an expert assessment can be made. However, it is important to be aware that if only dominant species have been counted, the number of species then obtained is considerably lower than if the correct method has been used, which affects the classification. A standardised counting of a sample in accordance with this method indicates on average 40-58 taxa with the exception of mountain lakes, where around 20 taxa occur.

The number of species thus shows how acidic the lake is, but it does not indicate whether the acidity is natural or results from anthropogenic acidification.

The ecological quality ratio is calculated as follows:

EQR = observed species number / reference value

Reference values and class divisions are given in Table 3.9.

3.8.2 Reference values and class boundaries

Table 3.9. Reference values and class boundaries for classification of the number of species (N) parameter, also stated as ecological quality ratios (EQR). The classes show different stages of acidity and do not relate to status. The acidity classes roughly correspond to the following pH intervals: nearly neutral 6-7, acidic 5.5-6, very acidic 5-5.5, extremely acidic <5. If the number of species ≥ the reference value, the EQR is set to 1.

Type	Acidity class	Number of species	Ecological quality ratio (EQR)
Mountains above tree-line	Reference value	25	
	Almost neutral	N≥20	EQR≥0.8
	Acidic	20>N≥15	0.8>EQR≥0.6
	Highly acidic	15>N≥10	0.6>EQR≥0.4
	Extremely acidic	N<10	EQR<0.4
Norrland, clear lakes, colour ≤30 mg Pt^{-1}. Southern boundary *Limes Norrlandicus*	Reference value	45	
	Almost neutral	N≥30	EQR≥0.67
	Acidic	30>N≥25	0.67>EQR≥0.56
	Highly acidic	25>N≥20	0.56>EQR≥0.44
	Extremely acidic	N<20	EQR<0.44
Norrland, humic lakes, colour >30 mg Pt^{-1}. Southern boundary *Limes Norrlandicus*	Reference value	45	
	Almost neutral	N≥40	EQR≥0.89
	Acidic	40>N≥30	0.89>EQR≥0.67
	Highly acidic	30>N≥20	0.67>EQR≥0.44
	Extremely acidic	N<20	EQR<0.44
Southern Sweden, clear lakes, colour ≤30 mg Pt^{-1}. Northern boundary *Limes Norrlandicus*	Reference value	50	
	Almost neutral	N≥45	EQR≥0.9
	Acidic	45>N≥35	0.9>EQR≥0.7
	Highly acidic	35>N≥20	0.7>EQR≥0.4
	Extremely acidic	N<20	EQR<0.4
Southern Sweden, humic lakes, colour>30 mg Pt^{-1}. Northern boundary *Limes Norrlandicus*	Reference value	45	
	Almost neutral	N≥40	EQR≥0.88
	Acidic	40>N≥30	0.88>EQR≥0.67
	Highly acidic	30>N≥15	0.67>EQR≥0.33
	Extremely acidic	N<15	EQR<0.33

3.9 Chlorophyll

3.9.1 Classification of status

See REG Annex 1, section 1.8

In cases where there is no available data to enable a classification to be made with the parameters stated in Sections 3.5 - 3.7, the water authority will have to make a classification by using chlorophyll alone. The biomass of planktonic algae can be gauged in a general way by analysing the algae's chlorophyll a content. However, this analysis gives no detailed information about the phytoplankton community structure.

The chlorophyll content is determined according to the standard method and the EQR is calculated as follows:

EQR = reference value / observed chlorophyll content

Reference values and class boundaries are given in table 3.10

3.9.2 Reference values and class boundaries

Table 3.10. Reference values and class boundaries for classification of status with regard to chlorophyll (Chla) in µg l-1 and as EQRs. If the chlorophyll content ≤ the reference value, the EQR is set to 1.

Type	Status	Chlorophyll content (Chla) (µg l^{-1})	Ecological quality ratio (EQR)
Mountains above the tree-line	Reference value	1.0	
	High	Chla≤1.5	≥0.75
	Good	1.5<Chla≤3.0	0.75>EQR≥0.33
	Moderate, poor, bad	Carry out a complete phytoplankton analysis to verify the status class	
Norrland, clear lakes, colour ≤30 mg Pt l^{-1}. Southern boundary *Limes Norrlandicus*	Reference value	2.0	
	High	Chla≤4.0	≥0.50
	Good	4.0<Chla≤6.0	0.50>EQR≥0.33
	Moderate, poor, bad	Carry out a complete phytoplankton analysis to verify the status class	
Norrland, humic lakes, colour >30 mg Pt l^{-1}. Southern boundary *Limes Norrlandicus*	Reference value	2.5	
	High	Chla≤5.0	≥0.50
	Good	5.0<Chla≤7.5	0.50>EQR≥0.33
	Moderate, poor, bad	Carry out a complete phytoplankton analysis to verify the status class	
Southern Sweden, clear lakes, colour ≤30 mg Pt l^{-1} Northern boundary *Limes Norrlandicus*	Reference value	2.5	
	High	Chla≤5.0	≥0.50
	Good	5.0<Chla≤8.5[1]	0.50>EQR≥0.30[1]
	Moderate, poor, bad	Carry out a complete phytoplankton analysis to verify the status class	
Southern Sweden, humic lakes, colour 30 mg Pt l^{-1}. Northern boundary *Limes Norrlandicus*	Reference value	3.0	
	High	Chla≤6.0	≥0.50
	Good	6.0<Chla≤10	0.50>EQR≥0.30
	Moderate, poor, bad	Carry out a complete phytoplankton analysis to verify the status class	

If a lake is assigned the status ‘moderate’ or worse, either a supplementary phytoplankton analysis must be carried out, especially if no other quality factors show a similar classification status, or an expert assessment has to be made. This applies particularly in humic lakes (AbsF420/5 >0.06 or water colour >30 mg Pt l^{-1}) in which the phytoplankton biomass can in certain cases be dominated by the flagellate *Gonyostomum semen*.

3.9.3 Comments

When evaluating chlorophyll data, it is important to keep in mind that the chlorophyll content gives only an estimate of the phytoplankton biomass and it cannot completely substitute phytoplankton analyses. These analysis methods are not completely comparable both because of uncertainties in the chlorophyll measurements and because different phytoplankton species contain varying quantities of chlorophyll a, and in many cases are also supplemented by other chlorophylls or other pigments. Since chlorophyll analyses are comparatively quick and cheap, they can be a good complement in, for example, screening studies or long-term monitoring. Any changes or divergent contents should nevertheless always be followed up by a supplementary and verifying phytoplankton analysis to investigate the cause of the change or divergence.

In comparisons between classifications as regards chlorophyll a and total phytoplankton biomass, it is obvious that the variation is large. As mentioned above, that is because of uncertainties in the chlorophyll analyses and because phytoplankton species contain different amounts of chlorophyll. Another important reason why there is a certain difference is that the analyses have often not been carried out on the same water sample. Chlorophyll analyses are often conducted on surface water samples (0.5 m), while phytoplankton analyses are commonly done on integrated samples that are intended to correspond to the water mass above the thermocline. Since phytoplankton are in general not homogeneously distributed in the water column, major differences can arise if integrated samples are compared to surface water samples. The difference is perhaps most obvious in calm weather during the summer when cyanobacteria often tend to accumulate in the surface water and there is then a risk that they are over-represented in a surface sample. Even so, any accumulation of e.g. Gonyostomum at the thermocline can give significantly higher biomasses compared with samples taken near the surface. This difference between surface water and integrated samples is nevertheless unavoidable and indeed reflects well the reality that status classification as regards chlorophyll content will primarily be conducted on surface samples.

A lake must nevertheless not be given the status ‘moderate’ or worse, however, solely on the classification of chlorophyll and instead supplementary analyses of, for example, phytoplankton must be made to ascertain the cause and guarantee the lake’s status before taking any necessary measures to maintain or achieve ‘good’ status.

3.10 Management of uncertainty

To make a good classification, the mean value from at least three years must be used. Several measurements give a more reliable classification and an uncertainty interval in the form of a standard deviation can be calculated for the parameter in the water body concerned. In cases where only a small volume of data is available, the fixed value for the method-bound uncertainty (standard deviation) for the respective parameters and types stated in Table 3.11 can be used. The standard deviation gives a measure of how unreliable a classification is. Variations in EQRs in reference lakes have been used as a general method to illustrate the spread of values. The spread is then presented as the median of standard deviations for the selected parameters. The starting-point for the calculations has been the yearly variations in EQR from those lakes where material of that kind has been available (Mountain region: 2 lakes, 12 August months; Norrland's clear water: 5 lakes 26 seasons; Norrland's humic lakes: 18 lakes 98 seasons; Southern Sweden clear water: 9 lakes 53 seasons; Southern Sweden humic lakes: 5 lakes 27 seasons). However, this method does not reflect the spread in other status classes. We might expect greater variation both within a lake type and between different years with increasing trophic level.

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See GG to Chapter 2 9 §

In cases where an uncertainty interval around the EQR overlaps any of the class boundaries between high and good status or between good and moderate status, the calculated EQR-value lies very close to a class boundary. This indicates that a reasonability assessment must be made, as described in Section 4.1.1 of the main handbook. See also Section 4.1.2 of the main handbook for more guidance on how to manage uncertainty.

Table 3.11. Mean values of the standard deviation in EQRs for reference lakes in the data material.

Indicator	**Mountain region**	**Norrland clear**	**Norrland humic**	**Southern Sw. clear**	**Southern Sw. humic**
Total biomass	0.05	0.09	0.13	0.19	0.12
Proportion of cyanobacteria	0	0.02	0.02	0.04	0
TPI	0.17	0.18	0.18	0.23	0.002
Number of species	0.14	0.05	0.03	0.07	0.07

3.11 Weighting of trophic status parameters

When weighted together, the parameters total biomass, trophic plankton index (TPI) and proportion of cyanobacteria, form the basis for the classification of the lake's status as regards nutrients.

See REG Annex 1, section 1.6

Since the TPI can only be used if at least four species in a sample have been assigned an indicator number, there will be lakes where the classification is based solely on total volumes and cyanobacterial proportion. For lakes characterised by Gonyostomum semen, the total biomass parameter may be unsuitable, particularly if the biomass is very large, which is not uncommon since this species often develops en masse. Such mass development is not necessarily a sign of eutrophication. It is therefore recommended that Gonyostomum lakes should be quality-classed using the TPI value and cyanobacterial proportion instead of by total biomass.

Parameters are weighted together as follows:

Step 1) The weighting must be based on the classified status for total biomass, cyanobacterial proportion and TPI. The status classes are given a numerical value in accordance with Table 3.12. A weighted class value for each parameter is calculated before the weighting is conducted in accordance with Step 2.

Table 3.12. Division of the status classes in numerical values.

Status	Numerical value
High status	4 - 4.99
Good status	3 - 3.99
Moderate status	2 - 2.99
Poor status	1 - 1.99
Bad status	0 - 0.99

The numerical class (N_{class}) for the respective parameters for the relevant EQR class interval (EQR_{lower}–EQR_{upper}) is calculated as follows:

$$(N_{class}) = (N_{lower}) + (EQR_{calculated} - EQR_{lower})/(EQR_{upper} - EQR_{lower})$$

Where
(N_{class}) = weighted status value for each parameter
N_{lower}= the first digit (integer) in the numerical values for the status class in accordance with Table 3.12
$EQR_{calculated}$= calculated EQR-value from the classification
EQR_{lower} and EQR_{upper} = EQR for lower and upper class boundary for the corresponding class, taken from Tables 3.3, 3.4 and 3.8 respectively.
EQR_{lower} for bad status = 0 and EQR_{upper} for high status = 1

Step 2) The mean value for the numeric classes (Nclass) of the three parameters is calculated, which becomes the weighted classification of phytoplankton. The status classification is determined by the mean value for the numerical classification in accordance with Table 3.12.

Example:
A southern Swedish clear water lake has a three-season total biomass mean value of 500µg l-1 (=0.5 mm3 l-1=0.5mg l-1).
The corresponding TPI value is -1 and the proportion of cyanobacteria is 15%.

Total biomass
The total biomass gives high status in accordance with Table 3.3, which is the numerical class 4–4.99. N_{lower} is thus 4.
$EQR_{calculated}$=400/500=0.8 (calculated for this parameter as the relation between the measured value and the reference value).
EQR_{lower} is shown in Table 3.3=0.67, EQR_{upper} must be <1, but is entered in the equation as 1.
4 + [(0.8-0.67)/(1-0.67)]=4.39 which is N_{class} for the total biomass.

Proportion of cyanobacteria
The value from the proportion of cyanobacteria gives 'good' status according to Table 3.4.
$EQR_{calculated}$ is calculated in accordance with the information in the table heading (100-15)/(100-5)=0.89
According to Table 3.4 EQR_{lower} is 0.80
EQR_{upper} is 0.95
3 + [(0.89-0.80)/0.95-0.80)]=3.6 which gives 'good' status with regard to cyanobacterial proportion.

TPI
The TPI-value gives high status according to Table 3.8
$EQR_{calculated}$ which is calculated using the EQR equation in Section 3.7.1 is 0.58. (The variation in TPI status class as it is expressed in Table 3.8, i.e. r 75 = -0.9, r50= -1.25, x= -1)
According to the table, EQR_{lower} is 0.5
EQR_{upper} should be <1 but is set to 1
4 + [(0.58-0.5)/1-0.5)]=4.16 which is N_{class} for evaluation in accordance with the TPI, which gives 'high' status.

The mean value of these calculations is (4.39+3.6+4.16)/3, i.e. 4.05. This would classify the lake as 'high' status.

A classification based on chlorophyll must only be used in cases when it is impossible to make a classification of the total biomass or TPI, e.g. because the necessary phytoplankton data is not available.

The total classification of phytoplankton in lakes is determined by the weighted status for the parameters which shows the nutrient conditions or the result of the classification of acidification in accordance with Section 3.12, depending on which is worse.

3.12 Human impact or natural acidity

If the lake is classified as one of the acidity classes acidic, highly acidic or extremely acidic, an assessment must be made as to whether this is a result of anthropogenic

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acidification or whether the lake is naturally acidic. A more thorough analysis should be made with the aid of the assessment criteria given in Chapter 14. The analysis can further be improved by making an assessment of the acidification impacts/pressures. Important supporting material for this is provided by deposition data, critical load calculations and the impact of forestry.

See GG to Chapter 2 10 §

If the assessment is that the lake to some degree is naturally acidic, a reference value for pH for the water body is calculated in accordance with Chapter 14. The calculated pH value for the lake is correlated with the aid of the line equations for mountains, Norrland or southern Sweden, as set out below, to a new reference value for the number of species.

See REG Annex 1, section 1.7.3

Mountains: number of species$_{ref}$ = -20.61+6.3 • pH$_{ref}$
Norrland: number of species$_{ref}$ = -28.98+11.1 • pH$_{ref}$
Southern Sweden: number of species$_{ref}$ = -87.53+21.7 • pH$_{ref}$

The observed number of species is divided by the new reference value and compared to the class boundaries in Table 3.9.

The acidity classes, according to the revised reference value or the original classification, are converted to status classes as follows:

Almost neutral – High status
Acidic – Good status
Highly acidic – Moderate status
Extremely acidic – Poor or bad status

When the status classification results in a moderate or worse status, and this is indicated by the parameters that show nutrient richness/eutrophication, it may be necessary to make an assessment as to whether this is a result of anthropogenic eutrophication or whether the lake is naturally rich in nutrients. However, it is not particularly common for lakes to have naturally high nutrient levels. In order to evaluate this, a comparison can be made with results for the assessment criterion for phosphorus. The assessment can further be improved by looking at the impacts/pressures on the water body. Important supporting material for this is source distribution data, historical data, etc. Supporting data for this is produced in connection with the characterisation. If the assessment is that the lake is naturally rich in nutrients, the water authority revises the reference value for the specific water body by means of an expert assessment.

See REG Chapter 2 11 §

See GG to Chapter 2 11 §

Background reports:

Phytoplankton: Willén, E., 2007. Växtplankton i sjöar Bedömningsgrunder [Phytoplankton in lakes - Assessment criteria]. Report 2007:6. Department of Environmental Assessment. Swedish University of Agricultural Sciences (SLU)

Chlorophyll: Sonesten, L., 2007. Reviderade bedömningsgrunder för klorofyll [Revised assessment criteria for chlorophyll]. Revidering och anpassning till den “nordiska” interkalibreringen av klorofyll i sjöar [Revision and adjustment of the “Nordic” intercalibration of chlorophyll in lakes] (NGIG). Report 2007:5. Department of Environmental Assessment. Swedish University of Agricultural Sciences (SLU)

4 Macrophytes in lakes

Parameter	Primarily shows the effets of	How often do measurements need to be taken?	At what time of year?
Trophim macrophyte index (TMI)	Nutrient impact	Once a year	Late summer

4.1 Introduction

The concept of macrophytes, i.e. aquatic vegetation, includes vascular plants (helophytes and hydrophytes), mosses and charophytes. Macrophytes have an impact on, and are themselves impacted by biological and hydro-biogeochemical processes in lakes. Macrophyte species show different preferences along gradients of nutrient status (primarily nitrogen and phosphorus), pH and alkalinity. It is these preferences (Figure 4.1) that have been used in many countries to develop macrophyte-based indicator values and that form the basis of the trophic macrophyte index (TMI) described here. Among vascular macrophytes, it is only the hydrophytes that are considered to reflect the nutrient status of lake-water. Helophytes are therefore excluded from many indicator systems.

In contrast to phytoplankton, macrophytes are regarded as being more inert in their reaction to changes in the nutrient status. The presence of macrophyte species should therefore be regarded as a measure of the spring/early summer nutrient status, rather than the prevailing nutrient status when the inventory is taken.

The assessment criteria for macrophytes in lakes should not be confused with potential assessment tools for biodiversity (e.g. according to the Habitat Directive). There are many factors that influence biological diversity and natural values in lakes. This means that it is extremely difficult to link changes in, for example, biological diversity with particular environmental changes. Nevertheless, it is a requirement under the WFD to make such a link. In this context, the presence of macrophytes has in earlier studies shown itself to be an important indicator for the nutrient status of lakes (primarily phosphorus). It is also important to emphasise that it is not the phosphorus levels as such that govern the division into status classes but the presence of certain macrophyte species and their presence along the phosphorus gradient.

In order to classify the status of lakes as regards macrophytes, a trophic macrophyte index (TMI) is calculated. This is based on giving all macrophyte species, other than helophytes, an indicator value along a total phosphorus gradient. The TMI thus corresponds to the nutrient status, in the first instance total phosphorus.

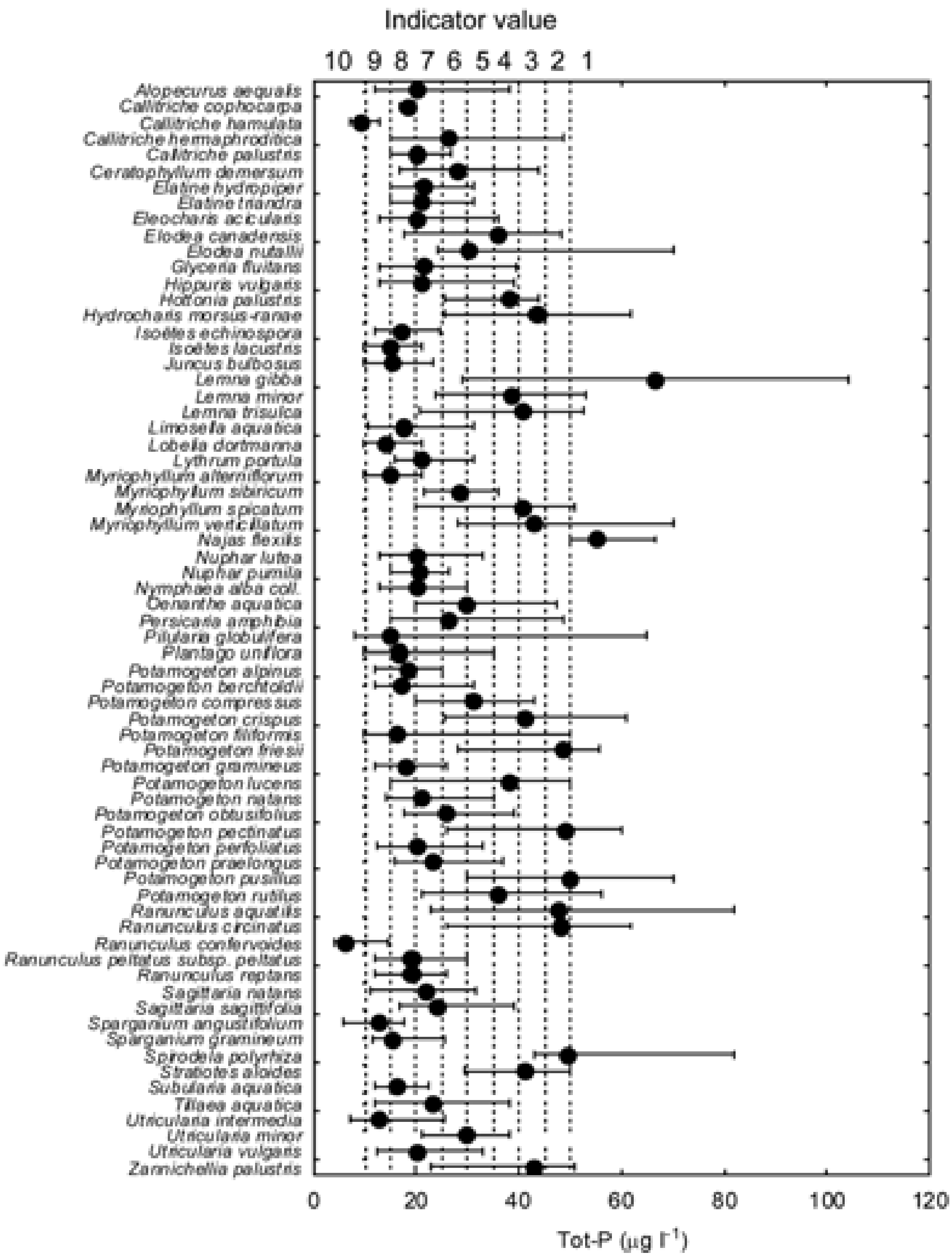

Figure 4.1. Macrophytes (vascular plants, other than helophytes, in alphabetical order): median values (± 25 and 75 percentiles) along the Tot-P gradient. Only species present in ≥3 lakes in the supporting data have been included in the figure.

4.2 Input parameters

In order to classify the status of lakes as regards macrophytes, a trophic macrophyte index (TMI) is calculated. This is based on giving all macrophyte species, other than helophytes, an indicator value along a total phosphorus gradient. The TMI thus corresponds to the nutrient status, in the first instance total phosphorus.

See REG Annex 1, Section 2.1

4.3 Requirements for supporting data

A precise description of the inventory methodology for ecological classification of lakes envisaged under the WFD is currently (2007) being produced and will be one of the Swedish EPA's survey types. Only the most central aspects are described below.

See REG Annex 1, Section 2.2

Application of the assessment criteria for macrophytes in lakes requires that the inventory must cover all macrophytes, including mosses and charophytes, with the exception of helophytes. Sampling must have been carried out during the late summer when aquatic vegetation is fully developed. The inventory is conducted both along the shoreline and from boats. Both underwater viewing tubes and rakes (e.g. Luther rakes) are used in carrying out inventories from a boat. A record should be made of the maximum depth at which each macrophyte species is found. It is therefore recommended that some form of transect inventory should be conducted. The presence of all existing macrophytes should be noted on a semi-quantitative scale (e.g. DAFOR, Palmer et al. 1992[5]) or on a binary scale (present, not present). To ascertain the lakes' TMIs, however, only binary data are required. It is preferable for the inventory to be carried out in different areas of the lake in order to obtain a complete macrophyte list and, above all, to ensure that any different bottom substrates in the lakes have been catalogued.

4.4 Typology

For the classification of macrophytes, lakes in Sweden are divided into three types, with different reference values (Table 4.1). These types are based on the ecoregions stated in the Agency's Regulations on Typology and Analysis, NFS 2006:1. The regulations contain a more precise division into limnic types but the current supporting data does not show other factors for limnic systems having had a significant impact on the macrophyte community. All the limnic types that match one of those for macrophyte systems are given the same reference value.

Table 4.1. Typology for status classification of macrophytes in relation to the ecoregions given in the regulations on Typology and Analysis NFS 2006:1.

Types for macrophytes	Ecoregin in accordance with NFS 2006:1
1 North of Limes Norrlandicus, above the highest coastline	Ecoregions 1 and 2
2 North o Limes Norrlandicus,	Ecoregion 3
3 South of Limes Norrlandius,	Ecoregions 4, 5, 6, and 7

[5] Palmer, M. A., S. L. Bell, and I. Butterfield. 1992. A botanical classification of standing waters in Britain: Applications for conservation and monitoring. Aquatic Conservation: Marine and Freshwater Ecosystems 2:125-143.

4.5 Classification of status

See REG Annex 1, Section 2.3

Step 1) Calculate the Trophic Macrophyte Index, TMI. The TMI of the lakes is a weighted mean value of the individual macrophyte indicator values and weight factors.

The calculation is carried out as follows:

$$\mathrm{TMI}_{Lake_x} = \frac{\sum_{i=1}^{n}\left(Indicator\ value_{Species_i} \times Weight\ \ factor_{Species_i}\right)}{\sum_{i=1}^{n} Weight\ \ factor_{Species_i}}$$

The macrophyte indicator values and weight factors are given in Table 4.2.

Table 4.2. The macrophyte indicator values (1-10) and weight factors (0.1-1), listed in the order of the Latin names of the species. The indicator values are based on the species preference (median value) along the tot-P gradient. A high indicator number indicates a preference for low tot-P concentrations and a high weight factor indicates narrow niches (low difference between the 75 and 23 percentiles) along the tot-P gradient.

Charophytes

Scientific name	Indicator value	Weight factor
Chara aspera	2	0.5
Chara contraria	2	0.6
Chara globularis	6	0.9
Chara hispida	1	0.4
Chara rudis	6	0.6
Chara tomentosa	7	0.6
Chara virgata	8	1.0
Nitella flexilis	10	1.0
Nitella opaca	10	1.0
Nitella wahlbergiana	7	0.9

Mosses

Scientific name	Indicator value	Weight factor
Bryum pseudotriquetrum	10	1.0
Calliergon cordifolium	7	0.9
Calliergon giganteum	9	0.9
Calliergon megalophyllum	8	1.0
Calliergonella cuspidata	8	0.4
Drepanocladus aduncus	7	0.8
Drepanocladus longifolius	8	0.9
Drepanocladus polygamus	8	1.0
Drepanocladus sordidus	7	1.0
Fissidens fontanus	8	1.0
Fontinalis antipyretica	8	0.7
Fontinalis dalecarlica	10	0.8
Fontinalis hypnoides	6	0.9
Leptodictyum riparium	8	0.9
Platyhypnidium riparoides	9	1.0
Pseudobryum cinclidioides	8	0.8
Riccia fluitans	2	0.5
Ricciocarpus natans	2	0.8
Scorpidium scorpioides	10	0.9
Sphagnum auriculatum	8	0.4
Sphagnum cuspidatum	10	1.0
Sphagnum platyphyllum	8	0.9
Sphagnum subsecundum	10	1.0
Warnstorfia exannulata	8	1.0
Warnstorfia fluitans	10	1.0
Warnstorfia trichofylla	10	1.0
Warnstorfia tundrae	8	1.0

Vascular plants

Scientific name	Indicator value	Weight factor
Alopecurus aequalis	8	0.8
Callitriche cophocarpa	8	1.0
Callitriche hamulata	10	1.0
Callitriche hermaphroditica	6	0.7
Callitriche palustris	8	0.9
Ceratophyllum demersum	6	0.8
Elatine hydropiper	7	0.9
Elatine triandra	7	0.9
Eleocharis acicularis	8	0.8
Elodea canadensis	4	0.7
Elodea nutallii	6	0.6
Glyceria fluitans	7	0.8
Hippuris vulgaris	7	0.8
Hottonia palustris	4	0.9
Hydrocharis morsus-ranae	3	0.7
Isoëtes echinospora	8	0.9
Isoëtes lacustris	9	0.9
Juncus bulbosus	8	0.9
Lemna gibba	1	0.3
Lemna minor	4	0.8
Lemna trisulca	3	0.7
Limosella aquatica	8	0.8
Lobelia dortmanna	9	0.9
Lythrum portula	7	0.9
Myriophyllum alterniflorum	9	0.9
Myriophyllum sibiricum	6	0.9
Myriophyllum spicatum	3	0.7
Myriophyllum verticillatum	3	0.6
Najas flexilis	1	0.9
Nuphar lutea	8	0.9
Nuphar pumila	7	0.9
Nymphaea alba coll.	8	0.9
Oenanthe aquatica	6	0.8
Persicaria amphibia	6	0.7
Pilularia globulifera	9	0.5
Plantago uniflora	8	0.8
Potamogeton alpinus	8	0.9
Potamogeton berchtoldii	8	.9
Potamogeton compressus	5	0.8
Potamogeton crispus	3	0.7
Potamogeton filiformis	8	0.7
Potamogeton friesii	2	0.8
Potamogeton gramineus	8	0.9
Potamogeton lucens	4	0.7
Potamogeton natans	7	0.8
Potamogeton obtusifolius	6	0.8
Potamogeton pectinatus	2	0.7

Scientific name	Indicator value	Weight factor
Potamogeton perfoliatus	8	0.8
Potamogeton praelongus	7	0.8
Potamogeton pusillus	2	0.7
Potamogeton rutilus	4	0.7
Ranunculus aquatilis	2	0.5
Ranunculus circinatus	2	0.7
Ranunculus confervoides	10	0.9
Ranunculus peltatus subsp. peltatus	8	0.9
Ranunculus reptans	8	0.9
Sagittaria natans	7	0.8
Sagittaria sagittifolia	7	0.8
Sparganium angustifolium	9	0.9
Sparganium gramineum	8	0.9
Spirodela polyrhiza	2	0.7
Stratiotes aloides	3	0.8
Subularia aquatica	8	0.9
Tillaea aquatica	7	0.8
Utricularia intermedia	9	0.9
Utricularia minor	6	0.9
Utricularia vulgaris	8	0.8
Zannichellia palustris	3	0.8

The nomenclature for vascular plants follows Karlsson 2004[6].

The nomenclature charophytes follows Blindow, Krause, Ljungstrand & Koistinen 2007[7].

The nomenclature for mosses follows Hallingbäck, Hedenäs & Weibull 2006.[8]

Step 2) The Ecological Quality Ratio, EQR, for the respective lake is calculated as follows:

$$\mathrm{EQR}_{Lake_x} = \frac{(Observed\ TMI_{Lake_x} - 1)}{(Reference\ value - 1)}$$

Reference values and class boundaries are given in Table 4.3.

[6] Karlsson, T. 2004. Checklista över Nordens kärlväxter [The vascular plants of Sweden – a checklist] – version 2004-01-19. URL: http://www2.nrm.se/fbo/chk/. The list is available in printed form, as a booklet of the Svensk Botanisk Tidskrift (booklet 5, 1997). Supplements have come out in the same journal (booklets 2, 3–4 and 5, 2002, and booklets 3–4, 2003)

[7] Hallingbäck, T, Hedenäs, L. & Weibull, H. 2006. Ny checklista för Sveriges mossor [New checklist of Swedish bryophytes]. Svensk Botanisk Tidskrift 100:96-148

[8] Hallingbäck, T, Hedenäs, L. & Weibull, H. 2006. Ny checklista för Sveriges mossor [New checklist of Swedish bryophytes]. Svensk Botanisk Tidskrift 100:96-148

4.6 Reference values and class boundaries

Table 4.3. Reference values and class boundaries for classification of macrophytes in lakes. There is no supporting data enabling class boundaries between poor and bad status to be developed.

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Type	**Status**	**TMI** **Ecological quality ratio (EQR)**
1 North of Limes Norrlandicus, above the highest coast line	Reference value	8.54
	High	≥0.97
	Good	≥0.90 and <0.97
	Moderate	≥0.83 and <0.90
	Poor, bad	<0.83
2 North of Limes Norrlandicus, below the highest coast line	Reference value	8.16
	High	≥0.97
	Good	
	Moderate	≥0.85 and <0.94
	Poor, bad	<0.85
3 South of Limes Norrlandicus	Reference value	8.27
	High	≥0.98
	Good	≥0.88 and <0.98
	Moderate	≥0.58 and <0.88
	Poor, bad	<0.58

4.7 Management of uncertainty

To make a good classification, it is appropriate to use data from a number of samplings. Repeated measurements give a more reliable classification and an uncertainty interval in the form of a standard deviation can be calculated for the parameter in the water body in question. In those cases where data from only one inventory are available, an estimate of the uncertainty may be made. If the calculated value for the Ecological Quality Ratio lies <0.05 units from one of the class boundaries between high and good status, or between good and moderate status, it means that the value lies close to a class boundary. That indicates that a reasonability assessment must be made, as described in Section 4.1.1 of the main part of the Handbook. As an aid, the list of species in Table 4.4 should be used to make a more reliable classification of the status of the macrophyte quality factor. See also Section 4.1.2 in the main handbook for more guidance on how to manage uncertainty.

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Table 4.4 Macrophyte species that should be used in combination with EQRs of the lakes when these lie close to a class boundary, in order to distinguish between different classes of status in the three types.

Type	Class boundary between:			
	high and good		**good and moderate**	
	Only in high	**In good and lower status**	**In good or high but not in moderate**	**In moderate, poor or bad, but not in good or high**
1	*Alopecurus aequalis*[1]	*Lemna trisulca*[2]	*Callitriche hamulata*[2]	
	Fontinalis antipyretica[1]	*Myriophyllum spicatum*[2]	*Lobelia dortmanna*[2]	
	Isoëtes lacustris[2]	*Potamogeton compressus*[1]	*Nitella opaca*[2]	
	Isoëtes echinospora[2]	*Potamogeton obtusifolius*[1]	*Ranunculus confervoides*[2]	
	Juncus bulbosus[2]		*Sparganium angustifolium*[2]	
	Persicaria amphibia[1]		*Utricularia intermedia*[2]	
	potamogeton berchtoldii[2]			
	Scorpidium scorpioides[1]			
	Warnstorfia fluitans[1]			
	Warnstorfia trichophyllus[1]			
2	*Isoëtes lacustris*[2]			*Lemna minor*[2]
	Juncus bulbosus[2]			*lemna trisulca*[2]
	Lobelia dortmanna[2]			*Potamogeton compressus*[2]
	Myriophyllum alterniflorum[2]			
	ranunculus reptans[2]			
	Sparganium angustifolium[2]			
	Utricularia minor[2]			
3	*Isoëtes lacustris*[2]	*Chara aspera*[2]	*Calliergonella cuspidata*[2]	*Chara contraria*[2]
	Isoëtes echinospora[2]	*hydrocharis morsus-ranae*[2]	*Callitriche hamulata*[2]	*Potamogeton friesii*[2]
	Juncus bulbosus[2]	*Lemna trisulca*[2]		*Spirodela polyrhiza*[2]
	Lobelia dortmanna[2]	*Myriophyllum spicatum*[2]		*Stratiotes aloides*[2]
	nitella opaca[2]	*Potamogeton filiformis*[2]		
	Scorpidium scorpioides[2]	*Ranunculus circinatus*[2]		
	Sparganium angustifolium[2]	*Ricciocarpus natans*[2]		
	Sparganium gramineum[2]	*Zannichellia palustris*[2]		
	subularia aquatica[2]			
	Utricularia intermedia[2]			
	Warnstorfia fluitans[1]			
	Warnstorfia trichophyllus[1]			

[1] Exists only in the respective status class

[2] Exists with ≥ 70 % but < 100 % in the respective status class

4.8 Human impact or natural

When the status classification results in a moderate or worse status, it may be necessary to make an assessment whether this is a result of anthropogenic eutrophication or whether the lake is naturally rich in nutrients. However, it is not particularly common for lakes to have naturally high nutrient levels. In order to evaluate this, a comparison can be made with results for the assessment criterion for phosphorus. The assessment can further be improved by looking at the impacts/pressures on the water body. Important supporting material for this is source distribution data, historical data, etc., produced in connection with the characterisation. If the assessment that the lake is naturally rich in nutrients is made on the basis of an expert assessment by the water authority, a revision of the reference value for the specific water body should be made.

See REG Chapter 2 11 §

See GG to Chapter 2 11 §

4.9 Comments

Data material for the development of macrophyte-based assessment criteria was qualitative, i.e. only presence was noted, not the degree of coverage or the frequency of occurrence of the respective species. For example, a lake that shows several signs of eutrophying, also contains a small, and possibly diminishing, amount of quillwort (*Isoëtes lacustris*, indicating nutrient-poor conditions). This small amount may help to make the trophic index for this lake good or even high. The trophic index in its present form does not take account of how much there is of a particular species (degree of coverage, individuals, etc). For future environment monitoring with the aid of macrophytes, it is nevertheless recommended that there should be a semi-quantitative inventory (Ecke, 2007[9]) .

4.10 Example

The following macrophytes were found in Abiskojaure, Torne Lappmark, (without helophytes): *Alopecurus aequalis, Hippuris vulgaris, Myriophyllum alterniflorum, Nitella opaca, Ranunculus confervoides, Ranunculus peltatus subsp. peltatus, Ranunculus reptans and Sparganium angustifolium*. According to Table 4.2 and the formula for calculating the trophic index (TMI), the TMI for Abiskojaure is 8.68.

The next step is to identify what type the lake belongs to, which can be done using Table 4.1. Abiskojaure belongs to type 1. The status of the lakes is identified with the aid of the class boundaries in Table 4.3.

The ecological ratio for Abiskojaure is (8.68 – 1)/(8.54 – 1) = 1.02.

Since the ecological quality ratio is greater than the critical value for the H/G class boundary (0.97), the established assessment criteria give Abiskojaure ‘high’ status. The EQR for Abiskojaure is 1.02, i.e. <0.05 units from the class boundary high/good (0.97). For the expert assessment, the species list from Table 4.4 is compared with the species list from Abiskojaure. Three species in Abiskojaure can be present in lakes with both high and good status, namely *Nitella opaca, Ranunculus confer-*

[9] Ecke, F. 2007. Utvärdering av metoder för makrofytinventering [Evaluation of macrophyte inventory methods]. Technical Report, 2007:02, Department of Chemistry and Geo-Science, Luleå University of Technology

voides and *Sparganium angustifolium*. However, one of the species present in Abiskojaure is *Alopecurus aequalis*, which is typical of lakes with high status only. Abiskojaure is therefore classified as a lake with high-status.

Background reports: Ecke, F., 2007. Bedömningsgrunder för makrofyter i sjöar - bakgrundsrapport [Assessment criteria for macrophytes in lakes - background report]. Research report, 2007:17. Luleå University of Technology, Department of Applied Chemistry and Geo-Science, Division of Applied Geology

5 Diatoms in watercourses

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what times of the year?
IPS	Nutrient impact and organic pollution	Once a year	Late summer/autumn
ACID	Acidity	Once a year	Late summer/autumn
%PT (support parameter)	*Organic pollution*	*Once a year*	*Late summer/autumn*
TDI (support parameter)	*Nutrient impact*	*Once a year*	*Late summer/autumn*

5.1 Introduction

Periphytic algae play an important role as primary producers, particularly in running water, and diatoms are often the dominant group in the periphyton community. Diatoms are good indicators of water quality and methods of classification and other evaluations of watercourses based on diatoms are in wide use in Europe and other parts of the world.

5.2 Input parameters

The parameters which must be classified for the diatom quality factor are the two indices IPS (Indice de Polluo-sensibilité Spécifique) and the acidity index ACID. The support parameters %PT (Pollution Tolerant valves) and TDI (Trophic Diatom Index) can also be assessed, to obtain better evidence in doubtful cases.

See REG Annex 1, Section 3.1

IPS shows the impact of nutrients and organic pollution. The support parameters %PT (indicates organic pollution) and TDI (indicates nutrient impact) may be used to obtain a more reliable classification. It is nevertheless IPS which must chiefly be used for the classification.

ACID indicates acidity. The acidity index, however, gives no status class but only groups the watercourse in a pH-regime. ACID thus does not distinguish between what is naturally acidic and what is anthropologically acidified. That must be determined by use of physico-chemical assessment criteria for acidification, as described in Chapter 15.

Classifications according to these two indices function throughout Sweden and the reference values and class boundaries are the same for the whole country.

5.3 Requirements for supporting data

The classification must be based on sampling and analyses in accordance with SS-EN 13946:2003 and SS-EN 14407:2005, or by another method which gives equivalent results. The latest version of the Agency's survey type: 'Periphyton in running water – diatom analysis' is also a good procedure to follow.

See REG Annex 1, Section 3.2

One sample per year, preferably taken in the late summer/autumn, is sufficient to classify the water quality, although several samples of course give a more reliable classification. It is important that the diatom analysis is carried out at the species level and also that the person conducting it has good knowledge of the species and makes use of sufficient taxonomic literature (described in the Swedish EPA's survey type: 'Periphyton in running water – diatom analysis'), since the most important source of error lies in the identification of species. The software program Omnidia, available through CLCI (Catherine Lecointe Conseil Informatique) (http://perso.club-internet.fr/clci/tour_guide.htm) facilitates the calculation of IPS, %PT, TDI and ACID.

5.4 IPS

5.4.1 Classification of status

IPS is calculated as follows:

$IPS = \Sigma A_j I_j V_j / \Sigma A_j V_j$

See REG Annex 1, Section 3.3

where
A_j = the relative abundance in percentage of taxon j
V_j = the indicator value of taxon j (1-3, where a high value means that a taxon only tolerates limited ecological variations, i.e. it is a strong indicator)
I_j = the pollution sensitivity of taxon j (1-5, where high values show a high pollution sensitivity).

Results obtained according to the above formula are recalculated on a scale of 1-20 according to
4.75 * original index value – 3.75.

The ecological quality ratio (EQR) is calculated as follows:

EQR = calculated IPS / reference value

Reference values and class boundaries are given in Table 5.1.

As a complement to the IPS index, it is suggested that a computation of TDI and %PT, which show the diatoms' tolerance of nutrient impact and organic pollution respectively, should be carried out. TDI is calculated in the same way as IPS using TDI-specific indicator values and sensitivity values respectively. Results obtained according to the above formula are recalculated on a scale of 1-100 according to 25 * original index value – 25.

%PT is the sum of the relative abundance of all diatom species that are classed as organic pollution tolerant.

These parameters are, however, only a support and it is IPS which indicates the status class. Class boundaries for TDI and %PT are given in Table 5.2.

Calculation of the index and support parameters can be carried out with the aid of the software program Omnidia. Indicator values and pollution sensitivity classifications for common diatoms in Sweden are also shown in the method description in the Agency's survey type: 'Periphyton in running water – diatom analysis'.

5.4.2 Reference values and class boundaries

Table 5.1. Reference values and class boundaries for IPS in all Swedish types. Method-bound measure of uncertainty: Margin of error +/- 0.5 unit if IPS > 13, margin of error +/- 1 unit if IPS < 13.

Status	IPS value	EQR value
Reference value	19.6	
High	≥17.5	≥0.89
Good	≥14.5 and <17.5	≥0.74 and <0.89
Moderate	≥11 and <14.5	≥0.56 and <0.74
Poor	≥8 and <11	≥0.41 and <0.56
Bad	<8	< 0.41

For status classification it is recommended to use the IPS values. Conversion to EQR values and use of these class boundaries gives the same result but can be an unnecessary step in the calculation in normal cases. If the assessment is nonetheless that the watercourse is naturally nutrient-rich, the reference value can be adjusted and in that case the EQR class boundaries are used to obtain the same deviation from the reference value as before. This is further described in Chapter 5.7.

Table 5.2. The class boundaries for the support parameters %PT and TDI may be used to distinguish the classes further in uncertain cases. It is however IPS that gives the main status classification.

Status	%PT	TDI
Reference value	-	-
High	< 10	< 40
Good	< 10	40-80
Moderate	< 20	40-80
Poor	20-40	> 80
Bad	> 40	> 80

5.5 ACID

5.5.1 classification of status

The acidity index ACID is calculated as follows:

ACID = [log((ADMI/EUNO)+0.003))+2.5] +
[log((circumneutral+alkaliphile+alkalibiont)/(acidobiont+acidophile))+0.003)+2.5]

See REG Annex 1, Section 3.4

A numerator or denominator = 0 is replaced by 1, when the relative abundance is expressed as a percentage. In Omnidia the relative abundance of van Dam groups is given per mille, and 0 is then replaced by 10.

The first part of the index is based on the ratio between the relative abundance of Achnanthidium minutissimum (ADMI) and the genus Eunotia (EUNO). The second part of the index takes into account all diatoms in the sample and is based on the following classification (van Dam et al. 1994[10]) , which is given in the software program Omnidia:

acidobiont	mainly present at pH <5.5
acidophile	mainly present at pH <7
circumneutral	mainly present at pH values around 7
alkaphile	mainly present at pH >7
alkalibiont	only present at pH >7

Class boundaries between the various acidity classes are given in Table 5.3.

5.5.2 Class boundaries

Table 5.3. Assessment of acidity in watercourse with the aid of diatoms (acidity index ACID). Division into five acidity classes. The classes show different stages of acidity and do not relate to status. Corresponding mean and minimum pH is also given. Method-bound measure of uncertainty: Margin of error ± 10%.

Acidity classes	Acidity index ACID	Corresponds to mean pH (mean value of the 12 months preceding sampling)	Corresponds to minimum pH (during the 12 months preceding sampling)
Alkaline	≥ 7.5	≥ 7.3	-
Almost neutral	5.8-7.5	6.5-7.3	-
Moderately acidic	4.2-5.8	5.9-6.5	< 6.4
Acidic	2.2-4.2	5.5-5.9	< 5.6
Highly acidic	< 2.2	< 5.5	< 4.8

The acidity classes relate to the reaction of diatoms to pH changes. For the quality factors benthic fauna in lakes and watercourses, and phytoplankton in lakes, there are also acidity classes bearing the same names. Since e.g. benthic fauna do not

[10] van Dam, H., Mertens, A. & Sinkeldam, J. (1994). A coded checklist and ecologicaltor values of freshwater diatoms from The Netherlands. 28(1): 117-133.

react as quickly as diatoms to a reduction in pH, their attribution to classes is somewhat different. That is fully in line with the Water Framework Directive. It is the biological response that must be measured. Since different quality factors have different sensitivities to impact they will in certain cases result in different status classes for the same body of water. Because the operating principle is that the worst quality factor determines the classification, this ensures that the most sensitive quality factor is also protected.

5.6 Management of uncertainty

To make a good classification, it is appropriate to use data from a number of samplings. Several readings give a more reliable classification and an uncertainty interval in the form of a standard deviant can be calculated for the parameter in the water body in question. In cases where only data from one year is available, the fixed value for method-bound uncertainty for IPS or ACID given in Tables 5.1 and 5.3 may be used. In cases where the uncertainty interval around the calculated value overlaps any of the class boundaries between high and good status, or between good and moderate status, it means that the calculated value lies very close to a class boundary. For this reason, a reasonability assessment should be made, as described in Chapter 4.1.1 of the main handbook. See also Chapter 4.1.2 in the main handbook for more guidance on how to handle uncertainty.

See REG
Chapter 2
9 §

See GG to
Chapter 2
9 §

5.7 Human impact or natural

If the watercourse is classified in one of the acidity classes 'moderately acidic', 'acidic' or 'highly acidic', an assessment must be made about whether the acidity conditions are anthropogenic in origin or whether the watercourse is naturally acidic. A more thorough analysis should be made with the aid of the assessment criteria for acidification in accordance with Chapter 15. The analysis can be further improved by making an assessment of the impact or stress caused by the acidification. The impact of forestry, for example, can provide important evidence about this. Furthermore, data on deposits may be used if analyses of large areas are to be made. If the assessment is that the watercourse is naturally acidic, a reference value for pH for the water body should be calculated in accordance with Chapter 15. The pH reference value is compared with the pH values which correspond with the acidity classes for diatoms (Table 5.3). The acidity class for which the interval for mean pH covers the calculated reference value for pH corresponds to high status. The subsequent classes correspond to good, moderate, poor and bad status following the order of descending pH values.

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Chapter 2
10 §

See GG to
Chapter 2
10 §

See REG
Annex 1,
section 3.4.3

See REG
Chapter 2
11 §

When the status classification results in a 'moderate', or worse, status it may be necessary to make an assessment whether that is a result of anthropogenic eutrophication or whether the lake is naturally nutrient-rich. However, it is not particularly common for watercourses to have naturally high nutrient content. In order to evaluate this, a comparison can be made with results for the assessment criterion for phosphorus. The assessment can further be improved by looking at the impacts/pressures on the water body. Source distribution data, historical data, etc., provide important supporting material, produced in connection with the characterisation. If the evaluation that the watercourse is naturally rich in nutrients is made, on the basis of an expert assessment by the water authority, a revision of the reference value for the specific water body should be made. In this case, the EQR class boundaries in Table

See GG to
Chapter 2
11 §

5.1 are used instead of the stated IPS values. The calculated IPS value for the water body is divided by the new reference value, to obtain an EQR that is then compared with the EQR class boundaries.

Background reports: Kahlert, M., Andrén, C. & Jarlman, A., 2007. Bakgrundsrapport för revideringen 2007 av bedömningsgrunder för Påväxt – kiselalger i vattendrag [Background report for revision 2007 of assessment criteria for periphyton - diatoms in watercourses]. Report 2007:23. Department of Environmental Assessment, Swedish University of Agricultural Sciences (SLU)

6 Benthic macroinvertebrate assemblages in lakes

Parameter	Shows primarily effects of	How often do measurements need to be taken?	At what time of year?
ASPT	Ecological quality (littoral)	Once a year	autumn
MILA	Acidity (littoral)	Once a year	autumn
BQI	Nutrient impact (profundal)	Once a year	autumn

6.1 Introduction

Different types of impact, such as eutrophication and acidity, result in a shift in the taxonomic composition of benthic macroinvertebrate assemblages (bottom-dwelling invertebrate animals) in lakes and watercourses, towards a greater dominance of tolerant species. Within Europe there is a long tradition of using benthic macroinvertebrate assemblages as an indicator of changes in aquatic systems, and many countries have developed their own national metrics. An index weights together information from several indicator taxa (or species) and thereby simplifies classification. In recent years, the development has tended to be towards 'multimetric' indices, in which information from several different individual indices or parameters are combined to a single index. Each of these simple indices shows a correlation with a specific impact and in that way a multimetric index can be constructed from several simple indices, each of which reflects different aspects of the benthic macroinvertebrate communities (e.g. species richness, diversity, function, pollution-tolerance). The Swedish University of Agricultural Sciences (SLU) has developed two multimetric macroinvertebrate indices for acidity, for both lakes and watercourses (respectively, MILA and MISA); in addition, a relatively new multimetric index for detecting the impact of eutrophication on watercourses (the DJ index) has been calibrated.

Index calculations can conveniently be carried out using the software program ASTERICS, which is freely available on the website http://www.aqem.de. Data files can be uploaded to ASTERICS (in Excel or ASCII format) if these contain sampled taxa equipped with AQEM codes (Shortcode, ID_ART or TAXON_NAME). The AQEM codes are described in the English manual (Manual for AQEM European Stream assessment program, version 2.3) and in the taxa lists found on the same website as the program. The out-file from ASTERICS contains many different indices in use in Europe. Some of these indices are part of these assessment criteria, but others are perhaps in use in other European countries and can for example be used to calculate additional multimetric indices. **Please note however that ASTERICS also gives a classification of indices in accordance with the old assessment criteria (Swedish EPA report 4913, 1999), and these must therefore not be used.**

6.2 Input parameters

ASPT (Average Score Per Taxon) (Armitage et al 1983)[11] is an index in which different families of benthic macroinvertebrates are scored according to their sensitivity to an environmental impact and which integrates the impact from eutrophication, pollution by oxygen-consuming substances and habitat-disturbing impact such as straightening/clearing (including turbidity).

See REG Annex 1, Section 4.1

BQI (Benthic Quality Index) (Wiederholm 1980)[12] exploits knowledge about the sensitivity of different species to low oxygen levels and is used to measure the condition in the profundal of lakes.

MILA (Multimetric Index for Lake Acidity) (Johnson & Goedkoop 2007)[13] is a multimetric acidity index for lakes which contains six parameters/indices based on the littoral macroinvertebrate assemblages of lakes.

6.3 Requirements for supporting data

To enable the assessment criteria for benthic macroinvertebrate assemblages in lakes to be applied, the sampling and analysis must be carried out according to SS EN-27828 or by another method that gives equivalent results for samples in the littoral, and according to SS-028190 or by another method that gives equivalent results for samples in the profundal. The remaining information in Table 6.1 is also recommended for optimum classification. The species determination must have been made in accordance with the standardised taxonomic list in regulations (NFS 2008:1), Annex 1, Table 4.6.

See REG Annex 1, Section 4.2

Table 6.1. Overview of sampling methods for benthic macroinvertebrate assemblages in lakes.

Habitat	Method	Sampling effort*	Mesh-size (mm)	Number of samples	Season
Exposed littoral	SSEN-27828	60 s x 1 m	0.5	5	Autumn
Profundal	SS 028190	**	0.5	5	Autumn

* refers to SPARK time and SPARK distance, ** not time-dependent

[11] Armitage, P.D., Moss, D. Wright, J.F. & M.T. Furse. 1983. The performance of a new biological water quality score system based on macroinvertebrate assemblages over a wide range of unpolluted running-waters. Water Research 17: 333–347.

[12] Dahl, J. & R.K. Johnson. 2004. A multimetric macroinvertebrate index for detecting organic pollution of streams in southern Sweden. Archiv für Hydrobiologie, 160: 487-513

[13] Johnson, R.K. and Goedkoop, W. 2007. Bedömningsgrunder för bottenfauna i sjöar och vattendrag – Användarmanual och bakgrundsdokument [Assessment criteria for benthic macroinvertebrate assemblages in lakes and watercourses - User manual and background document]. Report 2007:4.

6.4 Typology

For the classification of benthic macroinvertebrate assemblages, Swedish lakes are divided into three types, based on Illies ecoregions (Figure 6.1). Table 6.2 shows how they correspond to the limnic ecoregions given in the Swedish EPA's Regulations on Typology and Analysis, NFS 2006:1.

Figure 6.1. Illies ecoregions, the Central Plains (14), the Fenno-Scandian Shield (22) and the Borealic Uplands (20).

Table 6.2 Typology for status classification of benthic macroinvertebrate assemblages in relation to the ecoregions given in Regulations NFS 2006:1.

Types for benthic invertebrate assem-blages	Ecoregion in accordance with NFS 2006:1
Illies Ecoregion 20	Ecoregions 1 and 2 (partial)
Illies Ecoregion 22	Ecoregions 2 (partial) and 3
Illies Ecoregion 14	Ecoregions 4, 5, 6 and 7

6.5 ASPT

6.5.1 Classification of status

ASPT exploits the differences in tolerance among different families of benthic macroinvertebrates and also includes the Oligochaeta order (earthworms). Highly sensitive families give high indicator values, while those with high tolerance give low indicator values. The index value for ASPT is a mean value for included taxa and is calculated by adding indicator values and dividing them by the number of included taxa (families).

See REG Annex 1, Section 4.3

Table 6.3. Indicator values for ASPT for different families.

Indicator value	Family
10	Aphelocheiridae, Beraeidae, Brachycentridae, Capniidae, Chloroperlidae, Ephemeridae, Ephemerellidae, Goeridae, Heptageniidae, Lepidostomatidae, Leptoceridae, Leptophlebiidae, Leuctridae, Molannidae, Odontoceridae, Perlidae, Perlodidae, Phryganeidae, Potamanthidae, Sericostomatidae. Siphlonuridae, Taeniopterygidae
8	Aeshnidae, Astacidae, Agriidae, Cordulegasteridae, Corduliidae, Gomphidae, Lestidae, Libellulidae, Philopotamidae, Psychomyiidae
7	Caenidae, Limnephilidae, Nemouridae, Polycentropodidae, Rhyacophilidae (incl. Glossosomatidae)
6	Ancylidae, Coenagriidae, Corophiidae, Gammaridae, Hydroptilidae, Neritidae, Platycnemididae, Unionidae, Viviparidae
5	Chrysomelidae, Clambidae, Corixidae, Curculionidae, Dendrocoelidae, Dryopidae, Dytiscidae, Elminthidae, Gerridae, Gyrinidae, Haliplidae, Heledidae, Hydrophilidae (incl Hydraenidae), Hydropsychidae, Hygrobiidae, Hydrometridae, Mesoveliidae, Naucoridae, Nepidae, Notonectidae, Planariidae, Pleidae, Simuliidae, Tipulidae (inkl Pediciidae)
4	Baetidae, Piscicolidae, Sialidae
3	Asellidae, , Erpobdellidae, Glossiphoniidae, Hirudidae, Hydrobiidae, Lymnaeidae, Planorbidae, Physidae, Sphaeriidae, Valvatidae
2	Chironomidae
1	Oligochaeta

The ecological quality ratio (EQR) is calculated as follows:

EQR = calculated ASPT / reference value

Reference values and class boundaries are given in Table 6.4.

6.5.2 Reference values and class boundaries

Table 6.4. Reference values and class boundaries for classification of the ASPT parameter in lakes. SD is the standard deviation for the EQR. Illies ecoregions according to Figure 6.1.

Type	**Status**	**ASPT** **Ecological quality ratio (EQR)**
Illies Ecoregion 14 Central Plains	Reference value	5.85
	Uncertainty (SD of EQR)	0.057
	High	≥0.95
	Good	≥0.70 and <0.95
		≥0.50 and <0.70
	Poor	≥0.25 and <0.50
		< 0.25
Illies Ecoregion 22 Fenno-Scandian Shield	Reference value	5.80
	Uncertainty (SD of EQR)	0.070
	High	≥0.90
	Good	≥0.70 and <0.90
	Moderate	≥0.45 and <0.70
	Poor	≥0.25 and <0.45
	Bad	< 0.25
Illies Ecoregion 20 Borealic Uplands	Reference value	5.60
	Uncertainty (SD of EQR)	0.130
	High	≥0.60
	Good	≥0.45 and <0.60
	Moderate	≥0.30 and <0.45
	Poor	≥0.15 and <0.30
	Bad	< 0.15

6.6 BQI

6.6.1 Classification of status

BQI exploits knowledge of the varying tolerance of different species of midges to low oxygen levels at lake bottoms. BQI is calculated on the basis of the presence and population density of different indicator taxa of midge larvae in the samples. BQI is calculated as:

See REG Annex 1, Section 4.4

$$BQI = \sum_{i=0}^{5} \frac{(k_i \bullet n_i)}{N}$$

Where:
ki = 5 for *Heterotrissocladius subpilosus (Kieff.),*
ki = 4 for *Paracladopelma sp., Micropsectra sp., Heterotanytarsus apicalis* (Kieff.),

Heterotrissocladius grimshawi (Edw.),
Heterotrissocladius marcidus (Walker) and
Heterotrissocladius maeaeri (Brundin)
ki = 3 for *Sergentia coracina* (Zett.), *Tanytarsus sp.*
and *Stictochironomus sp.,*
ki = 2 for *Chironomus anthracinus* (Zett.),
ki = 1 for *Chironomus plumosus* L.,
ki = 0 if these indicator taxa are not present in the sample
ni = the number of individuals within the indicator group in
N = the total number of individuals in all indicator groups

The ecological quality ratio (EQR) is calculated as follows:

EQR = the calculated BQI / reference value

Reference values and class boundaries are given in Table 6.5.

6.6.2 Reference values and class boundaries

Table 6.5. Reference values and class boundaries for classification of the BQI parameter. SD is the standard deviation for the EQR. Illies ecoregions according to Figure 6.1.

Type	Status	BQI Ecological quality ratio (EQR)
Illies Ecoregion 14 Central Plains	Reference value	2.68
	Uncertainty (SD of EQR)	0.060
		≥0.75
	Good	≥0.60 and <0.75
	Moderate	≥0.40 and <0.60
	Poor	≥0.20 and <0.40
	Bad	< 0.20
Illies Ecoregion 22 Fenno-Scandian Shield	Reference value	3.00
	Uncertainty (SD of EQR)	0.067
	High	≥0.90
	Good	≥0.70 and <0.90
	Moderate	≥0.45 and <0.70
		≥0.25 and <0.45
	Bad	< 0.25
Illies Ecoregion 20 Borealic Uplands	Reference value	3.25
	Uncertainty (SD of EQR)	0.01
	High	≥0.95
	Good	≥0.70 and <0.95
	Moderate	≥0.50 and <0.70
	Poor	≥0.25 and <0.50
	Bad	< 0.25

6.7 MILA

6.7.1 Classification of status

MILA is constructed from six different simple indices and responds to acidity. The indices are (1) relative abundance (%) of mayflies (Ephemeroptera), (2) relative abundance (%) of true flies (Diptera), (3) the number of mollusc taxa (Gastropoda) (4) the number of mayfly taxa (5) the value for the British AWIC index, and (6) the relative abundance (%) of predators in the sample.

See REG Annex 1, Section 4.5

Values for these simple indices must be normalised so that each has a value (indexnorm) between 0 and 10 according to Table 6.6. The normalised values are then added together and re-scaled by dividing the sum of the normalised index values by the number of simple indices included (a mean value) and multiplying this mean value by 10 according to the following:

MILA = 10 * sum $\text{index}_{\text{norm}}$/6

MILA thus acquires a value that can vary between 0 and 100.

Table 6.6. Normalisation of index values (Indexnorm) for the six simple indices to values between 0 and 10. In the next step MILA is calculated as a mean value for these normalised indices. ”ASTERICS nomenclature” relates to the software program at http://www.aqem.de.

Index	ASTERICS-nomenclature	$\text{Index}_{\text{norm}}$=10 if the index	$\text{Index}_{\text{norm}}$=0 if the index	Otherwise $\text{Index}_{\text{norm}}$=
% mayflies (of total abundance)	-Ephemeroptera|%]	>27	<0.05	$\frac{\lvert Ephemeroptera[\%]-0{,}05\rvert}{\lvert 27-0{,}05\rvert}*10$
% true flies (of total abundance)	-Diptera|%|	<26	>86	$\frac{\lvert Diptera[\%]-86\rvert}{\lvert 26-86\rvert}*10$
Molluscs (number of taxa)	-Gastropoda	>8	<0	$\frac{\lvert Gastropoda-0\rvert}{\lvert 8-0\rvert}*10$
Mayflies (number of taxa)	-Ephemeroptera	>6	<1	$\frac{\lvert Ephemeroptera-1\rvert}{\lvert 6-1\rvert}*10$
$\text{AWIC}_{\text{family}}$ index	AWIC Index	>5.4	<4.8	$\frac{\lvert AWICIndex-4{,}8\rvert}{\lvert 5{,}4-4{,}8\rvert}*10$
% predators (of total abundance)	-|%| Predators	<8.7	>19	$\frac{\lvert [\%]Predators-19\rvert}{\lvert 19-8{,}7\rvert}*10$

MILA shows the benthic invertebrates’ response to acidity. It cannot be determined from the MILA classification whether the acidity is natural or of anthropogenic origin.

The ecological quality ratio (EQR) is calculated as follows:
EQR = calculated MILA /reference value

Reference values and class boundaries are given in Table 6.7.

6.7.2 Reference values and class boundaries

Table 6.7. Reference values and class boundaries for MILA. The classes show different stages of acidity and do not relate to status. SD is the standard deviation for the EQR. Illies ecoregions according to Figure 6.1.

Type	Acidity class	MILA Ecological quality ratio (EQR)
Illies Ecoregion 14 Central Plains	Reference value	77.5
	Uncertainty (SD of EQR)	0.166
	Almost neutral	≥0.85
	Moderately acidic	≥0.50 and <0.85
	Acidic	≥0.35 and <0.50
	Highly acidic	≥0.15 and <0.35
	Extremely acidic	<0.15
Illies Ecoregion 22 Fenno-Scandian Shield	Reference value	49.4
	Uncertainty (SD of EQR)	0.202
	Almost neutral	≥0.85
	Moderately acidic	≥0.60 and <0.85
	Acidic	≥0.40 and <0.60
	Highly acidic	≥0.20 and <0.40
	Extremely acidic	< 0.20
Illies Ecoregion 20 Borealic Uplands	Reference value	41.7
	Uncertainty (SD of EQR)	0.130
	Almost neutral	≥0.60
	Moderately acidic	≥0.45 and <0.60
	Acidic	≥0.30 and <0.45
	Highly acidic	≥0.15 and <0.30
	Extremely acidic	<0.15

6.8 Management of uncertainty

A mean value of several measurements gives a better and more reliable classification, and an uncertainty interval in the form of a standard deviation can be calculated for the parameter. In those cases where only one year's data are available, the fixed value for method-bound uncertainty (standard deviation) for the respective parameters and types shown in Tables 6.4, 6.5 and 6.7 may be used. The uncertainty is calculated for reference lakes. Greater variation is expected in polluted lakes, a fact which it is well to keep in mind in making an uncertainty assessment. The standard deviation is a measure of precision or the uncertainty associated with classification. In cases where an uncertainty interval around the EQR overlaps any of the class boundaries between high and good status or between good and moderate status, the calculated EQR-value lies very close to a class boundary. That indicates that a reasonability assessment must be made, as described in Section 4.1.1 of the

See REG Chapter 2 9 §

See GG to Chapter 2 9 §

main handbook. See also Section 4.1.2 in the main handbook for more guidance on how to manage uncertainty.

6.9 Weighting of parameters

ASPT shows general ecological quality in the littoral zone, BQI shows eutrophication in the profundal zone and MISA shows the impact of acidity. To assess the weighted status for the benthic invertebrate quality element, the index that has received the worst status class is used.

6.10 Human impact or natural

If the lake is classified, using MILA, as either moderately acidic or acidic, an assessment must be made about whether that is due to anthropogenic acidification or whether the lake is naturally acidic. A more thorough analysis should be made with the aid of the assessment criteria for acidification as shown in Chapter 14. The analysis can be further improved by making an assessment of the impacts or pressures caused by the acidification. Important supporting data for this is, for example, the impact of forestry, and in addition data on deposition may be useful if analyses of relatively large areas are to be carried out. If the assessment is that the lake is to some extent naturally acidic, a pH reference value for the water body should be calculated in accordance with Chapter 14. Using the line equation in Figure 6.2, the calculated pH value for the lake is correlated with a new reference value for MILA. The measured value for MILA is divided by the new reference value and compared with the class boundaries in Table 6.7.

See REG Chapter 2 10 §

See GG to Chapter 2 10 §

See REG Annex 2, Section 4.5.2

The acidity classes, according to the revised reference value or the original classification, are transposed to status classes in accordance with the following:

Almost neutral – High status
Moderately acidic – Good status
Acidic – Moderate status
Highly acidic – Poor status
Highly acidic – Bad status

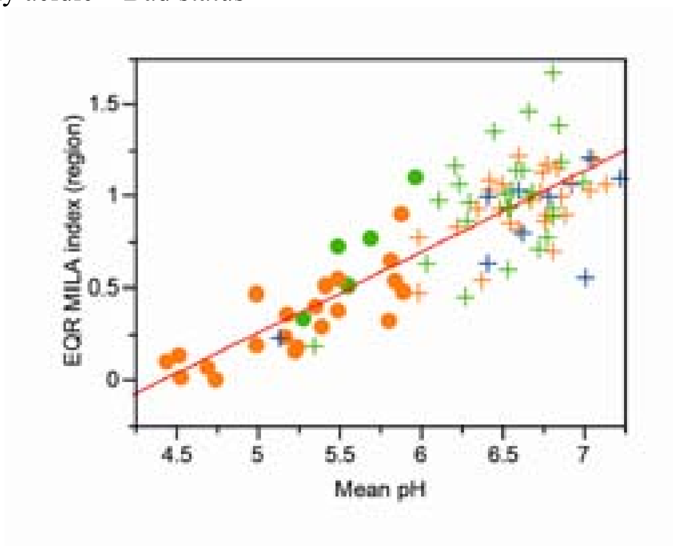

Figure 6.2. Correlation between mean-pH and the MILA index value. Borealic Upland (blue), Fenno-Scandian Shield (green) and Central Plains (orange). + = reference. The line equation gives the calculation of $MILA_{ref}$ in accordance with the following:

Region 14: $MILA_{ref} = -1.98 + 0.441\ pH_{ref}$

Region 22: $MILA_{ref} = -1.90 + 0.446\ pH_{ref}$

Region 20: $MILA_{ref} = -1.69 + 0.386\ pH_{ref}$

When the status classification results in a moderate or worse status, as indicated by the parameters showing nutrient impact, it may be necessary to determine if this is a result of anthropogenic eutrophication or whether the lake is naturally rich in nutrients. However, it is not particularly common for lakes to have naturally high nutrient levels. In order to evaluate this, a comparison should be made with the result for the phosphorus assessment criteria. The assessment can further be improved by looking at the impacts/pressures on the water body. Important supporting material for this is source distribution data, historical data, etc. produced in connection with the characterisation. If the evaluation shows that the lake is naturally rich in nutrients, a revision of the reference value for the specific water body should be made on the basis of an expert assessment by the water authority.

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See GG to Chapter 2 11 §

6.11 Comments

Assessment criteria for benthic macroinvertebrate assemblages are based on data acquired using standardized kick-sampling, autumn sampling, sorting of the whole sample (no sub-sampling!), and application of the operative taxonomic list of 517 taxa (see Regulations NFS 2008:1, Annex 1, Table 4.6). A precondition for application of the assessment criteria is therefore that these four criteria are met. Deviations can give a false depiction of the environmental quality.

The BQI index requires special taxonomic expertise as regards the 8 species and 4 families of midges included. Another disadvantage is that the abundances of oxygen-demanding species (for example, Heterotrissocladius species) can be low, which creates a certain risk that these indicator taxa are not collected by standardised sampling comprising five Ekman Grab samples.

In those cases where the classification for ASPT is better than that for BQI it may be because the lake has a shore habitat which has a good status, even if the lake has, for example, high phosphorus content and oxygen-free deep water. Nor is it uncommon that moderately-sized, brown and relatively nutrient-poor forest lakes have a natural oxygen-deficit in the deep water, particularly in the summer. These organic lakes frequently lie relatively well-protected in the forest landscape, resulting in relatively short circulation periods (and oxygenation) and thus long periods of thermal stratification.

Background reports: Johnson, R.K. and Goedkoop, W., 2007. Bedömningsgrunder för bottenfauna i sjöar och vattendrag – Användarmanual och bakgrundsdokument [Background report for benthic invertebrates in lakes and watercourses - User manual and background document]. Report 2007:4. Department of Environmental Analysis Swedish University of Agricultural Sciences (SLU).

7 Benthic macroinvertebrate assemblages in watercourses

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of year?
ASPT	Ecological quality	Once a year	autumn
DJ index	Nutrient impact	Once a year	autumn
MISA	Acidity	Once a year	autumn

7.1 Introduction

Different types of impact, such as eutrophication and acidity, result in a shift in the taxonomic composition of benthic macroinvertebrate assemblages (bottom-dwelling invertebrate animals) in lakes and watercourses, towards a greater dominance of tolerant species. Within Europe, there is a long tradition of using benthic macroinvertebrate assemblages as an indicator of changes in aquatic systems, and many countries have developed their own national metrics. An index weights together information from several indicator taxa (or species) and thereby simplifies classification. In recent years, the development has tended to be towards 'multimetric' indices, in which information from several different individual indices or parameters are combined to a single index. Each of these simple indices shows a correlation with a specific impact and in that way a multimetric index can be constructed from several simple indices, each of which reflects different aspects of the benthic macroinvertebrate communities (e.g. species richness, diversity, function, pollution-tolerance). The Swedish University of Agricultural Sciences (SLU) has developed two multimetric macroinvertebrate indices (respectively, MILA and MISA) for acidity, both for lakes and watercourses; in addition a relatively new multimetric index for detecting the impact of eutrophication on watercourses (the DJ index) has been calibrated.

Index calculations can conveniently be carried out using the software program ASTERICS, which is freely available on the website http://www.aqem.de. Data files can be uploaded to ASTERICS (in Excel or ASCII format) if these contain sampled taxa equipped with AQEM codes (Shortcode, ID_ART or TAXON_NAME). The AQEM codes are described in the English manual (Manual for AQEM European Stream assessment program, version 2.3) and in the taxa lists found on the same website as the programme. The out-file from ASTERICS contains many different indices in use in Europe. Some of these indices are part of these assessment criteria, but others are perhaps in use in other European countries and can for example be used to calculate additional multimetric indices. **Please note however that ASTERICS also gives a classification of indices in accordance with the old assessment criteria (Swedish Environmental Protection Agency report 4913, 1999), and these must therefore not be used.**

7.2 Input parameters

ASPT (Average Score Per Taxon) (Armitage et al 1983[14]) is an index in which different families of benthic macroinvertebrates are scored according to their sensitivity to an environmental impact and which integrates the impact from eutrophication, pollution by oxygen-consuming substances and habitat-degradation such as straightening/clearing (including turbidity).

See REG Annex 1, Section 5.1

The DJ index (Dahl & Johnson 2005[15]) is a multimetric index indicating eutrophication with five simple indices included.

MISA (Multimetric Index for Stream Acidity) (Johnson & Goedkoop 2005[16]) is a multimetric acidity index for watercourses which contains six simple indices.

7.3 Requirements for supporting data

To apply the classification criteria for macroinvertebrate assemblages in watercourses the sampling and analyses must have been carried out in accordance with SS-EN-27828 or by another method which gives equivalent results. The remaining information in Table 6.1 is also recommended for an optimal classification. The species determination must have been made in accordance with the standardised taxonomic list in the regulations (NFS 2008:1), Annex 1, Table 4.6.

See REG Annex 1, Section 5.2

Table 7.1. Overview of sampling methods for benthic macroinvertebrate assemblages in water-courses.

Habitat	Method	Sampling effort*	Mesh-size (mm)	Number of samples	Season
Stream sections	SSEN-27828	20 s x 1 m	0.5	5	Autumn

* relates to SPARK time and SPARK site

[14] Armitage, P.D., Moss, D. Wright, J.F. & M.T. Furse. 1983. The performance of a new biological water quality score system based on macroinvertebrate assemblages over a wide range of unpolluted running-waters. Water Research 17: 333–347.

[15] Dahl, J. & R.K. Johnson. 2004. A multimetric macroinvertebrate index for detecting organic pollution of streams in southern Sweden. Archiv für Hydrobiologie, 160: 487-513

[16] Johnson, R.K. and Goedkoop, W. 2007. Bedömningsgrunder för bottenfauna i sjöar och vattendrag – Användarmanual och bakgrundsdokument [Assessment criteria for benthic macroinvertebrate assemblages in lakes and watercourses - User manual and background document]. Report 2007:4.

7.4 Typology

For the classification of benthic macroinvertebrate assemblages, Swedish lakes are divided into three types, based on Illies ecoregions (Figure 7.1). Table 7.2 shows how they accord with the limnic ecoregions given in the Swedish EPA's Regulations on Typology and Analysis, NFS 2006:1.

Figure 7.1. Illies ecoregions, the Central Plains (14), the Fenno-Scandian Shield (22) and the Borealic Uplands (20).

Table 7.2. Typology for status classification benthic macroinvertebrate assemblages in relation to the ecoregions given in Regulations NFS 2006:1.

Types for benthic macro-invertebrate assemblages	Ecoregion in accordance with NFS 2006:1
Illies Ecoregion 20	Ecoregions 1 and 2 (partial)
Illies Ecoregion 22	Ecoregions 2 (partial) and 3
Illies Ecoregion 14	Ecoregions 4, 5, 6 and 7

7.5 ASPT

7.5.1 Classification of status

ASPT exploits the differences in tolerance among different families of benthic macroinvertebrates and the order Oligochaeta (earthworms). Very sensitive families give high indicator values, while those with high tolerance give low indicator values. The index value for ASPT is a mean value for included taxa and is calculated by adding indicator values and dividing them by the number of included taxa (families).

See REG Annex 1, Section 5.3

Table 7.3. Indicator values for ASPT for different families.

Indicator value	Family
10	Aphelocheiridae, Beraeidae, Brachycentridae, Capniidae, Chloroperlidae, Ephemeridae, Ephemerellidae, Goeridae, Heptageniidae, Lepidostomatidae, Leptoceridae, Leptophlebiidae, Leuctridae, Molannidae, Odontoceridae, Perlidae, Perlodidae, Phryganeidae, Potamanthidae, Sericostomatidae. Siphlonuridae, Taeniopterygidae
8	Aeshnidae, Astacidae, Agriidae, Cordulegasteridae, Corduliidae, Gomphidae, Lestidae, Libellulidae, Philopotamidae, Psychomyiidae
7	Caenidae, Limnephilidae, Nemouridae, Polycentropodidae, Rhyacophilidae (incl. Glossosomatidae)
6	Ancylidae, Coenagriidae, Corophiiidae, Gammaridae, Hydroptilidae, Neritidae, Platycnemididae, Unionidae, Viviparidae
5	Chrysomelidae, Clambidae, Corixidae, Curculionidae, Dendrocoelidae, Dryopidae, Dytiscidae, Elminthidae, Gerridae, Gyrinidae, Haliplidae, Heledidae, Hydrophilidae (incl Hydraenidae), Hydropsychidae, Hygrobiidae, Hydrometridae, Mesoveliidae, Naucoridae, Nepidae, Notonectidae, Planariidae, Pleidae, Simuliidae, Tipulidae (inkl Pediciidae)
4	Baetidae, Piscicolidae, Sialidae
3	Asellidae, , Erpobdellidae, Glossiphoniidae, Hirudidae, Hydrobiidae, Lymnaeidae, Planorbidae, Physidae, Sphaeriidae, Valvatidae
2	Chironomidae
1	Oligochaeta

The ecological quality ratio (EQR) is calculated as follows:

EQR = calculated ASPT / reference value

Reference values and class boundaries are given in Table 7.4.

7.5.2 Reference values and class boundaries

Table 7.4. Reference values and class boundaries for classification of the ASPT parameter I water-courses. SD is the standard deviation for the EQR. Illies ecoregions according to Figure 7.1.

Type	Status	ASPT Ecological quality ratio (EQR)
Illies Ecoregion 14 Central Plains	Reference value	5.37
	Uncertainty (SD of EQR)	0.075
	High	≥0.90
	Good	≥0.70 and <0.90
	Moderate	≥0.45 and <0.70
	Poor	≥0.25 and <0.45
	Bad	< 0.25
Illies Ecoregion 22 Fenno-Scandian Shield		6.53
	Uncertainty (SD of EQR)	0.045
	High	≥0.90
	Good	≥0.70 and <0.90
	Moderate	≥0.45 and <0.70
	Poor	≥0.25 and <0.45
	Bad	< 0.25
Illies Ecoregion 20 Borealic Uplands	Reference value	6.67
	Uncertainty (SD of EQR)	0.027
	High	≥0.90
	Good	≥0.70 and <0.90
	Moderate	≥0.45 and <0.70
	Poor	≥0.25 and <0.45
	Bad	< 0.25

7.6 DJ index

7.6.1 Classification of status

The multimetric DJ index (Dahl & Johnson 2005) for determining the effects of eutrophication on macroinvertebrate assemblages is constructed from five different simple indices. These are (1) the number of taxa of mayflies, stoneflies and caddis flies (Ephemeroptera, Plecoptera and Trichoptera), (2) the relative abundance (%) of Crustaceans (Crustacea), (3) the relative abundance (%) of mayflies, stoneflies and caddis flies, (4) ASPT, and (5) the Saprobic index according to Zelinka and Marvan (1961[17]) . Values for these simple indices must be normalised so that each has a value 1.2 or 3 according to the criteria in Table 7.5.

See REG Annex 1, Section 5.4

[17] Zelinka, M & P. Marvan. 1961. Zur präzisierung der biologischen klassifikation der reinheit fließender gewässer. - Arch. Hydrobiol. 57:389-407.

Table 7.5. Criteria for normalising simple index values to values of 1, 2 or 3 for calculation of the DJ index.

Index	**Criteria**		
Mayflies, stoneflies and caddis flies (Number of taxa)	≤ 5	5 – 12	> 12
% crustaceans (of total abundance)	≥ 22.2	0.5 – 22.2	≤ 0.5
% mayflies, stoneflies and caddis flies (of total abundance)	≤ 10.4	10.4 – 52.1	≥ 52.1
ASPT	≤ 5	5 – 6.3	≥ 6.3
Saprobic index	≥ 2.5	1.9 – 2.5	≤ 1.9
Index_{norm}	**= 1**	**= 2**	**= 3**

The DJ index is calculated by adding the normalised values and can assume a minimum value of 5 and a maximum value of 15.

The ecological quality ratio (EQR) is calculated as follows:

EQR = (calculated DJ index – 5) / (reference value – 5)

Reference values and class boundaries are given in Table 7.6.

7.6.2 Reference values and class boundaries

Table 7.6. Reference values and class boundaries for classification of the parameter DJ index in watercourses. SD is the standard deviation for the EQR. Illies ecoregions according to Figure 7.1.

Type	Status	DJ index Ecological quality ratio (EQR)
Illies Ecoregion 14 Central Plains	Reference value	10
	Uncertainty (SD of EQR)	0.219
	High	≥0.80
	Good	≥0.60 and <0.80
	Moderate	≥0.40 and <0.60
	Poor	
	Bad	< 0.20
Illies Ecoregion 22 Fenno-Scandian Shield	Reference value	14
	Uncertainty (SD of EQR)	0.061
	High	≥0.80
	Good	≥0.60 and <0.80
	Moderate	≥0.40 and <0.60
	Poor	
	Bad	< 0.20
Illies Ecoregion 20 Borealic Uplands	Reference value	14
	Uncertainty (SD of EQR)	0.070
	High	≥0.80
	Good	≥0.60 and <0.80
	Moderate	≥0.40 and <0.60
	Poor	≥0.20 and <0.40
	Bad	< 0.20

7.7 MISA

7.7.1 Classification of status

See REG Annex 1, Section 5.5

MISA is constructed from six different simple indices and responds to acidity. The ínput indices are (1) the number of families, (2) the number of mollusc taxa (Gastropoda), (3) the number of mayfly taxa (Ephemeroptera) (4) the ratio between the relative abundance (%) of mayflies and the relative abundance (%)of stoneflies (Plecoptera), (5) the AWIC index (Acid Waters Indicator Community index; Davy-Bowker et al (2005[18]) and (6) the relative abundance (%) of shredders.

Values for these simple indices must be normalised so that each has a value ($index_{norm}$) between 0 and 10 according to Table 7.7. The normalised values are then added together and re-scaled by dividing the sum of the normalised index values by

[18] Davy-Bowker, J., J.F. Murphy, G.P. Rutt, J.E.C. Steel & M.T. Furse. 005. The development and testing of a macroinvertebrate biotic index for detecting the impact of acidity on streams. Arch Hydrobiol. 163: 383-403.

the number of simple indices included (a mean value) and multiplying this mean value by 10 according to the following:

MILA = 10 * sum $\text{index}_{\text{norm}}/6$

MILA thus acquires a value that can vary between 0 and 100.

Table 7.7. Normalisation of index values ($\text{Index}_{\text{norm}}$) for the six simple indices to values between 0 and 10. In the next step MILA is calculated as a mean value for these normalised indices. ”ASTERICS nomenclature” relates to the software program at http://www.aqem.de.

Index	**ASTERICS- no-menclature**	**$\text{Index}_{\text{norm}}$=10 if the index**	**$\text{Index}_{\text{norm}}$=0 if the index**	**Otherwise $\text{Index}_{\text{norm}}$=**
Number of families	Number of Families	>43	<21	$\frac{\mid NumberofFa\ \ milies\ -21\mid}{\mid 43-21\mid}*10$
Molluscs (number of taxa)	- Gastropoda	>3	<0	$\frac{\mid Gastropoda\ \ -0\mid}{\mid 3-0\mid}*10$
mayflies (number of taxa)	- Ephemeroptera	>16	<3	$\frac{\mid Ephemeroptera-3\mid}{\mid 16-3\mid}*10$
Mayflies/stoneflies (% abundance)*	- Ephemeroptera [%] and - Plecoptera |%]	>7	<0	$\frac{\mid \frac{Ephemeropt\ era[\%]}{Plecoptera\ [\%]}-0\mid}{\mid 7-0\mid}*10$
$\text{AWIC}_{\text{family}}$ index	AWIC Index	>4.6	<3.8	$\frac{\mid AWICIndex-3{,}8\mid}{\mid 4{,}6-3{,}8\mid}*10$
% Shredders	- |%|Shredders	<1.4	>14	$\frac{\mid [\%]Shredders\ \ -14\mid}{\mid 14-1{,}4\mid}*10$

*Please note that the Mayflies/stoneflies (%abundance) index is not included in MISA in those cases where there are no stoneflies in the sample! The absence of stoneflies makes it impossible to calculate this simple index. When there are no stoneflies MISA is instead calculated as a mean value of 5 normalised index values.

MISA shows the benthic macroinvertebrates' response to acidity. It cannot be determined from the MISA acidity classification whether the acidity is natural or of anthropogenic origin.
The ecological quality ratio (EQR) is calculated as follows:

EQR = calculated MISA / reference value

Reference values and class boundaries are given in Table 7.8.

7.7.2 Reference values and class boundaries

Table 7.8. Reference values and class boundaries for MISA. The classes show different stages of acidity and do not relate to status. SD is the standard deviation for the EQR. Illies ecoregions according to Figure 7.1.

Type	**Acidity class**	**MISA** **Ecological quality ratio (EQR)**
Illies Ecoregion 14 Central Plains	Reference value	47.5
	Uncertainty (SD of EQR)	0.135
	Almost neutral	≥0.55
	Moderately acidic	≥0.40 and <0.55
	Acidic	≥0.25 and <0.40
	Highly acidic	<0.25
Illies Ecoregion 22 Fenno-Scandian Shield	Reference value	47.5
	Uncertainty (SD of EQR)	0.135
	Almost neutral	≥0.55
	Moderately acidic	
	Acidic	≥0.25 and <0.40
	Highly acidic	<0.25
Illies ecoregion 20 Borealic Uplands	Reference value	47.5
	Uncertainty (SD of EQR)	0.135
	Almost neutral	≥0.55
	Moderately acidic	≥0.40 and <0.55
	Acidic	≥0.25 and <0.40
	Highly acidic	<0.25

7.8 Management of uncertainty

A mean value of several measurements gives a better and more reliable classification, and if replicate samples are taken then an uncertainty interval in the form of a standard deviant can be calculated for the parameter. In those cases where only one year's data are available, the fixed value for method-bound uncertainty (standard deviant) for the respective parameters and types shown in Tables 6.4, 6.5 and 6.7 may be used. The uncertainty is calculated for reference lakes. Greater variation is expected in polluted lakes, a fact which it is well to keep in mind in making an un-

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certainty assessment. The standard deviation is a measure of precision or the uncertainty associated with classification. In cases where an uncertainty interval around the EQR overlaps any of the class boundaries between high and good status or between good and moderate status, the calculated EQR-value lies very close to a class boundary. That indicates that a reasonability assessment must be made, as described in Section 4.1.1 of the main handbook. See also Section 4.1.2 in the main part of the handbook for more guidance on how to manage uncertainty.

7.9 Weighting of parameters

ASPT shows general ecological quality, the DJ index is specific for eutrophication and MISA shows the impact of acidity. To assess the weighted status for the quality element benthic macroinvertebrate assemblages, the index that has the worst status is used.

7.10 Human impact or natural

If the lake is classified, using MISA, as acidic, highly acidic or extremely acidic, an assessment must be made about whether that is due to anthropogenic acidification or whether the lake is naturally acidic. A more thorough analysis should be made with the aid of the assessment criteria for acidification as shown in chapter 15. The analysis can be further improved by making an assessment of the impact or stress caused by the acidification. The impact of forestry, for example, is important evidence for this and, in addition, data on deposits may be useful if analyses of relatively large areas are to be carried out. If the assessment is that the lake is to some extent naturally acidic, a pH reference value for the water body should be calculated in accordance with Chapter 15. The calculated pH value for the watercourse is correlated with the aid of the line equation in Figure 7.2 to a new reference value for MISA. The measured value for MISA is divided by the new reference value and compared with the class boundaries in Table 7.8.

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See REG Annex 1, Section 5.5.3

The acidity classes, according to the revised reference value or the original classification, are transposed to status classes in accordance with the following:

Almost neutral – High status
Moderately acidic – Good status
Acidic – Moderate status
Highly acidic – Poor or bad status

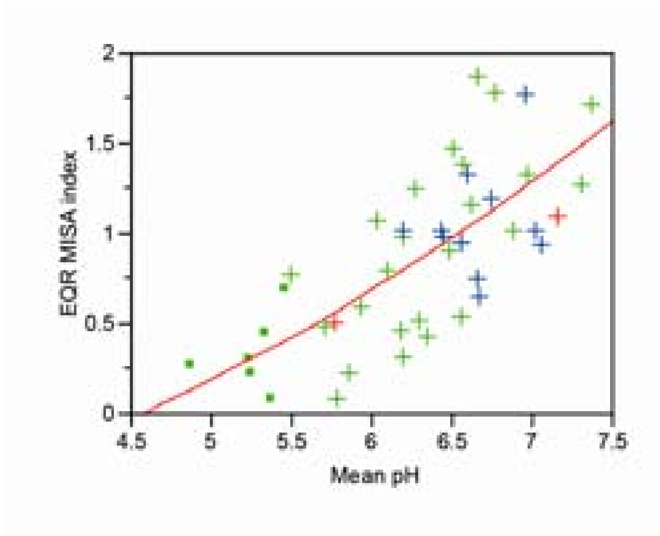


Figure 5.9. Correlation between mean-pH and the MISA index value. Borealic Upland (blue), Fenno-Scandian Shield (green) and Central Plains (red). + = reference. The line equation gives the calculation of MISAref in accordance with the following:
$MISA_{ref} = 1.21 - \sqrt{4.47-0.68\ pH_{ref}}$

When the status classification results in a moderate or worse status, as indicated by the parameters showing nutrient richness/eutrophication, it may be necessary to determine if this is a result of anthropogenic eutrophication or whether the watercourse is naturally nutrient-rich. However, it is not particularly common for watercourses to have naturally high nutrient content. In order to evaluate this, a comparison can be made with results for the assessment criterion for phosphorus. The assessment can further be improved by looking at the impacts/pressures on the water body. Important supporting material for this is the source distribution data, historical data, etc., produced in connection with the characterisation. If the evaluation that the watercourse is naturally rich in nutrients is made on the basis of an expert judgement by the water authority, a revision of the reference value for the specific water body should be made.

See REG Chapter 2 11 §

See GG to Chapter 2 11 §

Background reports: Johnson, R.K. and Goedkoop, W., 2007. Bedömningsgrunder för bottenfauna i sjöar och vattendrag – Användarmanual och bakgrundsdokument [Background report for benthic macroinvertebrate assemblages in lakes and watercourses - User manual and background document]. Report 2007:4. Department of Environmental Analysis Swedish University of Agricultural Sciences (SLU).

8 Fish in lakes

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of year?
EQR8	General impact	At least once	July - August

8.1 Introduction

Regional processes, such as historic events, species-formation and colonisation, determine which fish species are found in a region, while local processes determine which can establish themselves and live together at a given place. To distinguish the effects of human impact (e.g. acidification and eutrophication) one needs to know how different measures of the structure of the fish community also depend on natural preconditions. Geographical situation and a lake's size, depth and shape, pH and nutrient status are some of the natural variables that determine the conditions for the presence of fish in lakes.

Pure fish indices are often 'multimetric indices' of biological integrity. The objective is to obtain a measure of the ecosystem's capacity to maintain a balanced, integrated and well-adapted organism community, whose species composition, diversity and functional organisation is typical of the natural habitat in the region. A composite index is created via indicators/metrics for several different characteristics in individuals, populations and communities. Irrespective of which metrics one measures, a precondition is to know the intervals of the measured values that are to be expected in relatively un-impacted water with high integrity or status.

8.2 Input metrics

EQR8 is based on observed values in eight metrics, all of which are primarily calculated from the catch in standardised fishing with benthic gillnets. If any further species is caught in the pelagic gillnet, it is nevertheless counted in the number of native species. It is a pre-condition of several of the metrics that native species or species within the cyprinid family should be distinguished. The eight metrics are:

1. Richness of native fish species
2. Simpson's Dn (diversity index based on the number of individuals)
3. Simpson's Dw (diversity index based on biomass)
4. Relative biomass of native fish species
5. Relative number of native species
6. Mean weight in the total catch
7. Proportion of potential piscivorous percids (based on the biomass in the total catch)
8. The ratio perch / cyprinids (based on biomass)

8.3 Requirements for supporting data

See REG Annex 1, Section 6.2

1. The natural conditions of the lake must be such that it can harbour fish, an assumption that may be based on historical data or on expert judgement derived from knowledge of conditions in similar lakes.
2. Data from standardised monitoring using Nordic multi-mesh gillnets, in accordance with Standard SS-EN 14 757.
3. Available information about the lake's altitude, area, maximum depth, mean air-temperature values, and position in relation to the highest coast-line.

8.4 Classification of status

The National Board of Fisheries (Institute of Freshwater Research) will be able to carry out calculations for all standardised fish monitoring data, provided readings are delivered digitally to the National Register of Survey Test-Fishing (NORS).

See REG Annex 1, Section 6.3

Step 1) Calculation of lake characteristics:
1. the lake's altitude (m above sea-level)
2. area (hectares)
3. max depth (m)
4. annual mean air temperature values (°C)
5. position in relation to the highest coastline (0 = below, 1 = above)

The altitude is transformed using log10(x+1), while for area and maximum depth log10(x) is used.

Step 2) Calculation of reference values:
Use linear regression models, $Y = a + b_1 * X_1 + \ldots + b_n * X_n$

Where a is intercept and b1 - bn are regression coefficients for lake characteristics ($X_1 – X_n$) according to Table 8.1.

Step 3) Transformation of some of the observed metric values:
metrics 4-5 are transformed using log10(x+1) and for metrics 6 and 8 log10(x) is used.

1. **Richness of native fish species** (Table 8.2)
2. **Simpson's Dn** (diversity index based on the number of individuals) is calculated as $1 / (\Sigma\ Pi^2)$, where P_i = numerical proportion of species i , and all species in the catch are included in the sum.
3. **Simpson's Dw** (diversity index based on biomass): is calculated as $1 / (\Sigma\ P_i^2)$, where Pi = the proportion by weight of species i, and all species in the catch are included in the sum.
4. **Relative biomass of native fish species**: total weight (g) of all native species, number of gillnets.
5. **Relative abundance of native species**: total number of individuals of all native species, divided by the number of gillnets.
6. **Mean weight in the total catch**: all species are included, and their total biomass (g) is divided by the abundance.

7. **Proportion of potential piscivorous percids** (based on the biomass in the total catch): The proportion of potential piscivorous perch is assumed to be 0 for lengths less than 120 mm and 1 for lengths greater than 180 mm. For lengths in between the proportion is calculated as 1 – ((180 – length) / 60). The individual weights of perch are estimated as weight (g) = a * length (mm) ^{b}, where a = 3.377 * 10^{-6}, and ^{b} = 3.205. Each estimated individual weight is then multiplied by the length-dependent proportion of piscivorous perch, as above. The sum of the products is the biomass of piscivorous perch, which is then added to any biomass of pikeperch. Finally the total sum of piscivorous percids is divided by the total biomass of all species in the catch.
8. **The ratio perch / cyprinids** (based on biomass): total weight of perch divided by the total weight of all native cyprinids.

Step 4) Calculation of deviations from reference values (residuals):
For each metric the residual is calculated as the observed value minus the reference value (where appropriate, using the transformed values).

Step 5) Calculation of Z-values:
The residuals are converted to Z-values by dividing them by the metric-specific standard deviation (SD) of residuals in the reference distribution (Table 8.1).

Step 6) Conversion to P-values:
Obtain a double-sided P value for each Z-value through an optional statistics software (in SPPS P = 2 * CDF.NORMAL(-ABS(Z-value),0,1) is used).

Step 7) Calculation of multi-metric fish-index:
Calculate EQR8 as a mean value of the P-values for the 3-8 metrics that are calculable from a given fish-sampling catch.

Step 8) Determine the status class for EQR8 with the aid of the class boundaries in Table 8.3.

Table 8.1. Intercept and regression coefficients for calculation of the fish-metrics' reference values, and the standard deviations (SDresid) required for calculating Z- values.

Metric	Code	intercept	lgHoh	lgLakeArea	lgMaxz	Temp	HK	SDresid
1. Richness of native fish species	niart	-0.410		2.534		0.347	-0.916	1.538
2. Species diversity Simpson's D (number)	S Dn	2.537	-0.460	0.380				0.570
3. Species diversity Simpson's D (biomass)	S Dw	1.223		0.345		0.153		0.753
4. Relative biomass of native fish species	lgWiart	3.666	-0.202	0.121	-0.394			0.202
5. Relative abundance of native fish species	lgNiind	2.171	-0.397	0.081	-0.262	0.044		0.241
6. Mean weight in the total catch	lgMeanW	1.181	0.307			-0.038		0.234
7. Proportion of potential piscivorous percids	andpis	0.057			0.198			0.175
8. Ratio perch / cyprinids (biomass)	lgAbCyW	1.223				-0.186		0.472

Table 8.2. List of known fish species in Swedish freshwater. Fish species considered to be native in Sweden are denoted with X, as well as fish species occurring in lakes within the National Register of Survey Test-Fishing (NORS).

Familj	Vetenskapligt namn	Svenskt namn	Hotstatus	NORS
Petromyzontidae (nejoögon)	Petromyzon marinus	Havsnejonöga	Starkt hotad	
	Lampetra fluviatilis	Flodnejonöga	Missgynnad	X
	Lampetra planeri	Bäcknejonöga	Livskraftig	
Acipenseridae (störfiskar)	Acipenser oxyrinchus	Stör	Försvunnen	
Anguillidae (ålfiskar)	Anguilla anguilla	Ål	Akut hotad	X
Clupeidae (sillfiskar)	Alosa fallax	Staksill	Ej tillämplig	
Cyprinidae (karpfiskar)	Abramis ballerus	Faren	Livskraftig	X
	Abramis bjoerkna	Björkna	Livskraftig	X
	Abramis brama	Braxen	Livskraftig	X
	Vimba vimba	Vimma	Kunskapsbrist	X
	Alburnus alburnus	Löja	Livskraftig	X
	Aspius aspius	Asp	Sårbar	X
	Carassius carassius	Ruda	Livskraftig	X
	Cyprinus carpio	Karp	Inplanterad	X
	Gobio gobio	Sandkrypare	Livskraftig	X
	Leucaspius delineatus	Groplöja	Missgynnad	X
	Leuciscus idus	Id	Livskraftig	X
	Leuciscus leuciscus	Stäm	Livskraftig	X
	Pelecus cultratus	Skärkniv	Ej tillämplig	
	Phoxinus phoxinus	Elritsa	Livskraftig	X
	Rutilus rutilus	Mört	Livskraftig	X
.	Scardinius erythrophtalamus	Sarv	Livskraftig	X
	Squalius cephalus	Färna	Livskraftig	X
	Tinca tinca	Sutare	Livskraftig	X
Cobitidae (nissögefiskar)	Cobitis taenia	Nissöga	Livskraftig	X
Balitoridae (grönlingsfiskar)	Barbatula barbatula	Grönling	Livskraftig	
Siluridae (egentliga malar)	Silurus glanis	Mal	Akut hotad	X
Esocidae (gäddfiskar)	Esox lucius	Gädda	Livskraftig	X
Salmonidae (laxfiskar)	Oncorhynchus clarki	Strupsnittsöring	Inplanterad	
	Oncorhynchus mykiss	Regnbåge	Inplanterad	X
	Oncorhynchus nerka	Indianlax	Inplanterad	
	Salmo salar	Lax	Livskraftig **	X
	Salmo trutta	Öring	Livskraftig	X
	Salvelinus alpinus	Fjällröding	Livskraftig	X
	Salvelinus fontinalis	Bäckröding	Inplanterad	X
	Salvelinus namaycush	Canadaröding	Inplanterad	X
	Salvelinus umbla	Storröding	Livskraftig **	X
	Thymallus thymallus	Harr	Livskraftig	X
Coregonidae (sikfiskar)	Coregonus albula	Siklöja	Livskraftig	X
	Coregonus sp.	Sikar		X
	Coregonus maraena	Älvsik	Livskraftig	
	Coregonus maxillaris	Storsik	Livskraftig	
	Coregonus megalops	Blåsik	Livskraftig	
	Coregonus nilssoni	Planktonsik	Livskraftig	

Familj	Vetenskapligt namn	Svenskt namn	Hotstatus	NORS
	Coregonus pallasi	Aspsik	Livskraftig	
	Coregonus peled	Storskallesik	Akut hotad	
	Coregonus trybomi	Vårlekande siklöja	Akut hotad	
	Coregonus widegreni	Sandsik	Livskraftig	
Osmeridae (norsfiskar)	Osmerus eperlanomarinus	Bracknors	Ej bedömd	
	Osmerus eperlanus	Nors	Livskraftig	X
Lotidae (lakefiskar)	Lota lota	Lake	Livskraftig	X
Gasterosteidae (spiggfiskar)	Gasterosteus aculeatus	Storspigg	Livskraftig	X
	Pungitius pungitius	Småspigg	Livskraftig	X
Cottidae (simpor)	Cottus gobio	Stensimpa	Livskraftig	X
	Cottus koshewnikowi	Rysk simpa	Livskraftig	
	Cottus poecilopus	Bergsimpa	Livskraftig	X
	Triglopsis quadricornis	Hornsimpa	Livskraftig	X
Percidae (abborrfiskar)	Perca fluviatilis	Abborre	Livskraftig	X
	Sander lucioperca	Gös	Livskraftig	X
	Gymnocephalus cernua	Gärs	Livskraftig	X
Pleuronectidae (flundrefiskar)	Platichthus flesus	Skrubbskädda	Livskraftig	

** = lokalt starkt hotad

8.5 Class boundaries

Table 8.3. Class boundaries for status classification of EQR8

Status	EQR8
Uncertainty (SD of EQR8)	0.077
High	≥ 0.72
Good	≥ 0.46 and < 0.72
Moderate	≥ 0.30 and < 0.46
Poor	≥ 0.15 and < 0.30
Bad	< 0.15

See REG Annex 1, Section 6.4

8.6 Management of uncertainty

To make a good classification, it is appropriate to use data from a number of samplings. Several measurements give a more reliable classification and an uncertainty interval in the form of a standard deviation can be calculated for the metric in the water body concerned. It is difficult to state in general terms which years should be co-weighted for the classification. That depends on whether any environmental changes have occurred that can affect the status. If no specific environmental changes have been noted, it is recommended that a mean value of all values from the most recent six-year period should be used. In the event that the number of values is only two, it is recommended to take the latest available value unless it is known that the year in question was extreme as regards e.g. temperature or flux. In cases where data from only one reading (or only a small number) are available, the fixed value for method-bound uncertainty (standard deviation) for EQR8 given in Table 8.3 may be used. The standard deviation gives a measure of how unreliable a classification is. In cases where an uncertainty interval around the EQR overlaps any of the class

See REG Chapter 2. 9 §

See GG to Chapter 2 9 §

boundaries between high and good status or between good and moderate status, the calculated EQR-value lies very close to a class boundary. That indicates that a reasonability assessment must be made, as described in Section 4.1.1 of the main handbook. See also Chapter 4.1.2 for more guidance on how to manage uncertainty.

8.7 Causes of deterioration in status

If EQR8 shows moderate status or worse, an assessment should be made of which impact is the cause of the deterioration in status.

During the development and testing of EQR8, many of the metrics included reacted in opposite directions, depending on whether the impact criterion was acidity or high total phosphorus content (Table 8.4). Six out of eight metrics responded significantly to acidity stress. An equal number of metrics responded to nutritive salt stress. EQR8 had nevertheless significantly better ability to discover the effects of acidity than of nutrient stress.

Table 8.4. Description of which metrics within EQR8 show a significant response to acidity and eutrophy, and whether the response is negative (-) or positive (+).

Metric	Acidity	Eutrophy
1	-	+
2	-	
3	-	+
4	-	+
5	-	+
6		+
7	+	
8		-

The diversity-related metrics (1-3) had significant negative deviations in acidic lakes. High total phosphorus content instead gave positive deviations which were more or less significant. Relative biomass (4) and abundance (5) showed the same type of deviations as the diversity metrics in both groups of impacted lakes. The ratio in the biomass between perch and cyprinids (8) also reacted to both acidity and nutrient salt stress. Here, however, the direction of the deviation was reversed, with significantly higher values in acidic lakes. The mean weight (6) showed no significant response to acidity, but nutrient stress gave positive deviations. The proportion of piscivorous percids (7) was significantly higher in acidic lakes but, contrary to expectation, no significant effect was noted from high total phosphorus content. In summary, acidity and nutrient stress worked in opposite directions. Four of the metrics showed significant differences between non-limed and limed lakes in the reference dataset and the directions were then the same as for acidity stress. If those metrics that give a significant response to acidity show deviation in the direction which indicates acidity impact, it can be interpreted as showing that the lake has acquired a lower status in the classification of EQR8 because of acidic conditions.

As a free-standing complement an assessment can also be made of the presence of acidity-sensitive species and stages (Degerman & Lingdell 1993[19]) . The presence or absence of the most sensitive species can be predicted on the basis both of pH and other acidity-related variables (Holmgren & Buffam 2005[20]) with a precision that is acceptable at least in southern Sweden.

8.8 Human impact or natural

When the status classification results in a moderate or worse status, as indicated by the metrics showing acidity/acidification, it may be necessary to make an assessment whether the deterioration in status is a result of anthropogenic acidification or whether the lake is naturally acidic. A more thorough analysis should be made with the aid of the assessment criteria for acidification shown in Chapter 14. The analysis can further be improved by making an assessment of the acidification impacts/pressures. Deposition data and the impact of forestry, for example, can provide important supporting data about this. If the assessment is that the lake is naturally acidic, the water authority makes an expert assessment of the status for the specific water body.

See REG Chapter 2 10 §

See GG to Chapter 2 10 §

When the status classification results in a moderate or worse status, as indicated by the metrics indicating nutrient richness/eutrophication, it may be necessary to make an assessment whether that is a result of anthropogenic eutrophication or whether the lake is naturally rich in nutrients. However, it is not particularly common for lakes to have naturally high nutrient levels. In order to evaluate this, a comparison can be made with results for the assessment criterion for phosphorus. The assessment can further be improved by looking at the impacts/pressures on the water body. Important supporting data includes the source distribution data, historical data, etc., produced in connection with the characterisation. If the assessment is that the lake is naturally nutrient-rich, the water authority makes an expert judgement of the status for the specific water body.

See REG Annex 1, Section 6.5

See REG Chapter 2 11 §

See GG to Chapter 2 11 §

8.9 Comments

The general evaluations that have been made as regards how the different quality-factor class boundaries relate to one another show that the fish index often resulted in the lowest status and was thus decisive in the classification of ecological status. The need for harmonisation of the class boundaries for the different quality factors has been discussed but the National Board of Fisheries has concluded that the class boundaries for EQR8 should not be adjusted in the present state. The reasons for this are that

- in producing the boundary between good and moderate status for EQR8 the methodology used was the same as in the EU-common project FAME (Fish-based Assessment Method for the Ecological Status of European
- Rivers).

[19] Degerman, E. & Lingdell, P.-E. 1993. pHisces – fisk som indikator på låg pH [pHisces - fish as an indicator of low pH]. Information from Sötvattenslaboratoriet, Drottningholm 1993 (3): 37-54.

[20] Holmgren, K. & Buffam. 2005. Critical values of different acidity indices – as evaluated by fish communities of Swedish lakes. Verhandlungen Internationale Vereinigung für Theoretische und Angewandte Limnologie 29:654-660.

- the quality factors have different sensitivities to different kinds of impact, which makes it natural that they can result in different status classes. For example, fish are affected significantly more by hydromorphological impact than are other quality factors.
- the National Board of Fisheries concludes that changing the class boundaries would require more extensive
- supporting data than is currently available.

One shortcoming is that the connection between the fish metrics and the lake characteristics and the capacity of EQR8 to distinguish between reference lakes and impacted lakes could not be tested on an independent dataset. That should be done when several monitored lakes can be classed in accordance with the same reference-filter. With a larger dataset it also becomes more relevant to divide the impacted lakes into groups affected in different degrees. A more general, but more obvious, limitation is that the classifications become theoretically more uncertain for lakes closer to the boundaries of, and outside, the intervals that were included in the reference material: altitude 10 – 894 m above sea- level, lake area 2 – 4236 ha, maximum depth 1 – 65 m, and annual mean air temperature values -2 – 8 °C. The very large lakes are few in number and each has its own unique conditions. That argues that they require specially adapted assessment criteria. In the current situation an expert judgement may be carried out with the aid of the EQR8 result.

As a guarantee against uncertainty in monitoring individual metrics it is an advantage to have a weighted index with several metrics that respond similarly to the impact. That is also an argument for not making great effort to estimate relevant class boundaries for individual metrics.

In EQR8 only the native fish species are used when the deviations from expected values are calculated out. No attention is paid to translocations of native species, carried out to augment stocks, because it is difficult, not to say impossible, to distinguish their effects in the present state. Probably, the status classification in accordance with EQR8 will be lower if part of the fish community comprises non-native species (e.g. Salvelinus namaycush) since the density of native species then probably becomes lower.

As regards depth, maximum depth was used in the classification of EQR8 despite the fact that average depth is widely prescribed, both nationally and internationally. The reason was purely practical: many more lakes lack estimates of average depth compared to maximum depth. When the monitors come to a lake which has no information on depths, they are requested to make an estimate of the maximum depth (the deepest value they see on the echo-sounder as they travel across the lake). An estimate of the depth must by definition be made if one is to be able to carry out standardised fish sampling. Thus a half good estimate has been taken rather than no estimate at all.

Background reports: Holmgren, K., Kinnerbäck, A., Pakkasmaa, S., Bergquist, B. & Beier, U., 2007. Bedömningsgrunder för fiskfaunans status sjöar – utveckling och tillämpning av EQR8 [Assessment criteria for the status of fish fauna in lakes - development and application of EQR8] Fiskeriverket Informerar 2007:3

9 Fish in watercourses

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of year?
VIX (**W**atercourse **I**nde**X**)	Nutrient impact (incl. bottom sedimentation, overgrowth, low oxygen content), acidity impact, morphological and hydrological impact. VIX indicates older impact if obstacles to migration prevent fish recolonisation. VIX also indicates diffuse negative effects, including deterioration of habitat quality because of migration barriers, agriculture and forestry.	at least once	August - October
VIXsm (page index)	*Clearer acidity and/or morphological impact*	at least once	August - October
VIXh (page index)	*Clearer hydrological impact*	at least once	August - October

9.1 Introduction

The original fish fauna in running water are in practice primarily impacted by three interconnected factors: the immigration history of the fish after the Ice Age, different species adaptability to physical and chemical conditions and biological interactions.

Fish fauna are also impacted by human activity. Environmental disturbances such as acidification, eutrophication, physical interventions, canalisation, hydroelectric dams, forestry, etc have impacted and continue to impact fish just as they do other flora and fauna. The impact differs in strength for different species, depending on their adaptability. Observation of fish fauna in a given locality gives an indication of the extent of impact on the fauna of different environmental disturbances.

Chemical or toxic impact is often, for natural reasons, considerably more drastic for fish as compared with hydrological or morphological impact. In these assessment criteria, a main index is used to show general impact, while an attempt has also been made to show types of impacts using some collateral indices.

9.2 Input parameters

See REG Annex 1, Section 7.1

Six parameters are included in the Watercourse Index (VIX) to measure general impact:

1. Total abundance of trout and salmon
2. Proportion of tolerant individuals
3. Proportion of lithophilic individuals (lithophilic species = spawn on gravel and stone, i.e. hard bottom material)
4. Proportion of tolerant species
5. Proportion of intolerant species
6. Proportion of intolerant salmon fish species that reproduce at the site

The seventh parameter, Simpson's diversity index, is included only in the collateral index VIXh.

7. Simpson's diversity index

The parameters are converted to probability values with the aid of comparisons with expected values. The probabilities indicate how great the chance is for the site to be unimpacted. The mean value of these probabilities constitutes the VIX index.

To provide further demonstration of changes in specific impact factors, two collateral indices have been developed: VIXsm, which indicates more clearly the impact type acidity and/or morphological impact (the same parameters with the same expected direction, depending on the degree of impact for acidity and morphology, respectively) and VIXh for hydrological impact. These separate indices consist of mean values of those parameters which have significantly demonstrated the various impact factors, with the expected direction of the impact on each parameter. The result is, for example, that the collateral index for acidity and morphological impact shows somewhat more clearly the impact of acidity/acidification than the general index does.

- VIX for general impact, parameters 1, 2, 3, 4, 5 and 6
- VIXsm for acidity and/or morphological impact, parameters 1, 3, 5 and 6
- VIXh for hydrological impact, parameters 1, 2, 4, and 7

9.3 Requirements for supporting data

See REG Annex 1, Section 7.2

1. The site must have natural conditions to enable it to harbour fish permanently, an assumption that may be based on historical data or on expert assessment derived from knowledge of conditions in similar watercourses. If there is no local knowledge, it is suggested that the criteria for altitude (less than 800 m above sea level) and catchment area (more than 3 km2) should be used to apply VIX.

2. Standardised electric fishing in accordance with SS-EN 14011.

3. Local variables: the size of the catchment area (category according to Table 9.2), proportion of lake in the catchment area (category according to Table 9.3), minimum distance to the nearest lake upstream or downstream (if the distance is greater than 10 km, 10 km is recorded), height above sea-level (m), gradient (‰, height in metres per section in kilometres on the basis of a relief map, scale 1:50 000), annual mean air temperatures (Meteorological Office maps 1961-1990, showing long-term mean values), and mean air temperatures for July (Meteorological Office maps, long-term mean values), the width of the watercourse (m) and the sampled area (m2). The width of the watercourse and the area sampled are recorded on the occasion of the electric fishing operation.

9.4 Classification of status

The Institute of Freshwater Research will be able to carry out calculations on all standardised electric fishing data, provided that the readings are delivered digitally to

SERS (the Swedish Register of Electric Fishing). Figure 9.1 shows a path diagram for the basis of fish status classification.

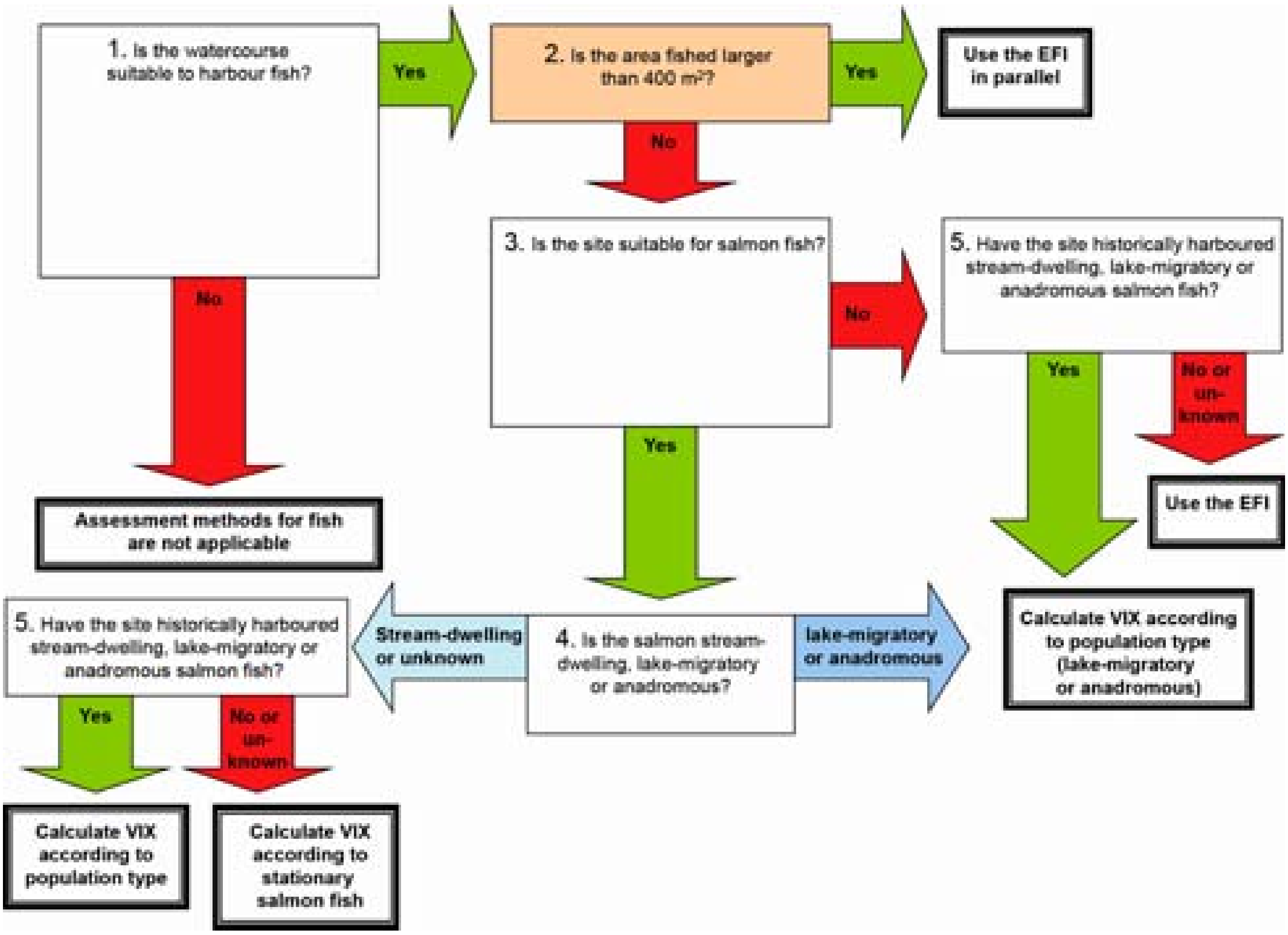


Figure 9.1. Flow chart showing the basis of status classification of fish fauna in running water.

1. First determine whether the watercourse is at all suitable to harbour fish. If it is not, the fish status cannot be classified.

2. When the area fished is sufficiently large it is also convenient to use the European Fish Index (EFI) (FAME consortium 2004[21]) . The preliminary limit is at least 400 m2, which was the mean in the database used for the development of the EFI and is represented by 19% of the electric fishing in the Swedish Register of Electric Fishing (SERS). The EFI is an assessment method developed for European conditions, primarily for major water-courses, which makes it easier to make comparisons with other countries for the same sorts of waters. A follow-up project to FAME is currently in progress, in which the EFI is being developed. One of the objectives is that the revised index, EFI+, should improve the assessment possibilities for major watercourses. According to the regulations (NFS 2008:1), it is classification with the aid of VIX that should in the first instance be applied in Sweden.

3. An assessment is made whether the site is suitable for salmon fish, since VIX is based on several parameters comprising salmon fish. The preliminary criterion is a gradient of 0-50 ‰ for the two smallest catchment area classes <10 and <100 km2. The alternative criterion is flow, classified as 'flowing/rushing', i.e. >0.2 m/s.

4. If the site is suitable for salmon fish, an assessment is made of the original population type of salmon fish (stream-dwelling, lake-migratory or anadromous). The assessment is based on historical information or expert assessment deriving from, for example, information about sites nearby, the topography of the site and abundances and size of sub-yearlings. In the development of VIX the present population types for the site have been used. In certain cases it means that the trout populations at sites in watercourses above a constructed migration barrier have been classed as "stream-dwelling" although they were previously "anadromous". It is possible to use the historical classification, where it is known, and thereby also consider the aspect of artificial migration barriers. A greater abundance of salmon fish is expected in a anadromous or lake-migratory population than in a stream-dwelling population. That will influence the VIX outcome, which in general should give a worse result for previously anadromous populations that have become stream-dwelling. For the time being, the prevailing (present) population types should be used in assessments, but there are possibilities of weighing in historical changes.

5. If the site is no longer suitable for salmon fish, an assessment is made whether nevertheless it could have had stream-dwelling, lake-migratory or anadromous salmon fish. VIX is based on observed values in seven pa-

[21] FAME consortium 2004. Manual for the application of the European Fish Index – EFI. A fish-based method to assess the ecological status of European rivers in support of the Water Framework Directive. Version 1.1. January 2005. (accessible at http://fame.boku.ac.at).

rameters, all of which are primarily calculated from the readings taken in electric fishing. Four of the seven parameters (no. 2-5) are based on functional groups (Table 9.1) which are the same as for EFI (FAME consortium 2004).

Table 9.1. List of existing fish species classified as intolerant, lithophilic, tolerant (FAME consortium 2004) and salmon fish species where the presence of sub-yearlings (0+) indicates reproduction.

Fish species	Intolerant	Lithophilic	Tolerant	Salmon fish species 0+ indicates reproduction
Perca fluviatilis			X	
Aspius aspius		X		
Alburnus alburnus			X	
Cottus poecilopus	X	X		
Blicca bjoerkna			X	
Abramis brama			X	
Lampetra planeri	X	X		
Salvelinus fontinalis	X	X		
Phoxinus phoxinus		X		
Abramis ballerus		X		
Lampetra fluviatilis	X	X		
Leuciscus cephalus		X		
Ctenopharyngodon idella			X	
Barbatula barbatula		X		
Thymallus thymallus	X	X		X
Petromyzon marinus	X	X		
Triglopsis quadricornis		X		
Salvelinus namaycush	X	X		
Cyprinus carpio			X	
Lota lota		X		
Salmo salar	X	X		X
Rutilus rutilus			X	
Oncorhynchus mykiss		X		
Carassius carassius			X	
Salvelinus alpinus	X	X		X
Coregonus sp.		X		
Coregonus albula	X	X		
Pungitius pungitius			X	
Cottus gobio	X	X		
Coregonus peled		X		
Gasterosteus aculeatus			X	
Leuciscus leuciscus		X		
Tinca tinca			X	
Vimba vimba		X		
Anguilla anguilla			X	
Salmo trutta	X	X		X

Calculation of VIX

See REG Annex 1, Section 7.3

Step 1) Transformation of site variables by log10(x+1):

1. size of catchment area (category according to Table 9.2)
2. proportion of lake in the catchment area (category according to Table 9.3)
3. 3minimum distance to nearest lake upstream or downstream (km) where 10 km is the maximum
4. height above sea-level (m),
5. gradient (m per km, ‰)
6. absolute value of annual mean air temperature (long-term mean values, SMHI)
7. mean air temperature for July (long-term mean values, SMHI)
8. width of watercourse (m) measured on the occasion of electric fishing
9. sampled area (m2) on the occasion of electric fishing

For variable 6, mean annual temperature, the transformed value is multiplied by -1 if the original value is <0. Squared values for site variables are also used in certain cases (Table 9.4).

Table 9.2. Limits for categories 1-5 for the site variable Size of catchment area.

Size of catchment area (km^2)	**Category**
<10	1
<100	2
<1 000	3
<10 000	4
>10 000	5

Table 9.3. Limits for categories 1-4 for the site variable *Proportion of lake*. Shows % of total area upstream of the site.

Proportion of lake (% lake area)	**Category**
<1	1
<5	2
<10	3
>10	4

Step 2) Observed values of parameters are calculated from the electric fishing data. The six parameters for the general VIX are:

1. Total abundance of trout and salmon (n individuals per 100 m^2)
2. Proportion of tolerant individuals
3. Proportion of lithophilic individuals
4. Proportion of tolerant species
5. Proportion of intolerant species
6. Proportion of salmon fish species that reproduce

For VIXh (hydrological impact, see Step 7) are added

7. Simpson's diversity index gave a significant reading for hydrological impact and is therefore included only in VIXh. $S = 1 - \Sigma ((n_i / N)^2)$, where ni is the number of individuals (calculated abundance per hectare) of an individual species and N is the total number of individuals.

The values are transformed:
Total abundance of trout and salmon are transformed by $\log_{10}(x+1)$, other parameters that are ratios between 0 and 1 are transformed by $\arcsin(\sqrt{x})$.

Step 3) Reference values of parameters for each electric fishing occasion are calculated by linear regression (Table 9.4) based on transformed values of the site variables. Models for certain reference values are selected in accordance with prevailing population type (Step 1). Calculation of reference values: Use linear regression models, $Y = a + b_1 * X_1 + \dots + b_n * X_n$, where a is the intercept and b_1 - b_n are regression coefficients for site factors ($X_1 – X_n$) in accordance with Table 9.4. The reference values correspond with transformed values in accordance with Step 2.

Step 4) Calculation of deviations from reference values (residuals): For each parameter the residual is calculated as observed value minus reference value.

Step 5) Calculation of Z-values: The residuals are recalculated as Z-values by division by parameter-specific standard-deviation (SD) from the reference material's residuals (Table 9.4).

Step 6) Conversion to P-values: Obtain a P value (probability value) for each Z-value via an optional statistics program. Depending on the expected response in each parameter because of the impact (Table 9.5) either a single-sided P-value is collected for a positive or negative response, or double-sided P-value for response with maximum or minimum for intermediate impact.

Step 7) Calculation of indices: Calculate VIX and the collateral indices VIXsm (acidity and/or morphological impact) and VIXsm (hydrological impact) as a mean value of the P-values for those parameters that are given as relevant (those placed in parenthesis are deleted from the respective indices) in Table 9.5. The P-values must be either single-sided or double-sided depending on the expected response to the respective impact types.

Step 9) Apply class limits valid for the general impact in accordance with Table 9.6 for status classification and, if necessary, the collateral indices VIXsm and VIXh in order to show more clearly acidity, morphological and hydrological impact or recovery after earlier impact.

Table 9.4. Constants for calculation of reference values for fish parameters for VIX with linear regression models. SDresid is the standard deviation for transforming residuals to Z-values.

Site values	1 Total abundance of trout and salmon	2 Proportion of tolerant individuals	3 Proportion of lithophilic individuals	4 Proportion of tolerant species	5 Proportion of intolerant species	6 Proportion of salmon fish species that reproduce	7 Simpson's diversity index	1a POTAMODRAMOUS Density trout and salmon	1b LAKE-MIGRATORY Density trout and salmon	1c OCEANODROMOUS Density trout and salmon	3a POTAMODRAMOUS Proportion of lithophilic individuals
intercept	1.6612	-0.0941	1.4814	-0.3804	1.6743	2.0105	-1.9028	-3.1468	2.0220	2.3956	-2.2575
catchment area class	-1.3934	0.4065				-2.1484	0.3597		-1.7749	-3.1389	
proportion of lake class					-0.4270						
min. distance lake		-0.3690	0.6081	-0.5692	0.1937		0.1356				0.3161
height above sea-level					0.4449			0.6388			3.2391
gradient								0.3440		-0.2581	0.1623
mean.ann. temp.	-0.8184				0.7936			0.7952	1.2151	-1.8217	
mean temp. July							1.3382				
width		-0.0637						-0.2250	-0.3411	0.5216	-0.1498
sampling area				0.1458			0.2702				
catchment area class.^{2}			-0.2838		-0.5358						
proportion of lake class.^{2}		0.1149	-0.2976	0.2662					-0.9735		-0.4396
min. distance lake^2	0.2496	0.2623	-0.3637	0.4539							

Site values	**1 Total abundance of trout and salmon**	**2 Proportion of tolerant individuals**	**3 Proportion of lithophilic individuals**	**4 Proportion of tolerant species**	**5 Proportion of intolerant species**	**6 Proportion of salmon fish species that reproduce**	**7 Simpson's diversity index**	**1a POTAMODRAMOUS Density trout and salmon**	**1b LAKE-MIGRATORY Density trout and salmon**	**1c OCEANODROMOUS Density trout and salmon**	**3a POTAMODRAMOUS Proportion of lithophilic individuals**
height above sea-level.[2]	-0.0436				-0.1601						-0.7175
gradient [2]	0.0970				0.0808		-0.0723				
mean.ann. temp[2]	1.4885	0.1396		0.4312	-1.3832					2.9676	
mean temp. July[2]								1.4363			
sampling area[2]					-0.0629						
SDresid	0.5080	0.1518	0.2756	0.2235	0.3966	0.7186	0.2861	0.4384	0.4435	0.4084	0.2567

Table 9.5. Expected response to general impact and separate impact types for parameters in VIX. Non-significant parameters in brackets. + - indicates that the parameter first increases then diminishes with degree of impact, - + indicates that the parameters first diminish and then increase with the degree of impact. + + indicates that the parameter increases and - - that it diminishes with the impact. For the impact type morphology there were only unimpacted and moderately impacted sites in the database.

	General	**Acidity**	**Nutrient salts/organic load**	**Morphology**	**Hydrology**	**Connectivity**
Total abundance of trout and salmon	- -	- -	- -	-	- -	(- +)
Proportion of tolerant individuals	+ +	(+ +)	+ +	(+)	+ +	- +
Proportion of lithophilic individuals	- -	- -	- -	-	(- +)	(+ -)
Proportion of tolerant species (number of species)	+ +	(- -)	+ +	(+)	+ -	- -
Proportion of intolerant species (number of species)	- -	- -	- -	-	(- +)	(+ -)
Proportion of salmon fish species that repro-duce	- -	- -	- -	-	(- -)	(+ -)
Simpson's diversity index	(+ -)	(- -)	(+ -)	(-)	+ -	(- +)

9.5 Class boundaries

See REG Annex 1, Section 7.4

Table 9.6. Class boundaries for VIX values.

General impact	
Status	**VIX value**
Uncertainty	Calculated in accordance with Ch. 9.6
High	≥ 0.749
Good	≥ 0.467 and < 0.749
Moderate	≥ 0.274 and < 0.467
Poor	≥ 0.081 and < 0.274
Bad	< 0.081

Collateral index	
Index	**Class boundary good - moderate**
VIXsm for acidity	≥ 0.432
VIXsm for morphological impact	≥ 0.430
VIXh	≥ 0.434

9.6 Management of uncertainty

See REG Chapter 2. 9 §

See GG to Chapter 2 9 §

Because of natural variation, the VIX value can vary between sampling events even if there has been relatively little effect on the environment (Table 9.7). The degree of variation is governed by, among other things, natural factors in the surroundings. To make a good classification, it is appropriate to use data from several sampling events. Several measurements give a more reliable classification and an uncertainty interval in the form of a standard deviation can be calculated for the parameter in the water body concerned. In cases where data from only one reading are available, the expected standard deviation for the site may be calculated. The standard deviation gives a measure of how unreliable a classification is. In cases where an uncertainty interval around the ecological quality ratio (EQR) overlaps any of the class boundaries between high and good status or between good and moderate status, the calculated EQR-value lies very close to a class boundary. That indicates that a reasonability assessment must be made, as described in Section 4.1.1 of the main handbook. See also Section 4.1.2 in the main handbook for more guidance on how to manage uncertainty.

Table 9.7. Natural variation in VIX – descriptive values in the distribution of uncertainty measures (observed SD) at 336 sites in the Swedish Register of Electric Fishing (SERS), with data from a minimum of three years. The sites have all been classed as relatively unimpacted (max class 2 of 5) for eutrophication, acidity and, respectively, morphological or hydrological impact.

Observed standard deviation		
Number of electric fishing operations		336
Mean value		0.097
Median		0.088
Minimum		0.000
Maximum		0.384
Percentiles	5	0.024
	10	0.039
	25	0.056
	50	0.088
	75	0.123
	90	0.178
	95	0.206

The Institute of Freshwater Research will be able to calculate the expected variation for all standardised electric fishing data, provided that the readings are delivered digitally to SERS (Swedish Register of Electric Fishing). The site-specific expected variation in the form of a standard deviation (SD) is calculated by the formula:

Predicted SD for VIX-index = 0.1318 + (0.0951 * transformed Proportion of lake in catchment area) + (-0.0039 * transformed, squared Altitude) +
(-0.0348 * transformed Minimum distance to lake) +
(-0.0400 * transformed Sampled area) +
(0.0988 * transformed Catchment Area's size class).

The variation (SD) for VIX is thus expected to increase by the proportion of the lake in the catchment area and the class size of the catchment area, but is expected to diminish with altitude, minimum distance to a lake and sampled area. That is because of the way in which fish communities function in running water. In certain years when the water-level is low, fish from lakes situated nearby can migrate in running water biotopes. The more lakes there are, and the nearer to the closest lake, the greater the chance is of finding more species than are normally found in lakes in running water biotopes.

An alternative way of using the calculated values for expected, site-specific standard deviation is to calculate the probability for the observed electric fishing results to correspond to the classification of VIX for each respective status class. An objective way of determining whether a value represents a boundary case or not is to base the classification on the difference between adjacent probabilities for good and moderate status. If the VIX value falls in the good or moderate class but the difference between probabilities for, respectively, the good/moderate class is less than, for example, 0.1 the VIX value can be considered a boundary case between good and moderate (Table 9.8).

Cumulative probability =
cumulative distribution function for normal distribution (observed value, specified mean value, specified standard deviation).

Cum-P for class high = 1 – (cum-funct-norm (0.749, o-VIX, p-SD)).
Cum-P for class high or good = 1 - (cum-funct-norm (0.467, o-VIX, p-SD)).
Cum-P for class high, good or moderate = 1 - (cum-funct-norm (0.274, o-VIX, p-SD)).
Cum-P for class high, good, moderate or poor = 1 - (cum-funct-norm (0.081, o-VIX, p-SD)).
Cum-P for class high, good, moderate, poor or bad = 1.

The probabilities that the electric fishing results correspond with individual classifications can then be calculated:
P-high: see Cum-P for class high above.
P-good=(Cum-P high or good0 – (P-high).
P-moderate=(Cum-P high, good or moderate) - (P-high) - (P-good).
P-poor=(Cum-P for class high, good, moderate or poor) - 1 (P-high) - (P-good) - (P-moderate).
P-bad =(Cum-P high, good, moderate, poor or bad) - (P-high) - (P-good) - (P-moderate) - (P-poor).
(Cum-P = Cumulative probability, P = probability, p-SD = local-specific predicted standard deviation, cum-funct-norm = cumulative distribution function m.a.p. normal distribution, o-VIX = observed VIX-value.)

Table 9.8. Examples of reports of uncertainty in status classification: Husörenbäcken, Uppströms vägen (main flood area of Bräkneån, Blekinge County), local coordinates 625192-145149, electric fishing during the period 1994-2005. Original parameters, P-values for these and mean value of P-values (VIX) stated. The probability that the observed electric fishing results correspond to the VIX classifications for each respective status class (high, good, moderate, poor or bad) is stated against the background of the expected standard deviation (SD). The highest probability for the respective years is marked in bold type. Boundary cases in the classification of VIX are defined as when the difference between the probability that the classification is, respectively, good or moderate is <0.1, which applied in the year 2005.

År	nölax	nandtol	nandlith	spproptol	sppropint	Kvot	p_VIX_nölax	p_VIX_nandtol	p_VIX_nandlith	p_VIX_spproptol	p_VIX_sppropintol	p_VIX_kvot	VIX	VIX klass	förväntad SD	Sannolikhet för klass hög	Sannolikhet för klass god	Sannolikhet för klass måttlig	Sannolikhet för klass otillfredsställande	Sannolikhet för klass dålig	Skillnad mellan klass god och måttlig	Gränsfall
1994	4,6	0	1	0	1	1	0,06	0,82	0,63	0,84	0,83	0,79	0,66	2	0,13	0,25	**0,67**	0,07	0	0	0,60	
1997	43,0	0	1	0	1	1	0,68	0,82	0,63	0,84	0,83	0,79	0,77	1	0,13	**0,55**	0,44	0,01	0	0	0,43	
1998	62,6	0	1	0	1	1	0,79	0,82	0,63	0,84	0,83	0,79	0,78	1	0,13	**0,60**	0,39	0,01	0	0	0,38	
2000	32,6	0	1	0	1	1	0,57	0,82	0,63	0,84	0,83	0,79	0,75	2	0,13	**0,50**	0,49	0,02	0	0	0,47	
2002	23,6	0	1	0	1	1	0,47	0,81	0,65	0,86	0,85	0,79	0,74	2	0,13	0,46	**0,52**	0,02	0	0	0,50	
2004	10,4	0	1	0	1	1	0,21	0,80	0,66	0,84	0,84	0,79	0,69	2	0,13	0,33	**0,63**	0,05	0	0	0,58	
2005	9,9	0	0,80	0	0,50	1	0,19	0,81	0,08	0,83	0,15	0,79	0,47	2	0,13	0,02	**0,49**	0,42	0,06	0	0,07	X

9.7 Causes of deterioration in status

If the general VIX value shows moderate status or worse, the collateral index and expert assessment should be used to determine which impact has caused the deterioration in status.

All parameters included in VIX show the impact of eutrophication. For morphological and hydrological impact, the separate indices, VIXsm and VIXh respectively, show somewhat more clearly the actual impact factors compared with the general index. VIX is roughly equally effective in distinguishing all different types of impact except connectivity, where the break-point has an unsatisfactorily low probability to classify the respective impacted sites correctly. Thus the index still lacks a clear demonstration of the impact on connectivity for fish in watercourses.

9.8 Human impact or natural

When the status classification results in a moderate or worse status, as indicated by VIXsm, an assessment must be made whether the deterioration in status is a result of anthropogenic acidification or whether the watercourse is naturally acidic. VIXsm contains indicators that can demonstrate both acidity and morphological impact. If one uses VIXsm to show acidity one should therefore be able to exclude a moderate or high morphological impact. A more thorough analysis should be made with the aid of the assessment criteria for acidification shown in Chapter 15 before the status and quality requirement level are established. The analysis can be further improved by making an assessment of the acidification impact/pressure. Important supporting data for this is provided by deposition data, calculations of critical load and the impact of forestry. If the evaluation is that the water body is naturally acidic, the water authority makes an expert assessment of the status of the specific water body.

See REG Chapter 2 10 §

See GG to Chapter 2 10 §

See REG Annex 1, Section 7.5

When the status classification results in a moderate or worse status, as indicated by the parameters showing nutrient richness/eutrophication, it may be necessary to make an assessment whether that is a result of anthropogenic eutrophication or whether the watercourse is naturally rich in nutrients. However, it is not particularly common for watercourses to have naturally high nutrient content. In order to evaluate this, a comparison can be made with results for the assessment criterion for phosphorus. The assessment can further be improved by looking at the impacts/pressures on the water body. The source distribution data, historical data, etc. provide important supporting material, produced in connection with the characterisation. If the assessment is that the water body is naturally nutrient-rich, the water authority makes an expert assessment of the status for the specific water body.

See REG Chapter 2 11 §

See GG to Chapter 2 11 §

9.9 Comments

The parameters were tested against general impact which includes the following factors: eutrophication, acidity, morphological and hydrological impact. The majority of the parameters included in the general index, plus Simpson's diversity index, were intercorrelated. The selection of these parameters was justified by the fact that they could demonstrate different types of impact, and that the accuracy of measurement for a whole index may be assumed to be worse if there are fewer parameters.

The general evaluations that have been made as regards how the different quality-factor class boundaries relate to one another show that the fish index often resulted in the lowest status and it was thus decisive in the classification of ecological status. The need for harmonisation of the class boundaries for the different quality factors has been discussed but the National Board of Fisheries, as the national fisheries experts, have concluded that the class boundaries for VIX should not be adjusted in the present situation.

In establishing the boundary between good and moderate status for VIX the methodology used was the same that applied in the EU-common project FAME (Fish-based Assessment Method for the Ecological Status of European Rivers). There are primarily four reasons, which can in part seem contradictory, why the VIX is in any cases more severe in the classification of status than the index for e.g. periphytic algae and benthic fauna.

- Fish have a relatively large movement area and live in a larger habitat with different requirements for various macro and micro habitats compared with, for example, periphytic algae, which remain attached to a limited habitat. Fish therefore integrate habitat quality to a greater extent. One of the most common disturbances in running water is the deterioration of habitat quality in combination with fragmentation and it is therefore important to be able to indicate this.
- Fish have a longer lifespan than the majority of other aquatic organisms. That means that despite the larger movement area of fish it can take a long time for fish to show recovery after improvements in the environment. Benthic fauna can re-colonise within months, but it can take years for fish. Fish can therefore to a higher degree show a long-term mean value for the condition of the site, while short-lived organisms more clearly indicate temporary variations.
- The re-colonisation and even survival of fish on the site are dependent on the possibilities of migration. Fish rarely re-colonise passively by drift from areas upstream as do, for example, periphytic algae and certain benthic fauna. In the course of a year the majority of individual fish move 100 – 300 m. For older fish it is a question of temporary movements over even greater distances. The water landscape is today greatly fragmented, which has a greater impact on the fish fauna than on periphytic algae and benthic animals.
- In addition to water quality, fish are also dependent on the bottom substrate for spawning and the search for food. The impact of agriculture is not only eutrophication. Despite the fact that water quality can be relatively good and sensitive benthic fauna can be found at the site, sediment transport and site-clearing and canalisation may render the amount of suitable bottom substrate insufficient for fish. Moreover, because of pesticides and insecticides, dense vegetation or lack of stabilising littoral vegetation

and inadequate screening by trees, fish can show even worse results in the classification.

Eels are a tolerant species (acidification, oxygen-deficiency, and eutrophication) and this means that when calculating the VIX, the presence of eels results in a lower status. This has attracted attention as a problem in certain parts of Sweden, where eels can be present in large quantities even in water-courses with good or high status. It is also the case that eels are an endangered species at the European level, as a consequence of fishing and hydro-electric power generation, which means that it may seem remarkable that they lower the VIX classification. Eels must therefore remain in the index as a tolerant species for the time being. The justification for this is that it has been used as a tolerant species in the development of the index and if eels are not used as a tolerant species, the whole index must be re-calibrated. There may in future be a need to review the results for other species in the index, and it is therefore proposed that it be left to future revision of the tools for classification of status. For the West Coast/South Coast Water, where eels are relatively common, it is recommended that any deviant abundance of eels should be considered and taken account of in the final classification.

In conclusion VIX also give indication of more diffuse negative effects on fish due to barriers to migration, agriculture and forestry, than is indicated by the water quality itself and by the hydromorphological quality.

9.10 Example

An example from Husörenbäcken, Uppströms vägen (the main flood area of Bräkneån, Blekinge County).
Site coordinates: 625192-145149, date of electric fishing 14.07.1994.

These observed values are calculated from the electric fishing data:

1. Number of trout and salmon (nölax) – combined calculated abundance per 100 m2 from the electric fishing register.
2. Proportion of tolerant individuals (nandtol) – ratio between 0 and 1, based on calculated abundances.
3. Proportion of lithophilic individuals (nandlith) – ratio between 0 and 1, based on calculated abundances.
4. Proportion of tolerant species (spproptol) – ratio between 0 and 1.
5. Proportion of intolerant species (sppropint) – ratio between 0 and 1.
6. Proportion of salmon fish species with reproduction (Ratio) – ratio between 0 and 1.
7. *Simpson's diversity (Simpson)* S = 1 - Σ ((ni / N) 2), where ni is the calculated individual density per hectare of an individual species and N is the total number of individuals. The diversity measure describes

"evenness" in the distribution between species. *Used only in the collateral index hydrology.*

Instead of the original parameter values the transformed values are used according to the practice:
Tnölax= log10 (nölax+1).
Tnandtol=arcsin (√(nandtol)).
Tnandlith= arcsin (√(nandlith)).
Tspproptol= arcsin (√(spproptol)).
Tsppropint= arcsin (√(sppropint)).
TRatio= arcsin (√(Ratio)).
TSimpson= arcsin (√(Simpson)).

These site variables are used to model the expected values:
Population type of trout: stream-dwelling
Category of catchment area size: 2 (>=10 km^2 <100 km^2)
Category of proportion of lake in catchment area: 3 (>=5% <10%)
Minimum distance to nearest lake upstream or downstream: 1.0 km1
Height above sea-level: 94 m
Gradient: 50.00 ‰ (per mille, height in m per km)
Annual mean temperature: 7°C
Mean temperature for July: 15.5°C
Width of watercourse: 2.0 m
Sampled area: 90 m^2
Site variables are transformed ($\log_{10}(x+1)$) and squared so that the transformed and the transformed, squared values can be used as constants (Table 9.4.)

These site variables are used to model the expected values:
Population type of trout: stream-dwelling
Category of catchment area size: 2 (>=10 km^2 <100 km^2)
Category of proportion of lake in catchment area: 3 (>=5% <10%)
Minimum distance to nearest lake upstream or downstream: 1.0 km1
Height above sea-level: 94 m
Gradient: 50.00 ‰ (per mille, height in m per km)
Annual mean temperature: 7°C
Mean temperature for July: 15.5°C
Width of watercourse: 2.0 m
Sampled area: 90 m^2
Site variables are transformed ($\log_{10}(x+1)$) and squared so that the transformed and the transformed, squared values can be used as constants (Table 9.4.)

These expected values are calculated (Table 9.4.):
Expected number of trout and salmon (nölax) – diminishes with impact.

Separate models for stream-dwelling (applies here), lake-migrant and anadromous trout.
Expected proportion of tolerant individuals (nandtol) – increases with impact.
Expected proportion of lithophilic individuals (nandlith) – diminishes with impact.
Separate model for stream-dwelling trout.
Expected proportion tolerant species (spproptol) – increases with impact.
Expected proportion intolerant species (sppropint) – diminishes with impact.
Expected proportion salmon fish species with reproduction (Ratio) – diminishes with impact.
Expected Simpson diversity (Simpson)—first increases, then diminishes with impact,
used only in the collateral index hydrology.

Example of calculation of expected value (Table 9.4):
Number of trout and salmon, expected value (applies to stream-dwelling trout here):

Expected value =-3.147 +
(-0.225 * transformed Watercourse width) +
(0.344 * transformed Gradient) +
(0.795 * transformed Annual mean temperature) +
(0.639 * transformed Height above sea-level) +
(1.436 * transformed, squared Mean temperature for July).

Expected value = 1.44. This value corresponds with a transformed value (log10(x+1)).

Expected values for the number of trout and salmon must be compared with observed (transformed) values: Observed (transformed) value Tnölax = 0,75. (The actual observed value is 4.6. If the expected value is converted to a non-transformed value, the expected value is 26.5 number of trout and salmon per 100 m^2. The observed abundance is thus one fifth of the expected.)

Example of the calculation of residual, standardised residual and P-value:
Residuals are calculated as the difference between the observed and expected value:
Residual = Tnölax – Expected value.
Residual = 0.75 – 1.44.
Residual = -0.70.
The standardised residual (Z-value) is calculated as the Residual divided by the parameter-specific standard deviation for the residuals (Table 9.4):
Indicator specific (nölax) standard deviation for the residuals = 0.438.
Standardised residual = Residual / 0.438.
Standardised residual = -1.59.

A transformation to probabilities is carried out to enable comparison of all parameters with one another, as equal tools to indicate impact. To obtain probability values between 0 and 1 different transformations are carried out, depending on the expected effect of the impact (Table 9.5). For each observed value it is possible to produce the cumulative probability of obtaining a lower value than the observed value (the area to the left of the value in a normal distribution curve), if the hypothesis is single-sided and the impact is expected to give a negative deviation (applies here). With expected positive deviations with the impact, the probability is that the value will be higher than the observed value (the surface to the right of the value in the normal distribution curve). With double-sided hypotheses the probability is that a value is lower in the case of expected negative deviation or higher the case of expected positive deviation with the impact. The lower the P-value is, the lower is the probability that the site is unimpacted.

The general formula to obtain the cumulative probability of obtaining a value lower than the observed value (in this case standardised residual) using Excel or other programs is:
Cumulative probability
= cumulative distribution function for normal distribution (observed value, specified mean value, specified standard deviation).

To obtain P-value when the hypothesis is single-sided and the impact is expected to give positive deviation:
The probability (P-value) that the site is unimpacted
= 1 - (cumulative distribution function m.a.p. normal distribution (Standardised residual,0.1)).

When the hypothesis is double-sided and increasing impact is expected to give first positive, then negative deviation, or the opposite:
The probability (P-value) that the site is unimpacted
= 2 - (cumulative distribution function m.a.p. normal distribution ((- absolute value of (Standardised residual,0.1)).
When the hypothesis is single-sided and the impact is expected to give negative deviation (applies here):

The probability (P-value) that the site is unimpacted
= (cumulative distribution function m.a.p. normal distribution (Standardised residual,0.1)).

P-value for the parameter Number of trout and salmon is then 0.056. P-values for the other parameters are calculated and VIX is the mean value of them. VIX was in total 0.66 – i.e. good status).

Examples of time series with parameter values and VIX values:

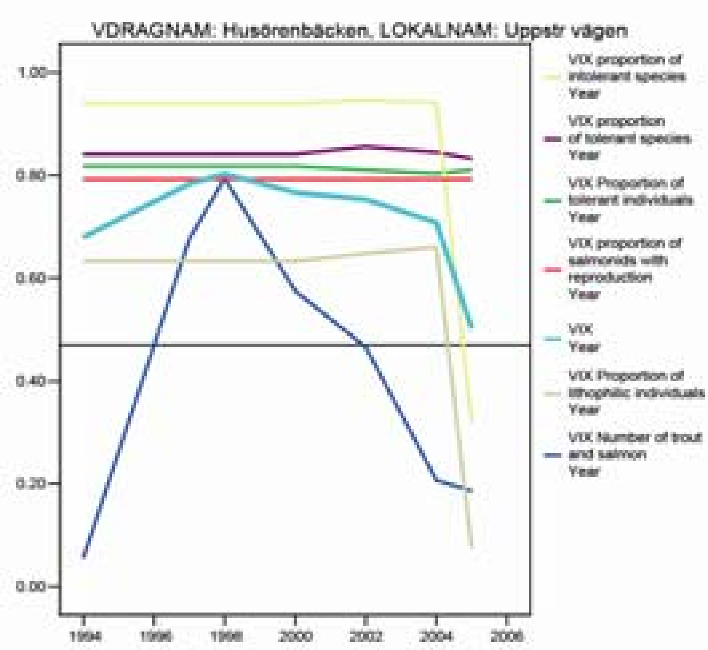


Figure 5.11 Example of time series with parameter values for the different parameters that form VIX.

Husörenbäcken, site Uppströms vägen. VIX (mean value of the six P-values for the parameters) lies for the whole period above the boundary between good and moderate status (0.467). The parameter that varied most is *Number of trout and salmon (nölax)*. In addition to trout, pike were caught at the site in the year 2005. Also the P-values for *Proportion of intolerant species (spproptol) and Proportion of lithophilic individuals (nandlith)* were significantly lower in 2005 than previously, which together with the P-value for *Number of trout and salmon (nölax)* has reduced the mean value (VIX) so that it lies just above the boundary.

Background reports: Beier, U., Degerman, E., Sers, B., Bergquist, B. & Dahlberg, M., 2007. Bedömningsgrunder för fiskfaunans status i rinnande vatten – utveckling och tillämpning av VIX [Assessment criteria for the status of fish fauna in running water - development and application of VIX]. Information from the National Board of Fisheries 2007:5

10 Nutrients in lakes

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of the year?
Total phosphorus	Nutrient impact	4 times/year	Twice in spring, twice in autumn

10.1 Introduction

The concentration of nutrients (primarily phosphorus and nitrogen) in a lake have a substantial impact on the lake's status. The phosphorus supply often regulates primary production considerably. The biomass of phytoplankton (and chlorophyll a) can be mentioned as one of the primary response factors for nutrients in lakes. Other primary producers in lakes are macrophytes and periphyton (here referring to algae growing on submerged objects).

Some lakes can be naturally rich in nutrients. In these assessment criteria, therefore, object-specific reference values for each water body have been developed. These take into account various local factors and chemical parameters and indicate the original phosphorus content in the lake.

10.2 Input parameters

See REG Annex 2, Section 1.1

Phosphorus and nitrogen are among the main nutrients that can cause or affect eutrophication. The main parameter to be used to classify nutrients is total phosphorus (tot-P).

Concentrations of total nitrogen, nitrate and/or ammonium have a significant impact on production regulation especially in relation to total phosphorus in such a way that a low nitrogen/phosphorus ration can benefit nitrogen-fixing blue-green algae and also regulate total production. There are indications that nitrogen can be restrictive in certain nutrient-poor lakes and watercourses (e.g. in the mountains) and in seriously eutrophied lakes and watercourses.

If there are clear indications that the nitrogen content is regulating growth and influencing the species composition in a water body where there is a significant anthropogenic nitrogen load, the water authority can make an expert assessment of a suitable nitrogen concentration as a boundary between good and moderate status for nitrogen. In such cases, the status for the 'nutrients in lakes' quality factor is determined by the status for tot-P or the status for nitrogen content depending on which is worse.

10.3 Requirements for supporting data

See REG Annex 2, Section 1.2

A surface sample is often used in extensive programmes (0.5 m deep). It is beneficial if the sample is taken during late summer (end of July-August). In order to obtain good supporting data for classification, it is recommended that sampling be performed at least four times a year, preferably more often. If only four samples are taken, it is beneficial, when it comes to dimictic lakes, to take them during the two periods of the year when stratification conditions are stable (late winter/early spring and late summer) and during the circulation periods in the spring and autumn. In order to even out intermediate-year variations, it is recommended to do calculations based on three-year periods rather than on annual mean values. In order to be able to apply the assessment criteria for nutrients in lakes, analyses of tot-P must have been performed in accordance with SS-EN ISO 6878 or SS-EN ISO 15681, or using another method that provides equivalent results. If nitrogen is classified, analyses of the different fractions, depending on which are used, must have been performed in accordance with the following standards or using a method that provides equivalent results. Ammonium nitrogen in accordance with SIS 028134, nitrate- and nitrite-nitrogen in accordance with SS-EN ISO 13395, and total nitrogen in accordance with SS-EN ISO 11905-1.

When calculating reference values, data on absorbance, height above sea-level and average depth is needed.

10.4 Classification of status

Predictions using the equations below are used.

See REG Annex 2, Section 1.3

Step 1) Calculating reference values

Calculate the reference value starting from the lake's absorbance, height above sea-level

and average depth.

$\log_{10}$ (ref-P) =1.627 + 0.246•$\log_{10}$AbsF – 0.139• $\log_{10}$Alt – 0.197• $\log_{10}$Depth

Where

ref-P = reference value (total-P µg/l)

AbsF = absorbance measured at 420 nm in 5 cm cuvettes

Alt = the lake's height above sea-level: (m)

Depth = the lake's average depth (m)

<u>Simplified method</u>

If there is no data on the lake's average depth, the following formula can be used to calculate the reference value.

$\log_{10}$(ref-P) = 1.561 + 0.295• $\log_{10}$AbsF – 0.146• $\log_{10}$Alt

Since this is a less reliable method, it may only be used for classification if the measured concentration of tot-P is more than 5 µg/l from any of the class boundaries calculated under Step 2. If the value is too close to a class boundary, the classification will be too unreliable. If this is the case, either an expert assessment must be performed or the lake's average depth must be ascertained so that the original formula for calculating the reference value can be used.

The calculation in the formulae above is based on measurements of absorbance at 420 nm using 5 cm cuvettes. If the measurement has been performed at 436nm per meter, the value of log10(ref-P) must be divided by the factor 15.72 to obtain a value corresponding to a measurement at 420 nm in 5 cm cuvettes. In the given standard, measurements must be performed at 436 nm per meter, but up until now, measurements in Sweden have mostly been performed at 420 nm using 5 cm cuvettes.

Step 2) Classification of status
This is done by dividing the reference value by the observed value. The obtained ecological quality ratio (EQR) is compared to the class boundaries in Table 10.1 and assigned to the right class. To classify a water body as high status, the measured concentration of tot-P must furthermore be less than 12.5 µg/l.

EQR = calculated reference value / observed tot-P

To obtain the class boundaries in µg/l, the reference value is divided by the EQR value for each class boundary respectively.

Class boundary (µg/l) = calculated reference value / class boundary (EQR value)

10.5 Class boundaries

Table 10.1 Status classification of tot-P in lakes.

Status	EQR value
High	≥ 0.7
Good	≥ 0.5 and < 0.7
Moderate	≥ 0.3 and < 0.5
Poor	≥ 0.2 and < 0.3
Bad	< 0.2

See REG Annex 2, Section 1.4

10.6 Comments

Calculations of status should be based on the best possible material. This means a high sampling frequency and a calculation period of at least three years in order to minimise the risk of classification error.

The strong link between total phosphorus content and the biological status of lakes led to the early development of what are known as “trophic scales”. Both nationally and internationally, a concentration of over 25-30 µg tot-P/l corresponds to a trophic level denoting that the lake is in a eutrophied state. According to national experts in the field, the concentration at which a lake’s ecosystem risks undergoing a functional change that accelerates eutrophication is about 25 µg tot-P/l and above. For most Swedish lakes, therefore, a concentration exceeding 25 µg tot-P/l over a long period of time is deemed to be at risk for the abovementioned changes.

Regarding most of the lakes in Sweden that have a phosphorus content of over 25 µg tot-P/l, most of these levels have been increased by human activities and the lakes show a changed biodiversity compared to the original. There may however be certain cases where levels of over 25 µg tot-P/l occur naturally.

The above-described assessment criteria take into account the fact that different lakes have widely varying conditions in the form of reference values and they therefore can tolerate different concentrations of total phosphorus before their biology is disrupted. In certain cases, the calculated boundary between good and moderate status may exceed 25 µg/. In these cases, it is appropriate to make an extra assessment of whether the lake really does have good status or whether it, despite everything, its ecosystem has changed as a result of eutrophication and should therefore be classified as having moderate status.

Using a comparator with coloured discs is an old way of assessing the water's colour. It was mostly used prior to the advent of absorbance measurements. For data where only colour equivalents are available, but where absorbance values are missing, colour numbers can be used if the measured concentration is a long way from a class boundary. The following method is proposed:
The comparator compares the sample to coloured discs that show various degrees of brown colour expressed in mg Pt/ in a telescope. The measurement provides discreet values corresponding to the colour number for one of the glass discs. The method is subject to human error and the margin for error can be assumed to be at least one step in each direction on the assessment scale. If the assessment is for example performed using the scale 20, 50, 100, 150, 200 mg Pt/l etc., and the sample gives a value of e.g. 100 mg/l, we can assume that the real value is between 50 and 150 mg Pt per litre. The reference value is calculated using the two extremities, i.e. 50 and 150 mg/l converted to absorbance (420 nm in 5 cm cuvettes) by dividing by 500. If the two assessments give the same answer regarding whether or not good status has been achieved, the old colour number values can be used and the deviation stated as the interval of the two assessments.

Background reports: Wilander, A., 2004. Förslag till bedömningsgrunder för eutrofierande ämnen [Proposals for assessment criteria for eutrophying substances]. Report 2004:19. Department of Environmental Assessment. Swedish University of Agricultural Sciences (SLU)

11 Nutrients in watercourses

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of year?
Nutrients	Nutrient impact	4 times/year	Twice in spring, twice in autumn

11.1 Introduction

The concentration of nutrients (mainly phosphorus and nitrogen) in a watercourse has a substantial effect on the water's status. Above all, it has a serious effect on primary production. Diatoms can be mentioned as one of the primary response factors for nutrients in watercourses.

Some watercourses can be naturally rich in nutrients. In these assessment criteria, therefore, object-specific reference values for each water body have been developed. The calculation takes into account various local factors and chemical parameters and the original phosphorus content of the watercourse is estimated.

11.2 Input parameters

The parameter on which these calculations are based is the total concentration of phosphorus. Reference values (natural values) are measured preferably in water bodies that are similar to the one being analysed, but they can also be calculated.

See REG Annex 2, Section 2.1

Since there is no general connection between nitrogen levels and effects on biological quality factors in inland waters, no assessment criteria for nitrogen levels have been developed. There are indications however that nitrogen can be restrictive in certain nutrient-poor watercourses (e.g. in the mountains) and in seriously eutrophied watercourses, which is why nitrogen levels might need to be considered in such individual cases. If there are clear indications that the nitrogen content is regulating growth in a water body where there is a significant anthropogenic nitrogen load, the water authority can make an expert assessment of a suitable nitrogen content as a boundary between good and moderate status for nitrogen. In such cases, the mean value of the status for nitrogen and phosphorus is used for the assessment.

11.3 Requirements for supporting data

Samples should be taken once a month. To avoid incorrect calculation values due to e.g. yearly variations, calculations should be made based on three-year periods instead of on annual mean values. In order to perform a classification using the assessment criteria for nutrients in watercourses, analyses of tot-P must have been conducted in accordance with SS-EN ISO 6878 or SS-EN ISO 15681, or using another method that provide equivalent results. If nitrogen is classified, analyses of the different fractions, depending on which are used, must have been performed in

See REG Annex 2, Section 2.2

accordance with the following standards or using a method that provides equivalent results: Ammonium nitrogen in accordance with SIS 028134, nitrate- and nitrite-nitrogen in accordance with SS-EN ISO 13395, and total nitrogen in accordance with SS-EN ISO 11905-1.

11.4 Classification of status

Predictions using the equation below shall be used. It is based on data for the period up until 2002. Regarding arable land areas, it is also based on model calculations from SMED 2007 (PLC5).

See REG Annex 2, Section 2.3

Step 1) Calculating reference values
Calculate the reference value starting from the sampling station's height above sea level, non-marine base cations and absorbance:

$$\log(ref-P) = 1{,}533 + 0{,}240 * \log(Ca^* Mg^*) + 0{,}301 * \log(AbsF) - 0{,}012\sqrt{stationshöjd}$$

Where
ref-P = reference value (total-P µg/l)
Ca*Mg* = non-marine base cations (meq/l)
AbsF = absorbance measured at 420 nm in 5 cm cuvettes
altitude = the height of the sampling station above sea level (m)

Non-marine base cations are calculated thus
Ca*Mg* = Ca + Mg – 0.235•Cl where all concentrations are given as meq/l

<u>Simplified method</u>
If there is no data for base cations and chloride ions for the water body, the following formula can be used to calculate the reference value.

$$\log(ref-P) = 1{,}380 + 0{,}240 * \log(AbsF) - 0{,}0143\sqrt{stationshöjd}$$

Since this is a less reliable method, it may only be used for classification if the measured concentration of tot-P is more than 8 µg/l from any of the class boundaries calculated under Step 2. If the value is too close to a class boundary, the classification will be too unreliable. If this is the case, either an expert assessment must be performed or new sampling has to be undertaken so that the original formula for calculating the reference value can be used.

The calculations above are based on measurements of absorbance at 420 nm using 5 cm cuvettes. If the measurements have been performed at 436nm per meter, the value of $\log_{10}$(ref-P) must be divided by the factor 15.72 to obtain a value corresponding to a measurement at 420 nm in 5cm cuvettes. In the given standard, measurements must be performed at 436 nm per meter, but up until now, measurements in Sweden have mostly been performed at 420 nm using 5 cm cuvettes.

For water bodies whose catchment areas are made up of more than 10% arable land, the reference value is calculated (ref-Pagr) as below. Alternatively, computed reference values are used that also take any retention upstream of the water body into account.

ref-Pagr = (Pagr • Aagr*0.5 + ref-P•(100-Aagr))/100

Where
ref-Pagr = the co-weighted reference value (total-P µg/l) in areas with agricultural land
Pagr = reference value (total-P µg/l) for agricultural land
Aagr = percentage (%) of agricultural land in the area
ref-P = reference value for "non-agricultural land" in accordance with the above
0.5 = a specific factor for weighting in the status classification

Example: In an areas with 30% agricultural land where ref-P is calculated as 20 µg/l and Pagr is 120 µg/l, ref-Pagr = (120*30•0.5+20•(100-30)))/100 = 32

The reference value for agricultural land Pagr are related to the soil type and leaching region and is equivalent to the leakage from an unfertilised, unharvested permanent grass-bank. To calculate ref-Pagr, information about the dominant soil type in the catchment area and which leaching region it belongs to is consequently needed.

Step 2) Classification of status
This is done by dividing the reference value by the observed value. The obtained EQR is compared to the class boundaries in Table 11.1 and assigned to the right class. To classify a water body as high status the measured concentration of total phosphorus has to be less than 12.5 µg/l.

EQR = calculated reference value (ref-P or ref-Pagr / observed tot-P

To obtain the class boundaries in µg/l, the reference value is divided by the EQR value for each class boundary respectively.

Class boundary (µg/l) = calculated reference value / class boundary (EQR value)

11.5 Class boundaries

Table 11.1. Status classification of total phosphorus in watercourses.

Status	**EQR value**	**Measured concentration of tot P**
High	≥ 0.7	and < 12,5
Good	≥ 0.5 and < 0.7	
Moderate	≥ 0.3 and < 0.5	
Poor	≥ 0.2 and < 0.3	
Bad	< 0.2	

See REG Annex 2, Section 2.4

11.6 Comments

Calculations of status should be based on the best possible material. This means that the sampling frequency for status calculations should be based on the best possible material involving a high sampling frequency and a calculation period of three years in order to minimise the risk of classification error.

Using a comparator with coloured discs is an old way of assessing the water's colour. It was mostly used prior to the advent of absorbance measurements. For data where colour numbers are available, but where absorbance values are missing, colour numbers can be used if the measured concentration deviates significantly from a class boundary. The following method is proposed:
The comparator compares the sample to coloured discs that show various degrees of brown colour expressed in mg Pt/ in a telescope. The measurement provides discrete values corresponding to the colour number for one of the glass discs. The method is subject to human error and the margin for error can be assumed to be at least one step in each direction on the assessment scale. If the assessment is for example performed using the scale 20, 50, 100, 150, 200 mg Pt/l etc., and the sample gives a value of e.g. 100 mg/l, we can assume that the real value is between 50 and 150 mg Pt per litre. The reference value is calculated using the two extremities, i.e. 50 and 150 mg/l converted to absorbance (420 nm in 5cm cuvettes) by dividing by 500. If the two assessments give the same answer regarding whether or not good status has been achieved, the old colour number values can be used and the deviation stated as the interval of the two assessments.

Background reports: Wilander, A., 2004. Förslag till bedömningsgrunder för eutrofierande ämnen [Proposals for assessment criteria for eutrophying substances]. Report 2004:19. Department of Environmental Assessment, Swedish University of Agricultural Sciences (SLU).

12 Transparency in lakes (Secchi depth)

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of year?
Transparency	water colour/impact of nutrients	Once a month or once a year	May-October or August

12.1 Introduction

Measurements of Secchi disc depth have an old tradition in limnology and provide a measurement of the water's optical properties and its content of organic material in various forms. The measurements give a simple characterisation of a water's transparency and can suitably be described in time series either seasonally or over a long period of time. The water's transparency is determined both by its own colour, mostly dissolved humic substances, by suspended material such as phytoplankton and detritus and in special cases by inorganic particulate matter (clay particles). Generally speaking, transparency decreases mostly due to the colour of the water, but it can also be significantly reduced by significant nutrient impact, which increases the amount of e.g. phytoplankton. Transparency can be used e.g. to assess the maximum depth at which benthic flora and phytoplankton can survive.

12.2 Requirements for supporting data

See REG Annex 2, Section 3.2

Transparency can be measured either with or without a water telescope. When taking samples, it is important to note down whether a telescope has been used or not since using one generally gives a slightly higher transparency value. Measurements are carried out on open water with a Secchi disc, i.e. a white-painted disk, 25cm in diameter, and sufficiently weighted down so that it hangs perpendicular on a graded line or rope. Measurements should preferably be taken monthly during the vegetation period (May-October) or in August. The calculation period is one year when more than four measurement values are available and three years when measurements are only taken in August. Sampling must be carried out in accordance with standard SS-EN 27027 (part 2, 2.2). More support can be found in the Swedish EPA's survey type: Water chemistry in lakes.

12.3 Classification of status

See REG Annex 2, Section 3.3

Three-year mean values for transparency or the mean value for one year when four or more samples have been taken are preferably used to calculate reference values. When calculating reference values, the values of current light absorbance are used, although a reference value for chlorophyll must also be used to compensate for the effects of eutrophication. Such a reference value must be developed with the aid of assessment criteria for chlorophyll (Sonesten & Wilander 2006).

Step 1) Calculate the reference value for transparency primarily by using transparency values for the lake from periods prior to any impact.

Otherwise, calculate as follows:

$\log_{10}(\text{SDref}) = 0.678 – 0.116*\log_{10}(\text{absF420}) – 0.471*\log_{10}(\text{chla})$

Where
VD = transparency (m)
AbsF420 = absorbance measured on filtered sample at 420 nm, in 5 cm cuvettes.
Chla = reference value for chlorophyll a concentration, chlorophyll µg/l (taken from the assessment criteria for phytoplankton, Chapter 3)

The calculation in the above formula is based on measurements of the absorbance at 420 nm using 5cm cuvettes. If the measurement has been performed at 436nm per meter, the value of log10(ref-P) must be divided by the factor 15.72 to obtain a value corresponding to a measurement at 420 nm in 5cm cuvettes. In the given standard, measurements must be performed at 436 nm per meter, but up until now, measurements in Sweden have mostly been performed at 420 nm using 5 cm cuvettes.

Step 2) Classification of transparency status

EQR is calculated as follows:

EQR = observed transparency / reference value

12.4 Class boundaries

Table 12.1. Status classification of transparency in lakes.

Status	EQR value
High	≥ 0.67
Good	≥ 0.50 and < 0.67
Moderate	≥ 0.33 and < 0.50
Poor	≥ 0.25 and < 0.33
Bad	< 0.25

See REG Annex 2, Section 3.4

12.5 Comments

Measurements of transparency as part of any sampling project are always very useful. However, since samples are most commonly taken in August in most monitoring programmes, these are most suitable for classification. Lakes that are naturally cloudy, e.g. glacier lakes and many flatland lakes that drain clayey areas, have a low transparency. The reference value calculation models do not take this into account. The CRM (Coordinated Recipient Monitoring) lakes that have been tested and that have turbidity values do not show any systematic effect of it.

Background reports: Wilander, A. and Sonesten, L., 2006. Underlag och förslag till reviderade bedömningsgrunder för siktdjup [Supporting data and proposals for revised assessment criteria for transparency]. Report 2006:8. Department of Environmental Assessment. Swedish University of Agricultural Sciences.

13 Dissolved oxygen in lakes

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of year?
Oxygen	Organic material/ nutrient impact	4 times/year	Late winter, spring circulation, summer stagnation, (Aug), autumnal circulation.

13.1 Introduction

Most aquatic animals and many bacteria must have access to oxygen dissolved in the water to survive. Both optimum concentrations and tolerance against low concentrations vary from one animal genus to the next and even between species. Low oxygen levels can occur naturally due to the oxygen consumption from the degradation of organic matter as humic substances in brown waters and in the sediments, especially in shallow lakes. The lowest oxygen levels occur during late summer in a stratified lake's isolated hypolimnion and during late winter if the lake water has been isolated due to ice-cover. The oxygen level depends partly on the oxygen consumption rate and partly on the length of ice-cover/summer stagnation. To differentiate this natural oxygen consumption from the degradation of organic matter due to anthropogenic impact, a model for calculating the natural uptake is used here. If the natural consumption leads to low oxygen concentrations, the anthropogenic impact must be more stringently restricted than if the natural uptake is only small. The status classification takes this into account.

13.2 Requirements for supporting data

Sampling must be conducted in accordance with SS EN 25813 or SS EN 25814, or using another method that provides equivalent results. More support can be found in the Swedish EPA's survey type: Water chemistry in lakes.

See REG Annex 2, Section 4.2

It is suggested that calculation of the deviation from the reference value be based on at least one sampling event at the end of the stagnation periods, i.e. late winter (when there is ice-cover) and in late summer. Furthermore, it is also recommended that samples be taken at the end of the circulation periods; i.e. in late spring and late autumn. The lake should be homothermal when sampling takes place. If the measured value falls below the class boundary for good oxygen content in accordance with Table 13.1, it is appropriate to increase the sampling frequency during the stratification periods (winter or summer) to establish the duration of low oxygen levels. During stagnation, it is suggested that samples be taken in deep water in accordance with the Swedish EPA's survey type: Water chemistry in lakes. Measurements are taken in deep water that is representative of larger water volumes/sediment surfaces and not just in the lake's deepest part/s, since these often only include a limited surface and volume of the lake. The lake's average depth and maximum depth must be known in order to calculate reference values. In addition to measurements of oxygen concentrations, the temperature and colour of the water are measured at every metre between the surface and the bottom.

13.3 Classification of status

To classify oxygen, concentrations of dissolved oxygen (mg O2/l) or oxygen consumption are used. The classification is based on deviations from normal oxygen levels and is divided into two different types of biotope: water where fish fauna consist of "common" warmwater species or water where there are more oxygen-demanding salmonids (salmon-like fish such as salmon, trout, char, rainbow trout and grayling). The natural oxygen concentration at a chosen point in time can be calculated by using the modelled oxygen consumption rate.

See REG Annex 2, Section 4.3

Step 1) Calculate status starting from the minimum value for the year's sampling according to Table 13.1 The table is divided into two types of biotope: water with warmwater biota and water with salmonids that are more oxygen-demanding.

Table 13.1. Status classification of oxygen concentration for lakes based on whether the fish fauna consist of "common" warmwater species or whether there are more oxygen-demanding salmonids (salmon-like fish such as salmon, trout, char, rainbow trout and grayling).

Status	Temp (°C)	Oxygen concentration (mg/l) Warmwater fish	Oxygen concentration (mg/l) Mostly salmonids
High	-	≥ 8	≥ 9
Good	0 – 5	≥7 and < 8	≥8 and < 9
‘	5 – 15	≥6 and < 7	≥7 and < 8
‘	> 15	≥5 and < 6	≥6 and < 7
Moderate	-	≥4 and < 5	≥5 and < 6
Poor	-	≥3 and < 4	≥3 and < 5
Bad	-	< 3	< 3

If the lake's status is moderate or worse regarding its oxygen conditions, the status shall be compared to reference values calculated in accordance with Step 2.

Step 2) Calculation of reference values for oxygen shall primarily be based on measurement values for the lake from the period prior to impact. Otherwise, the reference value is calculated using the formula below.

$$C_t = C_0 - \partial C / \partial t \times t$$

Where:
Ct = calculated reference value for oxygen concentration at sampling event (mg/l)
C0 = oxygen concentration during ice-cover/start of stratification (mg/l)
∂C/∂t = oxygen consumption rate (mg/l per 24 hours) for summer stagnation or ice-cover period as below

t = time between ice-cover or start of summer stratification and the sampling (days). If the stratification time is not known, it can be estimated with the aid of the ice-cover and ice-thaw maps available in the Swedish National Atlas - Climate, lakes and watercourses.

Oxygen concentration during ice-cover/initial stratification is primarily determined by taking measurements at the end of the circulation; i.e. prior to or at the start of thermal stratification in the spring or at the start of ice-cover. Otherwise, it is assumed that saturation is 90% at this time.

The saturation concentration for dissolved oxygen (mg/l) can be estimated based on the water temperature during homothermy (same temperature in the whole water profile) and is calculated as follows:

$$\textit{Saturation concentration} =$$
$$14.603 - 0.4021 \cdot (\textit{Temp}) + \frac{7.68703 \cdot (\textit{Temp})^2}{1\,000} - \frac{69.2575 \cdot (\textit{Temp})^3}{1\,000\,000}$$

Where:
Temp = water temperature at time of measurement (°C)

For summer stagnation (hypolimnion), the reference values for oxygen consumption rate (∂C/∂) are calculated as follows:

$$\textit{Oxygen consumption rate} = \frac{0.3}{\textit{maxdepth} - \textit{transp}} \cdot 1.047^{(temp-20)} + 0.01 \cdot 1.047^{(temp-20)} \cdot abs_{420/5} \cdot 79.4$$

Where:
Oxygen consumption rate (mg/l, 24 hours)
Maxdepth = the lake's maximum depth (m)
Transp = transparency (Secchi disc depth) during the summer (m)
Temp = water temperature in the hyperlimnion (mean value) (°C)
Abs420 = absorbance measured at 420 nm on a filtered sample (5 cm cuvette).
If the thickness of the hyperlimnion can be determined via temperature measurements, this value should be used instead of maxdepth - transparency

Regarding the ice-cover period, the reference values for oxygen uptake rate (∂C/∂) are calculated as follows:

$$\textit{Oxygen consumption rate} = \frac{0.3}{\textit{mean depth}} \cdot 1.11^{(temp-20)} + 0.01 \cdot 1.11^{(temp-20)} \cdot abs_{420/5} \cdot 79.4$$

Where:
Oxygen consumption rate (mg/l, day)
Mean depth = the lake's average depth (m)
Temp = mean value of the lake's water temperature during the winter (°C)
Abs420 = absorbance measured at 420 nm on a filtered sample (5 cm cuvette)

The calculation in the above formula is based on measurements of absorbance at 420 nm using 5 cm cuvettes. If the measurement has been taken at 436 nm per metre, the oxygen uptake rate value must be divided by the factor 15.72 to obtain a value corresponding to a measurement taken at 420 nm in 5 cm cuvettes. According to the specified standard, measurements shall be taken at 436 nm per metre but until now most measurements in Sweden have been taken at 420 nm using 5 cm cuvettes.

13.4 Class boundaries

The observed concentration of oxygen is compared to the class boundaries in Table 13.1. If the value indicates high or good status, this will be the final classification. However, if the value indicates moderate or worse status, an assessment must be made as to whether this is natural or is the result of anthropogenic impact by calculating a reference value in accordance with the description in Section 13.3. The result is compared to the class boundaries calculated using Table 13.1 in order to obtain the final classification.

See REG Annex 2, Section 4.4

Table 13.2. Lower class boundaries for calculating oxygen status. Ct = reference value calculated using formula 1.

Status	Lower class boundary
High	$= 1.19\ C_t - 0.0242\ C_t^2 - 0.418$
Good	$= 1.41\ C_t - 0.0476\ C_t^2 - 1.11$
Moderate	$= 1.08\ C_t - 0.0415\ C_t^2 - 0.202$
Poor	$= 0.674\ C_t - 0.0264\ C_t^2 - 0.577$
Bad	-

13.5 Comments

Oxygen concentrations can be at a critically low level during two periods: During late winter in ice-covered lakes and during late summer in thermally stratified lakes where the hyperlimnion is separated from the surface water and the air. During late winter, fish may have limited chances of finding water with a good oxygen content. During late summer, however, most species can flee from the oxygen-poor profundal areas, although naturally this has an impact on benthic fauna.

Models that divide the water column into several strata can probably give a more detailed picture of the conditions. Models that describe the oxygen conditions in a uniform water column require less extensive data, however, and will therefore be easier to apply. During ice-cover, the lake can in simple terms be described as a unit with the same water chemistry in the entire water column.

Background reports: Wilander, A. and Sonesten, L., 2006. Underlag och förslag till reviderade bedömningsgrunder för syrgas [Supporting data and proposals for revised assessment criteria for dissolved oxygen]. Report 2006:7. Department of Environmental Assessment. Swedish University of Agricultural Sciences.

14 Acidification in lakes

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of year?
Acidification	Acidification	Four times a year	Jan-Dec

14.1 Introduction

Acidification impact refers to the change in water chemistry caused by the anthropogenic deposition of sulphur and nitrogen and the acidifying impact of forestry caused by the uptake of base cations. Acidification impact is classified as deviation from a reference status calculated using the dynamic geochemical model MAGIC.

The classification is based on model calculations performed using the MAGIC model. In cases where there are no MAGIC model calculations, the water body is classified using the MAGIC Library tool. The tool is called the MAGIC Library because it is based on several hundred MAGIC model calculations performed on lakes and watercourses throughout Sweden. The basic idea of the library tool is that water bodies that are currently similar to each other with regard to acidification-relevant parameters have also undergone a similar development in water chemistry over the last hundred or two hundred years and will probably continue to develop in a similar way in the future.

See REG Annex 2, Section 5.1

14.2 Requirements for supporting data

The following data is required to classify a water body using the MAGIC Library.

- Water chemistry parameters; pH, SO4, Cl, Ca, Mg and DOC or TOC, for one year after 1990.
- X- and Y coordinates of the water body in Sweden's national network: RT90.
- The surface area of the lake
- Run-off to the water body in m/year catchment area. This data can be estimated from runoff maps.
- For lakes, the classification shall be done based on median concentration values.

See REG Annex 2, Section 5.2

The supporting data needed to classify a water body using the MAGIC Library depends on the aim of the classification and the degree of impact. If the aim is only to classify whether or not a lake is acidified according to the criteria dpH > 0.4, just a few samples are needed if the acidification impact in the lake is a long way from the limit value. If, on the other hand, the acidification impact is close to the limit value, samples from several years will be necessary in order to obtain a reliable result.

To make a reliable classification of lakes (e.g. if the acidification impact is very close to a class boundary), it is recommended that samples are taken four times a year (in each season) over a three-year period. The classification of acidification impact is done using median concentration values. A simple classification can be done based on four samples taken during a single 12-month period. A comparison is then drawn with nearby reference lakes to determine whether the sampled year deviates substantially from the normal situation. If this is the case, or if there is deemed to be major variation in the chemistry, the sampling is repeated. Autumn samples are recommended for single-sample inventories.

The dissolved organic carbon (DOC) content is used to assess acidification. In most acidification-sensitive lakes, this is equivalent to the content of total organic carbon (TOC). DOC and TOC are the same analysis performed on filtered and unfiltered samples respectively.

14.3 Classification of status

If MAGIC models of water bodies exist, the results for the year 1860 are compared to the current status and the obtained pH change is compared to the class boundaries in Table 14.1.

If there is no MAGIC model for a lake, the acidification impact can be classified from a similar water body in the web-based MAGIC Library tool: http://www.ivl.se/magicbibliotek. The assessment of acidification impact is made by deriving MAGIC results from the water body in the library that is most similar to the water body being classified. This is done on the assumption that if the lakes and watercourse are sufficiently similar, the acidification impact is also comparable. Applying the MAGIC Library to the status data results in an estimated status class.

If the intention is to make a more thorough assessment of the water body, a model can be developed using the MAGIC modelling tool. Guidance on how to do this is given at http://www.ivl.se/magicbibliotek.

In lakes with a short turnover time, an episodic assessment can be made using BDM or pBDM in accordance with the same methodology described for watercourses in Chapter 15.

14.3.1 Classification of limed lakes

Limed water shall be classified after the water chemistry has been corrected for liming impact using the ratio between non-marine Ca and Mg or using a method that produces similar results. The ratio between non-marine Ca and Mg can be derived from measurements taken prior to liming or from a nearby unlimed reference lake.

See REG Annex 2, Section 5.1

To correct limed water in order to classify the acidification impact, Ca*/Mg* for unlimed water (from the time prior to liming or for nearly waters) can be used as supporting data. The error in any individual case can however be too large for the assessment to be sufficiently thorough. This is particularly true when samples are taken from nearby unlimed references. The error in individual cases depends on the variation in time and space of Ca*/Mg*, which in turn depends on local natural conditions. It is hence not possible to give any general recommendation as to what supporting data is necessary. The guidelines presented here are to be used with considerable caution and taking in account local knowledge.

One sample is enough to determine Ca*/Mg* in an unlimed reference object as long as the following conditions have been met: Catchment area > 60 km2, Mg* > 50 µeq/l, alkalinity > 50 µeq/l, Cl < 200 µeq/l and precipitation < 800 mm/year. In other cases, two samples from different seasons under stable flow conditions are recommended. If these results differ by more than 14% or 0.35 with regard to Ca*/Mg*, two more samples are taken and the mean value is used. For samples from the time prior to liming, concentrations of variables that are not affected by liming, such as Mg and Cl, can be compared to later samples to assess how representative the samples are. Results from 2000-2002 should be avoided since Ca*/Mg* often deviated from the norm during this period.

Use data from the period prior to liming first of all. If there is no analysis of Cl, the concentration can be estimated from later measurements, provided that the marine proportion only constitutes a small part of Ca and Mg.

If there are no samples prior to liming, data from upstream points unaffected by the liming shall be used first of all. The sampling point should then represent a large part of the limed object's catchment area, which does not deviate from the rest in terms of its land use and geology.

In most cases, we are forced to use nearby reference waters that are not in the limed object's catchment area. Several objects should in this case be used and the variation in Ca*/Mg* between the reference objects must be taken into consideration. A rule of thumb is to take results for three objects or two references if they produce the same results, and then make two to three separate assessments of acidification for the limes object using Ca*/Mg* from the reference objects. If the assessments produce similar results with regard to acidified/not acidified according to MAGIC/MAGIC Library, the assessment can be deemed reliable. Otherwise, an expert assessment of the reference water's usability will be required; if more samples are necessary or if there is too much uncertainty to be able to classify the acidification impact for the limed object. More information on this methodology is described in Fölster and Wilander (2005[22]) .

[22] Fölster, J. and Wilander, A.: 2005, 'Försurningsbedömning in kalkade vatten med kvoten Ca*/Mg*. [Acidification assessments in limed waters using the Ca*/Mg* ratio. Department of Environmental Analysis, SLU. Report 2005:3'.

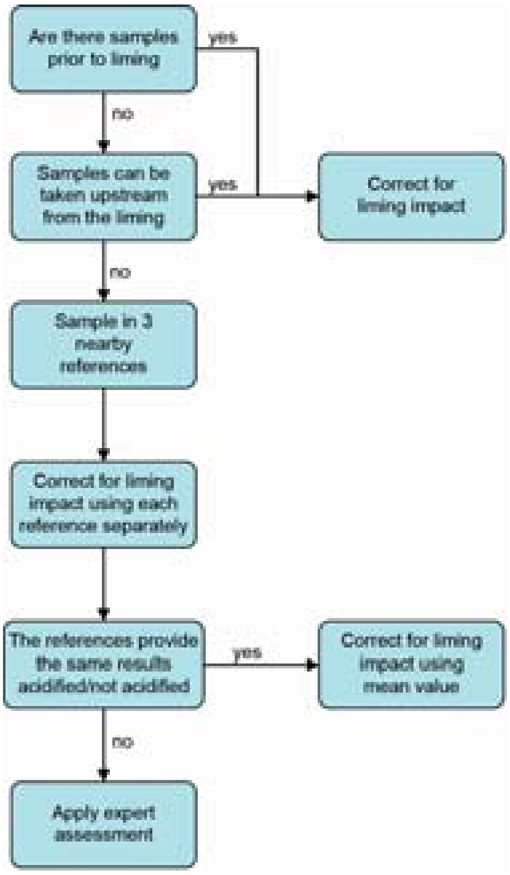


Figure 14.1. Schedule for how corrections for liming can be made

14.4 Reference values and class boundaries

The reference status is defined as the ANC in the pre-industrial era (the year 1860) calculated using the MAGIC model/MAGIC Library (reference values for pH from ANC can be calculated using the calculation in Section 14.4.1). To determine the deviation, the reference value is compared to the current value of ANC, ANCt. So that the deviation assessment reflects the biological impact, the ANC-change is converted into an equivalent change in pH using a chemical equilibrium calculation. This conversion is done because pH is more strongly linked to acidification-sensitive organisms than ANC.

The deviation from the reference value is expressed a change in pH assuming unchanged concentrations of naturally organic matter and a constant carbon dioxide pressure. Water bodies with a pH change >0.4 do not achieve “good status” (Table 14.1). The boundary between high and good status lies within the margin for error of the assessment tool. A lower limit value, 0.2 pH units, applies during the spring flood episode.

See REG Annex 2, Section 5.3

Table 14.1. Classification of status.

Class	pH-change	Status
1	<0.2	High status
2	0.2 – 0.4	Good status
3	0.4 – 0.6	Moderate status
4	0.6 – 0.8	Poor status
5	>0.8	Bad status

14.4.1 Calculating reference values for pH from ANC

Calculating reference values for pH when the change in ANC has been estimated using the MAGIC Library tool. This is only interesting when an adjustment of the reference values or class boundaries is to be done for the relevant biological parameters (see the Section on Human impact of natural under each biological quality factor respectively). A calculation model in Excel format to facilitate the calculation below can be accessed on the Swedish EPA's website (the model is described below).

See GG to Chapter 2 11 §

PH model

Input data:

ID: Numerical value giving the identity of the sampling station.

DOC: The concentration of carbon measured as DOC (filtered sample) or TOC (unfiltered sample) in mg/l

ANC: Acid Neutralising Capacity in the lake or watercourse for which the pH reference value is to be calculated. Calculated thus:

ANC = Ca+Mg+Na+K-SO4-Cl-NO3 (eq 1)

with all units in µeq/l. If the concentration of NO3 is deemed negligible, it can be omitted from the calculation.

deltaANC: Change in ANC in the matched lake or watercourse in the MAGIC Library measured in µeq/l.

Output data:

ID: Same as above.

pH reference: Calculated reference for pH. The reference value for ANC is calculated as the sum of ANC for the lake (or watercourse) to be classified and deltaANC for the lake in the MAGIC Library that is matched against the lake or watercourse to be classified:

ANCref = ANC + deltaANCMAGIC-library (eq 2)

The reference value for pH is then calculated from ANCref as below.

Calculation of reference pH from ANCref and DOC

The model for calculating pH from ANC and DOC has been developed at SLU by Stefan Köhler et al (1999).

The conversion of ANC to pH is based on the law of electroneutrality, namely that the sum of all the cations is equal to the sum of all the anions. For a normal natural water, the electrical charge balance, in simple terms, is as follows:

BC + H+ = SAA + HCO3- + CO32- + A- (eq 3)

where BC = base cations, SAA =strong acid anions, A- = organic anions (all concentrations in eq/l). BC and SAA can be replaced by ANC which is equal to BC - SAA giving:

ANC = HCO3- + 2CO32- + A- - H+ (eq 4)

HCO3- and CO32- is calculated from pH using known equilibrium equations for the carbonate system and a carbon dioxide pressure set at 4 times the CO2 pressure in the air (this constitutes the mean value of the CO2 pressure in 89 ion-poor reference lakes 1998-2002).

A- is calculated from pH and the DOC concentration using a triprotic model (Hruska et al, 2001).

The relationship between ANC and pH in accordance with the above is described using the following equations:

(eq 5-14)

H^+ = 10-pH
OH^- = 10-14 / H^+
HCO_3- = 10-6.35 * 10-1.45 * $_{pCO2}$ / H^+
CO_3^2- = HCO_3- * 10-10.33 / H^+
Atot = 10-6 * DOC * 10.2 / 3
H_3A = Atot / (1 + 10-3.04 / H^+ + 10-3.04 * 10-4.51 / $(H^+)^2$ + 10-3.04 * 10-4.51 *10-6.46 / $(H^+)^3$)
H_2A- = 10-3.04 * H_3A / H^+
HA^2- = 10-4.51 * H2A- / H^+
A^3- = 10-6.46 * HA^2- / H^+
ANC (µeq/l) = 10 6 * (OH^- + HCO_3^- + 2 * CO_3^{2-} + H_2A^- + 2 * HA^{2-} + 3 * A^{3-} - H+)

To calculate the reference value for pH, it is assumed that the TOC concentration and CO^2 pressure was the same during the reference period as it is now. To obtain pHref, we insert a low pH value, e.g. pH = 3, in equation 5 and calculate an ANC. If ANC is lower than ANCref, a new ANC value is calculated according to equation 5 using a slightly higher pH value, e.g. 3.01. This is repeated until the ANC value calculated from equation 5 gets closer to ANCref. The pH value used in equation 5 that gives the ANC value closest to ANC_{ref} is pH_{ref}.

The calculation of pH does not take into account the presence of inorganic aluminium, Ali, correctly. If there are high concentrations of Ali in the water, the model will produce a pH value that is too low. Since the concentrations of Ali were seldom high during the pre-industrial era, this probably has no bearing on the calculation of the pH reference value.

14.5 Comments

Meteorological variations are reflected in the chemistry of the surface water, where e.g. spring flood episodes constitute the extreme situation. This is particularly true in watercourses, but also in lakes. The lake's turnover time then has a major bearing on the extent of the variation. In wet conditions, the groundwater level is high. The residence time in the soil is then shorter, resulting in the water being diluted with regard to buffering decomposition products. This results in all acidity parameters having lower values. High groundwater levels also result in more superficial flow paths, which often results in higher concentrations of organic matter. This lowers both the alkalinity and pH, but not ANC. In addition, high concentrations of organic matter can often lead to a higher CO^2 pressure, which reduces pH, but not ANC and alkalinity. The CO^2 pressure in lakes increases after they freeze over, especially in humic lakes.

Lakes with a pre-industrial pH of under 6 are normally said to be naturally acidic. Since lake chemistry is so variable, especially pH in ion-poor lakes, estimates of the proportion of naturally acidic lakes in a group of lakes depends on the hydrological conditions when samples are taken. In the National Inventory if 1995, for example, only half as many lakes were naturally acidic compared to the inventory in 2000, when conditions were much wetter. If we look at how pH varies in 189 lakes from the "reference lakes" in Sweden's national environmental monitoring programme, we can see that nearly half the lakes may have pH values of both over and under 6, depending on when samples are taken.

Not just their acidity but also the acidification impact can vary depending on the meteorological conditions. This is particularly apparent during episodes of temporarily increased sulphate levels in connection with the spring thaw, or when water flow rises after a drought causing the groundwater level to drop and sulphate to be released as a result of oxidation. In addition, a specific acidification pressure has a greater pH-reducing effect when the water's buffering capacity is diluted during inundation. The limit value for acidification impact of 0.4 pH units based on median values includes a margin to cover occasions when the acidification impact is temporarily higher. The exception is the spring flood episodes in northern Sweden, where the acidification impact is assessed using the BDM or pBDM episode model.

When acidification is classified using the MAGIC Library, the water body to be classified is matched against water bodies in the library that have already been assessed using the MAGIC model. The water body to be classified is deemed to be just as acidified with respect to dpH as the matched lake. If data from different years from one water body is used in the classification, the water body to be classified can be matched against different water bodies for different years. The reason for this is that the water chemistry data in the MAGIC Library is made up of modelled data where yearly variation has been evened out, whilst the measured chemistry partly reflects the climatic conditions during the period before the sample was taken. In a wet year, therefore, a water body to be classified can be matched against a more acidified water body in the library compared to a drier year. This is a useful aspect of the assessment tool as it reflects the fact that a water body can be expected to be more affected by acidification during a wet year. One disadvantage of matching a water body against different water bodies for different years is that the temporal development of dpH will not be constant. When evaluating time series, therefore, it is recommended that the temporal development be described using e.g. ANC, whilst the acidification classification in the MAGIC Library be done based on medians stretching over several years for water chemistry.

Background reports: Fölster, J. (2007). Förslag till bedömningsgrunder för försurning i sjöar och vattendrag [Proposal for assessment criteria for acidification in lakes and watercourses]. Department of Environmental Assessment. Swedish University of Agricultural Sciences (SLU). Report 2007:9. 28.

15 Acidification in watercourses

Parameter	Primarily shows the effects of	How often do measurements need to be taken?	At what time of the year?
Acidification	Acidification	6 times/year	Jan-Dec

15.1 Introduction

See REG Annex 2, Section 6.1

To classify at base flow, the same method as specified in Chapter 14 on lakes shall be used except when pH is between 4.6 and 5.4 during the episode. In such cases, class 2 (0.2 -0.4 pH units) shall also be considered moderate status. In this pH interval, even such small pH changes can be crucial and lead to an increase in toxic inorganic aluminium levels.

Watercourses north of Limes Norrlandicus and in lakes with short turnover time shall be classified using the episode model BDM, Boreal Dilution Model, when there is a risk of episodic acidification during the spring flood. If there are no measurements during the spring flood, episodic acidification can be estimated based on the base flow chemistry using the pBDM model (one point Boreal Dilution Model). BDM and pBDM are available online at: http//:ccrew.sek.slu.se/bdm.

15.2 Requirements for supporting data

See REG Annex 2, section 6.2

For base flow, see Chapter 14, the classification shall be done based on flow-weighted mean values.

The following data is needed to classify acidification impact during the spring flood episode in northern Sweden (Norrland):

- The following data is needed for BDM: ANC and TOC or DOC during base flow and in time series during the spring flood.
- The following data is needed for pBDM: ANC and TOC or DOC during winter base flow.

In order to carry out a reliable classification of the acidification impact during base flow in watercourses, monthly sampling over a three-year period is recommended. A simple classification can be done based on six samples over a 12-month period. A comparison is then drawn with nearby reference stations to determine whether the sampled year deviates from the normal situation. If this is the case, or if there is deemed to be major variation in the chemistry, the sampling is repeated.

15.3 Classification of status

For base flow, see Section 15.1 and 14.4.

For watercourses north of Limes Norrlandicus and in lakes with short turnover times, the acidification impact during the spring flood shall be classified using the episodic model BDM, Boreal Dilution Model (http://ccrew.sek.slu.se/bdm) If there are no measurements during the spring flood, episodic acidification can be estimated based on the base flow chemistry using the pBDM model (one point Boreal Dilution Model).

15.4 Reference values and class boundaries

For base flow, see Chapter 14, Table 14.1. A lower limit value of 0.2 pH units is applied during the spring flood episode, however.

See REG Annex 2, section 6.3

15.5 Comments

The acidification status can vary considerably during the year, especially in water-courses, and it is often the status in the most acidic conditions that sets the boundary for species composition. The acidification impact does not vary quite as much however and the limit value for acidification impact, 0.4 pH units, is set in order to take the variation during the year into account. The acidification impact can therefore be classified based on the average situation, which is preferably from a follow-up perspective. The exception to this is during episodes when the natural acidification impact increases temporarily. An example of this is during the spring thaw, when some acidic deposition enters the watercourses with only minor buffering by the soil. In southern Sweden, acidification episodes mostly occur when the flow increases after drought. Episodic acidification has declined as a result of less deposition and is no longer considered to be a major problem. It is expected to decrease further as a result of this recovery.

Background reports: Fölster, J. (2007). Förslag till bedömningsgrunder för försurning i sjöar och vattendrag [Proposal for assessment criteria for acidification in lakes and watercourses]. Department of Environmental Assessment. Swedish University of Agricultural Sciences (SLU). Report 2007:9. 28.

16 Specific pollutants in lakes and watercourses

16.1 Introduction

In the Swedish Ordinance on Water Quality Management and the European Water Framework Directive (WFD), toxic chemical substances in the water environment are dealt with in two different categories. Substances that have common EU environmental quality standards (above all the priority substances but also a number of other substances regulated by EC fishing waters and crustacean directives) are included in the classification of surface water chemical status, see also Chapter 5 in the main handbook. In addition to these, specific pollutants shall be classified as one of the physico-chemical quality factors when classifying ecological status.

See REG Annex 2, Section 7

What these pollutants are may vary from one water body to the next depending on different types of impact. Annex V of the Water Framework Directive (WFD) states that the substances to be classified are any pollutants that are discharged into bodies of water in significant quantities.

16.2 Choice of specific pollutants

What is meant by a substance being discharged in significant quantities? In the EU Guidance no 3 (Analysis of pressures and impacts[23]) the concept of discharge is interpreted in a broad sense. It covers discharges from point sources in the river basin, leakage from diffuse sources and e.g. atmospheric deposition from other areas. One should therefore consider all the possible pathways by which the pollutant can reach the water body. The Swedish EPA interprets "significant quantity" as a quantity of a substance that can prevent the biological status/potential from being fulfilled by 2015.

The water authorities shall classify the specific pollutants discharged into the water body. Discharged substances are identified with the help of the supporting data produced when assessing impact (See the Handbook for Typology and Analysis). The EU Guidance describes the procedure for selecting the specific pollutants in each river basin and in particular water bodies. Here is a summary of the most important steps.

1. Starting-point

The indicative list of the main pollutants set out in Annex VIII of the WFD can be the starting-point of the selection process.

[23] Common Implementation Strategy for the Water Framework Directive (2000/60/EC) Guidance no 3 Analysis of pressures and impacts, produced by working group 2.1 – IPRESS, 2003

2. Screening of information
A screening of all available information on pollution sources, impacts of pollution and production and usage of pollutants in order to identify those pollutants that are being discharged into water bodies in the river basin district.

2a. Collation of data/information
Data from:
- Sources - Production, industrial processes, usage, treatment, emissions
- Impacts - Change in the occurrence of pollutants in the water body (water quality monitoring data)
- Pollutants - Intrinsic properties of the pollutants affecting their likely pathways into the water environment.

Information from existing programmes/registers, e.g.:
- Swedish Pollutant Release and Transfer Register (PRTR)
- C-EMIR (emissions from point sources)
- MIFO (contaminated areas)

2b. List of pollutants
Assessment of information collated under Step 2a will result in a list of those pollutants identified as being discharged into water bodies in the river basin district. Pollutants for which there is adequate confidence that they are not being discharged into water bodies in the river basin district may be excluded from further considerations.

3. Assessment for relevance
All the pollutants being discharged in the river basin district have been identified in Step 2. Step 3 tests which of these are relevant. In other words, those pollutants that are likely to cause, or are already causing, harm to the water environment. This will depend on the intrinsic properties of the pollutants, their fate and behaviour in the environment and the magnitude and form of their discharges. Selection should ideally be based on an assessment of the ecological relevance of the concentrations estimated for the pollutant or its metabolites in the water body. However, effect data or a modelling of critical loads may also be relevant in the selection process.

3a. Data on concentrations and loads
Obtaining data through monitoring and/or modelling.

3b. Comparing concentrations with threshold values
Pollutants identified under Step 2 may be excluded where their concentrations are estimated to be lower than the most relevant critical value such as estimated LC50, NOEC, PNEC, EQS or model estimations for e.g. critical load.

Natural background concentrations of non-synthetic pollutants (mostly metals) may exceed EQS without them necessarily being considered relevant.

Potential bioaccumulations of the pollutant in sediment or biota should be considered.

4. Safety net

A safety net is needed to ensure that pollutants that may be environmentally significant are not incorrectly excluded from the list of specific pollutants during Step 3. For example, the safety net should consider;

- whether a number of small (individually minor) pollution sources may be expected to have a significant combined effect,
- whether there is a trend indicating the increasing importance of a pollutant, even though the EQS is not currently exceeded, and
- whether pollutants are present that have similar toxic effects and hence via additive or synergetic effects may cause significant impacts.

5 Final outcome

The final outcome is a list of specific pollutants relevant to a river basin district or to particular water bodies within a river basin district.

It is therefore the water authorities that select the relevant specific pollutants for each water body. Class boundaries should be established for these pollutants in accordance with Annex V of the WFD so that the status of the specific pollutants quality element can be established.

16.3 Classification of status

Class boundaries should be established for water, sediment or biota matrices depending on which of the matrices the most sensitive organism is exposed through. If ecotoxicological studies indicate that aquatic organisms are affected at the lowest concentrations of a pollutant, class boundaries should be established for water. If sediment-dwelling organisms are the most sensitive, the class boundaries should instead be established for sediment and if it is birds, mammals or humans who feed of the water environment (e.g. fish or crustaceans) and who, via secondary poisoning, react at the lowest concentrations, class boundaries should be established for biota.

The water authorities shall establish class boundaries between high and good and between good and moderate status in accordance with the normative definitions in Annex V Tables 1.2.1 - 1.2.2 in the WFD. How to set the boundary between good and moderate status is described in detail in Annex V, Section 1.2.6 of the WFD.

To help when establishing class boundaries, the water authorities can use the values that have already been established in accordance with the methodology described in Annex V of the WFD. As an example, there is a report entitled “Proposals for limit values for specific pollutants - support for the water authorities when classifying status and establishing environmental quality standards”, in which the Swedish Chemicals Agency, on behalf of the Swedish EPA, have drawn up proposals for limit values which the water authorities can use as class boundaries for a number of chemical substances that are deemed problematic in certain areas in Sweden (available as a background report at www.naturvardsverket.se).

16.4 Classification of status

When classifying the status of specific pollutants, the measured concentration in the water, sediment or biota in the water body of the substances identified as being discharged in significant quantities is compared to the class boundaries established by the water authority. The substance with the lowest status determines the total status for the specific pollutant quality element. "One out all out" is therefore the principle being used.

16.4.1 Non-synthetic pollutants

Regarding non-synthetic pollutants (mostly metals), Tables 1.2.1 - 1.2.2 in Annex V of the WFD states that high status should correspond to undisturbed conditions, i.e. the natural background concentration in the water body. In this handbook, the background concentration is defined as the concentration found before industrialism had really started and before agriculture was rationalised and began using chemicals to a much greater extent. It is therefore not possible to simply use the concentration in a water body that currently has no direct discharges of the substance. Historical changes and contributions from diffuse sources, such as atmospheric deposition, should also be taken into consideration. The water authority makes an assessment of the natural background concentration for the water body based on all the information available. The class boundary between high and good status is set as the background concentration for the water body whilst the class boundary between good and moderate status is determined based on ecotoxicological data in accordance with the procedure laid down in Annex V, 1.2.6 of the WFD and is specified for the bioavailable concentration.

The measured filtered (0.45 µm filter) concentration is compared to the class boundaries. If any of the class boundaries are exceeded at this stage, a more-detailed analysis should be done to determine whether this is due to a significant environmental impact or whether the high concentration has natural causes. The analysis consists of:

1. ASSESSMENT OF THE BACKGROUND CONCENTRATION
If the background concentration is high, the water authority should consider this and assess the risks for biological effects based on the local conditions. The natural level of most metals in water can be assessed with acceptable accuracy based on analyses from upstream points or nearby water areas that are undisturbed by local emissions and acidification. In the absence of such analysis values, standardised values for background concentrations are available, see Table 16.1 (Swedish EPA, 1999[24]) . The table gives an estimate of the original natural concentrations. It

[24] Swedish EPA 1999. Assessment principles for environmental quality, lakes and watercourses, back-ground report 1 chemical and physical parameters, Swedish EPA Report 4920.

should be borne in mind, however, that the background concentrations can vary considerably depending on the local pre-conditions.

Natural concentrations in sediment are primarily determined based on specific, local values from deeper sediment layers. These sediments are normally found in lakes at a depth of 15-30 cm, but in more nutrient-rich water (in which sedimentation is very fast), 150-year-old sediment is to be found lower down. Standardised values for natural concentrations of metals in sediment can also be found in Table 16.1.

Table 16.1. Estimated background concentrations for metals in Sweden (Swedish EPA, 1999).

	Cu	Zn	Cd	Pb	Cr	Ni	Co	As	V	Hg
Larger water-courses ((µg/l)										
Estimated back-ground concentration	1	3	0.003	0.05	0.2	0.5	0.05	0.2	0.1	0.001
Smaller water-courses (µg/l)										
Estimated back-ground concentration	0.3	1	0.002	0.02	0.1	0.3	0.03	0.06	0.06	0.001
Lakes (µg/l)										
Estimated back-ground concentration	0.3	1	0.005	0.05	0.05	0.2	0.03	0.2	0.1	0.001
Sediment (mg/kg ds)										
Estimated back-ground concentration	15	100	0.3	5	15	10	15	8	20	0.08

2. ASSESSMENT OF BIOAVAILABILITY

One analysed sample of the total filtered concentration of a metal tells us rather little about its biological effect. It is the bioavailable concentration that is significant for the magnitude of the impact the pollutant has on organisms. What proportion of the concentration is bioavailable depends on a number of different factors. It depends firstly on the type of discharge. If the discharge consists of metals in mineral form, only a small proportion is available compared to if the discharge consists directly of metal ions, which gives a very high bioavailability. The availability also depends on the chemical properties of the water. Organic content, pH and hardness are important factors for inland water. Based on the factors described, the water authority should make an assessment of the bioavailable concentration that can be compared to the class boundary. Models that calculate the bioavailable concentration based on total concentrations and determinants are currently being developed at the EU level but have yet to be sufficiently verified for Swedish conditions to be used straight away. It is possible to use these in combination with expert assessments, however.

16.4.2 Synthetic pollutants

Synthetic pollutants are substances that should not occur in the environment in undisturbed conditions. Regarding these substances, it is stated in Tables 1.2.1 - 1.2.2 in Annex V of the WFD that high status should involve concentrations close to zero and at least lower than the detection limit when using the advanced analysis technique in operation. The class boundary between high and good status is hence consequently set to the detection limit. It is important, however, that the detection limit is defined for each relevant substance so that it is as low as possible in order to be measured using the current technology since different analysis methods can otherwise give rise to widely differing limits.

The class boundary between good and moderate status is determined based on ecotoxicological data in accordance with the procedure laid down in Annex V, Section 1.2.6 in the WFD.

16.5 Comments

Class boundaries for pollutants should be calculated using the method described in Annex V, Section 1.2.6 of the WFD, i.e. the methods which EU Member States have agreed to use. This means that the established class boundaries are based on ecotoxicological effects studies on different trophy levels, and for humans or predators that feed off the water environment, and take the most sensitive organisms into consideration. These methods are not comprehensive and any additives or synergy effects are for example not taken into account even though shortcomings in the supporting data have been corrected with safety factors. Due to this, it cannot be guaranteed that effects on biota will not occur as a result of the exposure of hazardous substances despite no class boundaries being exceeded. Such effects should, however, be discovered due to the fact that the biological quality factors must always be assessed. If the biology indicates an impact, the water body is classified as having moderate or worse status even if the physico-chemical status is good. The parameters currently assessed for the biological quality factors don’t specifically indicate a toxic impact but do give a clearer response to nutrient- or acidity stress or to hydromorphological impact. This will be developed in the future so that parameters are established which respond more clearly to a toxic impact.

In cases where class boundaries for a substance have been set for the water phase but measurement data is unavailable, data for the relevant substance in sediment or biota can be used to make an expert assessment of whether the class boundaries risk being exceeded or not. Conversion models can be used to estimate the equivalent of a sediment or biota concentration in water. Such a model is described in the report containing proposals for limit values from the Swedish Chemicals Agency. Furthermore, values for sediment that correspond to values for water have also been determined using the method described in Annex V, Section 1.2.6 of the WFD. These conversion models are rather unreliable and the results must be complemented with an expert assessment. If a value is deemed to be close to a class boundary, this can be seen as an indication of a need for sampling in the water phase.

Annex B - Assessment criteria for coastal and transitional waters

(This annex contains the text for all assessment criteria for coastal and transitional waters and can be downloaded as a separate document from the Swedish EPA's website at www.naturvardsverket.se. The reason for this is so that the user can avoid having to download files that are very big and hence difficult to handle).

Contents

1 Introduction

Assessment criteria for coastal and transitional waters has been compiled by research scientists from Stockholm University, the Swedish University of Agricultural Sciences, Umeå University, Göteborg University, SMHI (Swedish Meteorological and Hydrological Institute) and other consultants on behalf of the Swedish Environmental Protection Agency.

Within the EU, intercalibration of the class boundaries between high and good, and also between good and moderate, has been carried out for the biological quality elements in accordance with the standards laid down in the Water Framework Directive (WFD). Intercalibration work has been carried out within the Common Implementation Strategy (CIS) and has been based on a comparison between the different Member States' class boundaries for the respective parameters or quality elements and, where necessary, adjust the boundaries in order to guarantee an equivalent protection of the water environment. EU waters have been divided into different types to enable the comparison to be made between waters with the same preconditions. The work has been carried out in a series of different working groups and has involved a considerable number of experts.

Because of the lack of comparable data and classification systems, it was not possible to intercalibrate all parameters within the different quality elements. As far as Sweden is concerned, the coastal water quality elements and parameters intercalibrated until the end of 2007 are as follows:

Benthic invertebrate fauna – Benthic Quality Index (completed but not formally adopted)
Macroalgae - depth distribution (completed but not formally adopted)
Phytoplankton – chlorophyll a, abundance (completed but not adopted

Following intercalibration, certain boundaries have been adjusted slightly but in most cases the Swedish assessment of high, good and moderate status has corresponded well with assessments made by other Member States. Decisions on boundaries, both absolute values and Ecological Quality Ratios (EQR), will be taken in the course of 2008 for phytoplankton, macroalgae, diatoms and benthic invertebrate fauna. The decision will be taken at the EU level.

In the WFD, it is stated that the results of the status classification shall be given in Ecological Quality Ratios (EQR) to guarantee comparability between Member States. EQRs show the deviation from the reference value. In the course of the work on intercalibration, both nationally and internationally, it has become apparent that the extent of the acceptable deviation for the different status classes varies between different quality elements and parameters. Therefore the EQR values for, e.g. the class boundary between good and moderate status for the various quality elements and parameters differ and the EQR values cannot be directly compared between the quality elements or parameters. In cases where there are class boundaries based on values for the parameters themselves e.g. µg/l chlorophyll or metres of transparency, these class boundaries are also presented in this handbook. The purpose of this is to facilitate understanding of the class boundaries.

Section numbering can be the same in other annexes to the handbook, but a reference to a given section in the annex always refers to the relevant section of this annex.

1.1 Quality elements and parameters included

Table 1.1. Summary of the included parameters and quality elements for coastal and transitional waters.

	Biological quality elements			**Physico-chemical quality elements**[1]			
Quality element	**Macroalgae**	**Phytoplankton**	**Benthic invertebrate fauna**	**Transparency**	**Nutrients**	**Oxygen**	**Pollutants**
Parameter	Depth distribution	Chlorophyll a Biovolume	BQI_m-indexspecies compositionnumber of species, abundance)	Transparency	Tot-N Tot-P DIN DIP	Oxygen balance	Pollutants released in significant amounts

So far, the Swedish EPA has only developed assessment criteria for quality elements and parameters for which there is sufficient knowledge and background data. Fish are not a specified quality element for coastal waters but are only listed for transitional waters in Annex V of the WFD. As Sweden only has two relatively small areas that are classed as transitional waters, one of the west coast and one on the east coast, it has not been possible to develop any national assessment criteria for these. Instead, it will be a question of type-specific assessments which the water authority will have to make based on an expert judgement.

Non-native species are not covered by the current assessment criteria. Work is ongoing within the EU to draw up guidelines on how to deal with this as it is a widespread problem.

All background reports on the assessment criteria are presented in more detail online at www.naturvardsverket.se. There may be discrepancies between the background reports and in the handbook, since developments have occurred since the reports were written. The handbook is the most up-to-date and represents the Swedish EPA's position on the material.

[1] Annex V of the WFD also contains priority pollutants discharged into water bodies but with a quality element below ecological status. Under EU Guidance no. 13, the priority pollutants shall only be dealt with under surface water chemical status once EU-wide limit values have been developed. In these regulations, general advice and the handbook priority pollutants are dealt with only under chemical surface water status.

1.2 Cofactoring of quality elements

When classifying ecological status and potential, the biological quality elements shall be cofactored. In cases where the biological quality elements indicate good or high status, or good or maximum potential, physico-chemical quality elements shall also be cofactored.

See REG 2, Chapter 2, Section 2

In cases where the biological and physico-chemical quality elements indicate high status or maximum potential, hydromorphological quality elements shall also be cofactored. When cofactoring quality elements, the element which is classified as the worst status or potential is the decisive one.

Physico-chemical quality elements can only reduce the ecological status from high to good or from good to moderate and only reduce the ecological potential from maximum to good or from good to moderate. Hydromorphological quality elements can only reduce the ecological status from high to good and only reduce the ecological potential from maximum to good.

1.3 Type classification

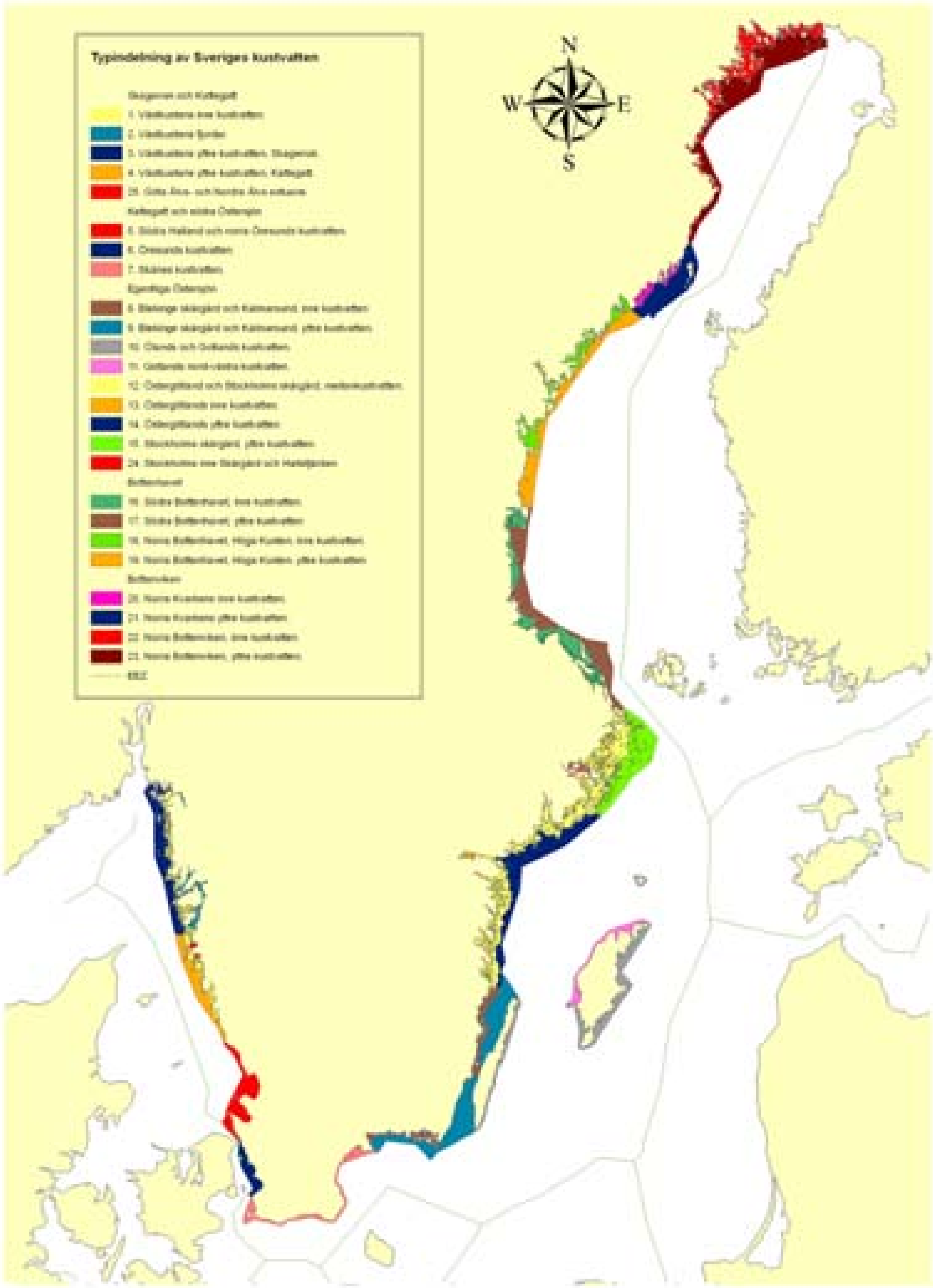
Sweden's coastal waters are classified into 25 types, two of which are transitional waters. A complete list and map can be found in the Swedish EPA's regulations on mapping and analysis (NFS 2006:1). The table below provides an overview of the type classification and a small-scale map can be seen in Figure 1.1.

Table 1.2. Overview of types of Swedish coastal and transitional waters (*) according to the classification in NFS 2006:1.

Type no	Area
1	Archipelago of the west coast, inner parts
2	Fjords of the west coast
3	Archipelago of the west coast, Skagerrak, outer parts
4	Archipelago of the west coast, Kattegat, outer parts
5	Coastal waters of south Halland and north Öresund
6	Coastal waters of Öresund
7	Coastal waters of Skåne
8	Archipelago of Blekinge and Kalmarsund, inner parts
9	Archipelago of Blekinge, and Kalmarsund, outer parts
10	Coastal waters of east Öland and south and east Gotland including Gotska sandön
11	Coastal waters of northwest part of Gotland
12	Archipelago of Östergötland and archipelago of Stockholm, middle parts
13	Archipelago of Östergötland, inner parts
14	Archipelago of Östergötland, outer parts
15	Archipelago of Stockholm, outer parts
16	Coastal waters of south Bothnian Sea, inner parts
17	Coastal waters of south Bothnian Sea, outer parts
18	Coastal waters of north Bothnian Sea, Höga kusten, inner parts
19	Coastal waters of north Bothnian Sea, Höga kusten, outer parts
20	Coastal waters of the Quark, inner parts

Type no	Area
21	Coastal waters of the Quark, outer parts
22	Coastal waters of north Bothnian Bay, inner parts
23	Coastal waters of north Bothnian Bay, outer parts
24 (*)	Archipelago of Stockholm, inner parts and Hallsfjärden
25 (*)	Göta Älv and Nordre Älv estuaries

Figure 1.1 Small-scale map typology for coastal and transitional waters according to NFS 2006:1



2 Benthic invertebrate fauna

Parameter	Shows primarily effects of	How often do measurements need to be taken?	At what time of the year?
Number of species, sensitivity and abundance in one index (BQI_m)	Eutrophication	Once a year	May-June

2.1 Introduction

The quality of the benthic environment can be assessed by analysing the benthic invertebrate fauna living in the sediment. These respond dramatically both to oxygen depletion (hypoxia) and to increasing or decreasing organic load. Benthic fauna are often stationary and relatively long-lived, which means that their composition reflects environmental conditions over a relatively long period of time. A benthic invertebrate assemblage includes both tolerant and sensitive species, and an analysis of the species composition normally provides a good basis for scientific assessment of environmental quality.

2.2 Parameters included

The status of benthic invertebrate fauna is classified based on an index (BQI_m), Benthic Quality Index) developed for soft bottoms. The index is based on three parameters; species composition (the ratio of sensitive to tolerant species), number of species and abundance (number of individuals), see formula 2.1. The index is based on the fact that these parameters change when the organic load on sea-bottoms increases. The emphasis in the index is on the sensitivity of the species to disturbances. BQI_m varies between 0 (dead bottoms) and about 22 (high status)

See REG Annex 4, Section 1.1

$$\mathrm{BQI_m} = \left[\sum_{i=1}^{S_{classified}} \left(\frac{N_i}{N_{totclassified}} * Sensitivityvalue_i \right) \right] * \log_{10}(S+1) * \left(\frac{N_{tot}}{N_{tot}+5} \right)$$

Formula 2.1. Formula for BQI_m. S = Total number of species, $S_{classified}$ = number of species classified as sensitive, N_{tot} = total abundance per 0.1 m^2, $N_{tot\ classified}$ = total number of individuals classified as sensitive, N_i = abundance of species *i*.

The ratio of sensitive to tolerant species varies between 1 and 15 and constitutes the main component of the index. Different species have different sensitivity values depending on their tolerance of or sensitivity to environmental disturbance. Low values indicate a large proportion of tolerant species and high values indicate a large proportion of sensitive species. In the Skagerrak and Kattegat, the species are classified based on their occurrence in different environments. A species occurring in species-poor (disturbed) environments is given a low sensitivity value whilst species that only occur in species-rich environments receive a high sensitivity value. In the Baltic Sea, with its naturally low number of benthic invertebrate species, this method does not work. The sensitivity values for different species in the Baltic are instead based on literature information and expert knowledge. Classes 1, 5, 10 and 15 are used, where 1 represents very pollution-tolerant and 15 very pollution-sensitive.

The second factor, based on the logarithm for number of species, varies between 0, when there is no life at all, to just under 2 in the most species-rich locations (with about 70 taxa) and lowers the index if the number of species is under 9 and raises the index if it is over 9.

The abundance factor, based on the number of individuals, is normally of only slight significance but lowers the index if it is less than 20 individuals in one sample. This factor is included in order to deal with situations where a few individuals of relatively sensitive species can give high index values. Such situations have been detected mainly on the east coast where species that can be found in undisturbed environments can sporadically occur in low abundances in stressed environments. The natural salinity gradient from the Skagerrak and Kattegat to the Bothnian Bay creates naturally large differences in number of species and abundance around the coast. The index cannot therefore reach the same levels all around the coast. It has not been considered necessary to standardise BQI_m for comparison between the sea basins as this is done when converting from the BQI_m value into EQR values. BQI_m - and EQR values for each type are given in Table 2.4.

2.3 Data requirements

2.3.1 Taxonomy

Lists of sensitivity values have been developed for this assessment criterion (see Table 2.1 for the west coast and 2.2 for the east coast). The taxonomy in these is mainly based on the International Taxonomic Information System (ITIS) recommended by ICES (International Council for the Exploration of the Sea).[2] Some taxa have been omitted from the collations as they are not considered to constitute a part of the fauna that can be sampled quantitatively with the methodology used. These omitted taxa are listed in Table 2.3.

See REG Annex 4, Section 1.5

It is important that benthic invertebrate fauna are identified at the lowest possible taxonomic level (mostly the species level) Midge larvae shall however be grouped as Chironomidae, regardless of species. The same is true of earthworms that are to be grouped as Oligochaeta and seed shrimps that are to be grouped as Ostracoda.

See REG Annex 4, Section 1.2

Data shall be based on samples taken with a grab with a 0.1 (±0.02) m^2 opening area, e.g. a van Veen grab or a Smith-McIntyre grab and sieved on a sieve with a 1 mm mesh size.

[2] www.itis.gov

Table 2.1. Sensitivity values for benthic invertebrate taxa on the west coast (types 1-6 and 25) (taxonomically sorted). In cases where there is no species name, the sensitivity values are used for the family or other higher taxonomic level if there is one. Species belonging to *Chironomidae**, *Ostracoda** or *Oligochaeta** are put together in each group immediately before calculating the factor for number of species. All other taxa shall be used ungrouped when calculating the factor for number of species in BQI_m. Explanatory text for *, ** and *** can be found under Table 2.2.

Taxon	Sensitivity value - West coast	Taxon	Sensitivity value - West coast
Oligochaeta *	5.10	Eumida bahusiensis	10.67
Tubificoides benedii	4.22	Eumida sanguinea	10.85
Paramphinome jeffreysii	9.80	Phyllodoce rosea	13.03
Ophryotrocha longidentata	12.82	Sige fusigera	11.44
Lumbrineris fragilis	6.89	Synelmis klatti	10.47
Lumbrineris gracilis	14.71	Bylgides sarsi	7.99
Lumbrineris impatiens	11.95	Enipo kinbergi	7.49
Lumbrineris scopa	9.54	Gattyana amondseni	7.71
Lumbrineris tetraura	12.50	Gattyana cirrosa	8.04
Drilonereis filum	11.99	Harmothoe antilopis	12.11
Onuphis quadricuspis	14.71	Harmothoe borealis	10.78
Aphrodita aculeata	9.91	Harmothoe elisabethae	5.23
Laetmonice filicornis	9.56	Harmothoe imbricata	5.25
Glycera alba	6.73	Harmothoe impar	6.74
Glycera lapidum	10.79	Lepidonotus squamatus	6.40
Glycera rouxi	10.92	Malmgreniella lunulata	11.76
Glycinde nordmanni	11.64	Panthalis oerstedi	12.68
Goniada maculata	9.27	Leanira tetragona	10.76
Gyptis rosea	13.74	Sthenelais limicola	6.97
Kefersteinia cirrata	7.51	Sphaerodoropsis philippi	9.95
Nereimyra punctata	8.73	Sphaerodorum flavum	11.06
Ophiodromus flexuosus	7.49	Sphaerodorum gracilis	7.49
Aglaophamus malmgreni	12.19	Exogone hebes	12.43
Nephtys caeca	6.01	Exogone verugera	12.56
Nephtys ciliata	8.78	Galathowenia oculata	6.53
Nephtys hombergii	5.04	Myriochele heeri	10.94
Nephtys incisa	7.99	Myriochele oculata	9.39
Nephtys longosetosa	8.75	Owenia fusiformis	7.70
Nephtys paradoxa	12.42	Chone duneri	6.56
Ceratocephale loveni	12.54	Chone infundibuliformis	10.96
Eunereis longissima	7.93	Euchone papillosa	9.83
Neanthes succinea	3.81	Laonome kroeyeri	8.29
Neanthes virens	4.58	Sabella pavonina	6.35
Hediste diversicolor	3.98	Apistobranchus tenuis	12.77
Pholoe baltica	9.41	Apistobranchus tullbergi	9.17
Pholoe inornata	9.66	Chaetopterus norvegicus	10.36
Pholoe longa	9.26	Spiochaetopterus typicus	10.71
Pholoe minuta	9.55	Magelona alleni	11.55
Pholoe pallida	12.27	Magelona minuta	12.06
Anaitides groenlandica	6.05	Magelona mirabilis	12.49
Anaitides longipes	10.68	Laonice bahusiensis	9.41
Anaitides maculata	6.75	Laonice cirrata	11.94

Taxon	Sensitivity value - West coast	Taxon	Sensitivity value - West coast
Anaitides mucosa	6.10	Malacoceros fuliginosus	2.16
Eteone barbata	10.46	Minuspio cirrifera	12.07
Eteone flava	4.72	Chone infundibuliformis	10.96
Eteone foliosa	11.12	Polydora caeca	8.13
Eteone longa	4.58	Polydora caulleryi	4.57
Polydora ciliata	4.99	Scionella lornensis	10.20
Polydora cornuta	5.94	Streblosoma bairdi	14.79
Polydora quadrilobata	6.74	Terebellides stroemi	8.29
Prionospio fallax	11.03	Trichobranchus glacialis	13.59
Prionospio dubia	11.64	Trichobranchus roseus	10.65
Prionospio multibranchiata	11.87	Arenicola marina	5.28
Pseudopolydora antennata	4.19	Capitella capitata	1.10
Pseudopolydora pulchra	8.01	Heteromastus filiformis	8.95
Pygospio elegans	4.85	Mediomastus **	5.39
Scolelepis tridentata	12.27	Notomastus latericeus	9.79
Spio armata	6.40	Cossura longocirrata	10.79
Spio filicornis	9.37	Maldane sarsi	7.45
Spiophanes bombyx	11.68	Praxillella praetermissa	10.61
Spiophanes kroeyeri	12.03	Rhodine gracilior	10.41
Trochochaeta multisetosa	6.75	Rhodine loveni	11.30
Ampharete acutifrons	8.20	Ophelia borealis	9.39
Ampharete baltica	8.21	Ophelina acuminata	9.44
Ampharete falcata	12.06	Ophelina cylindricaudata	15.42
Ampharete finmarchica	7.99	Ophelina modesta	13.58
Ampharete goesi	7.49	Ophelina norvegica	15.00
Ampharete lindstroemi	10.15	Orbinia norvegica	13.82
Amphicteis gunneri	11.73	Scoloplos armiger	6.24
Anobothrus gracilis	10.67	Aricidea jeffreysi	7.99
Eclysippe vanelli	14.35	Aricidea suecica	9.83
Melinna cristata	8.58	Cirrophorus lyra	11.73
Samytha sexcirrata	8.34	Levinsenia gracilis	9.23
Sosane sulcata	8.28	Paraonis fulgens	9.17
Aphelochaeta vivipara	9.37	Lipobranchus jeffreysii	11.29
Caulleriella **	6.22	Polyphysia crassa	6.38
Tharyx killariensis	11.83	Scalibregma inflatum	6.65
Chaetozone setosa	10.23	Anoplodactylus petiolatus	9.39
Cirratulus cirratus	9.76	Nephrops norvegicus	12.36
Aphelochaeta mcintoshi	14.71	Liocarcinus depurator	6.99
Brada villosa	10.46	Philocheras bispinosus	12.80
Diplocirrus glaucus	10.49	Calocaris macandreae	11.46
Pherusa plumosa	7.49	Callianassa tyrrhena	10.45
Pectinaria auricoma	9.73	Caprella linearis	6.40
Pectinaria belgica	10.16	Pariambus typicus	6.53
Pectinaria koreni	3.00	Phtisica marina	8.05
Amaeana trilobata	13.80	Ampelisca brevicornis	12.49
Artacama proboscidea	9.57	Ampelisca diadema	10.73
Lanassa venusta	10.51	Ampelisca macrocephala	9.58

Taxon	Sensitivity value - West coast	Taxon	Sensitivity value - West coast
Lanice conchilega	11.68	Ampelisca tenuicornis	9.99
Lysilla loveni	8.95	Byblis gaimardi	12.67
Neoamphitrite affinis	10.42	Haploops tubicola	9.37
Neoamphitrite figulus	6.40	Aora gracilis	11.63
Pista cristata	10.61	Lembos longipes	13.60
Microdeutopus gryllotalpa	6.91	Apseudes spinosus	12.56
Argissa hamatipes	12.51	Ostracoda *	10.30
Corophium affine	9.95	Pennatula phosphorea	11.40
Corophium bonnellii	5.00	Virgularia mirabilis	9.66
Corophium crassicorne	13.29	Cerianthus lloydii	8.68
Corophium insidiosum	9.30	Edwardsia danica	13.15
Corophium volutator	5.94	Edwardsia longicornis	11.52
Ericthonius difformis	11.47	Halcampa chrysanthellum	9.17
Neohela monstrosa	12.12	Brissopsis lyrifera	9.23
Atylus vedlomensis	12.76	Echinocardium cordatum	8.80
Dulichia monacantha	10.13	Echinocardium flavescens	9.17
Dulichia porrecta	8.85	Spatangidae **	13.75
Cheirocratus sundevallii	9.03	Echinocyamus pusillus	9.03
Eriopisa elongata	11.73	Labidoplax buski	10.66
Maera loveni	10.30	Cucumaria elongata	8.78
Protomedeia fasciata	11.36	Asterias rubens	5.82
Leucothoe lilljeborgi	10.44	Astropecten irregularis	5.33
Acidostoma obesum	13.05	Ophiura affinis	8.64
Arrhis phyllonyx	9.84	Ophiura albida	7.49
Bathymedon longimanus	13.33	Ophiura ophiura	3.00
Monoculodes packardi	13.35	Ophiura robusta	9.37
Monoculodes tenuirostratus	10.89	Ophiura sarsi	8.57
Perioculodes longimanus	11.74	Ophiura texturata	5.20
Synchelidium haplocheles	13.23	Amphilepis norvegica	14.71
Westwoodilla caecula	11.06	Amphiura chiajei	7.80
Harpinia **	11.74	Amphiura filiformis	7.80
Diastylis bradyi	9.54	Echiurus echiurus	9.04
Diastylis cornuta	5.38	Harrimania kupfferi	11.84
Diastylis laevis	6.53	Chaetoderma nitidulum	9.66
Diastylis lucifera	10.30	Hiatella arctica	3.95
Diastylis rathkei	8.12	Saxicavella jeffreysi	12.07
Diastylis tumida	10.49	Corbula gibba	4.58
Diastyloides biplicata	13.04	Mya arenaria	3.48
Diastyloides serrata	12.70	Mya truncata	6.24
Leptostylis longimana	13.07	Arctica islandica	5.92
Leptostylis villosa	12.20	Astarte elliptica	9.61
Hemilamprops rosea	9.32	Astarte montagui	9.24
Lamprops fasciata	10.79	Acanthocardia echinata	9.58
Eudorella emarginata	11.64	Cerastoderma edule	4.85
Eudorella truncatula	10.52	Cerastoderma glaucum	4.58
Leucon acutirostris	6.55	Parvicardium minimum	10.42
Leucon nasica	11.64	Parvicardium pinnulatum	10.05

Taxon	Sensitivity value - West coast
Campylaspis costata	13.98
Campylaspis rubicunda	12.99
Echinozone coronata	11.73
Montacuta tenella	10.77
Mysella bidentata	6.83
Kelliella miliaris	15.02
Lucinoma borealis	6.92
Myrtea spinifera	9.93
Mendicula ferruginosa	14.33
Thyasira equalis	10.96
Thyasira flexuosa	4.53
Thyasira obsoleta	14.71
Thyasira sarsii	7.47
Spisula subtruncata	6.43
Cultellus pellucidus	5.92
Abra alba	3.96
Abra nitida	9.26
Scrobicularia plana	4.33
Macoma balthica	5.23
Macoma calcarea	6.76
Tellina fabula	12.37
Tellina tenuis	7.44
Mysia undata	9.37
Petricola pholadiformis	3.81
Chamelea gallina	10.79
Clausinella fasciata	10.28
Venus gallina	9.01
Cuspidaria obesa	14.71
Thracia convexa	10.38
Thracia phaseolina	12.15
Nuculana minuta	9.53
Nuculana pernula	10.51
Ennucula tenuis	9.71
Nucula nitidosa	8.12
Nucula sulcata	10.40
Nucula tumidula	14.71
Yoldiella fraterna	14.71
Yoldiella lucida	14.33
Bathyarca pectunculoides	15.29
Modiolus modiolus	6.67
Musculus discors	9.70
Musculus niger	8.88
Mytilus edulis	7.05
Chlamys septemradiatus	10.79
Acteon tornatilis	7.56
Odostomia acuta	13.50
Aporrhais pespelicanis	4.65

Taxon	Sensitivity value - West coast
Parvicardium scabrum	5.91
Decipula tenella	13.88
Montacuta ferruginosa	9.55
Polinices pulchella	9.57
Alvania abyssicola	14.35
Hyala vitrea	10.12
Pusillina sarsi	7.00
Turritella communis	7.80
Akera bullata	4.50
Cylichna cylindracea	9.53
Diaphana minuta	11.85
Philine aperta	6.76
Philine scabra	9.43
Retusa obtusa	8.21
Retusa truncatula	9.83
Buccinum undatum	6.40
Mangelia attenuata	9.84
Mangelia brachystoma	11.62
Nassarius pygmaeus	10.84
Nassarius reticulatus	4.99
Entalina quinquangularis	14.98
Tubulanus linearis	6.85
Malacobdella grossa	8.59
Nemertea, övriga ***	7.99
Phoronis muelleri	8.34
Halicryptus spinulosus	6.29
Priapulus caudatus	7.96
Golfingia procera	8.56
Onchnesoma steenstrupi	14.71
Phascolion strombi	9.35
Oligochaeta *	5.10
Tubificoides benedii	4.22
Paramphinome jeffreysii	9.80
Ophryotrocha longidentata	12.82

Taxon	Sensitivity value - West coast	Taxon	Sensitivity value - West coast
Bittium reticulatum	7.41		
Hydrobia ulvae	2.60		
Euspira montagui	9.72		

Table 2.2. Sensitivity values for benthic invertebrate fauna (types 7-24) (taxonomically sorted). Using the table below, it should be possible to assign sensitivity values for most of the species found in the Baltic Sea system. The table does not, however, represent a complete list of the species that can be found in the Baltic Sea. In cases where there is no species name, the sensitivity values are used for the family or other higher taxonomic level if there is one. Species belonging to *Chironomidae**, *Ostracoda** or *Oligochaeta** are put together in each group immediately before calculating the factor for number of species. All other taxa shall be used ungrouped when calculating the factor for number of species in BQI_m.

Taxon	Sensitivity value - East coast	Taxon	Sensitivity value - East coast
Oligochaeta *	1	Idotea, other species ***	10
Nephtys**	10	Heterotanais oerstedi	5
Hediste diversicolor	5	Ostracoda*	15
Eteone**	10	Coleoptera**	10
Bylgides sarsi	15	Ceratopogonidae**	5
Fabricia sabella	10	Chaoboridae**	1
Manayunkia aestuarina	10	Chironomidae*	1
Marenzelleria**	5	Trichoptera**	15
Pygospio elegans	5	Ephemeroptera**	10
Spio filicornis	10	Mya arenaria	10
Streblospio benedicti	5	Arctica islandica	5
Trochochaeta multisetosa	5	Astarte borealis	15
Alkmaria rominji	5	Astarte elliptica	15
Terebellides stroemi	10	Astarte montagui	15
Arenicola marina	10	Cerastoderma edule	5
Capitella**	1	Cerastoderma glaucum	10
Heteromastus filiformis	5	Pisidium**	15
Scoloplos armiger	10	Sphaerium**	10
Aricidea jeffreysi	10	Macoma**	5
Aricidea suecica	10	Mytilus edulis	5
Levinsenia gracilis	10	Radix balthica	15
Crangon crangon	10	Lymnaeidae, other ***	10
Ampithoe rubricata	15	Valvata macrostoma	5
Leptocheirus pilosus	5	Valvata piscinalis	10
Microdeutopus gryllotalpa	10	Bithynia tentaculata	10
Corophium volutator	10	Potamopyrgus antipodarum	10
Gammarus**	10	Hydrobiidae, other ***	5
Bathyporeia pilosa	15	Littorina saxatilis	10
Melita palmate	15	Rissoa**	15
Phoxocephalus holbolli	15	Retusa truncatula	15
Monoporeia affinis	15	Limapontia**	15
Pontoporeia femorata	15	Theodoxus fluviatilis	15

Taxon	Sensitivity value - East coast
Diastylis rathkei	10
Cyathura carinata	5
Asellus aquaticus	5
Jaera**	15
Sphaeroma hookeri	10
Saduria entomon	10

Taxon	Sensitivity value - East coast
Micrura baltica	15
Nemertea, other***	10
Turbellaria**	10
Halicryptus spinulosus	15
Priapulus caudatus	10

* Sum up the abundance of all species belonging to *Chironomidae* and use the sensitivity value for *Chironomidae*. Species belonging to *Oligochaeta* and *Ostracoda* are put together in a similar way and the sensitivity values for *Oligochaeta* and *Ostracoda* respectively are used. The taxonomic groupings for *Chironomidae*, *Oligochaeta* and *Ostracoda* are ranked as species when calculating the factor for number of species in BQIm, i.e. *Chironomidae* is counted as one taxon regardless of how many *Chironomid* species occur.

** If one species in this group is obtained, the species name is specified along with the sensitivity value for the group. If two or more species in the group are obtained, each individual species name and the same sensitivity value are used for the species specified for the group.

*** Specify species name together with the sensitivity value specified for the "Other" group. The species should not be grouped as "other species" but is instead specified as individual species with their full name.

Table 2.3. The following taxa and sub-classes have been omitted as they are not considered to constitute part of the fauna that can be sampled quantitatively with the methodology used.

Rank	Taxa
Sub-class	Hirudinea
Sub-class	Acarina
Sub-order	Cladocera
Sub-family	Palaemoninae
Genus	Pandalus
Genus	Meganyctiphanes
Sub-order	Hyperiidea
Order	Mysida
Class	Maxillopoda
Genus	Acanthocephala
Phylum	Chaetognatha
Family	Branchiostomidae
Sub-phylum	Tunicata
Order	Myxiniformes
Infra-class	Teleostei
Family	Alcyoniidae
Genus	Urticina
Genus	Metridium
Genus	Clava
Genus	Dynamena
Genus	Sertularella
Genus	Sertularia
Phylum	Ectoprocta
Phylum	Nemata
Phylum	Nematomorpha
Class	Trematoda
Phylum	Porifera

2.3.2 Status is classified at the water body level

The assessment criterion has been formulated in order to classify status for whole water bodies instead of individual samples or sampling stations. To use the assessment criterion, data is needed from several stations in a water body, at least five and preferably more. A station is a point with a specific depth; some spread may occur around the point. If there is variation between stations, a more reliable estimate of the area's mean value can be obtained by spreading out the grabs in the area rather than employing two or more grabs at a small number of stations. The classification for the area will be more reliable the more stations are sampled in the survey. For classification we have chosen to follow the precautionary principle and to use the 20th percentile of the BQI_m mean value from a water body when comparing with class boundaries for the type. In this context, following the precautionary principle means choosing the 20% boundary so that the area's true mean value is over the chosen boundary with greater reliability than if, for example, the observed mean value had been used to classify status. The 20th percentile is calculated using a special method based on 9 999 mean values drawn randomly from the existing index values in a water body. In this case, the 20th percentile corresponds to a one-sided 80% confidence interval that gets smaller the more samples are taken. The lower part of the confidence interval used when classifying status, the 20% boundary, shifts upwards the greater the number of measurement values. Too few samples provide an unreliable estimate of the area's mean value and result in a wider confidence interval. It can also be detrimental as the lower confidence interval boundary ends up in an inferior status class.

See REG Annex 4, Section 1.2

Extrapolation of the results so that they apply to the whole water body is limited by any selection criteria or stratification of stations in the area. Extrapolation strictly applies only to bottom-types or depths included in the survey. For example, if only shallow areas, e.g. not deeper than 20 m, are included in a survey, the results can only be used to classify shallower bottoms. If the reason for the depth limit is because the deeper bottoms have no fauna due to oxygen depletion, this information must be included with the classification. For unrestricted extrapolation, completely random sampling is required.

A common problem might be a shortage of measurement values to classify status in a water body. Combining measurement values from several adjacent water bodies of the same type is one way of obtaining more background data for the classification. Consolidating data in this way is only recommended for areas where there are no local emissions from communities, factories or larger rivers. If there is no data available from a water body, results can be extrapolated from adjacent water bodies belonging to the same type and with a similar degree of anthropogenic disturbance. See Section 4.4.2 in the main part of the handbook for more details.

2.4 Status classification

2.4.1 Calculating status class

See REG Annex 4, Section 1.3

Status class for benthic invertebrate fauna in a water body shall be calculated as follows:

1. Calculate BQI_m based on species and abundance data from each individual sample (grab) in the water body (formula 2.1 and Tables 2.1 and 2.2). Taxa in Table 2.3 shall not be included in the data.
2. Calculate the mean value of BQI_m for each station (sampling point) and year.
3. Draw at random, with replacement, the same number of values as there are station mean values for BQI_m and calculate the mean value for these drawn values. Repeat the procedure 9 999 times. Calculate the 20th percentile for these 9 999 mean values (see Section 2.4.2 for calculation and Table 2.4 for any depth intervals).
4. Compare the value for the 20th percentile with the class boundaries for BQI_m for the relevant type and depth intervals in Table 2.4. This gives the status class.

The calculation is primarily done on a yearly basis, but data can also be aggregated over longer time periods, up to six years (one water cycle). A drawback of aggregation is that the dispersion tends to increase as a result of annual variation. A benefit is that more station mean values increase the background data, which reduces the measure of dispersion. Annual classifications also make it easier to detect trends.

2.4.2 Calculating the 20th percentile

An index value for BQI_m is calculated for each sample. If there are several samples from a single visit to a station, the mean value of all the index values from the visit are calculated. The percentile calculation is based on 9 999 mean values from randomly drawn observations from index values already calculated from the water body. There is software available for this calculation (www.naturvardsverket.se). This software is not applicable to benthic invertebrate fauna surveys with stratified sampling. For data from stratified sampling, the 20% boundary in the confidence interval is calculated manually according to the description of stratified sampling in Annex 9 of the background report regarding benthic invertebrate fauna[3].

For sampling programmes in which the stations were originally randomly selected, the calculation procedure is briefly described below. Randomize the same number of index values from the data-set to be evaluated as there are index values in the dataset. The replacement principle shall be used when randomizing, which means that the same value can be drawn several times. From each randomization series, the mean value for the index values obtained during randomization is calcu-

[3] Bedömningsgrund för kust och hav [Assessment criteria for coastal and sea waters] Bentiska evertebrater [Benthic invertebrates], M. Blomqvist, H. Cederwall, K. Leonardsson and R. Rosenberg, 2006

lated. Note the mean value and repeat the randomization procedure 9 999 times. Calculate the 20th percentile from the new data-set of 9 999 index mean values. One way of obtaining the 20th percentile among these values is to sort all 9 999 index mean values and note the value at position 2000 in the sorted list. Compare this index value with the class boundaries in Table 2.4. If the 20th percentile is over the class boundary between good and moderate but under the boundary between high and good, the status of the area is classified as good. If the 20th percentile is under the boundary between good and moderate, the status of the area is classified as moderate. The same principle applies when comparing the other class boundaries. A classification example based on annual data is illustrated graphically in Figure 2.1.

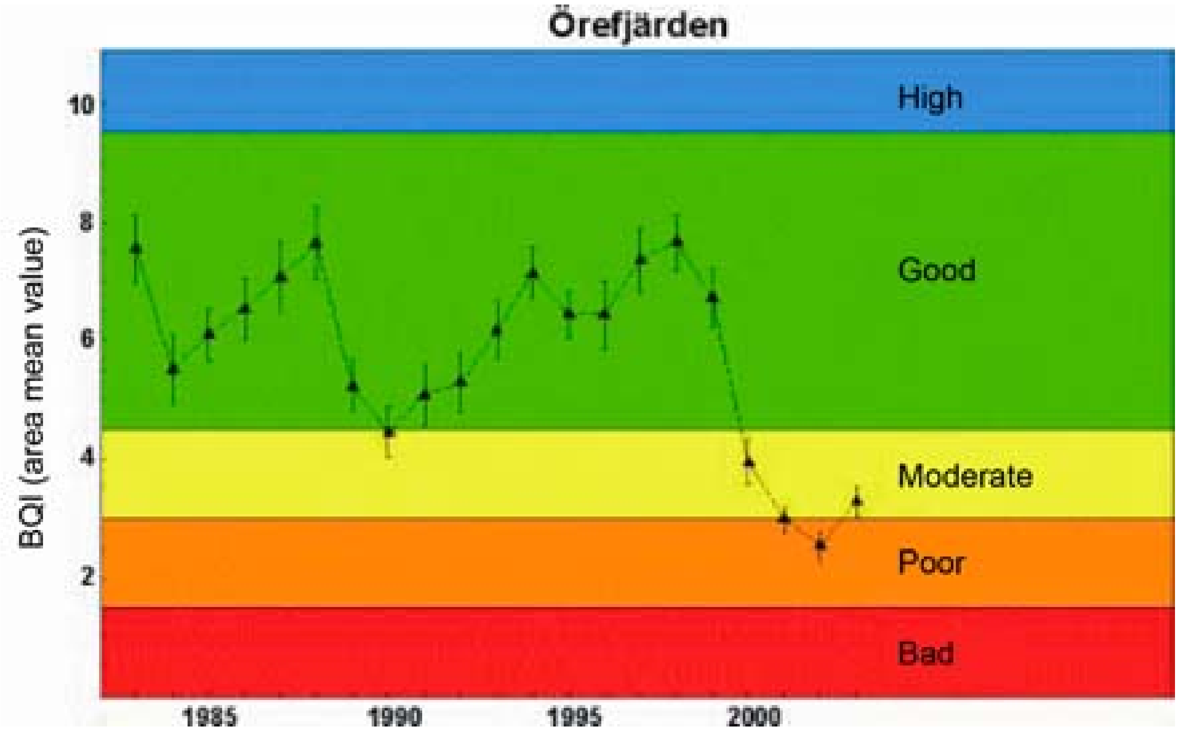


Figure 2.1. Examples of time series with annual BQI_m mean values from a sea area. The measure of dispersion is constituted by the 20th percentile (lower boundary) and the 80th percentile (upper boundary). Periods of good status can be differentiated in the figure where the lower part of the confidence interval finishes above the boundary between good and moderate status.

2.5 Class boundaries

The class boundaries for the Skagerrak and Kattegat differ depending on whether the bottoms are under or over the halocline, the lower disturbance limit of which has been set to 20 metres. Under this depth, salinity is relatively stable and the fauna found here generally give a higher BQI_m than above the halocline. Otherwise, the west coast is classified as a single water area down to the Öresund Bridge in the south. Areas of types 1-6 and 25 that are deeper than 20 m have been assigned common class boundaries. Similarly, shallower areas (5-20 m) in types 1-6 and 25 have been assigned common class boundaries regardless of type.

See REG Annex 4, Section 1.1

In the Baltic, the class boundaries refer to a limited depth interval, normally 5-60 m. Greater depths have been excluded since we then get down towards or go under the halocline where the risk of oxygen depletion and absence of benthic invertebrate fauna is considerable. Sampling at depths shallower than about 5 metres is not recommended in any area. The current class boundaries for each type are given in Table 2.4. EQR values are given in Table 2.4 but are not to be used when classifying status. They are given to enable comparison, e.g. when intercalibrating.

Table 2.4 Class boundaries for BQI_m for all types - to be used when classifying status. Numbering of types in accordance with Table 1.2.

			BQI_m					EQR[1]			
Basin	Type no.	Depth strata	HG	GM	MO	OD	max BQI_m	HG	GM	MO	OD
Skagerrak and Kattegat											
	1-6 and 25	5-20 m	13.9	10.3	6.9	3.4	15.7	0.89	0.66	0.44	0.22
	1-6 and 25	> 20 m	15.7	12.0	8.0	4.0	17.6	0.89	0.68	0.45	0.23
Baltic Sea											
	7	5-60 m	10.7	4.0	2.7	1.3	14.0	0.76	0.29	0.19	0.10
	8	5-60 m	10.5	3.5	2.3	1.2	14.0	0.75	0.25	0.17	0.08
	9	5-60 m	10.7	4.0	2.7	1.3	14.0	0.76	0.29	0.19	0.10
	10	5-60 m	9.3	4.0	2.7	1.3	12.0	0.78	0.33	0.22	0.11
	11	5-60 m	8.0	4.0	2.7	1.3	10.0	0.80	0.40	0.27	0.13
	12	5-60 m	10.7	4.0	2.7	1.3	14.0	0.76	0.29	0.19	0.10
	13	5-60 m	9.0	3.0	2.0	1.0	12.0	0.75	0.25	0.17	0.08
	14	5-60 m	10.7	4.0	2.7	1.3	14.0	0.76	0.29	0.19	0.10
	15	5-60 m	10.7	4.0	2.7	1.3	14.0	0.76	0.29	0.19	0.10
	24	5-60 m	7.7	3.0	2.0	1.0	10.0	0.77	0.30	0.20	0.10
Gulf of Bothnia											
	16	> 5 m	10.7	4.0	2.7	1.3	14.0	0.76	0.29	0.19	0.10
	17	> 5 m	10.0	4.0	2.7	1.3	13.0	0.77	0.31	0.21	0.10
	18	> 5 m	10.0	4.0	2.7	1.3	13.0	0.77	0.31	0.21	0.10
	19	> 5 m	10.0	4.0	2.7	1.3	13.0	0.77	0.31	0.21	0.10
	20	> 5 m	10.0	4.0	2.7	1.3	13.0	0.77	0.31	0.21	0.10
	21	> 5 m	10.0	4.0	2.7	1.3	13.0	0.77	0.31	0.21	0.10
	22	> 5 m	7.5	2.0	1.3	0.7	13.0	0.58	0.15	0.10	0.05
	23	> 5 m	6.3	1.5	1.0	0.5	11.0	0.57	0.14	0.09	0.05

[1] EQR is calculated by dividing the 20^{th} percentile by the max BQI_m.

2.6 Comments

2.6.1 In general

In some areas, the results have clearly deviated from the rest of the water bodies within the type. Such area include Rånefjärden in the northern Bothnian Bay, the area outside Söderhamn, Stockholm’s inner archipelago, the coastal areas around

Gotland and east of Öland (for more information, see the background report [4]). A more in-depth expert judgement of these areas may be justified.

The method employed on the west coast to determine the sensitivity values of species based on occurrence in environments of varying diversity and the number of species element gives a high correlation between the index and the number of species. The same conditions do not exist on the east coast, where the index may even decrease with an increasing number of species since several of the most common species are disturbance-tolerant, i.e. have low sensitivity values.

2.6.2 Gear

The assessment criterion is designed based on a sampling area of approx. 0.1m^2. This is the sampling area for a standard model van Veen or Smith-McIntyre grab. This gear is the standard equipment used in the Swedish national environmental monitoring programme. If gear is used that samples a different surface area, the results can of course be converted to 0.1 m^2 values, but this is incorrect statistically speaking. If four grab samples were taken using gear with a 0.025 m^2 sampling area, a larger number of taxa would most likely be obtained than if one sample was taken using a standard van Veen grab. The standard mesh size for sieves used in macrofauna monitoring in the marine environment is 1x1 mm. The abovementioned standard gear shall therefore be used.

See REG Annex 4, Section 1.2

Background report: Bedömningsgrund för kust och hav [Assessment criteria for coastal and sea waters] Bentiska evertebrater [Benthic invertebrates].
Authors: Mats Blomqvist (Hafok AB), Hans Cederwall (Stockholm University), Kjell Leonardsson (Umeå University), Rutger Rosenberg (Göteborg University).

[4] Bedömningsgrund för kust och hav [Assessment criteria for coastal and sea waters]) Bentiska evertebrater [Benthic invertebrates], M. Blomqvist, H. Cederwall, K. Leonardsson and R. Rosenberg, 2006

3 Macroalgae & angiosperms

Parameter	Shows primarily effects of	How often do measurements need to be taken?	At what time of the year?
Depth distribution	Nutrients /eutrophication and turbidity	Once a year	July-September

3.1 Introduction

Macroalgae take up nutrients directly from the water mass and therefore reflect the availability of nutrients and how disturbed the environment is by emissions from e.g. wastewater treatment plants and run-off from forest and agricultural land use. The species are also affected by turbidity, sedimentation and various environmental toxins (industrial emissions). Both the abundance and occurrence of species are affected. The advantage of attached vegetation is that they stay in one place and therefore give an integrated measure of what has happened in the water mass in the area over a long period of time (months to years). This means that sampling can take place at longer time intervals, preferably once a year, and still provide a good indicator of the state of the environment.

3.2 Parameters included

See REG Annex 4, Section 2.1

Status of a type is calculated based on the maximum depth distribution of a number of perennial macroalgae and a few aquatic angiosperms. The species selected for each type represent common, easily identifiable species that occur over a relatively large coastal area. The assessment criterion applies primarily to hard bottoms. For some types with limited access to hard-bottom substrate, where there is little variety in attached macroalgae species, a number of angiosperms for soft bottoms are included. The profile shall consist of a hard bottom when macroalgae are used for classification and of a soft bottom when characeans and angiosperms are used. The assessment criterion is based on the relationship between the depth distribution of the macrovegetation and the availability of light for macroalgal and aquatic angiosperm growth. Light availability can in turn be correlated to the effects of eutrophication, such as reduced transparency, increased epiphytic growth and turbidity in the water mass caused by phytoplankton bloom. The maximum depth distribution of attached vegetation in an area is a good indicator of how disturbed the environment is by high nutrient load.

There are assessment criteria for types 1-12 and 14-23. It has not been possible to develop assessment criteria for types 13, 24 and 25 since the natural salinity level in these water areas fluctuates considerably, which in turn gives far too large inter-annual variations and macroalgae is not considered to be a good indicator for anthropogenic disturbance in these areas.

The level of nutrients, the water's natural salinity and wave exposure affect the composition of macroalgal and angiosperm species. Since the same species don't occur in the whole gradient, classification cannot be performed based on the depth distribution of a single species. A combination of the maximum depth distribution of several species must therefore be used to assess the degree of disturbance.

Section 3.7 describes how the vegetation generally changes with an increasing nutrient load, decreased transparency, more epiphytes and increased deposits of sedimentary particles. This qualitative description is only intended for use as a complement, support and explanation to the quantitative status calculation. It can provide good support to expert judgements when there is a lack of data in accordance with mandatory requirements. The qualitative description of different degrees of disturbance specifies maximum depth distribution for some of the species included in the quantitative assessment criterion. The aim is to facilitate understanding of how total species composition is affected when a species is to be considered eradicated in one transect.

3.3 Data requirements

3.3.1 Sampling methodology

The methodology that has been used and is still used in Swedish national monitoring programmes differs slightly between the east coast and the Skagerrak and Kattegat. The method used to determine the maximum depth distribution of macroalgae can, however, be found in both programmes. The selected species required for this assessment criterion can be found in Table 3.3. Changes in the species composition of different types along the coast depend on the salinity conditions in each type, which are a precondition for these species to occur.

See REG Annex 4, Section 2.2

The number of transects needed is dependent on the size of the water body and on how varied the natural environmental conditions are. A minimum of three transects are needed within a similar area to enable a statistical evaluation of the variation in depth distribution to be performed. The more transects are included in the data used to classify the status of an area, the more reliable the classification will be.

The recommended sampling frequency for investigation of the distribution of perennial macroalgae is once a year. The sampling shall be carried out in late summer (July- September). If a rolling sampling programme is set up (in which the location is only visited every second or third year) for an individual water body or type, annual samples must be taken at several locations (e.g. as part of a national or regional programme in the vicinity) in order to be able to assess inter-annual variation.

A detailed description of established sampling methodology and strategies for choosing locations as referred to in the regulations (NFS 2008:1) can be found in the Swedish EPA Handbook for Environmental Monitoring on the Swedish east coast[5] with two supporting documents[6] on the same web-page and correspondingly for the Swedish west coast[7]. A survey from 2005 deals with spatial variation in depth distribution within a site and classification reliability[8]. To facilitate the development of new methodology, it may also be useful to include the species' coverage of the substrate in accordance with the scale described by the method in the national programme.

3.3.2 Criteria for transect location

The following criteria must be fulfilled to enable macroalgae and angiosperm species and their maximum depth distribution from a transect to be used to calculate an EQR value:

- depth distribution of at least three selected species must be present in the transect,
- salinity must be within the specified interval for the type in question (see typology[9]),
- the transect shall have hard substrates when macroalgae are used for classification and soft substrates when characeans and angiosperms are used. In some areas, a mixture of macroalgae and soft-bottom species can be used together and this is particularly true in the northern-most types in the Bothnian Bay. In all cases, the right substrate down to the maximum depth for high status must be available, i.e. hard substrates for macroalgae and soft substrates for when characeans and/or angiosperms are used.
- the depth of the transect must be greater than the maximum depth for the selected species at high status. A transect does not have to be deeper than 20 metres, however (the transects for type 3 can be deeper than 20 m).

See REG Annex 4, Section 2.2

If any of the points above are not fulfilled, collected data can still be used as the basis of an expert judgement together with Section 3.7

3.3.3 Taxonomy

For identification of the selected algae in the Baltic Sea, we suggest "Alger vid Sveriges östersjökust [Algae along Sweden's Baltic coast][10]. Otherwise, "Norsk Algeflora" [Norwegian Algal Flora][11] is suitable for identification of macroalgae species. To identify characean species, we recommend "Charophytes of the Baltic

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[5] Vegetationsklädda bottnar, ostkust [Vegetation-clad bottoms, Swedish east coast (www.naturvardsverket.se)

[6] Kautsky 1999 and 2000 (www.naturvardsverket.se)

[7] Vegetationsklädda bottnar, västkust, [Vegetation-clad bottoms, Swedish west coast], Karlsson, 2005 (www.naturvardsverket.se)

[8] Kullen-Paradishamn, Collection of macroalgal vegetation data at Kullen. Landskrona, September 2005. Toxicon AB, Report 118-05.

[9] NFS 2006:1

[10] Tolstoy and Österlund, 2003

[11] Rueness, 1977

Sea[12], The Baltic Marine Biologists Publication No 19[13] and an article on characeans by Willén and Tolstoy in Svensk Botanisk Tidskrift [Swedish Journal of Botany][14]. The correct Latin names as well as previous names and synonyms for macroalgae species can be found at www.algaebase.org.

Since many filamentous red, green and brown algae species can be difficult to identify, only two species in the *Cladophora* family are included in the assessment criterion. It is important to note that these two species differ in their depth distribution and that species determination is therefore necessary. *Phyllophora pseudoceranoides* and *Coccotylus truncatus* are difficult to distinguish in the Baltic Sea and shall therefore be treated as one species.

3.4 Status classification

The status class for macroalgae and angiosperms is calculated as follows:

See REG Annex 4, Section 2.3

1. Assess the maximum depth distribution (m) of the selected species in the transect. To be able to classify a transect, the maximum depth distribution of at least three selected species must be included.
2. Study Table 3.3 to see which score class it matches for each species in the current type and convert to the equivalent score (5, 4, 3, 2 or 1). A species shall be seen as eradicated only if it is proven that it has previously been found in the area and has been eradicated as a result of anthropogenic disturbance. (There is a qualitative description of status, Section 3.7, as a support to how species composition and depth distribution change when nutrient load increases).
3. Add up the score for all selected species in the transect, calculate the mean score and divide by five. The value obtained is the EQR value for the *transect*.
4. The EQR value for *the water body* is calculated as the mean value of all (at least three) transect EQR values and is specified with standard deviation.
5. Study Table 3.2 to see which status class the EQR value corresponds to for the water body. The standard deviation shows the uncertainty in the status classification.

The standard deviation gives a measure of how uncertain the assessment is. In cases where an uncertainty interval around the EQR value overlaps any of the class boundaries between high and good status or good or moderate status, this means that the calculated EQR value is very close to a class boundary.

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[12] Blindow, 1994

[13] Schubert H, Blindow I eds., 2003

[14] Willén and Tolstoy, Svensk Botanisk Tidskrift no 3-4 2007

This causes a reasonability assessment to be performed, as described in Section 4.1.1 in the main part of the handbook. See also Section 4.1.2 in the main part of the handbook for more guidance on how to manage uncertainty.

3.4.1 An example calculation for macroalgae and angiosperms

Table 3.1 presents depth distribution data from three studied transects and an EQR value for each transect. This value is obtained by giving an alga a score of 5 if it occurs at or below the minimum depth for high status. Furthermore, if it occurs within the depth interval specified for good status, it is given a score of 4, for moderate 3, for poor 2 and finally for bad status 1. The score for the six species are added together and the mean value calculated and then divided by 5 (the maximum score). An example is presented below showing how this calculation can be done for three transects in Gullmarsfjorden (type 2). Class boundaries for EQR values can be found in 3.2 and the point scales for each species are given in Table 3.3.

Table 3.1. Example status classification in type 2. The observed maximum depth distribution for each species is given in brackets in metres which corresponds for reference conditions/high status=5, good=4, moderate=3, poor=2 and bad=1. These are used to calculate the EQR value for the three transects.

Macrovegetation Species	**Reference score**	**Observed value (in metres) and score**		
		Transect 1	**Transect 2**	**Transect 3**
Chondrus crispus	5	(15) = 5	(4.5) =3	(5.5) =3
Furcellaria lumbricalis	5	(6.9) = 3	(4.5) = 3	(5) = 3
Halidrys siliquosa	5	(5.4) = 3	exists, but not in the transect	(3.9)=2
Saccharina latissima	5	(5.2) = 3	exists, but not in the transect	exists, but not in the transect
Phyllophora Pseudoceranoides	5	(6.7) = 3	(8) = 4	(7.5) = 4
Rhodomela Confervoides	5	(15) = 5	(1.5) = 2	(3.5) = 2
Mean score	5+5+5+5+5+5 = 30/6=5	5+3+3+3+3+5 = 22/6=3.66	3+3+4+2= 12/4=3	3+3+2+4+2= 14/5=2.8
EQR= obs. value / ref.value	**5/5 = 1**	**3.66/5 = 0.73**	**3/5 = 0.60**	**2.8/5 = 0.56**

The result is that transect 1 receives a value of 0.73, transect 2 a value of 0.60 and transect 3 a value of 0.56, with a mean value of 0.63 for the three transects. This means that the area is classified as good status (EQR interval 0.61-0.80 according to Table 3.2).

It is important to note that species that are found in the area but do not occur in the transect should not be included in the status calculation. This assessment must be made in connection with the survey before data is stored for analysis. Similarly, it is only once the entire macroalgae community has drastically changed, when most of the species left grow in shallow water, that missing species are to be included and classed as eradicated. The qualitative descriptions in Section 3.7 provides good guidance on what the environmental conditions are like in general and can be helpful in any reasonability assessments made. A species shall however be classed as bad, i.e. given a score of one, when it has been found in the transect in previous years and then disappeared and where there is a suspicion of anthropogenic disturbance.

3.5 Reference values and class boundaries

Class boundaries for status classification can be found in Table 3.2. Scores for the species included in each type respectively, to calculate EQR values, can be found in Table 3.3. It has not been possible to develop any reference values or class boundaries for three types (13, 24 and 25). These have clear salinity gradients, considerable natural variation and there is no information and data on depth distribution of macroalgae in an undisturbed state.

Table 3.2 Class boundaries for macroalgae and angiosperms. The EQR scale for macroalgae and angiosperms is the same for all types.

Status	EQR interval
High status	0.81-1.0
Good status	0.61-0.80
Moderate status	0.41-0.60
Poor status	0.21-0.40
Bad status	0-0.20

See REG Annex 4, Section 2.4

The intervals between the different class boundaries for the species are approximately the same size. For all species, bad status means that the species is no longer present and that the entire community has been eradicated. Poor status in a water area means that the selected species are still there but they are growing only sparsely and in shallow waters. Some of them may have disappeared altogether.

The definition of a species' maximum depth distribution limit differs, both between different methods and for different species. It is recommended that the specimen found at the greatest depth be used to classify the maximum depth distribution of a species.

The qualitative description of status for macroalgae and angiosperms (Section 3.7) is intended as an aid to determining whether a species has been eradicated in a water body or not depending on the impact of deteriorated transparency, increased sedimentation or other effects of eutrophication or of natural causes, such as a variable environment and is not just missing in the analysed transect. This qualitative description is important in order to be able to determine status and to determine whether the species is to be considered eradicated as a result of anthropogenic disturbance. Knowledge about the species' environmental requirements and tolerance of various eutrophication effects is needed to make the right assessment. It is in this context that the qualitative descriptions in Section 3.7 can be used.

Table 3.3. Boundaries (m) for maximum depth distribution of selected macroalgae species and angiosperms. There are no boundaries for types 13, 24 and 25. If the species has previously been found at the location but is now missing, a score of 1 is given. A qualitative description of changes in species composition can be found in Section 3.7, to help assess whether a species has been eradicated. Numbering of types in accordance with the type classification in NFS 2006:1. *Phyllophora pseudoceranoides* also covers the similar *Coccotylus truncatus*.

See REG Annex 4, Section 2.4

Type	**Taxa**	**5 points if > than:**	**4 points if > than:**	**3 points if > than:**	**2 points if ≤ than:**	**1 points if the species:**
1 Archipelago of the west coast, inner parts						
	Furcellaria lumbricalis	10	7	4	4	eradicated
	Phyllophora pseudoceranoides	18	12	6	6	eradicated
	Rhodomela confervoides	12	7	4	4	eradicated
	Zostera marina	8	6	3	3	eradicated
	Chondrus crispus	8	5	3	3	eradicated
	Delesseria sanguinea	18	12	6	6	eradicated
	Halidrys siliquosa	8	5	3	3	eradicated
	Saccharina latissima	10	7	4	4	eradicated
	Phycodrys rubens	15	10	5	5	eradicated
2 Fjords of the west coast						
	Furcellaria lumbricalis	12	8	4	4	eradicated
	Phyllophora pseudoceranoides	10	8	4	4	eradicated
	Rhodomela confervoides	12	8	4	4	eradicated
	Zostera marina	8	6	3	3	eradicated
	Chondrus crispus	10	7	4	4	eradicated
	Delesseria sanguinea	13	9	5	5	eradicated
	Halidrys siliquosa	10	7	4	4	eradicated
	Saccharina latissima	8	6	3	3	eradicated
	Phycodrys rubens	13	8	4	4	eradicated

3 Archipelago of the west coast, Skagerrak, outer parts						
	Furcellaria lumbricalis	12	9	5	5	eradicated
	Phyllophora pseudoceranoides	22	18	9	9	eradicated
	Rhodomela confervoides	12	9	5	5	eradicated
	Chondrus crispus	13	9	5	5	eradicated
	Delesseria sanguinea	22	18	9	9	eradicated
	Halidrys siliquosa	10	8	4	4	eradicated
	Saccharina latissima	12	9	5	5	eradicated
	Phycodrys rubens	22	17	9	9	eradicated
	Even species with a greater depth distribution than 20 m at high status can be included in the calculation in type 3.					

4 Archipelago of the west coast, Kattegat, outer parts						
	Furcellaria lumbricalis	12	8	4	4	Eradicated
	Phyllophora pseudoceranoides	12	8	5	5	Eradicated
	Rhodomela confervoides	12	8	4	4	Eradicated
	Chondrus crispus	12	8	4	4	Eradicated
	Delesseria sanguinea	16	8	5	5	Eradicated
	Halidrys siliquosa	8	5	3	3	Eradicated
	Saccharina latissima	10	6	4	4	Eradicated
	Phycodrys rubens	16	8	5	5	Eradicated

5 Coastal waters of south Halland and north Öresund						
	Furcellaria lumbricalis	10	7	4	4	Eradicated
	Phyllophora pseudoceranoides	12	7	4	4	Eradicated
	Rhodomela confervoides	10	7	3	3	Eradicated
	Zostera marina	8	6	3	3	Eradicated
	Chondrus crispus	8	5	2	2	Eradicated
	Delesseria sanguinea	12	8	5	5	Eradicated
	Halidrys siliquosa	8	5	3	3	Eradicated
	Saccharina latissima	6	4	2	2	Eradicated
	Phycodrys rubens	12	8	5	5	Eradicated

6 Coastal waters of Öresund						
	Furcellaria lumbricalis	10	6	3	3	Eradicated
	Phyllophora pseudoceranoides	10	6	3	3	eradicated
	Rhodomela confervoides	10	7	3	3	eradicated
	Zostera marina	8	6	3	3	eradicated
	Chondrus crispus	8	5	2	2	eradicated
	Halidrys siliquosa	8	5	3	3	eradicated
	Saccharina latissima	6	4	2	2	eradicated

7 Coastal waters of Skåne						
	Furcellaria lumbricalis	10	6	3	3	eradicated
	Phyllophora pseudoceranoi-des	10	6	3	3	eradicated
	Rhodomela confervoides	8	5	3	3	eradicated
	Zostera marina	8	6	3	3	eradicated

8 Archipelago of Blekinge and Kalmarsund, inner parts						
	Fucus serratus	8	4	2	2	eradicated
	Fucus vesiculosus	8	4	2	2	eradicated
	Furcellaria lumbricalis	10	6	3	3	eradicated
	Phyllophora pseudoceranoi-des	6	4	3	3	eradicated
	Rhodomela confervoides	6	4	2	2	eradicated
	Sphacelaria arctica	10	7	4	4	eradicated

9 Archipelago of Blekinge and Kalmarsund, outer parts						
	Fucus vesiculosus	10	6	3	3	eradicated
	Furcellaria lumbricalis	10	6	3	3	eradicated
	Phyllophora pseudoceranoi-des	8	6	4	4	eradicated
	Rhodomela confervoides	8	5	2	2	eradicated
	Sphacelaria arctica	12	7	3	3	eradicated

10 Coastal waters of east Öland and south and east Gotland including Gotska sandön						
	Fucus vesiculosus	7	5	2	2	eradicated
	Furcellaria lumbricalis	10	7	4	4	eradicated
	Phyllophora pseudoceranoi-des	15	11	6	6	eradicated
	Rhodomela confervoides	15	11	6	6	eradicated
	Sphacelaria arctica	15	11	6	6	eradicated
	Zostera marina	6	4	2	2	eradicated

11 Coastal waters of northwest part of Gotland						
	Fucus vesiculosus	7	5	2	2	eradicated
	Furcellaria lumbricalis	10	7	4	4	eradicated
	Phyllophora pseudoceranoi-des	15	11	6	6	eradicated
	Rhodomela confervoides	15	11	6	6	eradicated
	Sphacelaria arctica	15	11	6	6	eradicated
	Zostera marina	8	4	2	2	eradicated

12 Archipelago of Östergötland and archipelago of Stockholm, middle parts						
	Fucus vesiculosus	6	4	2	2	eradicated
	Furcellaria lumbricalis	10	6	3	3	eradicated
	Phyllophora pseudoceranoi-des	8	5	2	2	eradicated
	Rhodomela confervoides	10	6	3	3	eradicated
	Sphacelaria arctica	10	6	3	3	eradicated

14 Archipelago of Östergötland, outer parts						
	Fucus vesiculosus	8	5	3	3	eradicated
	Furcellaria lumbricalis	10	6	3	3	eradicated
	Phyllophora pseudoceranoi-des	10	6	4	4	eradicated
	Potamogeton perfoliatus	7	4	2	2	eradicated
	Rhodomela confervoides	10	6	4	4	eradicated
	Sphacelaria arctica	12	8	4	4	eradicated
	Tolypella nidifica	6	3	1	1	eradicated
	Zostera marina	7	4	2	2	eradicated

15 Archipelago of Stockholm, outer parts						
	Fucus vesiculosus	8	5	3	3	eradicated
	Furcellaria lumbricalis	10	6	3	3	eradicated
	Phyllophora pseudoceranoi-des	10	6	4	4	eradicated
	Potamogeton perfoliatus	7	4	2	2	eradicated
	Rhodomela confervoides	10	6	4	4	eradicated
	Sphacelaria arctica	12	8	4	4	eradicated
	Tolypella nidifica	6	3	1	1	eradicated
	Zostera marina	7	4	2	2	eradicated

16 Coastal waters of south Bothnian Sea, inner parts						
	Cladophora aegagropila	7	5	2	2	eradicated
	Cladophora rupestris	7	5	2	2	eradicated
	Fucus vesiculo-sus/F.radicans	7	5	3	3	eradicated
	Furcellaria lumbricalis	7	5	3	3	eradicated
	Phyllophora pseudoceranoi-des	10	6	4	4	eradicated
	Rhodomela confervoides	10	6	4	4	eradicated
	Sphacelaria arctica	11	8	4	4	eradicated
	Tolypella nidifica	6	3	1	1	eradicated

17 Coastal waters of south Bothnian Sea, outer parts					
Cladophora aegagropila	12	8	4	4	eradicated
Cladophora rupestris	12	8	4	4	eradicated
Fucus vesiculosus/ F.radicans	8	6	3	3	eradicated
Furcellaria lumbricalis	10	6	3	3	eradicated
Phyllophora pseudoceranoi-des	10	6	4	4	eradicated
Rhodomela confervoides	10	6	4	4	eradicated
Sphacelaria arctica	12	8	4	4	eradicated
Tolypella nidifica	6	3	1	1	eradicated

18 Coastal waters of north Bothnian Sea, Höga kusten, inner parts					
Cladophora aegagropila	7	5	2	2	eradicated
Cladophora rupestris	6	4	2	2	eradicated
Fucus vesiculosus/ F.radicans	6	4	2	2	eradicated
Furcellaria lumbricalis	6	4	2	2	eradicated
Potamogeton perfoliatus	5	3	2	2	eradicated
Sphacelaria arctica	9	6	3	3	eradicated
Tolypella nidifica	6	3	1	1	eradicated

19 Coastal waters of north Bothnian Sea, Höga kusten, outer parts					
Cladophora aegagropila	9	6	3	3	eradicated
Cladophora rupestris	8	5	3	3	eradicated
Fucus vesiculosus/ F.radicans	7	4	2	2	eradicated
Furcellaria lumbricalis	8	5	3	3	eradicated
Sphacelaria arctica	9	6	3	3	eradicated
Tolypella nidifica	6	3	1	1	eradicated

20 Coastal waters of the Quark, inner parts					
Cladophora aegagropila	10	8	4	4	eradicated
Fucus vesiculosus/ F.radicans	5	4	2	2	eradicated
Sphacelaria arctica	9	7	3	3	eradicated

21 Coastal waters of the Quark, outer parts					
Cladophora aegagropila	10	8	4	4	eradicated
Fucus vesiculosus/ F.radicans	6	4	2	2	eradicated
Potamogeton perfoliatus	5	3	2	2	eradicated
Sphacelaria arctica	9	7	3	3	eradicated

22 Coastal waters of north Bothnian Bay, inner parts						
	Cladophora aegagropila	8	6	4	4	eradicated
	Potamogeton perfoliatus	4	2	1	1	eradicated
	Tolypella nidifica	5	3	1	1	eradicated
	Nitella	10	6	3	3	eradicated
	Chara baltica/Chara aspera	10	6	3	3	eradicated

23 Coastal waters of north Bothnian Bay, outer parts						
	Cladophora aegagropila	8	6	4	4	eradicated
	Potamogeton perfoliatus	4	2	1	1	eradicated
	Tolypella nidifica	5	3	1	1	eradicated
	Nitella	10	6	3	3	Eradicated
	Chara baltica/Chara aspera	10	6	3	3	Eradicated

3.6 Comments

In addition to the mandatory measurements (NFS 2008:1), the following measurements are also useful (though not compulsory):

- transparency measured at the transect or even better at several different times in the water area during the survey period,
- the maximum depth distribution of species other than those currently included in the assessment criteria is registered and the degree of coverage is also noted in accordance with the scale described by the method in the national programme. By collecting further data, the assessment criteria can be improved and new methodology can be developed in the future,
- salinity and wave exposure data provide useful supplementary information about the conditions at the location as is data on nutrient conditions, chlorophyll *a* and biological data on e.g. the proportion of annual/perennial species, floating algae masses, number of grazers, etc.
- An important factor when choosing a new sampling location is wave exposure. Wave exposure affects the occurrence of the selected macroalgae species. It may therefore be useful to select survey locations based on a sea-chart and search for locations with moderate wave exposure. In protected locations, deep hard bottoms are less common and depth distribution is more likely to be limited by access to bottom substrate rather than by access to light. This means that the transect cannot be used to classify status. Wave exposure shall therefore be seen as an important supporting factor when selecting suitable sampling locations.

A precondition for calculating an EQR value for each water body is that the specified criteria are fulfilled in the transects included. If the survey is to be performed at a location close to land, e.g. in the inner archipelago, and the location is affected by discharge from a large river, its salinity will at certain times of the year be under the specified salinity interval. This may lead to certain species being eradicated or not managing to reproduce causing them to be absent from the area. Such locations cannot be used to classify status.

The developed quantitative assessment criterion for macroalgal vegetation can only be used in areas and at locations where there are hard bottoms at a depth where it is the access to hard bottom and not to light that restricts the depth distribution of the various species. For angiosperms and characeans, it is instead the access to soft bottoms and not to light that restricts the depth distribution of the species. Since various species can grow and survive in different amounts of lights, transparency is only a rough measure of the maximum depth distribution of macroalgal vegetation. Generally speaking, attached macroalgae, mainly red algae, can survive and grow at a depth that is the equivalent of twice the transparency. If there are no deep hard bottoms in the type or the water body, there are a number of other possibilities that can be used for the expert judgement. One is to only use some of the more shallow-living species for a calculation, or to use a smaller number of the proposed species. Another option is to only make an assessment of whether the status is good or worse. By following such a procedure, hard bottom profiles with a depth that is at least one at which these species have at the boundary between good and moderate status can be used. Biological expertise is required when choosing where to locate transects so that they reflect the status of the area.

One problem is determining whether a species has been eradicated due to high nutrient load and various effects of eutrophication, such as increased sedimentation and/or substantial epiphytic growth of filamentous algae or whether it is present in the area but was not registered in the transect during the survey. This is mostly a problem during the analysis of previously collected data which does not contain this information.

Because all the selected species are easy to identify and commonly occurring in each of the types, it is likely that they are to be found in the vicinity or in one of the other transects studied, even if they are not registered in the current transect. In addition, the qualitative description of the status at different disturbances (Section 3.7) is also available as a support. When a second survey of the same profile is performed, it is possible not only to document either an increase or decrease in the depth distribution of a species but also to show an eradication of one or more species.

3.7 Qualitative descriptions of macroalgal vegetation - supportive guidance

Type 1. Archipelago of the west coast, inner parts

High - Algal vegetation undisturbed or insignificantly disturbed. Dense populations of bladderwrack *(Fucus vesiculosus)* and/or knotted wrack *(Ascophyllum nodosum)*. Any epiphytes consists mainly of brown and red algae and only in a few cases of green algae or a small number of filter feeders. The sub-vegetation is varied. At exposed locations, the bladderwrack can lack bladders. Deeper down there is toothed wrack (*Fucus serratus*), *Halidrys siliquosa* and kelp *(Laminaria spp)*. Deeper still there is either a belt of red seaweed *(Furcellaria lumbricalis)* alternatively a species-rich red algae community with species such as the red algae *Coccotylus truncatus/Phyllophora pseudoceranoides*, the red macroalgae *Delesseria sanguinea, Phycodrys rubens* and *Rhodomela confervoides*).

Good - Algal vegetation is slightly disturbed. Continued dense populations of bladderwrack *(Fucus vesiculosus)* and/or knotted wrack (*Ascophyllum nodosum)*. The epiphytes consist of brown and red algae as well as a small amount of green algae and filter feeders. The sub-vegetation is varied. At exposed locations, the bladderwrack can lack bladders. Deeper down there is toothed wrack (*Fucus serratus*), *Halidrys siliquosa* and kelp *(Laminaria spp.).* Deeper still there is either a belt of red seaweed *(Furcellaria lumbricalis)* alternatively a species-rich red algae community with species such as the red algae *Coccotylus truncatus* and *Phyllophora pseudoceranoides*, the red macroalgae *Delesseria sanguinea, Phycodrys rubens* and *Rhondomela confervoides*. Kelp species, sea oak (*Halidrys siliquosa*) and red algae species don't extend quite as deep as in high-status areas (no or only negligible disturbance).

Moderate - Clearly disturbed algal vegetation. Sparse populations of bladderwrack and/or knotted wrack grow together with green algae. Purple laver (*Porphyra purpurea*) can be common at certain times of the year. Seaweed plants are overgrown with green algae and/or filter feeders. The deepest-growing plants of sugar wrack (*Laminaria saccharina*) are found at 4-5 m. The maximum depth distribution of the red algae species, *Rhodomela confervoides* and *Furcellaria lumbricalis*, is around 7 m. Filamentous and leafy species are more common, e.g. *Ceramium nodulosum* and *Polysiphonia stricta*. The total number of species is lower and several sensitive species have disappeared compared to good status.

Poor - Substantially disturbed algae communities. The deep species-rich red algae community has been eradicated. A few specimens of bladderwrack grow in shallow waters, often very overgrown with algae and filter-feeding animals. The most common epiphytic algae are various green algae species, *Enteromorpha spp*. and *Cladophora spp*. Among filter-feeding epiphytic animals, different bryozoans, blue mussels and barnacles sometimes dominate. Floating algae mats may be common. Number of species has decreased drastically compared to moderate status.

Bad - The perennial brown algae community has been eradicated. Very species-poor community. The algal vegetation is dominated by green algae. Common genera are *Enteromorpha spp.* and *Cladophora spp*. Floating algae mats are common. In certain cases there are only cyanobacteria (often mistakenly called blue-green algae) and other bacteria.

Types 3 and 4. Archipelago of the west coast, outer parts

High - Algal vegetation is undisturbed or only negligibly disturbed. The upper algal vegetation consists of short, annual, filamentous macroalgae. Different species replace each other during the year. During the summer, the upper belt consists of several red algae species. Kelp species (*Laminaria spp.*) occur in deeper water. Deeper down there is either a belt of red seaweed *(Furcellaria lumbricalis)* alternatively a species-rich red algae community with species such as the red algae *Coccotylus truncatus/Phyllophora pseudoceranoides,* the red macroalgae *Delesseria sanguinea, Phycodrys rubens* and *Rhodomela confervoides.* Erect macroalgae found deeper than 25 m.

Good - Algal vegetation is slightly disturbed. The upper algal vegetation consists of short, annual, filamentous macroalgae. Different species replace each other during the year. During the summer, the upper belt consists of several red algae species. Kelp species (*Laminaria spp*.) occur in deeper water. Deeper down there is either a belt of red seaweed (*Furcellaria lumbricalis*) or a species-rich red algae community with species such as the red algae *Coccotylus truncatus*/ *Phyllophora* p*seudoceranoides*, the red macroalgae *Delesseria sanguinea*, *Phycodrys rubens* and *Rhodomela confervoides.* Kelp species and red algae species don't extend quite as deep as in high-status areas (no or only negligible disturbance). Erect macroalgae found down to a depth of 20 m.

Moderate - Algal vegetation is clearly disturbed. The upper algal vegetation consists of short, annual, filamentous macroalgae. Different species replace each other during the year. During the summer, the upper belt consists of several red algae species. The deepest growing plants of sugar wrack (*Laminaria saccharina*) found at 4-6 m. The maximum depth distribution of the red algae species, *Rhodomela confervoides* and *Furcellaria lumbricalis*, is around 8 m. Erect macroalgae occur down to a depth of 10-15 metres. The total number of species is lower than in good-status areas.

Poor - Algal vegetation is substantially disturbed. Erect macroalgae occur down to a depth of 5 metres. The number of species has decreased drastically, especially perennial brown and red algae species. Short-lived, filamentous and leafy species dominate.

Bad - The perennial brown algae community has been eradicated. The number of species is low. Short-lived, filamentous and leafy species occur to a maximum depth of 1-2 metres.

Types 8, 10 and 12. Hard bottoms in the coastal waters of the Baltic Proper, middle parts

High - Algal vegetation is undisturbed or only negligibly disturbed. Dense community-forming *Fucus* populations occur from a depth of 0.5 to 3-4 metres at normal water levels. The maximum depth distribution for bladderwrack is around 6-8 metres. In the southern Baltic Sea (Blekinge archipelago and Kalmarsund), toothed wrack (*Fucus serratus*) normally make up the lower boundary of the seaweed belt. The sub-vegetation consists of red and brown algae and closest to the surface of the green algae genera *Enteromorpha spp.* and *Cladophora spp.* During summer-autumn, the red macroalgae *Ceramium tenuicorne* and *Polysiphonia fucoides* are common. Under the seaweed belt, there is a species-rich community consisting of the red algae *Coccotylus* and *Phyllophora*, *Rhodomela confervoides*, the brown algae *Sphacelaria spp*. and *Chorda spp*. and the green alga *Cladophora rupestris*. Macroalgal vegetation extends down to a depth of about 12-15 metres.

Good - Algae community is slightly disturbed. The bladderwrack is community-forming from 0.5 to 2-3 metres deep. The maximum depth distribution for bladderwrack is around 6-8 metres deep. The seaweed plants have brown algae epiphytes that mostly remain throughout the year. Common epiphytic fauna are *Electra crustulenta* and *Balanusimprovisus*. Under the seaweed, there is a species-rich community consisting of the red algae *Coccotylus* and *Phyllophora, Rhodomela confervoides* , the brown algae *Sphacelaria spp.* and *Chorda spp*. and the green alga *Cladophora rupestris*. *Coccotylus*, *Phyllophora* and *Furcellaria lumbricalis* are less common than in high-status areas. Macroalgal vegetation extends down to a depth of about 10-12 metres.

Moderate - Algal vegetation is clearly disturbed. Sparse populations of bladderwrack from about 0.5 to 2-3 metres deep. The seaweed is substantially overgrown with *Electra crustulenta, Balanus improvisus* and blue mussels (*Mytilus edulis*). The algal epiphytes consist of filamentous brown, red and green algae. Green algae dominate from the surface to a depth of a few metres. There are less macroalgae species than in good-status areas. Several of the more sensitive red algae species, e.g. *Coccotylus/Phyllophora*, are less common as is *Sphacelaria arctica*, whilst *Enteromorpha* become more common. The maximum depth distribution of many of the perennial algae species is about 5-6 metres.

Poor - Algal vegetation is substantially disturbed. Bladderwrack found only in a sparse population in very shallow waters or has completely disappeared. Filamentous green algae (*Cladophora* g*lomerata*) and *Enteromorpha spp.* are common and partially replace each other during the growing season. The number of macroalgae species has decreased even further. The vegetation extends down to a depth of 3-4 metres.

Bad - No perennial macroalgae species. Very few macroalgae species found. Filamentous “fluff” of green algae and cyanobacteria, which lie in loose layers over the bottom. Rich in loose-lying algae. There is often a white powder/white mat of sulphur bacteria, especially in crevices containing dead algal material.

Types 14 and 15. Archipelago of the Baltic Proper, outer parts

High - Algal vegetation undisturbed or only negligibly disturbed. Dense, short bladderwrack *(Fucus vesiculosus)* at the surface, community-forming with 25-100% coverage. In the southern Baltic Sea (Blekinge archipelago and Kalmarsund), toothed wrack (*Fucus serratus*) normally make up the lower boundary of the seaweed community. No or very sparse epiphytes. The red alga *Ceramium tenuicorne* is common close to the surface and downwards together with *Furcellaria lumbricalis* that make up the normal sub-vegetation of bladderwrack. Maximum depth distribution of bladderwrack down to a depth of 7-9 metres. Under the seaweed, there is a species-rich community consisting of the red algae *Coccotylus* and *Phyllophora, Rhodomela confervoides,* the brown algae *Sphacelaria spp., Chorda spp.* and *Dictyosiphon foeniculaceus* and the green alga *Cladophora rupestris* also occur but are less common. The maximum depth distribution of the macroalgal vegetation is about 12-14 metres.

Good - Algal vegetation is slightly disturbed. The bladderwrack often has epiphytes of red and/or brown filamentous algae. The brown alga *Elachista fucicola* is particularly common. The maximum depth distribution of the seaweed is down to a depth of about 6-8 metres and many other common perennial species don't grow deeper than 10-12 metres.

Moderate - Algal vegetation is clearly disturbed. The bladderwrack is substantial overgrown with *Electra crustulenta* and *Balanus improvisus* and with filamentous algae in its upper regions. Brown algae and threadlike red algae dominate. Rich in blue mussels (*Mytilus edulis*) and *Hydrobia spp*. are common on the bladderwrack plants. Blue mussels are common from a depth of 3-5 metres and partially outcompete the macroalgae. The number of macroalgae species is less than in good-status areas. Several of the more sensitive red algae species have disappeared. The maximum depth distribution of many of the perennial algae species is about 6-8 metres.

Poor - Algal vegetation is substantially disturbed. Bladderwrack found only in a sparse population in very shallow waters or has completely disappeared. The filamentous brown algae *Pylaiella* and *Ectocarpus*, *Cladophora glomerata* and various *Enteromorpha* species dominate. The number of macroalgae species has decreased further. The vegetation extends down to a depth of 3-4 metres.

Bad -No perennial macroalgae species. Very few macroalgae species found. Filamentous "fluff" of green algae and cyanobacteria, which lie in loose layers over the bottom. Rich in loose-lying algae. There is often a white powder/white mat of sulphur bacteria, especially in crevices containing dead algal material.

Types 16, 17, 18 and 19. Coastal waters of the Bothnian Sea, inner and outer parts

High - Algal vegetation is undisturbed or only negligibly disturbed. The bladderwrack *(Fucus vesiculosus)* form a community from about 2 to about 6 metres. The deepest-growing plants are found at a depth of about 7-11 metres. Shallow-growing seaweed plants are found in crevices and in places without ice-scraping. Filamentous green algae such as *Cladophoraglomerata, Cladophora aegagrophila* and *Cladophora rupestris* dominate at the surface. *Chorda filium* also occurs here. Other common species are the red alga *Ceramium tenuicorne,* especially in outer wave-exposed areas and the brown alga *Pylaiella littoralis. Furcellaria lumbricalis* and *Coccotylus* occur. The brown alga *Sphacelaria arctica* grow deepest down to a depth of about 12-15 metres.

Good - Algal vegetation is slightly disturbed. The amount of filamentous brown, green and red algae increases and the species have a dense cover of diatoms. The maximum depth distribution of the bladderwrack decreases slightly as does *Sphacelaria arctica*, which occurs at a maximum depth of about 7-12 metres.

Moderate - Algal vegetation is clearly disturbed. The bladderwrack community has thinned out and the deepest-growing plants occur at about 2-6 metres. The number of macroalgae species is less than in good-status areas. Filamentous green algae substantially overgrown with diatoms dominate. *Sphacelaria arctica* also with diatom epiphytic growth occur to a maximum depth of 3-8 metres.

Poor - Algal vegetation is substantially disturbed. Bladderwrack found only in a sparse population in very shallow waters (0-3 metres) or has completely disappeared. The filamentous green algae *Cladophora glomerata* and *Cladophora aegagrophila* dominate, substantially overgrown with filamentous fluff and diatoms. Various *Enteromorpha* also occur. The number of macroalgae species has decreased further. The vegetation extends down to a depth of 3-4 metres.

Bad - Few macroalgae species found. The bottom is covered in long fluffy veils of filamentous green algae, e.g. various *Cladophora*, *Enteromorpha* and cyanobacteria.

Background report: Förslag till och vidareutveckling av bedömningsgrunder för kust och hav enligt krav i ramdirektivet vatten – Makroalger och några gömfröiga växter [Proposals for and further development of assessment criteria for coastal and sea areas in accordance with requirements in the WFD - Macroalgae and certain angiosperms]. Authors: Lena Kautsky, Cecilia Wibjörn and Hans Kautsky (STOCKHOLM UNIVERSITY)

4 Phytoplankton

Parameters	Shows primarily effects of	How often do measurements need to be taken?	At what time of the year?
Biovolume	Nutrient level/eutrophication	3-5 times/year	June – August
Chlorophyll *a*	Nutrient level/eutrophication	3-5 times/year	June – August

4.1 Introduction

Phytoplankton react rapidly to changes in nutrient load and are well suited for use as indicators of water quality change. Changes in the phytoplankton community are often the primary cause of disturbances in other biotopes. The amount of phytoplankton affects transparency and subsequently the vertical distribution of benthic algae communities. Greater production of phytoplankton is the main cause of impact to sediment-dwelling organisms, directly as a result of an increased availability of food and/or indirectly as a result of poorer oxygen conditions. They constitute the basis of all secondary production in the marine environment and changes in phytoplankton production affect e.g. fish and crustacean production, as a result of e.g. toxic algae bloom.

4.2 Parameters included

Status is classified based on the biomass of autotrophic and mixotrophic phytoplankton expressed as biovolume (mm^3/L) and chlorophyll *a* (µg/L). As biovolume and chlorophyll data is available, they should be cofactored into one standardised status classification for phytoplankton (Section 4.4.2). If there is no data for any of these parameters, the classification is based on the remaining parameter. Data for types 8, 12, 13 and 24 shall be corrected according to salinity prior to classification, see further in Section 4.4.4.

See REG Annex 4, Section 3.1 and 3.2

4.3 Data requirements

4.3.1 Biovolume

Classification of phytoplankton biovolume shall be based on data from integrated samples (tube sampling or composite samples taken with a water sampler at various depths from the surface layer 0-10m, or discrete samples from the surface (0.5 m) if the water depth is <12 m. Data from other depth intervals can be converted to 0-10m using the conversion factors in Table 4.1

See REG Annex 4, Section 3.3

4.3.2 Chlorophyll a

Chlorophyll shall be classified based on data from the same depth as the biovolume samples for the Skagerrak and Kattegat (types 1-7 and 25) and the Gulf of Bothnia (types 16-23). For the Baltic Proper (types 8-15 and 24), the classification shall be based on data from a depth of 0.5 m (see also comments in Section 4.6). Chlorophyll data from deviating sample depths needs to be corrected with empirical relationships so that they correspond to the above-specified depths and depth intervals (Table 4.1).

Table 4.1. The equations for converting chlorophyll a (C) from a depth of 0 m (surface, S) to integrated tube sample (T).

Conversion	Equation	R^2	Data
From 0 m to tube sample 0-10 m	C_T=0.7146×C_S+0.7903	0.66	Type 1, June to August
From 0 m to tube sample 0-10 m	C_T=0.6829×C_S+0.5565	0.87	Type 5, June to August
From 0 m to tube sample 0-10 m	C_T=0.7515×C_S+0.5107	0.82	Type 6, June to August
From tube sample 0-14/20 m to 0 m	C_S=1.2164×C_T+0.0422	0.72	Himmerfjärden (type area 12n) and Askö B1 (type 14), June to August
From 0 m to tube sample 0-14/20 m	C_T=0.5937×C_S+0.6753	0.72	Himmerfjärden (type area 12n) and Askö B1 (type 14), June to August
From tube sample 0-10 m to 0 m	C_S=1.2204×C_T-0.2865	0.92	Svealand Coastal Water Protection Society surveys July-Aug 2004-07, types 12n, 15, 16
From 0 m to tube sample 0-10 m	C_T=0.7573×C_S+0.51	0.92	Svealand Coastal Water Protection Society surveys July-Aug 2004-07, types 12n, 15, 16

4.3.3 Sampling frequency and methods

The assessment criterion applies to the period June-August. Samples shall be taken at least three but preferably five times a year, distributed equally over this time period. If data is only available for a limited period, it can, if it is important for classification purposes, be corrected in accordance with known conditions for how biovolume and chlorophyll normally vary over the period. For example, chlorophyll is often slightly lower in June than in July-August. Classification shall be performed based on data from at least three years from the latest six-year period in order to take annual variations into consideration. The necessary sampling frequency depends both on the size of the natural variation in the water body in question, and on the status of the water body. In general, the closer a value is to a class boundary, the more samples are required to guarantee a statistically reliable classification. Sampling intensity should be greatest when the value is close to the boundary between good and moderate. If the conditions are good or the disturbance is very clear, there is little point in being able to say this in even more certain terms. The possibility of using information from adjacent water bodies with similar conditions must also be considered.

See REG Annex 4, Section 3.3

The sampling station shall be representative of the water body. If there are gradients within a water body, several sampling points may be necessary, at least until supporting data for locating a single representative sampling point has been obtained. A detailed description of accepted sampling methods referred to in the regulations (NFS 2008:1) can be found in the Swedish EPA Environmental Monitoring Manual[15].

The assessment criterion for phytoplankton biovolume is based on the quantification and species identification of phytoplankton in Lugol's-preserved samples. The analysis is carried out using an approved light microscope in accordance with the Swedish EPA's survey types or HELCOM's COMBINE manual [16], both of which are based on the Utermöhl method. The method provides a lower size limit for counted phytoplankton of about 2 µm, which excludes single-cell picoplankton from the analysis. Colonies or filaments of small picoplankton (e.g. *Cyanodictyon, Snowella, Pseudanabaena*) shall, however, be counted. The biovolume is obtained by using the size-classes in Biovolumes and Size-Classes of Phytoplankton in the Baltic Sea[17] with the latest version of the associated Excel file. The Excel file will be regularly updated with relevant species names, new species and size-classes and will be available on the national host's (SMHI's) website under the name "Phytoplankton PEG biovolumes". Obligatory heterotrophic organisms in this list shall not be included when calculating biovolumes.

Standard methods are used to analyse chlorophyll *a*: Swedish standard (SS 02 81 46), that prescribes acetone as an extraction solvent, or HELCOM's COMBINE manual which prescribes ethanol for this purpose. In both methods, water is filtered through glass-fibre filters and extracted with the solvent before absorbance is measured in a spectrophotometer or fluorescence in a fluorometer, calibrated to a spectrophotometer. The values need not be converted but a method reference shall be submitted when reporting data.

4.4 Classification of status

4.4.1 Calculation of EQR and classification of status

Status classes for biovolume and chlorophyll *a* are calculated as follows:

1) For all types apart from 8, 12, 13 and 24, the ecological quality ratios (EQR) is calculated for each individual sample based on the reference values in tables 4.4-4.5, in accordance with EQR= (Reference value)/(Observed value). 0≤ EQR ≤ 1, i.e. EQR is set to a maximum of 1. For types 8, 12, 13 and 24, EQR is calculated for each individual sample according to Section 4.4.4.
2) The mean EQR value is calculated for each year and for each *sampling station*.
3) The mean EQR value is calculated for each year and *water body* based on representative stations.

See REG Annex 4, Section 3.4.1

[15] Survey types, Phytoplankton (www.naturvardsverket.se)

[16] www.helcom.fi

[17] Olenina et al. 2006, se the HELCOM website at http://www.helcom.fi/groups/monas/en_GB/biovolumes

4) Mean EQR value is calculated based on data from at least 3 years from the latest six-year period.
5) Status is classified by comparing the multi-year mean EQR value with the specified EQR class boundaries in tables 4.4-4.5.
6) If EQR is calculated for both biovolume and chlorophyll, the EQR is cofactored in accordance with the description below (Section 4.4.2) for final status classification.

4.4.2 Cofactoring of EQRs for biovolume and chlorophyll *a*

Step 1) The cofactoring shall be based on the classified status for biovolume and chlorophyll *a*. The status classes are given a numerical value in accordance with Table 4.2. A weighted class value is calculated for each parameter using formula 4.1 before the EQR is cofactored in accordance with Step 2.

See REG Annex 4, Section 3.4.3

Table 4.2. Division of the status classes into numerical values.

Status	Numerical value
High status	4 - 4.99
Good status	3 - 3.99
Moderate status	2 - 2.99
Unsatisfactory status	1 - 1.99
Bad status	0 - 0.99

The numerical class (N_{class}) for the respective parameters for the relevant EQR class interval (EQR_{lower}–EQR_{upper}) is calculated using formula 4.1.

$$(N_{class}) = (N_{lower}) + (EQR_{calculated} - EQR_{lower})/(EQR_{upper} - EQR_{lower})$$

Formula 4.1.
(N_{class}) = weighted status class value for each parameter.
N_{lower}= the first integer in the numerical values for the status class in accordance with Table 4.2.
$EQR_{calculated}$= EQR value calculated from the classification.
EQR_{lower} and EQR_{upper} = EQR for lower and upper class boundary for the corresponding class, taken from Tables 4.4-4.5 below. EQR_{lower} for bad status = 0 and EQR_{upper} for high status = 1.

Step 2) Calculate the mean value of the numerical classifications (N_{class}) for biovolume and chlorophyll *a*, which results in the cofactored phytoplankton classification. The status classification is determined by the mean value for the numerical classification according to Table 4.2.

4.4.3 A calculation example for phytoplankton in type 9

1) Reference value in type 9 for biovolume is 0.18 and for chlorophyll a 1.2. Assume that the observed values for chlorophyll and biovolume are as follows:

	Biovolume			Chlorophyll *a*		
	June	**July**	**August**	**June**	**July**	**August**
2003	0.27	0.25	0.24	2.2	1.7	1.9
2004	0.29	0.25	0.24	2.3	1.7	1.9
2005	0.28	0.23	0.23	2	1.6	1.8

Translate all observed biovolume and chlorophyll values to EQR values. Biovolume EQR for type 9 is calculated as EQR=0.18/(observed biovolume). For chlorophyll, EQR is calculated as EQR=1.2/(observed chlorophyll value).

2a) Biovolume values converted to EQR values
2003: 0.67; 0.71 and 0.75 gives a mean value of 0.71
2004: 0.62; 0.72 and 0.76 gives a mean value of 0.70
2005: 0.64; 0.77 and 0.77 gives a mean value of 0.73

3) Mean biovolume EQR value for a three-year period: (0.71+0.70+0.73)/3 = 0.71

4) 0.71 corresponds to good status (Table 4.4) which gives N_{lower} = 3 (Table 4.2)

5) Numerical class biovolume: $(N_{class})=(N_{lower})+(EQR_{estimated}-EQR_{lower})/(EQR_{upper}-EQR_{lower})$
EQR boundaries for good status class in type 9 is 0.56-0.72, which gives:

$$(N_{class}) = 3 + (0.71-0.56)/(0.72-0.56) = 3.94$$

2b) Chlorophyll *a* values converted to EQR values
2003: 0.54; 0.71 and 0.64 gives a mean value of 0.63
2004: 0.52; 0.69 and 0.62 gives a mean value of 0.61
2005: 0.59; 0.73 and 0.67 gives a mean value of 0.66

3) Mean chlorophyll a EQR value for a three-year period: 0.63+0.61+0.66 = 0.63

4) EQR 0.63 corresponds to moderate status (Table 4.5) which gives N_{lower} = 2 (Table 4.2)

5) Numerical class chlorophyll *a* (N_{class})= 2 + (0.63-0.35)/(0.67-0.35) = 2.87

6) **Cofactoring**: Mean value of the numerical classes for biovolume and chlorophyll *a*: (3.94+2.87)/2= **3.4** which results in the water body being classified as good status for the phytoplankton quality element (good status=3-3.99 according to Table 4.2).

4.4.4 Calculation when classifying in salinity gradients – in types 8, 12, 13 and 24

In types 8, 12, 13 and 24, data on salinity is always required together with the chlorophyll and biovolume data that is to be classified. A reference value for total nitrogen is calculated based on the degree of freshwater impact, determined from the salinity level, and the reference values for total nitrogen in the sea and in freshwa-

See REG Annex 4, Section 3.4.2

ter discharge. Using empirical relationships with total nitrogen, corresponding reference values for chlorophyll and biovolume are calculated. These are used to calculate EQR values for the salinity-dependent types 8, 12, 13 and 24. The calculations are described theoretically step-by-step under points 1-4 below. An Excel application has been developed (available at www.naturvardsverket.se) to facilitate calculations from a purely practical point of view. Measurement data is entered into the application, the type is chosen and the results can be seen in the form of EQR values for the measurements.

Alternatively, observed values for chlorophyll and biovolume can be directly compared to class boundaries for various salinity intervals for each type respectively according to Tables 4.6 - 4.7 in Section 4.5.3. These tables are, however based on nominal (approximate) open sea salinity levels (point 1 below) which causes uncertainty about the degree of freshwater impact but can be used when there is no measurement data on open sea salinity levels.

The procedure for classifying chlorophyll and biovolume in salinity gradients described in points 3-4 below has been adapted to the procedure for classifying nutrients. The method is based on an upward adjustment of the class boundaries for outer coastal waters or the open sea in order to obtain the same EQR at the class boundaries in the entire salinity gradient. This results in more tolerant class boundaries than if the measurement values are instead corrected for the actual natural nitrogen discharge and then classified in accordance with the boundaries for the outer coastal waters. How to classify using this "correction method" alternative is described in connection with the Excel application (www.naturvardverket.se) that can be used to calculate the outcome of both nutrients and phytoplankton. The correction method can also be used to calculate open sea corrections. Using these, the generally deteriorated status in the open sea can be considered to provide a basis for where local measures can be necessary to achieve good status. The method for calculating open-sea corrections is described under point 5 below.

1. Calculation of freshwater impact coefficients

The degree of freshwater impact is calculated based on the observed salinity level (S) in the water body to be classified and on a comparative salinity level (S_{sea}) from the open sea or outer coastal waters that are affected only negligibly by local freshwater discharge. The comparative salinity level should be measured simultaneously. A nominal (approximate) salinity level can be used if there is no observed comparative salinity level. ($S_{sea} \approx 7$ for type 8, and $S_{sea} \approx 6$ for type 12, 13 and 24, calculated values can be found tabulated in Tables 4.6 - 4.7, Section 4.5.3). The nominal open-sea salinity level means, however, that the freshwater impact estimate will be more unreliable than when observed open-sea salinity levels are used.

The salinity correction coefficient, S_f, is calculated for each measurement occasion (for each chlorophyll and biovolume value) where:

$$S_f = (S_{sea}-S)/S_{sea} \qquad 0 \leq S_f \leq 1$$

If salinity in the area to be classified is as high as the comparative salinity level, the salinity correction coefficient will be 0. If it is pure freshwater, the coefficient will be 1. If the observed salinity level is higher than the comparative salinity level, the salinity correction coefficient will be set to 0. A higher salinity level in inner coastal waters may depend on the upwelling of deep waters and this should be noted when classifying since it may explain e.g. increased phosphorus levels.

2. Calculation of reference values for total nitrogen

The reference level for TN at a certain salinity level (TN_{refSf}) is calculated as follows:

$$TN_{ref\,Sf} = TN_{ref\,sea} + S_f \times (TN_{ref\_freshw} - TN_{ref\,sea})$$

where TN_{ref_sea} is the reference value for total nitrogen in the open sea, TN_{ref_freshw} is the reference value in freshwater, and S_f is the salinity correction coefficient.

3. Calculation of reference values for chlorophyll a, biovolume and transparency

Reference values for chlorophyll at certain salinity levels ($CPHYLL_{refSf}$) is calculated as follows:

$$CPHYLL_{refSf} = A \times (TN_{refSf})^{B}$$

where TN_{refSf} is the reference value for TN at certain salinity levels and A and B are from empirically found relationships between chlorophyll and total nitrogen. The reference values for transparency ($TRANS_{refSf}$) and phytoplankton biovolume ($BIOV_{refSf}$) are calculated using the corresponding empirical relationships (Table 4.3).

Table 4.3. Equations used when correcting reference values These are assumed to apply to types 8, 12, 13 and 24 in the Baltic Proper. Fixed boundaries within each type are used for other types, i.e. no correction is made for nutrient discharge corresponding to reference levels in inflowing freshwater. A and B are inserted into the equations under the heading "Relationship" at the given position.

Relationship	A	B	Reference
Chlorophyll *a* (µg/l)= A×TN (µmol/l) ^{B}	0.0051	1.9974	Larsson et al. 2006
Transparency (m) = A×TN (µmol/l) ^{B}	1023.3	-1.696	Larsson et al. 2006
Biovolume (mm^3/l) = A×TN (µmol/l) ^{B}	1.05×10^{-4}	2.6878	Larsson et al. 2006

4. ***Calculation of EQR for chlorophyll a, biovolume and transparency***

EQR for chlorophyll is calculated as follows:

$$EQR_{Cphyll} = CPHYLL_{refSf}/CPHYLL_{obs}$$

where $CPHYLL_{obs}$ is the observed chlorophyll level to be classified. The corresponding calculation for biovolume will be:

$$EQR_{Biov} = BIOV_{refSf}/BIOV_{obs}$$

and for transparency

$$EQR_{Trans} = TRANS_{obs} / TRANS_{refSf}$$

$0 \leq EQR \leq 1$, i.e. EQR is set to a maximum of 1.

5. Calculation of open-sea corrections for the planning of measures

A general deterioration in the open sea will also have an effect on the conditions in the archipelago. To facilitate the assessment of local impact on coastal waters, the open sea reading can be ignored. Areas within the drainage basin where measures are needed to achieve good status can thereby be identified. This shall not be applied in the assessment criterion when classifying status, but can be used as an aid when designing a programme of measures. At full salinity, the open-sea correction coefficient corresponds to the difference between <u>observed</u> TN in the open sea (TN_{obs_sea}) and the reference level in the open sea (TN_{ref_sea}). For freshwater-affected areas, with salinity level S, the open-sea correction ($TN_{sea\_correction}$) will be less and can be calculated as follows:

$$TN_{sea\_correction} = S/S_{sea} \times (TN_{obs\_sea} - TN_{ref\_sea})$$

where S_{sea} is the open-sea salinity level.

For chlorophyll, biovolume and transparency, the open-sea corrections are calculated as follows:

$$CPHYLL_{sea\_correction} = A(TN_{obs})^B - A(TN_{obs} - TN_{sea\_correction})^B$$

$$BIOV_{sea\_correction} = A(TN_{obs})^B - A(TN_{obs} - TN_{sea\_correction})^B$$

$$TRANS_{sea\_correction} = A(TN_{obs} - TN_{sea\_correction})^B - A(TN_{obs})^B$$

where the values for A and B come from corresponding empirical relationships with total nitrogen according to Table 4.3.

<u>The open-sea corrections are subtracted from observed values (TN_{obs}, $CPHYLL_{obs}$, $BIOV_{obs}$, $TRANS_{obs}$) prior to classification.</u>

This open-sea corrected classification provides an indication of the extent to which local discharge affects water quality and can constitute a basis for the design of any measures.

4.5 Reference values and class boundaries

The tables below give reference values and class boundaries for biovolume and chlorophyll and salinity-related values for the salinity-dependent types 8, 12, 13 and 24. The values in Tables 4.6-4.7 are based on nominal (approximate) open-sea salinity levels which leads to uncertainty about the degree of freshwater impact but can be used when there is no measurement data on open-sea salinity levels. Reference values (Rv) and EQR values shall be used to calculate status classes. Class boundaries for absolute values are given below only to obtain a general idea of where the status boundaries are.

4.5.1 Biovolume

Table 4.4. Reference values (Rv) and class boundaries (HG, GM, MO, OD) and corresponding EQRs for summertime (June-Aug) biovolumes of phytoplankton (mm^3/L). Grey figures signify that the reference values shall be corrected based on observed salinity levels before calculating EQRs and comparing them with EQR class boundaries (see Table 4.6 a-c).

See REG Annex 4, Section 3.5.1

Type area	Biovolume ^{3}/l)					Biovolume EQR			
	Rv	**HG**	**GM**	**MO**	**OD**	**HG**	**GM**	**MO**	**OD**
Skagerrak and Kattegat									
1n	0.8	1.2	1.55	3.1	6.1	0.67	0.52	0.26	0.13
1s	0.9	1.3	1.7	3.3	6.6	0.69	0.53	0.27	0.14
2	1.35	2.0	3.0	4.5	7.95	0.68	0.45	0.3	0.17
3	0.8	1.2	1.55	3.1	6.1	0.67	0.52	0.26	0.13
25	1.4	2.1	2.75	4.8	8.35	0.67	0.51	0.29	0.17
4	0.5	0.75	1.1	2.25	6.1	0.67	0.45	0.22	0.08
5	0.7	1.2	2.1	4.2	7.3	0.58	0.33	0.17	0.1
6	0.25	0.4	0.75	2.4	4.9	0.63	00.33	0.1	0.05
Baltic Proper									
7	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
8	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
9	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
10	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
11	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
12	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
13	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
14	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
15	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
24	0.18	0.25	0.32	0.74	2.26	0.72	0.56	0.24	0.08
Bothnian Sea									
16	0.21	0.32	0.47	0.87	2.64	0.66	0.45	0.24	0.08
17	0.18	0.27	0.4	0.74	2.26	0.67	0.45	0.24	0.08
18	0.21	0.32	0.47	0.87	2.64	0.66	0.45	0.24	0.08
19	0.18	0.27	0.4	0.74	2.26	0.67	0.45	0.24	0.08

Bothnian Bay									
20	0.16	0.25	0.37	0.67	2.05	0.64	0.43	0.24	0.08
21	0.15	0.27	0.4	0.74	2.26	0.56	0.38	0.2	0.07
22	0.16	0.25	0.37	0.67	2.05	0.64	0.43	0.24	0.08
23	0.15	0.27	0.4	0.74	2.26	0.56	0.38	0.2	0.07

4.5.2 Chlorophyll *a*

Table 4.5. Reference values (Rv), class boundaries (HG, GM, MO, OD) and corresponding EQR for summer levels for chlorophyll a (µg/L). Grey figures signify that the reference values shall be corrected based on observed salinity levels before calculating EQRs and comparing them with the EQR class boundaries (see Table 4.7 a-c).

See REG Annex 4, Section 3.5.2

Type area	**Chlorophyll *a* (µg /l)**					**Chlorophyll *a* EQR**			
	Rv	**HG**	**GM**	**MO**	**OD**	**HG**	**GM**	**MO**	**OD**
Skagerrak and Kattegat									
1n	1.3	1.7	2.1	3.7	6.7	0.76	0.62	0.35	0.19
1s	1.6	2.1	2.8	4.6	8.0	0.76	0.57	0.35	0.2
2	1.9	2.4	3.6	5.6	8.3	0.79	0.53	0.34	0.23
3	1.1	1.4	1.8	3.5	6.2	0.79	0.63	0.31	0.18
25	1.8	2.1	2.7	4.1	6.5	0.86	0.67	0.44	0.28
4	1.0	1.2	1.5	3.0	6.0	0.83	0.67	0.33	0.17
5	1.0	1.2	1.5	3.0	6.0	0.83	0.67	0.33	0.17
6	0.9	1.1	1.5	2.4	4.9	0.82	0.59	0.37	0.18
Baltic Proper									
7	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
8	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
9	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
10	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
11	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
12	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
13	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
14	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
15	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
24	1.2	1.5	1.8	3.4	8.0	0.8	0.67	0.35	0.15
Bothnian Sea									
16	1.4	1.8	2.3	4.3	10.1	0.78	0.61	0.33	0.14
17	1.2	1.5	2.0	3.7	8.7	0.8	0.6	0.32	0.14
18	1.4	1.8	2.3	4.3	10.1	0.78	0.61	0.33	0.14
19	1.2	1.5	2.0	3.7	8.7	0.8	0.6	0.32	0.14

Bothnian Bay									
20	1.2	1.8	2.3	4.3	10.1	0.67	0.52	0.28	0.12
21	1.1	1.5	2.0	3.7	8.7	0.73	0.55	0.3	0.13
22	1.2	1.8	2.3	4.3	10.1	0.67	0.52	0.28	0.12
23	1.1	1.5	2.0	3.7	8.7	0.73	0.55	0.3	0.13

4.5.3 Approximate reference values and class boundaries in salinity gradients based on nominal open-sea salinity levels

The tables below can be used to roughly classify chlorophyll and biovolume in types 8, 12, 13 and 124 based on salinity levels. The tables are based on the assumption of certain nominal open-sea salinity levels which means that the freshwater impact estimate will be more uncertain than if the calculation is made based on observed open-sea salinity levels. The tables can be used when there is no measurement data for open-sea salinity levels. Calculations using both observed and nominal open-sea salinity levels can be performed in the Excel application (mentioned in Section 4.4.4). The values are based on the assumption that the same EQR boundaries apply in the entire gradient.

Biovolume

Table 4.6 a. Salinity intervals with reference values and class boundaries for biovolume (mm^3/l) in **types 12n and 24**, based on an assumed nominal open-sea salinity level of 6 and a reference value for total nitrogen in freshwater of 23 µmol/l. The calculations have been performed in accordance with Section 4.4.4. The class boundaries for EQR in Table 4.4 apply to classification.

Salinity intervals		Rv	HG	GM	MO	OD
0-	1	0.44	0.62	0.84	2.15	5.78
1-	2	0.38	0.53	0.71	1.82	4.90
2-	3	0.32	0.44	0.59	1.53	4.11
3-	4	0.26	0.37	0.49	1.27	3.40
4-	5	0.21	0.30	0.40	1.03	2.78
5-	6	0.17	0.24	0.32	0.83	2.23
>6		0.18	0.25	0.32	0.74	2.26

Table 4.6 b. Salinity intervals with reference values and class boundaries for biovolume ($mm^3/$) in **types 12s and 13**, based on an assumed nominal open-sea salinity level of 6 and a reference value for total nitrogen in freshwater of 34 µmol/l. The calculations have been performed in accordance with Section 4.4.4. The class boundaries for EQR in Table 4.4 apply to classification.

Salinity interval		Rv	HG	GM	MO	OD
0-	1	1.21	1.69	2.28	5.86	15.75
1-	2	0.92	1.28	1.73	4.45	11.95
2-	3	0.67	0.94	1.27	3.27	8.78
3-	4	0.48	0.67	0.90	2.31	6.20
4-	5	0.32	0.45	0.60	1.55	4.16
5-	6	0.20	0.28	0.38	0.97	2.60
>6		0.18	0.25	0.32	0.74	2.26

Table 4.6 c. Salinity intervals with reference values and class boundaries for biovolume (mm^3/l) in **type 8**, based on an assumed nominal open-sea salinity level of 7 and a reference value for total nitrogen in freshwater of 59 µmol/l. The calculations have been performed in accordance with Section 4.4.4. The class boundaries for EQR in Table 4.4 apply to classification.

Salinity interval		Rv	HG	GM	MO	OD
0-	1	5.21	5.73	6.28	8.79	13.62
1-	2	3.78	4.21	4.66	6.77	10.93
2-	3	2.63	2.97	3.33	5.07	8.60
3-	4	1.72	1.98	2.27	3.67	6.60
4-	5	1.04	1.24	1.45	2.53	4.93
5-	6	0.56	0.70	0.85	1.65	3.55
6-	7	0.25	0.34	0.43	0.99	2.44
>7		0.18	0.25	0.32	0.74	2.26

Chlorophyll *a*

Table 4.7 a. Salinity level intervals with reference values and class boundaries for chlorophyll (µg/l) in **types 12n and 24**, based on an assumed nominal open-sea salinity level of 6 and a reference value for total nitrogen in freshwater of 23 µmol/l. The calculations have been performed in accordance with Section 4.4.4. The class boundaries for EQR in Table 4.4-5 apply to classification.

Salinity interval		Rv	HG	GM	MO	OD
0-	1	2.5	3.2	4.0	8.2	17.0
1-	2	2.2	2.9	3.6	7.2	15.0
2-	3	2.0	2.5	3.1	6.3	13.2
3-	4	1.7	2.2	2.7	5.5	11.5
4-	5	1.5	1.9	2.3	4.7	9.9
5-	6	1.2	1.6	2.0	4.0	8.4
>6		1.2	1.5	1.8	3.2	8.0

Table 4.7 b. Salinity level intervals with reference values and class boundaries for chlorophyll (µg/l) in **types 12s and 13**, based on an assumed nominal open-sea salinity level of 6 and a reference value for total nitrogen in freshwater of 34 µmol/l. The calculations have been performed in accordance with Section 4.4.4. The class boundaries for EQR in Table 4.4-5 apply to classification.

Salinity interval		Rv	HG	GM	MO	OD
0-	1	5.3	6.8	8.5	17.2	35.8
1-	2	4.3	5.6	6.9	14.0	29.2
2-	3	3.4	4.4	5.5	11.1	23.2
3-	4	2.7	3.4	4.3	8.6	17.9
4-	5	2.0	2.5	3.2	6.4	13.3
5-	6	1.4	1.8	2.2	4.5	9.4
>6		1.2	1.5	1.8	3.2	8.0

Table 4.7 c. Salinity level intervals with reference values and class boundaries for chlorophyll (µg/l) in **type 8**, based on an assumed nominal open-sea salinity level of 7 and a reference value for total nitrogen in freshwater of 59 µmol/l. The calculations have been performed in accordance with Section 4.4.4. The class boundaries for EQR in Table 4.4-5 apply to classification.

Salinity interval		Rv	HG	GM	MO	OD
0-	1	15.7	20.2	25.2	50.9	106.2
1-	2	12.4	15.9	19.9	40.1	83.7
2-	3	9.5	12.2	15.2	30.6	63.8
3-	4	6.9	8.9	11.1	22.4	46.6
4-	5	4.8	6.1	7.6	15.4	32.2
5-	6	3.0	3.9	4.8	9.8	20.4
6-	7	1.7	2.1	2.7	5.4	11.2
>7		1.2	1.5	1.8	3.2	8.0

4.6 Comments

4.6.1 Status based on species composition

Even if only biovolume status is to be classified, valuable information about species composition is also obtained when analysing phytoplankton. This information relates primarily to the occurrence of potentially toxic algae and algal blooms. Knowledge about species composition provides support to the interpretation of the status classification performed based on biovolume and chlorophyll. For example, the occurrence of large amounts of *Nodularia spumi gena* is often a sign that blooms from the open Baltic Proper affect the coastal waters, whilst substantial occurrence of e.g. *Planktothrix agardhii* often points to local nutrient disturbance. This is important to bear in mind when designing measures. Species composition analysis is also important in order to compile the necessary data for a future assessment criterion also based on species composition. It is beneficial to report (as abundance per species and size class) and input species composition data from the biovolume analysis into a database, i.e. send it to data hosts. The appurtenant regulations (NFS 2008:1) do not require information on species composition to be reported but it can also be of help when making an expert judgement.

4.6.2 Sampling

Integrated sampling of phytoplankton biovolume using a 0-10 m tube is justified given the fact that phytoplankton is not equally distributed in the trophogenic zone and given the importance of knowledge about species composition. There is a considerable risk of incorrectly estimating abundance and missing species that mainly live deeper in the water mass if only a surface water sample (0.5 m) is taken.). The reason why 10 metres is chosen as the lower limit is because most of the water column's phytoplankton can be found at between 0 and 10 metres but this is a compromise since many species occur deeper, some only at depths below 10 m.. Some stations with long data series (e.g. certain national environmental monitoring stations) have therefore sampled the entire trophogenic zone (e.g. 0-20, 0-14 m). It is recommended that the sampling depth at these stations remains unchanged in order not to break the continuity in valuable series and to obtain information about species that live at greater depths. Data from these is instead converted to apply to 0-10 metres.

The assessment criterion for chlorophyll currently applies to the 0-10 m layer in the Skagerrak and Kattegat and in the Gulf of Bothnia but to the surface zone (0.5 m) in the Baltic Proper, based on the data available when the criterion was developed. Since phytoplankton samples are taken in the Baltic Proper, it is desirable to supplement chlorophyll samples at the surface with samples at the same depth intervals as for biovolume. This provides empirical data for the estimation of biovolume at stations that only measure chlorophyll and data on which to base studies of suitable future sampling depths.

The sampling frequency in the chapter's initial table is based on simple statistical analyses of a limited amount of data. The assessment criteria for the different types are based on data collected using different methods, which is unfortunate, but has been necessary in order to obtain a sufficient volume of data. Before new sampling programmes are started in the next water planning cycle, it is important to review sampling designs and methods (depth, frequencies, etc.) for chlorophyll and phytoplankton together with other related parameters, e.g. nutrients and oxygen. This will allow data to be obtained that can form the basis of assessment criteria revisions. It is important the sampling is designed so that all the information in each sample is used in the best possible way in order to minimise the number of samples and the sampling frequency.

Even if it is desirable to collect data at the specified frequency and in the specified way, data collected less frequently is also useful. To facilitate the use of data collected using different methods, factors for converting data between the different tube lengths and surface samples have been developed (Table 4.1). Data collected at a frequency of as little as once per summer has, in a small number of cases where comparison has been possible, tallied relatively well with classifications based on the above-specified frequency. The results are significantly more unreliable, however. This causes problems, especially close to the boundary between good and moderate status, since there is a significantly greater risk of misclassification.

4.6.3 Classification in salinity gradients (according to Section 4.4.4)

When classifying based on salinity in certain inner types, model-estimated reference values for total nitrogen in freshwater are used. These reference values vary considerably for different drainage basins and must be considered unreliable. High reference values in freshwater involve significant corrections of all the reference values in the salinity gradient at freshwater impact. If the reference values for freshwater are overestimated or if there is local variation within the drainage basin, this can give rise to overcorrection and thereby underestimation of the local impact on water quality and vice versa. A reasonability assessment of the results must be performed as a result of this uncertainty before classifications are established, e.g. based on observed total nitrogen levels in inflowing freshwater and yearly variation. It is appropriate to develop locally adapted reference values in freshwater and an uncertainty estimate for the reference values in freshwater to be used as an aid when performing this reasonability assessment.

Background report: Bedömningsgrunder för kust och hav - Växtplankton [Assessment criteria for coastal waters and seas - Phytoplankton]
Authors: Ulf Larsson, Susanna Hajdu, Jakob Walve, (Stockholm University) Agneta Andersson,
Peder Larsson (Umeå University) and Lars Edler (SMHI)

5 Transparency

Quality element	Shows primarily effects of	How often do measurements need to be taken?	At what time of the year?
Transparency	Nutrient level/eutrophication	Once/month	June – August

5.1 Introduction

In general there is a clear link between transparency and chlorophyll levels. A lower transparency during the summer is often caused by an increased amount of particles in the water in the form of plankton in the upper water mass. In many areas, transparency can therefore provide a good indication of the biomass in the surface layer. Reduced transparency can also be caused by high levels of humus and particulate matter carried in dense run-off from land. Transparency should therefore be used with a certain amount of caution in areas where there is substantial freshwater impact, especially types 25 and 2 and should be compared with salinity and chlorophyll in order to determine the origin of the water mass and the effect from phytoplankton. Poor transparency can also occur in shallow areas due to resuspension of bottom material, which is dependent on meteorological conditions.

5.2 Data requirements

Transparency status shall be classified based on monthly data from the period June-August. If there is no data from this period, data from September can also be used. Transparency measurements are sensitive to meteorological conditions in rough seas and it is therefore inappropriate to specify precision with less certainty than 0.5 metres. Classification shall be performed based on data that has been sampled in accordance with HELCOM's COMBINE Manual[18]. Transparency can also be measured monthly as a support to other quality elements.

See REG Annex 5, Section 1.1

5.3 Classification of status

For transparency, EQR is calculated as follows:

See REG Annex 5, Section 1.2 and 1.3

$$\mathrm{EQR} = \frac{observedvalue}{referencevalue}$$

The reference value for each type can be derived from Table 5.1. Salinity correction of types 8, 12, 13 and 24 shall be performed prior to classification (see Section 4.4.4 for calculation, or alternatively Table 5.2 a-c). Classification shall be performed based on the mean value of all EQR values for the water body, which are

[18] www.helcom.fi

then compared to the EQR class boundaries in Table 5.1 in order to determine status.

5.4 Reference values and class boundaries

To calculate status classification, use the reference values (Rv) and EQR values in Table 5.1. For those types with strong salinity gradients (types 8, 12, 13 and 24), salinity-related reference values and class boundaries are specified in Table 5.2 a-c, so that the reference values for the specific salinity level observed when sampling are used. Alternatively, salinity-specific reference values can be calculated based on methods described in Section 4.4.4 if reference open-sea salinity levels are available. The class boundaries for absolute values are only given below so that a rough idea of boundary definitions and status can be obtained.

See REG Annex 5, Section 1.4

Table 5.1. Reference values (Rv), class boundaries (HG, GM, MO, OD) and corresponding EQRs for transparency (m). Grey figures indicate that the reference values shall be corrected based on observed salinity levels prior to calculating EQRs and comparing them to the EQR class boundaries.

Type area	Transparency (m)					Transparency EQR			
	Rv	HG	GM	MO	OD	HG	GM	MO	OD
Skagerrak and Kattegat									
1n	10.5	8.5	7.0	5.0	3.0	0.81	0.67	0.48	0.29
1s	8.0	6.5	5.5	4.0	3.0	0.81	0.69	0.50	0.38
2	8.0	6.5	5.0	3.5	2.5	0.81	0.63	0.44	0.31
3	12	10	8.0	5.0	3.5	0.83	0.67	0.42	0.29
25	4.5	4.0	3.0	2.0	0.5	0.89	0.67	0.45	0.11
4	10.5	9.5	8.0	5.0	3.5	0.90	0.76	0.48	0.33
5	10.5	9.5	8.0	5.0	3.5	0.90	0.76	0.48	0.33
6	10	8.0	7.5	4.5	3.0	0.80	0.75	0.45	0.30
Baltic Proper									
7	10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20
8	(10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20)
9	10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20
10	10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20
11	10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20
12	(10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20)
13	(10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20)
14	10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20
15	10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20
24	(10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20)
Bothnian Sea									
16	7.0	5.8	4.9	2.8	1.4	0.83	0.70	0.40	0.20
17	10	8.3	7.0	4.0	2.0	0.83	0.70	0.40	0.20
18	7.0	4.7	3.1	2.1	1.4	0.67	0.44	0.30	0.20
19	9.0	6.0	4.0	2.1	1.7	0.67	0.44	0.23	0.19

Type area	Transparency (m)					Transparency EQR			
	Rv	HG	GM	MO	OD	HG	GM	MO	OD
Bothnian Bay									
20	6.3	4.2	2.8	1.9	1.2	0.67	0.44	0.30	0.19
21	8.8	5.9	3.9	2.6	1.7	0.67	0.44	0.30	0.19
22	5.4	3.6	2.4	1.6	1.1	0.67	0.44	0.30	0.20
23	7.5	5.0	3.3	2.2	1.5	0.67	0.44	0.29	0.20

Class boundaries for salinity-dependent types (8, 12, 13 and 24)

Table 5.2 a. Salinity intervals with reference values and class boundaries for transparency (m) in **types 12n and 24,** based on an assumed nominal open-sea salinity level of 6 and a reference value for total nitrogen in freshwater of 23 µmol/l. The calculations have been performed in accordance with Section 4.4.4. The EQR class boundaries in table 5.1 apply to classification.

Salinity interval		Rv	HG	GM	MO	OD
0-	1	5.3	4.3	3.5	1.9	1.0
1-	2	5.9	4.7	3.9	2.2	1.2
2-	3	6.5	5.3	4.4	2.4	1.3
3-	4	7.4	6.0	4.9	2.7	1.5
4-	5	8.4	6.8	5.6	3.1	1.7
5-	6	9.6	7.8	6.4	3.6	1.9
>6		10	8.3	7	4	2

Table 5.2 b. Salinity intervals with reference values and class boundaries for transparency (m) in **types 12s and 13,** based on an assumed nominal open-sea salinity level of 6 and a reference value for total nitrogen in freshwater of 34 µmol/l. The calculations have been performed in accordance with section 4.4.4. The EQR class boundaries in table 5.1 apply to classification.

Salinity intervals		Rv	HG	GM	MO	OD
0-	1	2.8	2.3	1.9	1.0	0.6
1-	2	3.3	2.7	2.2	1.2	0.7
2-	3	4.1	3.3	2.7	1.5	0.8
3-	4	5.0	4.1	3.4	1.9	1.0
4-	5	6.5	5.3	4.4	2.4	1.3
5-	6	8.7	7.1	5.9	3.2	1.7
>6		10	8.3	7	4	2

Table 5.2 c. Salinity level intervals with reference values and class boundaries for transparency (m) in **type 8**, based on an assumed nominal open-sea salinity level of 7 and a reference value for total nitrogen in freshwater of 59 µmol/l. The classifications have been performed in accordance with Section 4.4.4 The EQR class boundaries in Table 5.1 apply to classification.

Salinity interval		Rv	HG	GM	MO	OD
0-	1	1.1	0.9	0.7	0.4	0.2
1-	2	1.4	1.1	0.9	0.5	0.3
2-	3	1.7	1.4	1.1	0.6	0.3
3-	4	2.2	1.8	1.5	0.8	0.4
4-	5	3.1	2.5	2.1	1.1	0.6
5-	6	4.5	3.7	3.0	1.7	0.9
6-	7	7.5	6.1	5.0	2.8	1.5
>7		10	8.3	7.0	4.0	2.0

5.5 Comments

The highest reference values in the Skagerrak and Kattegat can be found in the outer areas of Skagerrak (type 3) and of the Kattegat (types 4 and 5). The lowest reference values have been determined for the area around the estuaries of the Göta Älv and Nordre Älv rivers. The transparency in this area is naturally lower due to the large volume of suspended material discharged into the coastal area. In the Baltic Sea, the reference value is high for the outer types whilst it is adjusted downwards in the inner areas based on the degree of freshwater impact. In the Bothnian Sea and Bothnian Bay, the reference values are lower than in the Baltic Sea due to the discharge of humus substances from the rivers.

Background report: Bedömning av syrgashalt i kustvatten enligt vattendirektivet - metodbeskrivning [Assessment of oxygen levels in coastal waters in accordance with the WFD - method description]
Authors:
Skagerrak and Kattegat: Martin Hansson and Bertil Håkansson (SMHI)
Bothnian Sea & Bothnian Bay: Agneta Andersson (UMF)
The Baltic Proper: Jakob Walve and Ulf Larsson (STOCKHOLM UNIVERSITY)

6 Nutrients

Quality element	Shows primarily effects of	How often do measurements need to be taken?	At what time of the year?
Nutrients	Nutrient level/eutrophication	Once/month	Winter and summer

6.1 Introduction

Nutrients discharged into the sea are a natural precondition for all marine life and do not normally constitute an environmental problem in themselves. Problems occur however when nutrients are discharged in such volumes and proportions as to have a negative effect on the function and character of marine ecosystems. Several shoreline and open-sea areas around Sweden are currently adversely affected by eutrophication.

Eutrophication is caused by the increased discharge of nutrients that are otherwise inhibiting for production in coastal and sea areas. A clear sign of increased nutrient levels in the sea is a rise in the occurrence of filamentous, annual, fast-growing green and red algae, which overgrow and outcompete common seaweed. Even eel grass meadows, which are important breeding grounds for fish fry, are badly affected by filamentous algae.

The increased production in the surface layer can lead to large amounts of organic matter falling to the bottom below the pycnocline. To degrade, this matter needs oxygen and since the water exchange between the upper and lower layers is limited, oxygen can be in very short supply. When all the oxygen is used up, toxic hydrogen sulphide forms when organic material decomposes and nutrients such as phosphate and ammonium are released from the sediment into the water mass.

6.2 Parameters included

See REG Annex 5, Section 2.1

The parameters included in the nutrient quality element are total amounts of nitrogen and phosphorus, summertime and wintertime. During the winter period, dissolved inorganic nitrogen and phosphorus are also assessed.

6.2.1 Total amounts of nitrogen and phosphorus

Total nitrogen (tot-N) and total phosphorus (tot-P) measure all the nitrogen and phosphorus in the water, both dissolved and bound in particles and biomass. The total concentrations vary moderately during the year. The variation during the year is greater in the Skagerrak and Kattegat than in the Baltic Sea. Both winter and summer values provide an indication of how much nitrogen and phosphorus there is in the system and act therefore as a measure of eutrophication disturbance.

6.2.2 Dissolved inorganic nitrogen and phosphorus

Regarding the inorganic nutrients Dissolved Inorganic Nitrogen - DIN and Dissolved Inorganic Phosphorus - DIP, there is a very clear annual cycle. During the growing season, the levels drop rapidly as a result of the nutrition being consumed by phytoplankton and being bound to biomass. During the winter period, however, levels of DIN and DIP increase, since production is low and nutrients are discharged from land, as a result of mineralisation, air deposition and the intermix of nutrient-rich deep water into the surface layer. Levels are normally at their highest just before the spring bloom begins and provide a measure of the nutrient pool that is available for production and thereby the eutrophication disturbance. Since DIN and DIP levels are normally bound up in biological matter during the summer period, only the winter levels of DIN and DIP are assessed.

Definitions

Nutrients = Dissolved inorganic nutrients of nitrogen and phosphorus, i.e. nitrite (NO_2^-), nitrate (NO_3^-), ammonium (NH_4^+) and phosphate (PO_4^{3-})

DIN = Dissolved Inorganic Nitrogen. ∑ (nitrite (NO_2^-) + nitrate (NO_3^-) + ammonium (NH_4^+))

DIP = Dissolved Inorganic Phosphorus. Phosphate (PO_4^{3-})

Total amounts of N and P = total concentrations of nitrogen and phosphorus in the water, i.e. both dissolved and bound in particles and biomass (tot-N and tot-P).

Surface water = 0-10 m, or the upper water column if the pycnocline is shallower than 10 m

Summer values = Data from samples taken during the period June-August.

Winter values = General data from samples taken during the period December-February or immediately before the spring bloom begins. In the Skagerrak and Kattegat, the spring bloom can start earlier than February in mild winters. Measurement data that is affected by the spring bloom is not suitable for use in status classification. In such circumstances, data from November can be used instead. In the Gulf of Bothnia, data from November-February can be used.

Type-1n, Type-1s = Type 1 has been divided up into a northern and a southern component in order to function properly in this assessment criterion. Type-1s belongs to the northern Kattegat and Type 1n belongs to the Skagerrak. The boundary between these sub-types is somewhere around Åstol, south of Tjörn.

Type-12n, Type-12s = Type 12 has been divided up into a northern and a southern component in order to function properly in this assessment criterion. Type 12n belongs to northern Baltic Proper and Type 12s belong to western Baltic Proper. The boundary between these sub-types is somewhere near the Bråviken estuary.

Conversion factors: Nitrogen compounds (nitrite, nitrate, ammonium, tot-N) in µmol/l are multiplied by the factor 14.0 (the atomic weight of nitrogen) to give the concentration in µg/l. The corresponding factor for phosphorus and tot-P is 31.0 (the atomic weight of phosphorus).

6.3 Data requirements

6.3.1 Background to the methodology

To reflect naturally occurring gradients between discharged freshwater, coastal water and open-sea water, a holistic view needs to be taken in the assessment criterion for nutrients. Reference values and class boundaries shall be set so that they consider dilution effects and biochemical processes and their variation in different water bodies.

The starting-point is that the coastal area pelagic zone comprises a mixture of freshwater and open-sea water, whose mixture coefficient can be determined based on the specific salinity of each water body (station or sample). A simple water conversion model is used to calculate the salinity dependence of included parameters after their reference values in freshwater discharge and in the open sea have been determined. This provides a uniform approach that is applied to all coastal areas.

In coastal areas where salinity is close to zero, the freshwater reference values apply and in outer areas with high salinity levels close to the open sea, the open-sea reference values consequently apply. The water bodies are thereby given their reference values and classes in accordance with their characteristic salinity level. In order to relate the reference values to a salinity level, the easiest possible stationary water exchange model is used (Knudsen's relations) with conservation of salinity, tot-N, tot-P, DIN, DIP and transport.

Assessment criteria in a shoreline salinity gradient

In many Swedish coastal zones with freshwater discharge, the surface water is mixed with underlying water that has salinity levels similar to the sea area outside. As the water's salinity changes, the concentrations of other substances in the water also change. A conservative substance is mixed in the same way as the water's salinity. The dilution effect influences the distribution of the water's substance content and is a natural process that needs to be considered when classifying status. The simplest possible model describing the dilution and how it affects the assessment criterion is presented here.

It is assumed that properties such as salinity (S); substances such as total nitrogen (TN) and transport (Q) from river mouths to the sea as well as compensatory flow as a mean are conservative, i.e. don't change over time. Nutrients can be assumed to be conservative especially during the winter when biological production is low.

The reference values for total nitrogen are determined both in freshwater and in the sea area off the coast. The distribution of the reference values can be described as a function of the salinity, they will then change linearly between the reference values of the freshwater and the sea water as follows:

$$TN^{ref} = TN_{sea}^{ref} + S_f * (TN_{freshw}^{ref} - TN_{sea}^{ref}) \qquad (1)$$

for $0 \le S \le S_{sea}$

$$S_f = (S_{sea} - S) / S_{sea}$$

Here the reference value in the sea area off the coast is TN_{sea}^{ref} . TN_{freshw}^{ref} is the reference value in freshwater discharge, and S_f is the salinity coefficient.

When assessing the status, the observations are firstly normalised with the reference value at the salinity level indicated by the observations. Normalisation is performed using the EQR value:

$$EQR_{TN} = TN^{ref} / TN_{obs} \quad \text{for } S = S_{obs} \qquad (2)$$

where

$$0 \leq EQR_{TN} \leq 1$$

The classes in the assessment criterion are given as EQR values.

The class boundaries for tot-N, tot-P, DIN and DIP follow the salinity gradient and are calculated for each salinity level respectively with the assumption that the relationship between the reference value and the class boundaries in the open sea and in the entire sanity gradient is the same. This means that the class boundary interval for tot-N and DIN increases slightly with reduced salinity, which can be seen as reasonable since the variation in data increases with higher concentrations. This is normally the case for DIN and tot-N in areas with high freshwater impact, i.e. low salinity levels. For DIP and tot-P the opposite applies, i.e. that the class boundary interval increases slightly with higher salinity since the levels are generally somewhat higher in the open sea than in the inflowing freshwater.

There is normally no clear salinity gradient in the open sea and the concentration of nitrogen and phosphorus are independent of salinity. It is therefore possible to apply a fixed classification since the salinity exceeds the highest salinity level presented in Tables 6.2-6.7 (see highest salinity value in each table respectively).

Certain areas have no clear salinity gradient. This is because the reference values and class boundaries in freshwater discharge and in the open sea are the same or because the impact from inflowing freshwater is non-existent compared to the impact from the open sea (e.g. around Gotland and Öresund).

6.3.2 Sampling methodology

Samples are to be taken at discrete standard depths, for example; 0 m, 5 m, 10 m, 15 m, 20 m, 30 m, 40 m and so on down to the bottom (water samples are taken as close to the bottom as possible and no more than one metre above it). A more precise depth graduation might be necessary when sampling is done at shallow stations (depth of less than 10 m). When classifying status, values from the surface water shall be used (0-10 metres or from the upper water mass if the pycnocline is shallower than 10 metres.). Sampling and analysis must have been performed monthly and by an accredited laboratory and follow the recommendations in the HELCOM COMBINE manual[19].

See REG Annex 5, Section 2.2

It can be desirable to take samples in a profile from the surface to the bottom in order to obtain a holistic view of the surface layer levels compared to the rest of the depth profile, since the upwelling of nutrient-rich deep water to the surface water can affect the classification. From the profile, it is possible to estimate the total amount of nutrition available in the system both beneath and above the pycnocline. Data from the profile also indicates the local load to the bottoms which can be linked with any oxygen stress in deep water. Measurements from the entire profile are also valuable in order to control the quality of the data, for future research and to validate models.

[19] www.helcom.fi

6.4 Classification of status

Status classification based on measurement data shall be performed for every water body where data is available. Classification shall be performed for the winter period for DIN, DIP, tot-N and tot-P and shall be based on data from the surface water (0-10 m, or above the pycnocline if it is shallower than 10 m) under a specified period for each type respectively, see Tables 6.2-6.7. Only tot-N and tot-P are classified for the summer period.

See REG Annex 5, Section 2.3

For each measurement value to be classified, there must also be an observed salinity level. Based on this observed salinity level, a reference value and classification applicable to the specific measurement value are determined. In other words, each separate measurement is classified based on its position in the salinity gradient. The ecological quality ratio (EQR value) shall be calculated from each measurement. These EQR values can also be used to study trends and the development of a specific parameter independent of salinity. The following equation is used to calculate the EQR value (for tot-N, tot-P DIN and DIP):

$$\mathrm{EQR} = \frac{referencevalue}{observedvalue}$$

If measurements on a single occasion are performed at discrete depths, e.g. 0, 5 and 10 metres, the EQR value shall be calculated for each measurement and a mean EQR be created for the three depths.

6.4.1 Calculation of status class for tot-N, tot-P DIN and DIP

See REG Annex 5, Section 2.3.1

1) Calculate EQR for each separate sample based on the reference values in Tables 6.2-6-7. The relevant reference value is obtained from the salinity level observed when each separate sample is taken. If measurements are taken at discrete depths, calculate the EQR value for each measurement and then a mean-EQR for each specific measurement occasion.
2) The mean EQR value for every parameter is calculated for each year.
3) The mean EQR value for every parameter and water body is calculated for at least a three-year period.
4) The status classification for each parameter respectively is performed by comparing the mean EQR value with the EQR class boundaries given in Tables 6.2-6.7.
5) The EQRs are cofactored for the parameters included (tot-N, tot-P, DIN and DIP) according to the description below (6.4.2) for final status classification of the entire quality element.

6.4.2 Cofactoring of nutrients

In order to classify the nutrients quality element, the individual parameters must be cofactored. The cofactoring shall be based on the status classes for winter values of DIN, DIP, tot-N and tot-P and the status classes for summer values of tot-N, tot-P. The cofactoring shall be performed on data collected over three years. An example of the principle of cofactoring can be found in Section 4.4.3.

See REG Annex 5, Section 2.3.2

Step 1) The status classes are given a numerical value in accordance with Table 6.1. For each parameter, a weighted class value is calculated using formula 6.1 before the cofactoring is done in accordance with Step 2.

Table 6.1. Division of the status classes into numerical values.

Status	Numerical value
High status	4 - 4.99
Good status	3 - 3.99
Moderate status	2 - 2.99
Poor status	1 - 1.99
Bad status	0 - 0.99

The numerical class (N_{class}) for the respective parameters for the relevant EQR class interval (EQR_{lower} –EQR_{upper}) is calculated using formula 6.1.

$$(N_{class}) = (N_{lower}) + (EQR_{calculated} - EQR_{lower})/(EQR_{upper} - EQR_{lower})$$

Formula 6.1.

(N_{class}) = weighted status class value for each parameter.
N_{lower}= the first integer in the numerical values for the status class in accordance with Table 6.1.
$EQR_{estimated}$ = estimated EQR value from the classification.
EQR_{lower} and EQR_{upper} = EQR for lower and upper class boundary for the corresponding class, taken from Tables 6.2-6.7 below. EQR_{lower} for bad status = 0 and EQR_{upper} for high status = 1.

Step 2) A mean value of the numerical classifications (N_{class}) is calculated for DIN, DIP, tot-N, tot-P during the winter and a mean value for tot-N, tot-P during the summer. The mean value of summer and winter is then calculated and this will be the cofactored classification of nutrients. The reason why a mean value is first calculated for the winter and then one for the summer and then a common one for both is to avoid the winter values having greater weight despite the fact that there are four parameters measured for the winter as opposed to just two for the summer. The status classification is determined by the mean value for the numerical classification in accordance with Table 6.1.

In cases where the cofactored classification of nitrogen and phosphorus is under the boundary for good and moderate status, it is appropriate to check all parameters (winter: tot-N, tot-P, DN and DIP, and summer; tot-N and tot-P) individually in order to ascertain the cause of the classification. A more detailed analysis of the relevant parameters and comparisons with other quality elements can also be performed if they are available in order to determine the disturbance and show whether measures in the water body or in its vicinity are necessary.

6.5 Reference values and class boundaries

Tables 6.2-6.7 show the salinity-dependent reference values for the various types and the class boundaries for the different nutrients. The table also shows which parameter/s, time period, depth interval and type are referred to. The equation that gives the slope of the line representing each class boundary in the salinity gradient is enclosed if more exact calculations are required. The EQR values are constant across the salinity gradient. The values presented for each salinity interval are concentrations given in µmol/l. The tabulated concentrations are to be used when classifying status. Conversion factors from mg/l to µmol/l can be found under Definitions in Section 6.2 above.

See REG Annex 5, Section 2.4

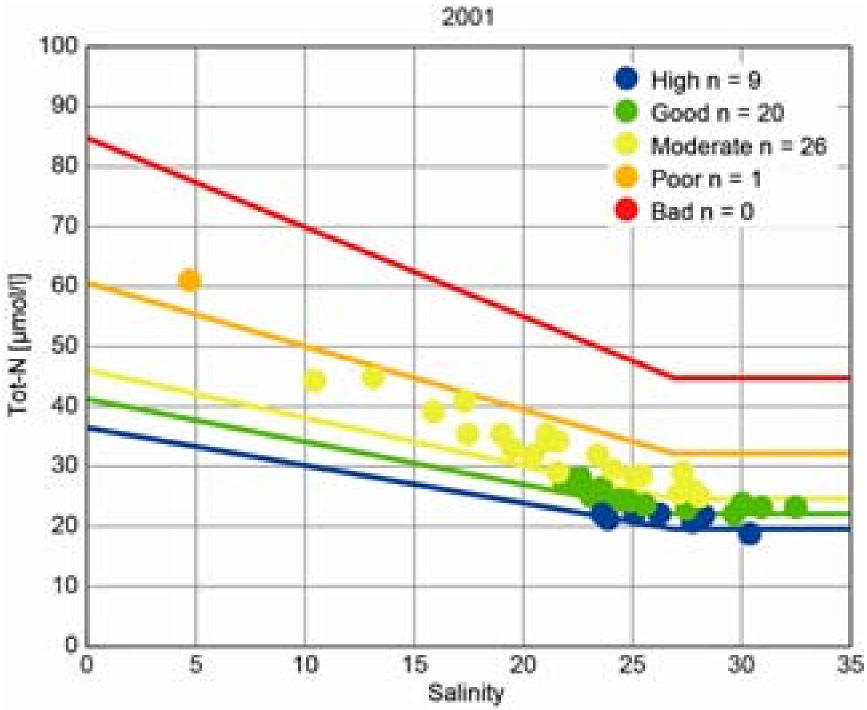

Fig 6.1. Illustration of classified data from type 2, in the salinity gradient. The lines illustrate the class boundaries and can be calculated using the equations in the tables. The measurement points are colour-coded depending on how they have been classified (n is the number of measurement points in each status class respectively).

6.5.1 Total nitrogen - winter

Table 6.2 Reference values and class boundaries for tot-N winter. The values presented for each salinity interval are concentrations given in µmol/l.

Tot-N, Winter, Nov-Feb, 0-10m						
Types 22 & 23		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-1*s+21	-1.09*s+22.89	-1.18*s+24.78	-1.45*s+30.45	-1.9*s+39.9
EQR		1.0	0.93	0.85	0.68	0.51
Salinity interval		Concentrations in µmol/l				
0	<1	21	23	25	31	41
1	<2	20	21	23	29	39
2	<3	19	20	22	28	37
≥3		18	20	22	27	36

Tot-N, Winter, Nov-Feb, 0-10m						
Types 20 & 21		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.6*s+6	-0.66*s+23.1	-0.72*s+25.2	-0.9*s+31.5	-1.2*s+2
EQR		1.0	0.91	0.83	0.67	0.50
Salinity interval		Concentrations in µmol/l				
0	<1	21	23	25	31	41
1	<2	20	22	24	30	40
2	<3	20	21	23	29	39
3	<4	19	21	23	28	38
4	<5	18	20	22	27	37
≥5		18	20	22	27	36

Tot-N, Winter, Nov-Feb, 0-10m						
Types 18 & 19		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.4*s+4	-0.44*s+44	-0.48*s+48	-0.6*s+6	-0.8*s+8
EQR		1.0	0.91	0.83	0.66	0.50
Salinity interval		Concentrations in µmol/l				
0	<1	20	22	24	30	40
1	<2	19	21	23	29	39
2	<3	19	21	23	29	38
3	<4	19	20	22	28	37
4	<5	18	20	22	27	36
≥5		18	20	22	27	36

Tot-N, Winter, Nov-Feb, 0-10m						
Types 16 & 17		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-1*s+23	-1.1*s+25.3	-1.2*s+27.6	-1.5*s+34.5	-2*s+46
EQR		1.0	0.93	0.85	0.68	0.51
Salinity interval		Concentrations in µmol/l				
0	<1	23	25	27	34	45
1	<2	22	24	26	32	43
2	<3	21	23	25	31	41
3	<4	20	21	23	29	39
4	<5	19	20	22	28	37
≥5		18	20	22	27	36

Tot-N, Winter, Dec-Feb, 0-10m					
Types 24, 12n & 15	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	-1*s+23	-1.1*s+25.3	-1.2*s+27.6	-1.5*s+34.5	-2*s+46
EQR	1.0	0.93	0.85	0.68	0.51
Salinity interval	Concentrations in µmol/l				
0 <1	23	25	27	34	45
1 <2	22	24	26	32	43
2 <3	21	23	25	31	41
3 <4	20	21	23	29	39
4 <5	19	20	22	28	37
5 <6	18	19	21	26	35
≥6	17	19	20	26	34

Tot-N, Winter, Dec-Feb, 0-10m					
Types 12s, 13, 14	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	-2.833*s+833	-3.1167*s+37.4	-3.4*s+40.8	-4.25*s+25	-5.6667*s+6667
EQR	1.0	0.91	0.83	0.66	0.50
Salinity interval	Concentrations in µmol/l				
0 <1	33	36	39	49	65
1 <2	30	33	36	45	60
2 <3	27	30	32	40	54
3 <4	24	26	29	36	48
4 <5	21	23	26	32	43
5 <6	18	20	22	28	37
≥6	17	19	20	26	34

Tot-N, Winter, Dec-Feb, 0-10m					
Types 10 & 11	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	0*s+17	0*s+18.7	0*s+20.4	0*s+25.5	0*s+34
EQR	1.0	0.89	0.85	0.65	0.50
Salinity interval	Concentrations in µmol/l				
- -	17	19	20	26	34

- No clear salinity gradient in types 10 and 11. The classification does not therefore depend on the salinity level.

Tot-N, Winter, Dec-Feb, 0-10m						
Types 7, 8 & 9		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-6*s+59	-6.6*s+64.9	-7.2*s+70.8	-9*s+88.5	-12*s+118
EQR		1.0	0.91	0.84	0.67	0.50
Salinity interval		Concentrations in µmol/l				
0	<1	56	62	67	84	112
1	<2	50	55	60	75	100
2	<3	44	48	53	66	88
3	<4	38	42	46	57	76
4	<5	32	35	38	48	64
5	<6	26	29	31	39	52
6	<7	20	22	24	30	40
≥7		17	19	20	26	34

Tot-N, Winter, Dec-Feb, 0-10m						
Types 5 & 6		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0*s+17	0*s+19.295	0*s+21.59	0*s+28.475	0*s+39.95
EQR		1.0	0.89	0.77	0.61	0.43
Salinity interval		Concentrations in µmol/l				
-	-	17	19	22	28	40

- No clear salinity gradient in types 5 and 6. The classification does not therefore depend on the salinity level.

Tot-N, Winter, Dec-Feb, 0-10m						
Types 1s, 4 & 25		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.65*s+65	-0.738*s+34.05	-0.8255*s+38.1	-1.0888*s+50.25	-1.528*s+70.5
EQR		1.0	0.88	0.79	0.60	0.43
Salinity interval		Concentrations in µmol/l				
0	<1	30	34	38	50	70
1	<2	29	33	37	49	68
2	<3	28	32	36	48	67
3	<4	28	31	35	46	65
4	<5	27	31	34	45	64
5	<6	26	30	34	44	62
6	<7	26	29	33	43	61
7	<8	25	29	32	42	59
8	<9	24	28	31	41	58
9	<10	24	27	30	40	56
10	<11	23	26	29	39	54
11	<12	23	26	29	38	53
12	<13	22	25	28	37	51
13	<14	21	24	27	36	50
14	<15	21	23	26	34	48
15	<16	20	23	25	33	47
16	<17	19	22	24	32	45
17	<18	19	21	24	31	44
18	<19	18	20	23	30	42

Tot-N, Winter, Dec-Feb, 0-10m						
Types 1s, 4 & 25		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.65*s+65	-0.738*s+34.05	-0.8255*s+38.1	-1.0888*s+50.25	-1.528*s+70.5
EQR		1.0	0.88	0.79	0.60	0.43
Salinity interval		Concentrations in µmol/l				
19	<20	17	20	22	29	41
≥20		17	19	22	28	40

Tot-N, Winter, Dec-Feb, 0-10m						
Types 1n, 2 & 3		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		- 0.630*s+630	-0.715*s+40.86	-0.799*s+45.72	-1.0546*s+60.3	-1.480*s+84.6
EQR		1.0	0.88	0.79	0.60	0.43
Salinity interval		Concentrations in µmol/l				
0	<1	36	41	45	60	84
1	<2	35	40	45	59	82
2	<3	34	39	44	58	81
3	<4	34	38	43	57	79
4	<5	33	38	42	56	78
5	<6	33	37	41	54	76
6	<7	32	36	41	53	75
7	<8	31	36	40	52	74
8	<9	31	35	39	51	72
9	<10	30	34	38	50	71
10	<11	29	33	37	49	69
11	<12	29	33	37	48	68
12	<13	28	32	36	47	66
13	<14	28	31	35	46	65
14	<15	27	30	34	45	63
15	<16	26	30	33	44	62
16	<17	26	29	33	43	60
17	<18	25	28	32	42	59
18	<19	24	28	31	41	57
19	<20	24	27	30	40	56
20	<21	23	26	29	39	54
21	<22	22	25	29	38	53
22	<23	22	25	28	37	51
23	<24	21	24	27	36	50
24	<25	21	23	26	34	48
25	<26	20	23	25	33	47
26	<27	19	22	25	32	45
≥27		19	22	24	32	45

6.5.2 DIN = Dissolved Inorganic Nitrogen

Table 6.3. Reference values and class boundaries for DIN (Dissolved Inorganic Nitrogen) winter-time. The values presented for each salinity interval are concentrations given in µmol/l.

DIN, Winter, Nov-Feb, 0-10m						
Types 22 & 23		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-1.333*s+333	-1.667*s+11.25	-2*s+13.5	-3*s+20.25	-4.667*s+31.5
EQR		1.0	0.80	0.67	0.44	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	8.3	10.4	12.5	18.8	29.2
1	<2	7.0	8.8	10.5	15.8	24.5
2	<3	5.7	7.1	8.5	12.8	19.8
≥3		5.0	6.3	7.5	11.3	17.5

DIN, Winter, Nov-Feb, 0-10m						
Types 20 & 21		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.76*s+76	-0.95*s+95	-1.14*s+14	-1.71*s+71	-2.66*s+66
EQR		1.0	0.80	0.67	0.44	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	7.6	9.5	11.4	17.1	26.7
1	<2	6.9	8.6	10.3	15.4	24.0
2	<3	6.1	7.6	9.2	13.7	21.4
3	<4	5.3	6.7	8.0	12.0	18.7
4	<5	4.6	5.7	6.9	10.3	16.0
≥5		4.2	5.3	6.3	9.5	14.7

DIN, Winter, Nov-Feb, 0-10m						
Types 18 & 19		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.2*s+2	-0.25*s+6.25	-0.3*s+7.5	-0.45*s+11.25	-0.7*s+17.5
EQR		1.0	0.80	0.66	0.44	0.28
Salinity interval		Concentrations in µmol/l				
0	<1	4.9	6.1	7.4	11.0	17.2
1	<2	4.7	5.9	7.1	10.6	16.5
2	<3	4.5	5.6	6.8	10.1	15.8
3	<4	4.3	5.4	6.5	9.7	15.1
4	<5	4.1	5.1	6.2	9.2	14.4
≥5		4.0	5.0	6.0	9.0	14.0

DIN, Winter, Nov-Feb, 0-10m						
Types 16 & 17		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.4*s+4	-0.5*s+6.25	-0.6*s+7.5	-0.9*s+11.25	-1.4*s+17.5
EQR		1.0	0.80	0.67	0.44	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	4.8	6.0	7.2	10.8	16.8
1	<2	4.4	5.5	6.6	9.9	15.4
2	<3	4.0	5.0	6.0	9.0	14.0
3	<4	3.6	4.5	5.4	8.1	12.6
4	<5	3.2	4.0	4.8	7.2	11.2
≥5		3.0	3.8	4.5	6.8	10.5

DIN, Winter, Dec-Feb, 0-10m						
Types 24, 12n & 15		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.75*s+75	-0.9375*s+8.75	-1.125*s+10.5	-1.6875*s+15.75	-2.625*s+24.5
EQR		1.0	0.80	0.67	0.44	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	6.6	8.3	9.9	14.9	23.2
1	<2	5.9	7.3	8.8	13.2	20.6
2	<3	5.1	6.4	7.7	11.5	17.9
3	<4	4.4	5.5	6.6	9.8	15.3
4	<5	3.6	4.5	5.4	8.2	12.7
5	<6	2.9	3.6	4.3	6.5	10.1
≥6		2.5	3.1	3.8	5.6	8.8

DIN, Winter, Dec-Feb, 0-10m						
Types 12s, 13 & 14		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-1.0833*s+0833	-1.354*s+11.25	-1.625*s+13.5	-2.4375*s+20.25	-3.792*s+31.5
EQR		1.0	0.80	0.66	0.44	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	8.5	10.6	12.7	19.0	29.6
1	<2	7.4	9.2	11.1	16.6	25.8
2	<3	6.3	7.9	9.4	14.2	22.0
3	<4	5.2	6.5	7.8	11.7	18.2
4	<5	4.1	5.2	6.2	9.3	14.4
5	<6	3.0	3.8	4.6	6.8	10.6
≥6		2.5	3.1	3.8	5.6	8.8

DIN, Winter, Dec-Feb, 0-10m						
Types 10 & 11		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0*s+2.5	0*s+3.125	0*s+3.75	0*s+5.625	0*s+8.75
EQR		1.0	0.81	0.66	0.45	0.28
Salinity interval		Concentrations in µmol/l				
-	-	2.5	3.1	3.8	5.6	8.8

- No clear salinity gradient in types 10 and 11. The classification does not therefore depend on the salinity level.

DIN, Winter, Dec-Feb, 0-10m						
Types 7, 8 & 9		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-4.928*s+928	-6.1618*s+46.2	-7.3929*s+55.5	-11.089*s+83.25	-17.25*s+25
EQR		1.0	0.80	0.67	0.45	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	34.5	43.2	51.8	77.7	120.9
1	<2	29.6	37.0	44.4	66.6	103.6
2	<3	24.7	30.8	37.0	55.5	86.4
3	<4	19.8	24.7	29.6	44.4	69.1
4	<5	14.8	18.5	22.2	33.3	51.9
5	<6	9.9	12.4	14.8	22.3	34.6
6	<7	5.0	6.2	7.4	11.2	17.4
≥7		2.5	3.1	3.8	5.6	8.8

DIN, Winter, Dec-Feb, 0-10m						
Types 5 & 6		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.125*s+1.5	0.1563*s+1.88	0.1875*s+2.25	0.2813*s+3.375	0.4375*s+5.25
EQR		1.0	0.80	0.67	0.44	0.29
Salinity interval		Concentrations in µmol/l				
<8		2.5	3.1	3.8	5.6	8.8
8	<9	2.6	3.2	3.8	5.8	9.0
9	<10	2.7	3.4	4.0	6.0	9.4
10	<11	2.8	3.5	4.2	6.3	9.8
11	<12	2.9	3.7	4.4	6.6	10.3
12	<13	3.1	3.8	4.6	6.9	10.7
13	<14	3.2	4.0	4.8	7.2	11.2
14	<15	3.3	4.1	5.0	7.5	11.6
15	<16	3.4	4.3	5.2	7.7	12.0
16	<17	3.6	4.5	5.3	8.0	12.5
17	<18	3.7	4.6	5.5	8.3	12.9
18	<19	3.8	4.8	5.7	8.6	13.3
19	<20	3.9	4.9	5.9	8.9	13.8
≥20		4.0	5.0	6.0	9.0	14.0

The salinity gradient between land and coastal water is negligible compared to the gradient between SW. Baltic Proper and S Kattegat.

DIN, Winter, Dec-Feb, 0-10m						
Types 1s, 4 & 25		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.525*s+15	-0.656*s+18.75	-0.7875*s+22.5	-1.1813*s+33.75	-1.838*s+52.5
EQR		1.0	0.80	0.67	0.44	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	14.7	18.4	22.1	33.2	51.6
1	<2	14.2	17.8	21.3	32.0	49.7
2	<3	13.7	17.1	20.5	30.8	47.9
3	<4	13.2	16.5	19.7	29.6	46.1
4	<5	12.6	15.8	19.0	28.4	44.2
5	<6	12.1	15.1	18.2	27.3	42.4
6	<7	11.6	14.5	17.4	26.1	40.6
7	<8	11.1	13.8	16.6	24.9	38.7
8	<9	10.5	13.2	15.8	23.7	36.9
9	<10	10.0	12.5	15.0	22.5	35.0
10	<11	9.5	11.9	14.2	21.3	33.2
11	<12	9.0	11.2	13.4	20.2	31.4
12	<13	8.4	10.5	12.7	19.0	29.5
13	<14	7.9	9.9	11.9	17.8	27.7
14	<15	7.4	9.2	11.1	16.6	25.9
15	<16	6.9	8.6	10.3	15.4	24.0
16	<17	6.3	7.9	9.5	14.3	22.2
17	<18	5.8	7.3	8.7	13.1	20.3
18	<19	5.3	6.6	7.9	11.9	18.5
19	<20	4.8	6.0	7.1	10.7	16.7
≥20		4.5	5.6	6.8	10.1	15.8

DIN, Winter, Dec-Feb, 0-10m						
Types 1n, 2 & 3		Reference	High/Good	Good/Moderate	Modera-te/Poor	Poor/Bad
Equations		-0.51852*s+20	-0.64815*s+25	-0.77778*s+30	-1.1667*s+45	-1.8148*s+70
EQR		1.0	0.80	0.66	0.44	0.28
Salinity interval		Concentrations in µmol/l				
0	<1	20	25	30	44	69
1	<2	19	24	29	43	67
2	<3	19	23	28	42	65
3	<4	18	23	27	41	64
4	<5	18	22	27	40	62
5	<6	17	21	26	39	60
6	<7	17	21	25	37	58
7	<8	16	20	24	36	56
8	<9	16	19	23	35	55
9	<10	15	19	23	34	53
10	<11	15	18	22	33	51
11	<12	14	18	21	32	49
12	<13	14	17	20	30	47
13	<14	13	16	20	29	46

DIN, Winter, Dec-Feb, 0-10m						
Types 1n, 2 & 3		Reference	High/Good	Good/Moderate	Modera-te/Poor	Poor/Bad
Equations		-0.51852*s+20	-0.64815*s+25	-0.77778*s+30	-1.1667*s+45	-1.8148*s+70
EQR		1.0	0.80	0.66	0.44	0.28
Salinity interval		Concentrations in µmol/l				
14	<15	12	16	19	28	44
15	<16	12	15	18	27	42
16	<17	11	14	17	26	40
17	<18	11	14	16	25	38
18	<19	10	13	16	23	36
19	<20	10	12	15	22	35
20	<21	9	12	14	21	33
21	<22	9	11	13	20	31
22	<23	8	10	13	19	29
23	<24	8	10	12	18	27
24	<25	7	9	11	16	26
25	<26	7	8	10	15	24
26	<27	6	8	9	14	22
≥27		6	8	9	14	21

6.5.3 Total phosphorus

Table 6.4. Reference values and class boundaries for tot-P winter. The values presented for each salinity interval are concentrations given in µmol/l.

Tot-P, Winter, Nov-Feb, 0-10m						
Types 22 & 23		Reference	High/Good	Good/Moderate	Modera-te/Poor	Poor/Bad
Equations		-0.057*s+0.4	-0.073*s+0.512	-0.089*s+0.624	-0.137*s+0.96	-0.217*s+1.52
EQR		1.0	0.78	0.64	0.42	0.26
Salinity interval		Concentrations in µmol/l				
0	<1	0.37	0.48	0.58	0.89	1.41
1	<2	0.31	0.40	0.49	0.75	1.19
2	<3	0.26	0.33	0.40	0.62	0.98
≥3		0.20	0.26	0.31	0.48	0.76

Tot-P, Winter, Nov-Feb, 0-10m						
Types 20 & 21		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.02*s+0.4	-0.026*s+0.512	-0.031*s+0.624	-0.048*s+0.96	-0.076*s+1.52
EQR		1.0	0.78	0.64	0.42	0.26
Salinity interval		Concentrations in µmol/l				
0	<1	0.39	0.50	0.61	0.94	1.48
1	<2	0.37	0.47	0.58	0.89	1.41
2	<3	0.35	0.45	0.55	0.84	1.33
3	<4	0.33	0.42	0.51	0.79	1.25
4	<5	0.31	0.40	0.48	0.74	1.18
≥5		0.30	0.38	0.47	0.72	1.14

Tot-P, Winter, Nov-Feb, 0-10m					
Types 16, 17, 18 & 19	Reference	High/Good	Good/Moderate	Modera-te/Poor	Poor/Bad
Equations	0*s+0.4	0*s+0.478	0*s+0.556	0*s+0.79	0*s+1.18
EQR	1.0	0.83	0.71	0.51	0.34
Salinity interval	Concentrations in µmol/l				
- -	0.40	0.48	0.56	0.79	1.18

- The reference value in freshwater discharge and in the open sea is the same, which means the classification can be performed independent of salinity.

Tot-P, Winter, Dec-Feb, 0-10m					
Types 24, 12n, 12s, 13, 14 & 15	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	0*s+0.4	0*s+0.504	0*s+0.608	0*s+0.92	0*s+1.44
EQR	1.0	0.80	0.66	0.43	0.28
Salinity interval	Concentrations in µmol/l				
- -	0.40	0.50	0.61	0.92	1.44

Tot-P, Winter, Dec-Feb, 0-10m					
Types 10 & 11	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	0*s+0.4	0*s+0.496	0*s+0.592	0*s+0.88	0*s+1.36
EQR	1.0	0.80	0.68	0.45	0.29
Salinity interval	Concentrations in µmol/l				
- -	0.40	0.50	0.59	0.88	1.36

- No clear salinity gradient in types 10 and 11. The classification does not therefore depend on the salinity level

Tot-P, Winter, Dec-Feb, 0-10m					
Types 7, 8 & 9	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	0.014*s+0.4	0.017*s+0.488	0.021*s+0.576	0.03*s+0.84	0.0457*s+1.28
EQR	1.0	0.82	0.69	0.47	0.31
Salinity interval	Concentrations in µmol/l				
0 <1	0.41	0.50	0.59	0.86	1.30
1 <2	0.42	0.51	0.61	0.89	1.35
2 <3	0.44	0.53	0.63	0.92	1.39
3 <4	0.45	0.55	0.65	0.95	1.44
4 <5	0.46	0.57	0.67	0.98	1.49
5 <6	0.48	0.58	0.69	1.01	1.53
6 <7	0.49	0.60	0.71	1.04	1.58
≥7	0.50	0.61	0.72	1.05	1.60

Tot-P, Winter, Dec-Feb, 0-10m						
Types 5 & 6		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.017*s+0.367	0.0191*s+0.42	0.022*s+0.473	0.0288*s+0.633	0.041*s+0.898
EQR		1.0	0.88	0.78	0.58	0.41
Salinity interval		Concentrations in µmol/l				
<8		0.50	0.57	0.65	0.86	1.23
8	<9	0.51	0.58	0.66	0.88	1.25
9	<10	0.53	0.60	0.68	0.91	1.29
10	<11	0.54	0.62	0.70	0.93	1.33
11	<12	0.56	0.64	0.72	0.96	1.37
12	<13	0.58	0.66	0.74	0.99	1.41
13	<14	0.59	0.68	0.76	1.02	1.45
14	<15	0.61	0.70	0.78	1.05	1.49
15	<16	0.63	0.72	0.81	1.08	1.53
16	<17	0.64	0.73	0.83	1.11	1.57
17	<18	0.66	0.75	0.85	1.14	1.61
18	<19	0.68	0.77	0.87	1.16	1.65
19	<20	0.69	0.79	0.89	1.19	1.69
≥20		0.70	0.80	0.90	1.21	1.72

The salinity gradient between land and coastal water is negligible compared to the gradient between SW Baltic Proper and S Kattegat.

Tot-P, Winter, Dec-Feb, 0-10m						
Types 1s, 4, & 25		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.015*s+0.4	0.017*s+0.458	0.019*s+0.516	0.02588*s+0.69	0.0368*s+0.98
EQR		1.0	0.87	0.78	0.58	0.41
Salinity interval		Concentrations in µmol/l				
0	<1	0.41	0.47	0.53	0.70	1.00
1	<2	0.42	0.48	0.55	0.73	1.04
2	<3	0.44	0.50	0.56	0.75	1.07
3	<4	0.45	0.52	0.58	0.78	1.11
4	<5	0.47	0.54	0.60	0.81	1.15
5	<6	0.48	0.55	0.62	0.83	1.18
6	<7	0.50	0.57	0.64	0.86	1.22
7	<8	0.51	0.59	0.66	0.88	1.26
8	<9	0.53	0.60	0.68	0.91	1.29
9	<10	0.54	0.62	0.70	0.94	1.33
10	<11	0.56	0.64	0.72	0.96	1.37
11	<12	0.57	0.66	0.74	0.99	1.40
12	<13	0.59	0.67	0.76	1.01	1.44
13	<14	0.60	0.69	0.78	1.04	1.48
14	<15	0.62	0.71	0.80	1.07	1.51
15	<16	0.63	0.72	0.82	1.09	1.55
16	<17	0.65	0.74	0.84	1.12	1.59
17	<18	0.66	0.76	0.85	1.14	1.62
18	<19	0.68	0.78	0.87	1.17	1.66
19	<20	0.69	0.79	0.89	1.19	1.70
≥20		0.70	0.80	0.90	1.21	1.72

Tot-P, Winter, Dec-Feb, 0-10m						
Types 1n, 2 & 3		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.011*s+0.4	0.0131*s+0.47	0.015*s+0.54	0.02083*s+0.75	0.03056*s+1.1
EQR		1.0	0.85	0.74	0.53	0.36
Salinity interval		Concentrations in µmol/l				
0	<1	0.41	0.48	0.55	0.76	1.12
1	<2	0.42	0.49	0.56	0.78	1.15
2	<3	0.43	0.50	0.58	0.80	1.18
3	<4	0.44	0.52	0.59	0.82	1.21
4	<5	0.45	0.53	0.61	0.84	1.24
5	<6	0.46	0.54	0.62	0.86	1.27
6	<7	0.47	0.55	0.64	0.89	1.30
7	<8	0.48	0.57	0.65	0.91	1.33
8	<9	0.49	0.58	0.67	0.93	1.36
9	<10	0.51	0.59	0.68	0.95	1.39
10	<11	0.52	0.61	0.70	0.97	1.42
11	<12	0.53	0.62	0.71	0.99	1.45
12	<13	0.54	0.63	0.73	1.01	1.48
13	<14	0.55	0.65	0.74	1.03	1.51
14	<15	0.56	0.66	0.76	1.05	1.54
15	<16	0.57	0.67	0.77	1.07	1.57

Tot-P, Winter, Dec-Feb, 0-10m						
Types 1n, 2 & 3		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.011*s+0.4	0.0131*s+0.47	0.015*s+0.54	0.02083*s+0.75	0.03056*s+1.1
EQR		1.0	0.85	0.74	0.53	0.36
Salinity interval		Concentrations in µmol/l				
16	<17	0.58	0.69	0.79	1.09	1.60
17	<18	0.59	0.70	0.80	1.11	1.63
18	<19	0.61	0.71	0.82	1.14	1.67
19	<20	0.62	0.72	0.83	1.16	1.70
20	<21	0.63	0.74	0.85	1.18	1.73
21	<22	0.64	0.75	0.6	1.0	1.76
22	<23	0.65	0.76	0.88	1.22	1.79
23	<24	0.66	0.78	0.89	1.24	1.82
24	<25	0.67	0.79	0.91	1.26	185
25	<26	0.68	0.80	0.92	1.28	1.88
26	<27	0.69	0.82	0.94	1.30	1.91
≥27		0.70	0.82	0.95	1.31	1.93

6.5.4 DIP = Dissolved Inorganic Phosphorus

Table 6.5. Reference values and class boundaries for DIP (Dissolved Inorganic Phosphorus) wintertime. The values presented for each salinity interval are concentrations given in µmol/l.

DIP, Winter, Nov-Feb, 0-10m						
Types 22 & 23		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.029*s+0.2	-0.0357*s+0.25	-0.04286*s+0.3	- 0.06429*s+0.45	-0.1*s+0.7
EQR		1.0	0.80	0.67	0.45	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	0.19	0.23	0.28	0.42	0.65
1	<2	0.16	0.20	0.24	0.35	0.55
2	<3	0.13	0.16	0.19	0.29	0.45
>3		0.10	0.13	0.15	0.23	0.35

DIP, Winter, Nov-Feb, 0-10m						
Types 20 & 21		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.01*s+0.2	-0.0125*s+0.25	-0.015*s+0.3	-0.0225*s+0.45	-0.035*s+0.7
		1.0	0.82	0.68	0.45	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	0.20	0.24	0.29	0.44	0.68
1	<2	0.19	0.23	0.28	0.42	0.65
2	<3	0.18	0.22	0.26	0.39	0.61
3	<4	0.17	0.21	0.25	0.37	0.58
4	<5	0.16	0.19	0.23	0.35	0.54
≥5		0.15	0.19	0.23	0.34	0.53

DIP, Winter, Nov-Feb, 0-10m					
Types 18 & 19	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	0*s+0.2	0*s+0.25	0*s+0.3	0*s+0.45	0*s+0.7
EQR	1.0	0.80	0.67	0.44	0.29
Salinity interval	Concentrations in µmol/l				
- -	0.2	0.25	0.3	0.45	0.7

- The reference value in freshwater discharge and in the open sea is the same, which means the classification can be performed independent of salinity.

DIP, Winter, Nov-Feb, 0-10m					
Types 16 & 17	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	0.02*s+0.1	0.025*s+0.125	0.03*s+0.15	0.045*s+0.225	0.07*s+0.35
EQR	1.0	0.80	0.65	0.44	0.28
Salinity interval	Concentrations in µmol/l				
0 <1	0.11	0.14	0.17	0.25	0.39
1 <2	0.13	0.16	0.20	0.29	0.46
2 <3	0.15	0.19	0.23	0.34	0.53
3 <4	0.17	0.21	0.26	0.38	0.60
4 <5	0.19	0.24	0.29	0.43	0.67
≥5	0.20	0.25	0.30	0.45	0.70

DIP, Winter, Dec-Feb, 0-10m					
Types 24, 12n, 15, 12s, 13, 14	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	0.008*s+0.2	0.0104*s+0.25	0.0125*s+0.3	0.01875*s+0.45	0.0292*s+0.7
EQR	1.0	0.79	0.66	0.44	0.29
Salinity interval	Concentrations in µmol/l				
0 <1	0.20	0.26	0.31	0.46	0.71
1 <2	0.21	0.27	0.32	0.48	0.74
2 <3	0.22	0.28	0.33	0.50	0.77
3 <4	0.23	0.29	0.34	0.52	0.80
4 <5	0.24	0.30	0.36	0.53	0.83
5 <6	0.25	0.31	0.37	0.55	0.86
≥6	0.25	0.31	0.38	0.56	0.88

DIP, Winter, Dec-Feb, 0-10m					
Types 10 & 11	Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations	0*s+0.25	0*s+0.3125	0*s+0.375	0*s+0.5625	0*s+0.875
EQR	1.0	0.81	0.66	0.45	0.28
Salinity interval	Concentrations in µmol/l				
- -	0.25	0.31	0.38	0.56	0.88

- No clear salinity gradient in types 10 and 11. The classification does not therefore depend on the salinity level.

DIP, Winter, Dec-Feb, 0-10m						
Types 7, 8 & 9		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.007*s+0.2	0.0089*s+0.25	0.0107*s+0.3	0.0161*s+0.45	0.025*s+0.7
EQR		1.0	0.81	0.66	0.45	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	0.20	0.25	0.31	0.46	0.71
1	<2	0.21	0.26	0.32	0.47	0.74
2	<3	0.22	0.27	0.33	0.49	0.76
3	<4	0.23	0.28	0.34	0.51	0.79
4	<5	0.23	0.29	0.35	0.52	0.81
5	<6	0.24	0.30	0.36	0.54	0.84
6	<7	0.25	0.31	0.37	0.55	0.86
≥7		0.25	0.31	0.38	0.56	0.88

DIP, Winter, Dec-Feb, 0-10m						
Types 5 & 6		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.013*s+0.15	0.016*s+0.188	0.019*s+0.225	0.0281*s+0.338	0.0438*s+0.53
EQR		1.0	0.80	0.67	0.44	0.29
Salinity interval		Concentrations in µmol/l				
<8		0.25	0.31	0.38	0.56	0.88
8	<9	0.26	0.32	0.38	0.58	0.90
9	<10	0.27	0.34	0.40	0.60	0.94
10	<11	0.28	0.35	0.42	0.63	0.98
11	<12	0.29	0.37	0.44	0.66	1.03
12	<13	0.31	0.38	0.46	0.69	1.07
13	<14	0.32	0.40	0.48	0.72	1.12
14	<15	0.33	0.41	0.50	0.75	1.16
15	<16	0.34	0.43	0.52	0.77	1.20
16	<17	0.36	0.45	0.53	0.80	1.25
17	<18	0.37	0.46	0.55	0.83	1.29
18	<19	0.38	0.48	0.57	0.86	1.33
19	<20	0.39	0.49	0.59	0.89	1.38
≥20		0.40	0.50	0.60	0.90	1.40

The salinity gradient between land and coastal water is negligible compared to the gradient between SW Baltic Proper and S Kattegat.

DIP, Winter, Dec-Feb, 0-10m						
Types 1s, 4 & 25		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.01*s+0.2	0.0125*s+0.25	0.015*s+0.3	0.0225*s+0.45	0.035*s+0.7
EQR		1.0	0.81	0.68	0.45	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	0.21	0.26	0.31	0.46	0.72
1	<2	0.22	0.27	0.32	0.48	0.75
2	<3	0.23	0.28	0.34	0.51	0.79
3	<4	0.24	0.29	0.35	0.53	0.82
4	<5	0.25	0.31	0.37	0.55	0.86
5	<6	0.26	0.32	0.38	0.57	0.89
6	<7	0.27	0.33	0.40	0.60	0.93
7	<8	0.28	0.34	0.41	0.62	0.96
8	<9	0.29	0.36	0.43	0.64	1.00
9	<10	0.30	0.37	0.44	0.66	1.03
10	<11	0.31	0.38	0.46	0.69	1.07
11	<12	0.32	0.39	0.47	0.71	1.10
12	<13	0.33	0.41	0.49	0.73	1.14
13	<14	0.34	0.42	0.50	0.75	1.17
14	<15	0.35	0.43	0.52	0.78	1.21
15	<16	0.36	0.44	0.53	0.80	1.24
16	<17	0.37	0.46	0.55	0.82	1.28
17	<18	0.38	0.47	0.56	0.84	1.31
18	<19	0.39	0.48	0.58	0.87	1.35
19	<20	0.40	0.49	0.59	0.89	1.38
≥20		0.40	0.50	0.60	0.90	1.40

DIP, Winter, Dec-Feb, 0-10m						
Types 1n, 2 & 3		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.011*s+0.2	0.0139*s+0.25	0.01667*s+0.3	0.025*s+0.45	0.03889*s+0.7
EQR		1.0	0.80	0.66	0.44	0.29
Salinity interval		Concentrations in µmol/l				
0	<1	0.21	0.26	0.31	0.46	0.72
1	<2	0.22	0.27	0.33	0.49	0.76
2	<3	0.23	0.28	0.34	0.51	0.80
3	<4	0.24	0.30	0.36	0.54	0.84
4	<5	0.25	0.31	0.38	0.56	0.88
5	<6	0.26	0.33	0.39	0.59	0.91
6	<7	0.27	0.34	0.41	0.61	0.95
7	<8	0.28	0.35	0.43	0.64	0.99
8	<9	0.29	0.37	0.44	0.66	1.03
9	<10	0.31	0.38	0.46	0.69	1.07
10	<11	0.32	0.40	0.48	0.71	1.11
11	<12	0.33	0.41	0.49	0.74	1.15
12	<13	0.34	0.42	0.51	0.76	1.19
13	<14	0.35	0.44	0.53	0.79	1.23
14	<15	0.36	0.45	0.54	0.81	1.26
15	<16	0.37	0.47	0.56	0.84	1.30

DIP, Winter, Dec-Feb, 0-10m						
Types 1n, 2 & 3		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.011*s+0.2	0.0139*s+0.25	0.01667*s+0.3	0.025*s+0.45	0.03889*s+0.7
EQR		1.0	0.80	0.66	0.44	0.29
Salinity interval		Concentrations in µmol/l				
16	<17	0.38	0.48	0.58	0.86	1.34
17	<18	0.39	0.49	0.59	0.89	1.38
18	<19	0.41	0.51	0.61	0.91	1.42
19	<20	0.42	0.52	0.63	0.94	1.46
20	<21	0.43	0.53	0.64	0.96	1.50
21	<22	0.44	0.55	0.66	0.99	1.54
22	<23	0.45	0.56	0.68	1.01	1.58
23	<24	0.46	0.58	0.69	1.04	1.61
24	<25	0.47	0.59	0.71	1.06	1.65
25	<26	0.48	0.60	0.73	1.09	1.69
26	<27	0.49	0.62	0.74	1.11	1.73
≥27		0.50	0.63	0.75	1.13	1.75

6.5.5 Total nitrogen summer

Table 6.6. Reference values and class boundaries for tot-N summer. The values presented for each salinity interval are concentrations given in µmol/l.

Tot-N, Summer, Jun-Aug, 0-10m						
Types 22 & 23		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-1.333*s+333	-1.547*s+24.36	-1.76*s+27.72	-2.4*s+37.8	-3.467*s+54.6
EQR		1.0	0.86	0.76	0.55	0.39
Salinity interval		Concentrations in µmol/l				
0	<1	20	24	27	37	53
1	<2	19	22	25	34	49
2	<3	18	20	23	32	46
>3		17	20	22	31	44

Tot-N, Summer, Jun-Aug, 0-10m						
Types 20 & 21		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-1*s+21	-1.16*s+24.36	-1.32*s+27.72	-1.8*s+37.8	-2.6*s+54.6
EQR		1.0	0.88	0.78	0.57	0.39
Salinity interval		Concentrations in µmol/l				
0	<1	21	24	27	37	53
1	<2	20	23	26	35	51
2	<3	19	21	24	33	48
3	<4	18	20	23	32	46
4	<5	17	19	22	30	43
≥5		16	19	21	29	42

Tot-N, Summer, Jun-Aug, 0-10m						
Types 18 & 19		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.8*s+8	-0.928*s+23.2	-1.056*s+26.4	-1.44*s+44	-2.08*s+08
EQR		1.0	0.85	0.75	0.55	0.38
Salinity interval		Concentrations in µmol/l				
0	<1	20	23	26	35	51
1	<2	19	22	25	34	49
2	<3	18	21	24	32	47
3	<4	17	20	23	31	45
4	<5	16	19	22	30	43
≥5		16	19	21	29	42

Tot-N, Summer, Jun-Aug, 0-10m						
Types 16 & 17		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-1.4*s+4	-1.624*s+26.68	-1.848*s+30.36	-2.52*s+41.4	-3.64*s+59.8
EQR		1.0	0.86	0.76	0.56	0.39
Salinity interval		Concentrations in µmol/l				
0	<1	22	26	29	40	58
1	<2	21	24	28	38	54
2	<3	20	23	26	35	51
3	<4	18	21	24	33	47
4	<5	17	19	22	30	43
≥5		16	19	21	29	42

Tot-N, Summer, Jun-Aug, 0-10m						
Types 24, 12n, & 15		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-1.333*s+23	-1.5468*s+26.6	-1.72*s+29.67	-2.4*s+41.4	-3.467*s+59.8
EQR		1.0	0.87	0.78	0.56	0.38
Salinity interval		Concentrations in µmol/l				
0	<1	22	26	29	40	58
1	<2	21	24	27	38	55
2	<3	20	23	25	35	51
3	<4	18	21	24	33	48
4	<5	17	20	22	31	44
5	<6	16	18	20	28	41
≥6		15	17	19	27	39

Tot-N, Summer, Jun-Aug, 0-10m						
Types 12s, 13, 14		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-3.167*s+34	-3.721*s+39.95	-4.085*s+43.86	-5.9375*s+63.75	-8.708*s+93.5
EQR		1.0	0.87	0.78	0.56	0.39
Salinity interval		Concentrations in µmol/l				
0	<1	32	38	42	58	84
1	<2	29	34	38	53	76
2	<3	26	30	34	47	68
3	<4	23	27	30	41	60
4	<5	20	23	25	36	51
5	<6	17	19	21	30	43
≥6		15	17	19	27	39

Tot-N, Summer, Jun-Aug, 0-10m						
Types 10 & 11		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0*s+15	0*s+17.4	0*s+19.35	0*s+27	0*s+39
EQR		1.0	0.88	0.79	0.56	0.38
Salinity interval		Concentrations in µmol/l				
-	-	15	17	19	27	39

No clear salinity gradient in types 10 and 11. The classification does not therefore depend on the salinity level.

Tot-N, Summer, Jun-Aug, 0-10m						
Types 7, 8 & 9		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-6.286*s+59	-7.291*s+68.44	-8.109*s+76.11	-11.314*s+106.2	-16.3*s+153.4
EQR		1.0	0.86	0.77	0.55	0.38
Salinity interval		Concentrations in µmol/l				
0	<1	56	65	72	101	145
1	<2	50	58	64	89	129
2	<3	43	50	56	78	113
3	<4	37	43	48	67	96
4	<5	31	36	40	55	80
5	<6	24	28	32	44	64
6	<7	18	21	23	33	47
≥7		15	17	19	27	39

Tot-N, Summer, Jun-Aug, 0-10m						
Types 5 & 6		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.25*s+25	-0.288*s+19.55	-0.325*s+22.1	-0.4375*s+29.75	-0.625*s+42.5
EQR		1.0	0.87	0.77	0.57	0.40
Salinity interval		Concentrations in µmol/l				
<8		15	17	20	26	38
8	<9	15	17	19	26	37
9	<10	15	17	19	26	37
10	<11	14	17	19	25	36
11	<12	14	16	18	25	35
12	<13	14	16	18	24	35
13	<14	14	16	18	24	34
14	<15	13	15	17	23	33
15	<16	13	15	17	23	33
16	<17	13	15	17	23	32
17	<18	13	15	16	22	32
18	<19	12	14	16	22	31
19	<20	12	14	16	21	30
≥20		12	14	16	21	30

The salinity gradient between land and coastal water is negligible compared to the gradient between SW Baltic Proper and S Kattegat.

Tot-N, Summer, Jun-Aug, 0-10m						
Types 1s, 4 & 25		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.9*s+30	-1.035*s+34.5	-1.17*s+39	-1.575*s+52.5	-2.25*s+75
EQR		1.0	0.87	0.77	0.57	0.40
Salinity interval		Concentrations in µmol/l				
0	<1	30	34	38	52	74
1	<2	29	33	37	50	72
2	<3	28	32	36	49	69
3	<4	27	31	35	47	67
4	<5	26	30	34	45	65
5	<6	25	29	33	44	63
6	<7	24	28	31	42	60
7	<8	23	27	30	41	58
8	<9	22	26	29	39	56
9	<10	21	25	28	38	54
10	<11	21	24	27	36	51
11	<12	20	23	26	34	49
12	<13	19	22	24	33	47
13	<14	18	21	23	31	45
14	<15	17	19	22	30	42
15	<16	16	18	21	28	40
16	<17	15	17	20	27	38
17	<18	14	16	19	25	36
18	<19	13	15	17	23	33
19	<20	12	14	16	22	31
≥20		12	14	16	21	30

Tot-N, Summer, Jun-Aug, 0-10m						
Types 1n, 2 & 3		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.963*s+963	-1.088*s+40.68	-1.213*s+45.36	-1.5889*s+59.4	-2.215*s+82.8
EQR		1.0	0.88	0.79	0.60	0.43
Salinity interval		Concentrations in µmol/l				
0	<1	36	40	45	59	82
1	<2	35	39	44	57	79
2	<3	34	38	42	55	77
3	<4	33	37	41	54	75
4	<5	32	36	40	52	73
5	<6	31	35	39	51	71
6	<7	30	34	37	49	68
7	<8	29	33	36	47	66
8	<9	28	31	35	46	64
9	<10	27	30	34	44	62
10	<11	26	29	33	43	60
11	<12	25	28	31	41	57
12	<13	24	27	30	40	55
13	<14	23	26	29	38	53
14	<15	22	25	28	36	51
15	<16	21	24	27	35	48
16	<17	20	23	25	33	46
17	<18	19	22	24	32	44
18	<19	18	21	23	30	42
19	<20	17	19	22	28	40
20	<21	16	18	20	27	37
21	<22	15	17	19	25	35
22	<23	14	16	18	24	33
23	<24	13	15	17	22	31
24	<25	12	14	16	20	29
25	<26	11	13	14	19	26
26	<27	10	12	13	17	24
≥27		10	11	13	17	23

6.5.6 Total phosphorus summer

Table 6.7. Reference values and class boundaries for tot-P summer. The values presented for each salinity interval are concentrations given in µmol/l.

Tot-P, Summer, Jun-Aug, 0-10m						
Types 22 & 23		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.083*s+0.4	-0.102*s+0.49	-0.121*s+0.58	-0.177*s+0.85	-0.271*s+1.3
EQR		1.0	0.83	0.69	0.47	0.31
Salinity interval		Concentrations in µmol/l				
0	<1	0.36	0.44	0.52	0.76	1.16
1	<2	0.28	0.34	0.40	0.58	0.89
2	<3	0.19	0.23	0.28	0.41	0.62
≥3		0.15	0.18	0.22	0.32	0.49

Tot-P, Summer, Jun-Aug, 0-10m						
Types 20 & 21		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.044*s+0.4	-0.054*s+0.49	-0.064*s+0.58	-0.094*s+0.85	-0.144*s+1.3
EQR		1.0	0.81	0.69	0.47	0.31
Salinity interval		Concentrations in µmol/l				
0	<1	0.38	0.46	0.55	0.80	1.23
1	<2	0.33	0.41	0.48	0.71	1.08
2	<3	0.29	0.35	0.42	0.61	0.94
3	<4	0.24	0.30	0.35	0.52	0.79
≥4		0.20	0.25	0.29	0.43	0.65

Tot-P, Summer, Jun-Aug, 0-10m						
Types 18 & 19		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.03*s+0.4	-0.037*s+0.49	-0.044*s+0.58	-0.064*s+0.85	-0.098*s+1.3
EQR		1.0	0.83	0.70	0.48	0.31
Salinity interval		Concentrations in µmol/l				
0	<1	0.39	0.47	0.56	0.82	1.25
1	<2	0.36	0.43	0.51	0.75	1.15
2	<3	0.33	0.40	0.47	0.69	1.06
3	<4	0.30	0.36	0.43	0.63	0.96
4	<5	0.27	0.32	0.38	0.56	0.86
≥5		0.25	0.31	0.36	0.53	0.81

Tot-P, Summer, Jun-Aug, 0-10m						
Types 16 & 17		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.03*s+0.4	-0.036*s+0.48	-0.042*s+0.56	-0.06*s+0.8	-0.09*s+1.2
EQR		1.0	0.84	0.72	0.51	0.34
Salinity interval		Concentrations in µmol/l				
0	<1	0.39	0.46	0.54	0.77	1.16
1	<2	0.36	0.43	0.50	0.71	1.07
2	<3	0.33	0.39	0.46	0.65	0.98
3	<4	0.30	0.35	0.41	0.59	0.89
4	<5	0.27	0.32	0.37	0.53	0.80
≥5		0.25	0.30	0.35	0.50	0.75

Tot-P, Summer, Jun-Aug, 0-10m						
Types 24, 12n, 12s, 13, 14 & 15		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.017*s+04	-0.0196*s+0.47	-0.023*s+0.54	-0.0313*s+0.75	-0.0458*s+1.1
EQR		1.0	0.86	0.74	0.54	0.36
Salinity interval		Concentrations in µmol/l				
0	<1	0.39	0.46	0.53	0.73	1.08
1	<2	0.38	0.44	0.51	0.70	1.03
2	<3	0.36	0.42	0.48	0.67	0.99
3	<4	0.34	0.40	0.46	0.64	0.94
4	<5	0.33	0.38	0.44	0.61	0.89
5	<6	0.31	0.36	0.42	0.58	0.85
≥6		0.30	0.35	0.41	0.56	0.83

Tot-P, Summer, Jun-Aug, 0-10m						
Types 10 & 11		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0*s+0.3	0*s+0.3525	0*s+0.405	0*s+0.5625	0*s+0.825
EQR		1.0	0.86	0.73	0.54	0.36
Salinity interval		Concentrations in µmol/l				
-	-	0.30	0.35	0.41	0.56	0.83

- No clear salinity gradient in types 10 and 11. The classification does not therefore depend on the salinity level.

Tot-P, Summer, Jun-Aug, 0-10m						
Types 7, 8 & 9		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.013*s+0.4	-0.0157*s+0.47	-0.018*s+0.54	-0.025*s+0.75	-0.0367*s+1.1
EQR		1.0	0.85	0.74	0.53	0.36
Salinity interval		Concentrations in µmol/l				
0	<1	0.39	0.46	0.53	0.74	1.08
1	<2	0.38	0.45	0.51	0.71	1.05
2	<3	0.37	0.43	0.50	0.69	1.01
3	<4	0.35	0.42	0.48	0.66	0.97
4	<5	0.34	0.40	0.46	0.64	0.94
5	<6	0.33	0.38	0.44	0.61	0.90
6	<7	0.31	0.37	0.42	0.59	0.86
≥7		0.30	0.35	0.41	0.56	0.83

Tot-P, Summer, Jun-Aug, 0-10m						
Types 5 & 6		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		0.008*s+0.2	0.01*s+0.28	0.012*s+0.327	0.0167*s+0.467	0.025*s+0.7
EQR		1.0	0.82	0.71	0.50	0.33
Salinity interval		Concentrations in µmol/l				
<8		0.30	0.36	0.42	0.60	0.90
8	<9	0.30	0.37	0.43	0.61	0.91
9	<10	0.31	0.38	0.44	0.63	0.94
10	<11	0.32	0.39	0.45	0.64	0.96
11	<12	0.33	0.40	0.46	0.66	0.99
12	<13	0.34	0.41	0.47	0.68	1.01
13	<14	0.35	0.42	0.48	0.69	1.04
14	<15	0.35	0.43	0.50	0.71	1.06
15	<16	0.36	0.44	0.51	0.73	1.09
16	<17	0.37	0.45	0.52	0.74	1.11
17	<18	0.38	0.46	0.53	0.76	1.14
18	<19	0.39	0.47	0.54	0.78	1.16
19	<20	0.40	0.48	0.55	0.79	1.19
≥20		0.40	0.48	0.56	0.80	1.20

The salinity gradient between land and coastal water is negligible compared to the gradient between SW Baltic Proper and S Kattegat.

Tot-P, Summer, Jun-Aug, 0-10m						
Types 1n, 1s, 2, 3, 4 & 25		Reference	High/Good	Good/Moderate	Moderate/Poor	Poor/Bad
Equations		-0.006*s+0.4	-0.007*s+0.48	-0.008*s+0.56	-0.012*s+0.8	-0.018*s+1.2
EQR		1.0	0.83	0.71	0.50	0.33
Salinity interval		Concentrations in µmol/l				
-	-	0.4	0.48	0.56	0.8	1.2

6.6 Comments

In the Skagerrak and Kattegat, the levels of DIN and DIP are normally at their highest in February. The spring bloom can, however, during calm winters with stable stratification, start as early as in December-January. To avoid status classification from being based on measurement data taken when the spring bloom has already started to consume DIN and DIP, concentration trends should be followed during the period December-March. If the levels increase in relation to measurements in earlier months, it is assumed that the nutrient pool is still being built up and the measurements can be used for status classification. If instead the levels drop in relation to measurements in earlier months, it can be assumed that the spring bloom has begun and these measurements are therefore not suitable for use in status classification. Alternatively, measurement data can be filtered so that samples with chlorophyll values of e.g. >1µg/l or/and oxygen saturation over 100% are not included in the status classification, as this is a rough indication of the fact that the spring bloom in underway. In the Baltic Proper, the spring bloom normally starts slightly later and seldom influences winter data from January and February. In the Bothnian Sea and Bothnian Bay, levels of DIN and DIP are normally at their highest in February-March and the spring bloom starts even later in March-April and April-May respectively.

Phytoplankton blooms bind nutrients in their biomass and can therefore have an effect on the levels of tot-N and tot-P. This is particularly true in the Baltic Proper during the summer when blooms of *Nodularia* and *Aphanizomenon* can increase the values. Measurement values obtained during intense blooms are not suitable for use in status classification.

Background report: Förslag till vattendirektivets bedömningsgrunder för pelagiala vintertida näringsämnen och sommmartida effektrelarerade näringsämnen i kust- och övergångsvatten [Proposals for the WFD assessment criteria for pelagic wintertime nutrients and summertime effect-related nutrients in coastal and transitional waters]
Authors: Martin Hansson and Bertil Håkansson (SMHI)

7 Oxygen balance

Quality element	Shows primarily effects of	How often do measurements need to be taken?	At what time of the year?
Oxygen balance	Nutrient level/eutrophication	Once/month	Year-round

7.1 Introduction

Oxygen is a key parameter for all biological life. In many Swedish coastal and sea areas, the oxygen depletion is palpable and inhibiting for the ecological system. Oxygen depletion is affected by both physical and biological factors and can occur quite naturally. The oxygen level in deep water directly affects biological life in the bottom water and in the sediment. Oxygen is used in respiration and the degradation of organic matter causing it to be depleted. The oxygen level is therefore a good indicator of eutrophication as long as the deep retention time in the area in question is taken into consideration. The critical limit for how much load the ecosystem can tolerate before oxygen levels are affected varies from one water body to another. The bottom water in outer, open areas often has good access to oxygen. There are exceptions, however. For example, in the deep areas of the open Baltic Sea, oxygen depletion (hypoxia) or even anoxia (complete lack of oxygen) occur, mainly caused by the Baltic Sea being cut off from other seas and oceans. Oxygen depletion can also occur in shallow, open areas of the southern Kattegat where the depth of the pycnocline only allows a small amount of deep water to be at the bottom. The oxygen in this thin bottom layer is used up very quickly and oxygen depletion occurs. As a result of rapid oscillations in the halocline's position, the variations can be considerable during a short period of time.

Definitions 1

Hypoxia = Oxygen depletion. There is no exact limit for when hypoxia actually occurs. It depends on how well various flora and fauna groups adapt to surviving low oxygen levels. In the assessment criterion, the limit for oxygen depletion is set at 3.5 ml/l.

Anoxia = Totally oxygen-free conditions.

Hydrogen sulphide = When anoxia occurs, hydrogen sulphide is produced during the microbial degradation of organic matter, when sulphate is used as an energy source and converted into hydrogen sulphide. Hydrogen sulphide is toxic to all higher organisms. The occurrence of hydrogen sulphide leads to dead bottoms.

Bottom water = The water just at or very close to the bottom. Bottom water is sampled using a special bottom water sampler, from just above the bottom (0.5-1.0 m)).

Deep water = defined in this manual as the water found under the halocline that delimits the oxygenated surface layer and where problems with oxygen levels most often occur.

Retention time = The time, in days, it takes for all deep water in the water body to be exchanged.

Undisturbed period = January to May. The period when the oxygen conditions are mostly determined by the water body's natural properties.

Disturbed period = June to December. Period when the oxygen conditions are determined by both natural properties and by anthropogenic/natural load.

Oxygen depletion in Swedish coastal waters is most widespread during the growing season between June and December, when large amounts of biological matter is discharged into the bottom water and is broken down at the same time at water exchange is inhibited by a pycnocline. Between January and May, before the spring bloom has managed to sediment and the seasonal oxygen-demanding degradation of particulate organic matter has begun, the oxygen conditions can be good and it is mostly other factors that determine the oxygen level, e.g. meteorological conditions and morphological obstacles such as threshold depth and maximum depth which in turn determine the retention time and the supply of oxygen-rich deep water. The oxygen concentration during January-May reflects a kind of background value determined by the water body's natural properties.

7.2 Data requirements

There are different variants of hypoxia: seasonal, perennial and constant hypoxia (see Definitions 2 below). These differ in various ways, not least in their duration which can cause different effects. It is therefore difficult to create general reference values and class boundaries that can be applied to all water bodies. It is hence necessary to determine a water body's affiliation (which of the categories below it belongs to) before its oxygen status can be established. A water body belongs to one of the five categories defined in the Definitions below.

Definitions 2

A water body is assigned to one of the following five categories:

1) **Seasonal hypoxia** - Occurs in late summer and autumn as a result of the degradation of organic matter discharged into the deep water during the year. Conditions return to normal during the winter and early spring when there is only slight discharge of organic matter and the absence of a pycnocline facilitates the vertical intermix of the deep water.

2) **Perennial hypoxia/anoxia** - Oxygen levels under the reference value (<3.5 ml/l) occur all year round. Deep retention time is < 12 months.

3) **Constant hypoxia/anoxia** - Can occur in water bodies with very limited water exchange. Deep retention time is > 12 months. Environmental improvement measures will have little or no effect on the oxygen conditions. Examples include enclosed fjords/inlets.

4) **Oxygenated deep water** - The oxygen level is over the reference value (> 3.5 ml/l) all year round and several years consecutively.

5) **No data** - No measurements taken or insufficient in time and space.

See REG Annex 5, Section 3.2

Each water body is assigned to one of the above categories so that its status can be classified. Categorisation is done through a series of tests, the outcomes of which determine how the water body is to be treated. A flowchart of the approach is presented in Figure 7.2 and the different tests and their outcomes are described in more detail below.

7.2.1 Sampling methodology

To be able to statistically evaluate the oxygen conditions, oxygen levels need to be measured frequently (once a month) during a consecutive period of at least three years (preferably more). Since regrowth for bottoms affected by severe hypoxia can be estimated at about 12 months (the time it takes larvae to recolonise an oxygenated bottom), it is inappropriate to use a single 12-month period as the basis for the status classification of a water body.

See REG Annex 5, Section 3.1

Measurements shall be taken monthly in a profile from the surface to the bottom at standard depths (0 m, 5 m 10 m, 15 m, 20 m, 30 m, 40 m, ...etc down to the bottom, with the last sample being taken as close to the bottom as possible (less than one metre from the bottom) in the deepest part of the water body. At shallow stations (depth <10m), a more precise depth graduation is required (e.g. every 2.5 m).

It may be appropriate to measure the level of hydrogen sulphide when it is suspected of occurring. To assure the quality of the data, sampling and analysis shall be performed by an accredited laboratory in accordance with the HELCOM COMBINE Manual[20].

7.2.2 Test 1 - Is hypoxia a problem in the water body?

Test 1: Determine the station mean value based on the lower quartile of observed oxygen levels in the bottom water recorded every month during a three-year period (January-December).

To be able to carry out Test 1, the oxygen data from the bottom water must be available from a representative measuring station in the water body in question. The station shall be located in the deepest part of the water body. If there are several stations in the same water body, data from all the stations should be used. Alternatively, data from the most representative station should be used. The data shall cover the whole year, preferably with monthly measurements. If there is no data, model data can be used and as a last resort, an expert judgement can be applied to determine the status of the water body. The test is based on the specified reference value (3.5 ml/l) which guarantees that the oxygen level does not have a negative impact on the water body's ecosystems.

See REG Annex 5, Section 3.1

A "Box and whisker" diagram is calculated for each water body (measuring station). A "Box and whisker" diagram illustrates the spread of the number of data points within the data set; for the lower quartile (the lowest 25% of the data points),

[20] www.helcom.fi

inter quartile (containing 50% of the data) and the upper quartile (the highest 25% of the data points). The “Box and whisker” diagram is explained in Figure 7.1.

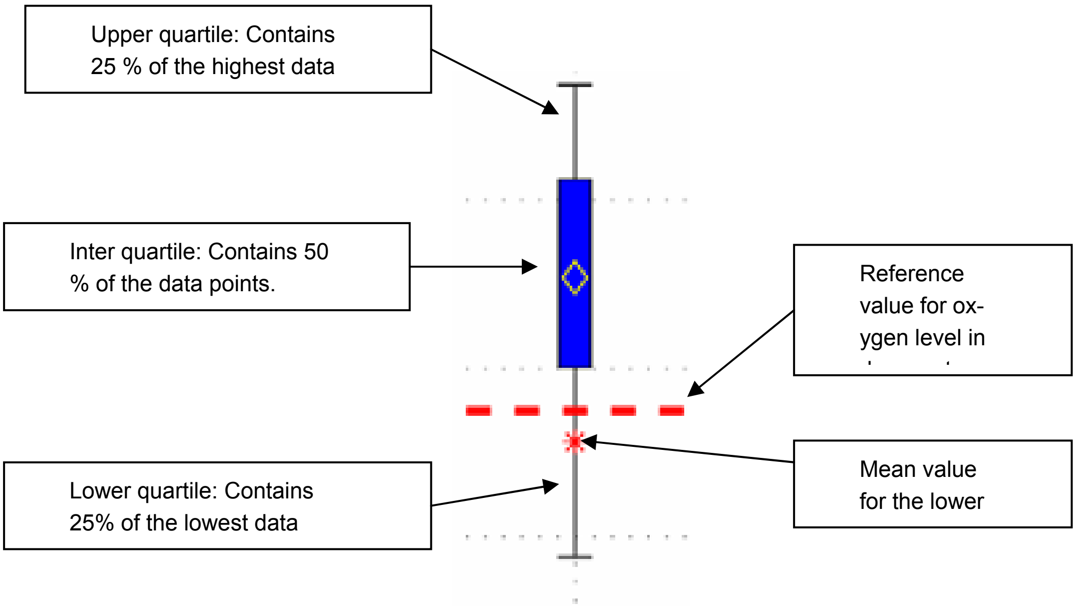

Figure 7.1. Explanation of “Box and whisker” diagram. The data set consists of three-years’ of data from the bottom water in the water body. In this case, the mean value of the lower quartile is under the reference value (Outcome 1b) and it is necessary to perform test 2 to determine whether the water body is affected by seasonal, perennial or constant hypoxia.

From test 1, it is possible to obtain two outcomes as presented below:

Test 1 - Outcome 1a – No oxygen depletion

The station mean value for January-December in the lower quartile exceeds the reference value (>3.5 ml/l). The water body does not show signs of oxygen depletion and can be considered to have oxygenated deep water. The water body’s oxygen level status can be directly set to high.

Test 1 - Outcome 1b - Oxygen depletion occurs

The station mean value for January-December in the lower quartile is less than the reference value (<3.5 ml/l). The water body shows signs of oxygen depletion and it is necessary to perform test 2 to determine whether this depletion is seasonal, perennial or constant.

7.2.3 Test 2 - Is the oxygen depletion seasonal, perennial or constant?

See REG Annex 5, Section 3.2

In water bodies where the oxygen level is less than 3.5 ml/l, it shall be determined whether the resulting depletion is seasonal, perennial or constant, based on the station mean value for the period January-May during three consecutive years.

Test 2 is limited in time to the months January to May inclusive as they are considered to represent the undisturbed period and take the deep water retention time into consideration if such data is available. In water bodies for which there is no data or for which the existing data is insufficient, model-generated data (e.g. using the SMHI coastal zone model) can be used instead. Alternatively, an expert judgement can be made to determine the oxygen status of the water body.

Test 2: Determine the station mean value (undisturbed period, January-May, during a consecutive three-year period) in the lower quartile of observed oxygen levels in the bottom water. If possible, the deep water retention time in the water body shall be determined. Data on retention times can be found in literature or be calculated (see e.g. Engqvist, 1999 and Engqvist, 2002).

Test 2 - Outcome 2a – Seasonal oxygen depletion

If the station mean value for January-May exceeds the reference value (>3.5 ml/l) and the deep water retention time is < 12 months, the oxygen depletion is seasonal. The water body shows no signs of oxygen depletion problems during the undisturbed period. The depletion is limited to the autumn period and is therefore seasonal. The oxygen status of the water body is determined using method 1 (Section 7.3.1).

Test 2 - Outcome 2b - Perennial oxygen depletion

If the station mean value for January-May is less than the reference value (<3.5 ml/l) and the deep water retention time is < 12 months, the oxygen depletion is perennial. The depletion occurs all year round, even during the undisturbed period, and is therefore perennial. The oxygen status of the water body is determined using method 2 (Section 7.3.2).

Test 2 - Outcome 2c - Constant oxygen depletion

If the station mean value for January-May is less than the reference value (<3.5 ml/l) and the deep water retention time is > 12 months, the oxygen depletion is constant. The depletion is a result of limited deep water renewal and any environmental improvement measures will have no or very little positive effect on the oxygen levels in the bottom water. The status of the water body is determined using method 2 (Section 7.3.2).

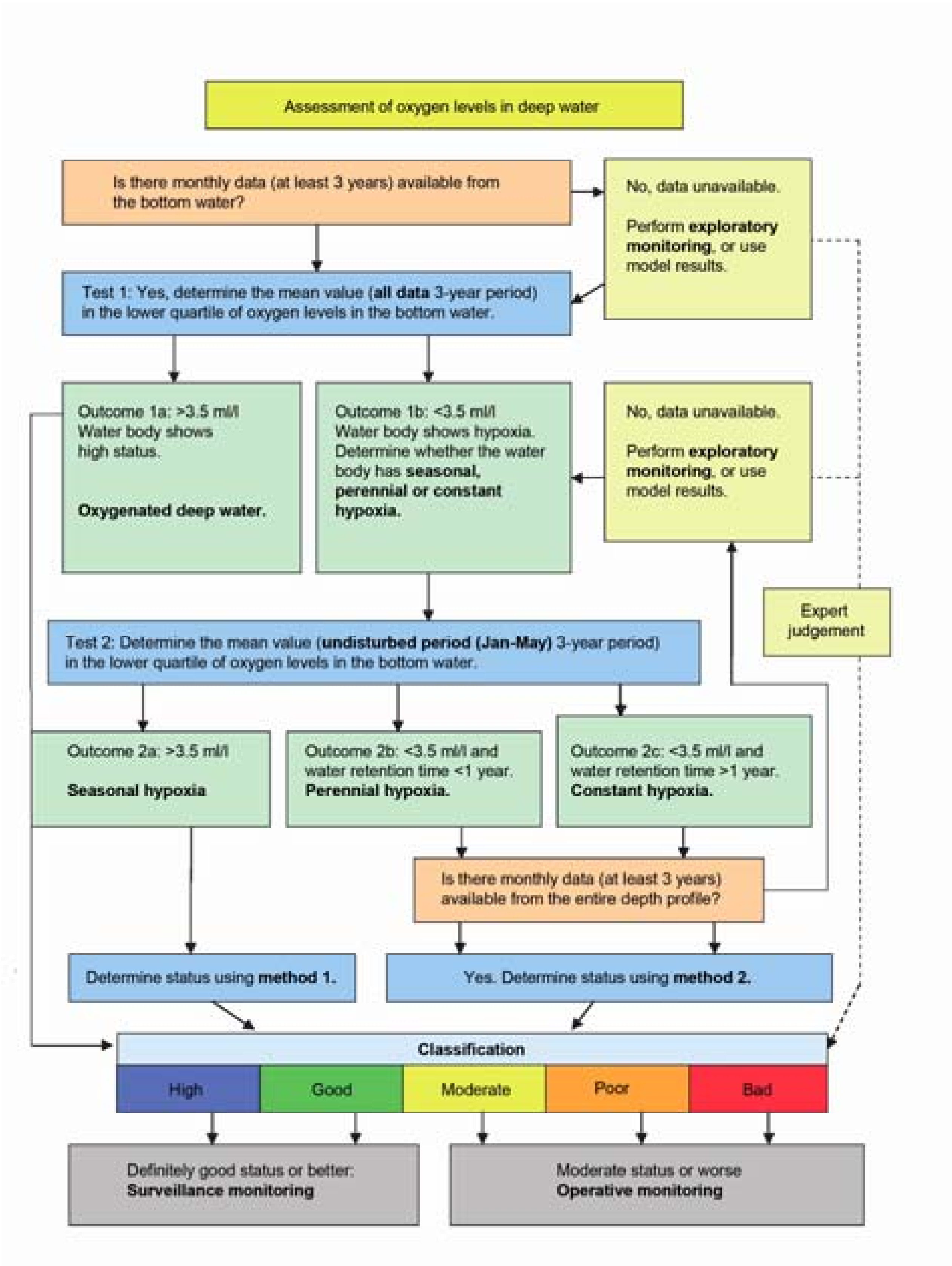

Figure 7.2. Flowchart for oxygen status classification in coastal waters.

7.3 Classification of status

There are two methods of determining the oxygen status in a water body. Which method is used depends on the outcome of test 1 and test 2 (Sections 7.2.2 and 7.2.3). The status of water bodies affected by seasonal oxygen depletion according to tests 1 and 2 shall be classified based on method 1. The status of water bodies showing signs of perennial or constant oxygen depletion shall be classified based on how much bottom area is affected by depletion, i.e. method 2.

7.3.1 Status according to method 1 (for water bodies with seasonal oxygen depletion)

For water bodies with seasonal oxygen depletion, classification shall be performed based on the results of test 1 (Section 7.2.2). Status is determined based on the station mean value in the lower quartile (the lowest 25% of the concentrations) of observed oxygen levels in the bottom water from January - December during a three-year period. The value is compared with the class boundaries given in Table 7.1 to obtain the status classification.

See REG Annex 5, Section 3.2

7.3.2 Status according to method 2 (affected surface area of the bottom, for water bodies with perennial or constant oxygen depletion)

For water bodies that show signs of perennial or constant oxygen depletion, status classification shall be based on the mean value of oxygen levels for the months June-December during a three-year period, and is expressed as a proportion of the total bottom area exposed to the depletion. When classifying using method 2, measurement data must have been collected from the entire water profile, from the surface to the bottom - every fifth meter in water that is shallower than 20m, and every tenth meter in water that is deeper than 30 m and samples using a bottom water sampler to determine the oxygen level just above the bottom. The level of oxygen depletion on the bottom is calculated using the vertical distribution of the oxygen level and the hypsographic curve. The depth where the oxygen level is 3.5 ml/l is calculated for each oxygen profile and we obtain the exposed bottom area for the depth in question from the hypsographic curve. This value is then compared to the class boundaries in Table 7.3 to obtain the status classification.'

See REG Annex 5, Section 3.2

In water bodies exposed to perennial or constant oxygen depletion, the depletion is most often caused by both increased load and natural morphological obstacles that inhibit water exchange. To set reasonable reference values, we calculate how much of the bottom surface in the water body (applicable to basins that have a limited surface area) is disturbed. The oxygen level in water bodies with perennial or constant oxygen depletion during the disturbed period cannot be better than the prevailing conditions during the undisturbed period (January-May). The objective is to identify the anthropogenically induced problem areas.

Hypsographs have been developed for all the basins in the Swedish Water Archive at SMHI.[21] Hypsographs describe how the surface area of a basin varies depending on its depth, for each metre from the surface to the maximum depth (data files with hypsographs for all water bodies can be found at www.smhi.se). By linearly interpolating oxygen profiles from discrete depths to include the whole water column, it is possible to find the critical depth at which oxygen levels of less than or equal to 3.5 ml/l are first encountered. On occasions when the critical depth is under the deepest measurement, the two deepest measurements are used to linearly extrapolate the oxygen levels down to the maximum depth in order to identify the critical depth (Figure 7.3). Using this method, we can obtain the proportion of affected bottom surface in a basin.

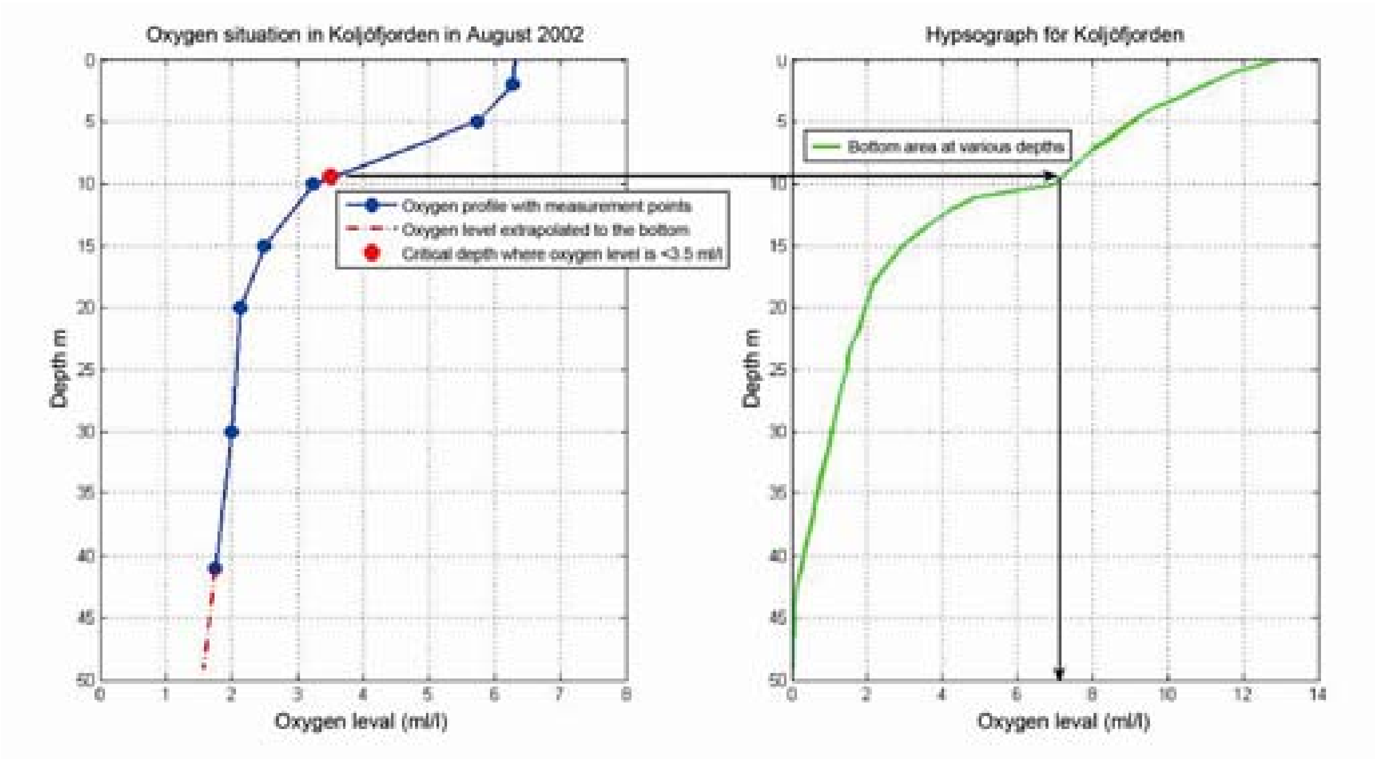

Figure 7.3. To the left: Oxygen levels in Koljöfjorden during August 2002. Interpolation has been performed between the data points and at the bottom, the two deepest data points have been used to extrapolate oxygen levels down to the bottom. The critical depth at the 3.5 ml/l level is highlighted. To the right: Hypsograph for Koljöfjorden illustrating how the bottom surface various with depth and the arrow shows the surface of the basin that is affected by oxygen levels of <3.5 ml/l.

[21] Lindkvist et al. 2003

7.4 Reference values and class boundaries

7.4.1 Water bodies with seasonal oxygen depletion

The general reference value for oxygen levels in Swedish deep waters has been set to >3.5 ml/l, lower values indicate depletion. The limit for acute depletion has been set to 2.1 ml/l, the limit at which several benthic flora and fauna display acute hypoxia. The boundary between moderate and poor status has been set to 1 ml/l. The boundary for bad status has been set to the point at which anoxic conditions occur and hydrogen sulphide (H_2S) has formed.

Table 7.1. Oxygen status in bottom waters according to method 1. The boundary between good and moderate status have been set at 2.1 ml/l, the limit at which several benthic flora and fauna display acute hypoxia.

Status	Limit value
High	>3.5 ml/l
Good	<3.5 ml/l - 2.1 ml/l
Moderate	<2.1 ml/l - 1 ml/l
Poor	<1 ml/l - H_2S
Poor	H_2S

See REG Annex 5, Section 3.3

Table 7.2 below presents water bodies that have been assessed as suffering from seasonal oxygen depletion and for which sufficient data exists.

Table 7.2. Water bodies disturbed by seasonal oxygen depletion. (Bottom surface with less than 1% oxygen depletion)

Stockholm archipelago	Laholmsbukten, Skälderviken & Öresund	Himmerfjärden	West coast
Strömmen (Blockhusudden)	Laholmsbukten (L9)	Himmerfjärden (H4)	Brofjorden
Askrikefjorden (Halvkakssundet)			Halsefjord (Galterö)
Strömmen (Kastellholmen)			Stigfjorden
Kallskärsfjärden (S. Möja)			

7.4.2 Water bodies affected by perennial or constant oxygen depletion

Table 7.3 presents reference values generated for water bodies that are affected by perennial or constant depletion. The reference values have been calculated from monthly mean values for the undisturbed period January-May.

In coastal areas where the perennial depletion is irregular, above all a number of water bodies in the Stockholm archipelago, an expert judgement of the oxygen situation in a longer term perspective may be necessary. In cases where perennial depletion is irregular, it may be necessary to abandon the status classification of disturbed bottom surfaces after a reasonability assessment and an expert judgement can be made based on the outcome of test 1 and on Table 7.1.

Table 7.3. Class boundaries for water bodies affected by perennial oxygen depletion. The following water bodies are deemed to affected by perennial depletion and shall be classified based on the proportion of disturbed bottom surface.

Water body (station)	**Class boundaries for proportion (%) of bottom area affected by oxygen depletion.**				
	High	Good	Moderate	Poor	Bad
Stockholm archipelago					
Tranholmen area (Ekhagen)	≤ 22	> 22-33	> 33-38	> 38-43	> 43
Kanholmsfjärden (Kanholmsfjärden)	≤ 14	> 14-21	> 21-48	> 48-75	> 75
Skurusundet (Lännerstadssundet)	≤ 30	> 30-45	> 45-48	> 48-50	> 50
Askrikefjärden (Älvvik)	≤ 2	> 2-3	> 3-35	> 35-67	> 67
Laholmsbukten, Skälderviken & Öresund					
Laholmsbuktens coastal waters (Hallands väderö)	≤ 11	> 11-16	> 16-55	> 55-93	> 93
Northern Oresund coastal waters (Kullen)	≤ 4	> 4-6	> 6-42	> 42-77	> 77
Skälderviken (S2)	≤ 8	> 8-12	> 12-45	> 45-78	> 78
Skälderviken (S5)	≤ 29	> 29-44	> 44-61	> 61-78	> 78
Northern and central Öresund coastal waters (W-Landskrona)	≤ 7	> 7-11	> 11-46	> 46-80	> 80
West coast					
Havstensfjord (Havstensfjord)	≤ 11	> 11-16	> 16-28	> 28-40	> 40
Koljöfjord (Koljöfjord)	≤ 14	> 14-20	> 20-27	> 27-33	> 33
Gullmarn central basin (Alsbäck)	≤ 16	>16-24	> 24-53	> 53-82	> 82

Table 7.4. Class boundaries for water bodies considered to be affected by natural oxygen depletion.

Water body (station)	**Class boundaries for proportion (%) of the bottom area affected by oxygen depletion**				
	High	Good	Moderate	Poor	Poor
Byfjorden (Byfjorden)	≤ 40	> 40-60	> 60-64	> 64-68	> 68

The inner Gamlebyviken may also be classified as a water body affected by constant oxygen depletion, but the available data is not sufficient to set a reference value.

7.4.3 A calculation example for oxygen

The data for this example comes from the Orust and Tjörn fjord system.

Test 1 - Does oxygen depletion occur?

Determine the station mean value (all months during a three-year period) in the lower quartile of observed oxygen levels in the bottom water to ascertain whether oxygen depletion occurs.

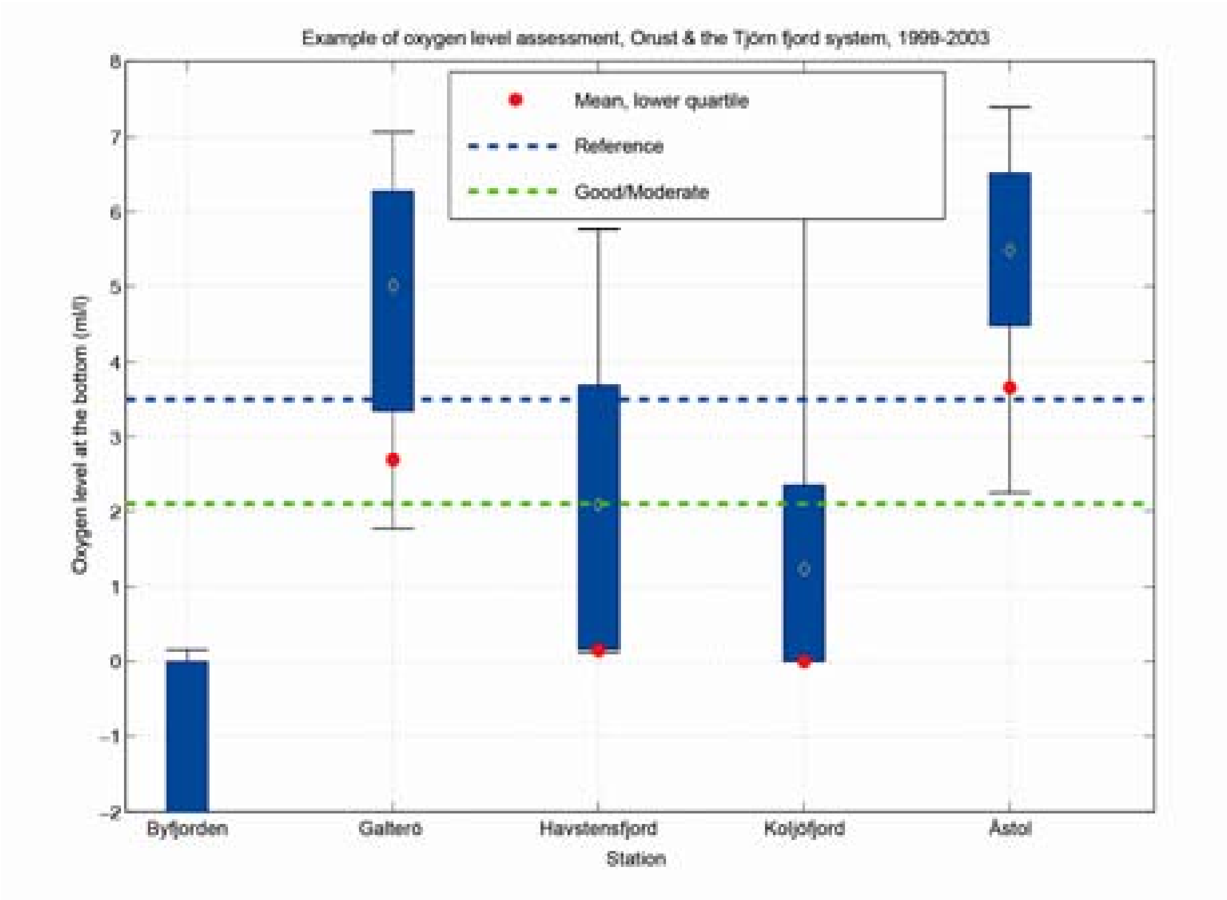

Figure 7.4. Outcome from test 1. Åstol demonstrates oxygen levels over the reference value, i.e. high status, whilst other stations show oxygen depletion and must undergo test 2. The green dotted line is only used when the outcome of test 2 is 2a (seasonal depletion) and indicates the boundary between good and moderate status.

Åstol is the station that can be directly considered to have oxygenated deep water (high status). All the other stations in the fjord system have a mean value in the lower quartile that is less than 3.5 ml/l and shall therefore undergo test 2 to determine whether the water body is affected by seasonal, perennial or constant oxygen depletion.

Test 2 - Is the oxygen depletion seasonal, perennial or constant?

In test 2, only data from the undisturbed period, January-May, is analysed in order to ascertain the type of depletion present. Data on deep water retention time is also useful if it is available. The outcome of test 2 is presented in Figure 7.5 below.

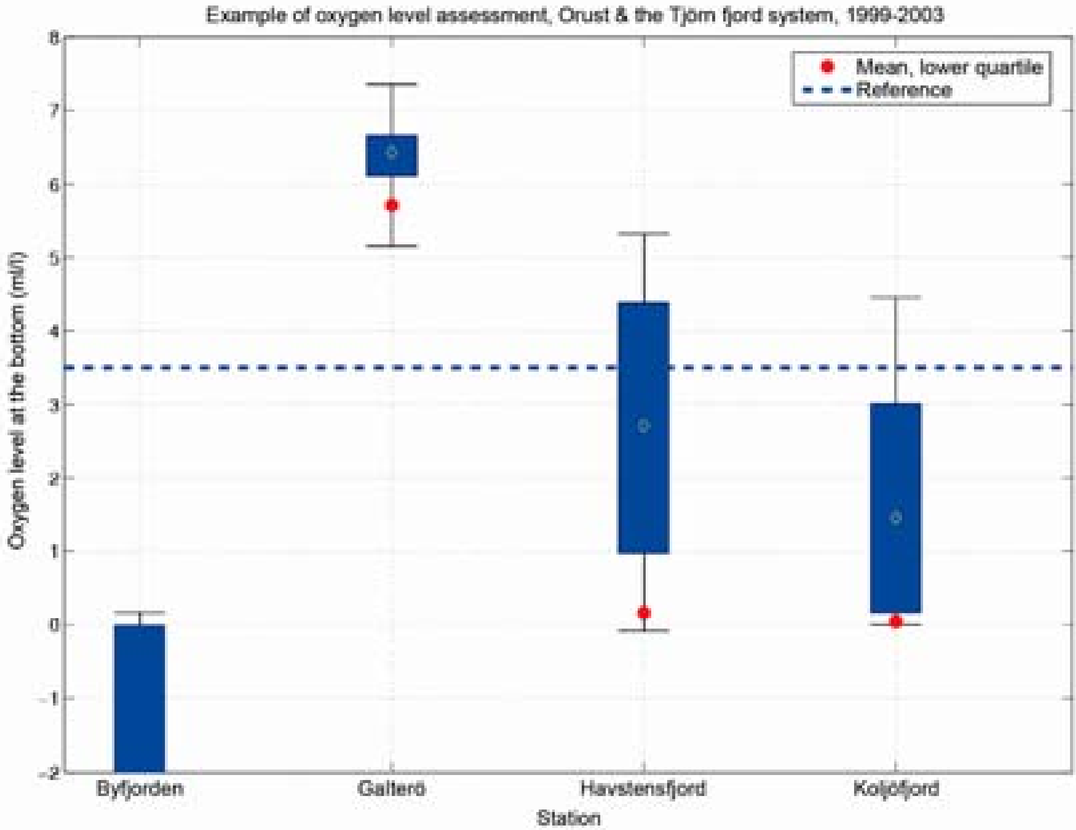

Figure 7.5. Outcome from test 2, for a discussion see below.

Galterö has a long retention time but its bottom water is still oxygenated during the undisturbed period, January-May, and can therefore be considered to have seasonal depletion. Its status shall therefore be classified using method 1 and the results from test 1 can then be used. The mean value in the lower quartile is between 2.1 ml/l and 3.5 ml/l and is therefore assessed to have good oxygen status. See Figure 7.4 in test 1 above.

Both Koljöfjord and Havstensfjord have long deep water retention times (>160 days) though not in excess of 12 months. The oxygen level in the lower quartile during the undisturbed period is close to zero. Both water bodies can therefore be considered to be affected by perennial depletion. Their status shall therefore be classified using method 2. Reference values and class boundaries are presented in Table 7.3.

The proportion of disturbed bottom surface using method 2 (based on the months June-December) for Havstensfjord is calculated at 49% and at 50% for Koljöfjord, which, according to Table 7.3 classifies their status as bad.

Byfjorden is known for being affected by perennial oxygen depletion and anoxia. The long deep water retention time (> 12 months) means that it is considered to be affected by natural hypoxia. Byfjorden shall be classified using method 2. Class boundaries for the classification of Byfjorden are given in Table 7.4. The proportion of disturbed bottom surface in Byfjorden during the period 1999-2003 was calculated at 66%, which, according to Table 7.4, classifies it as poor status.

7.5 Comments

There is no indication that oxygen depletion is a problem along the Halland coast or in the coastal waters of Hanöbukten and Blekinge. The data from several stations along the east coast is insufficient. It is therefore currently not possible to examine whether seasonal or perennial anoxia/hypoxia occurs nor what proportion of the bottom surface is affected. There is no data from the central and southern coast of Sörmland. The Gulf of Bothnia has no oxygen problems in general. In some water bodies, however, oxygen consumption can also occur during the winter. This is particularly true of coastal areas of the Gulf of Bothnia where large volumes of organic matter are discharged via rivers and streams. Unfortunately, the data from the Gulf of Bothnia has been limited during the development of the assessment criteria and has therefore not been included.

If perennial depletion were to be detected in a water body that has no class boundaries, new reference conditions can be created if there is sufficient data available from the deepest part of the water body. Reference values and class boundaries for the proportion (%) of the bottom surface that is affected by hypoxia based on data collected over the last 10 years. Similar to when classifying status, the surface area affected by oxygen levels of <3.5 ml/l is calculated, the difference being that data from the “undisturbed period” January-May is used instead. The mean value for the 10-year period will then become the reference value and the boundary between good and moderate is set to the reference value of * 1.5. The boundary for bad status is set to the maximum surface area that can be affected in the water body, i.e. the surface area that is circumscribed by the approximate depth distribution of the pycnocline. Other class boundaries, good and poor, are evenly distributed between the reference value, good-moderate and bad status.

If there is insufficient data, a shorter period can be used, although at least 5 years, to support an expert judgement based on the results of test 1.

Background report: Bedömning av syrgashalt i kustvatten enligt Vattendirektivet – metodbeskrivning [Assessment of oxygen content in coastal water in accordance with the Water Framework Directive
- method description
Authors: Martin Hansson and Bertil Håkansson (SMHI)

8 Specific pollutants in coastal and transitional waters

8.1 Introduction

In the Swedish Ordinance on Water Quality Management and the European Water Framework Directive (WFD), toxic chemical substances in the water environment are dealt with in two different categories. Substances that have common EU environmental quality standards (above all the priority substances but also a number of other substances regulated by EC fishing waters and crustacean directives) are included in the classification of surface water chemical status, see also Chapter 5 in the main handbook. In addition to these, specific pollutants shall be classified as one of the physico-chemical quality elements when classifying ecological status.

See REG Annex 2, Section 7

What these pollutants are may vary from one water body to the next depending on different types of impact. Annex V of the Water Framework Directive (WFD) states that the substances to be classified are any pollutants that are discharged into bodies of water in significant quantities.

8.2 Choice of specific pollutants

What is meant by a substance being discharged in significant quantities? In the EU Guidance no 3 (Analysis of pressures and impacts)[22] the concept of "being discharged" is interpreted in a wide sense. It covers discharges from point sources in the drainage basin, leakage from diffuse sources and e.g. atmospheric deposition from other areas. We should therefore consider all the possible pathways by which the pollutant can reach the water body. The Swedish EPA interprets "significant quantity" as a quantity of a substance that can prevent the biological status-/potential from being fulfilled by 2015.

The water authorities shall classify the specific pollutants discharged into the water body. Discharged substances are identified with the help of the supporting data produced when assessing disturbance (See survey and analysis handbook). The EU Guidance describes the procedure for selecting the specific pollutants in each drainage basin and in particular water bodies. Here is a summary of the most important steps.

[22] Common Implementation Strategy for the Water Framework Directive (2000/60/EC) Guidance no 3 Analysis of pressures and impacts, produced by working group 2.1 – IPRESS, 2003

1. Starting-point
The indicative list of the main pollutants set out in Annex VIII of the WFD can be the starting-point of the selection process.

2. Screening of information
A screening of all available information on pollution sources, impacts of pollution and production and usage of pollutants in order to identify those pollutants that are being discharged into water bodies in the drainage basin.

2a. Collation of data/information
Data from:
- Sources - Production, industrial processes, usage, treatment, emissions
- Impacts - Change in the occurrence of pollutants in the water body (environmental monitoring data)
- Pollutants - Intrinsic properties of the pollutants affecting their likely pathways into the water environment.

Information from existing programmes/registers, e.g.:
- Swedish Chemicals Emissions Register (KUR)
- C-EMIR (emissions from point sources)
- MIFO (contaminated areas)

2b. List of pollutants
Assessment of information collated under Step 2a will result in a list of those pollutants identified as being discharged into water bodies in the drainage basin. Pollutants that with sufficient certainty are not being discharged into water bodies in the drainage basin can now be excluded from further considerations.

3. Test for relevance
All the pollutants being discharged in the drainage basin have been identified in Step 2. Step 3 tests which of these are relevant. In other words, those pollutants that are likely to cause, or are already causing, harm to the water environment. This will depend on the intrinsic properties of the pollutants, their fate and behaviour in the environment and the magnitude and form of their discharges. Selection should ideally be based on an assessment of the environmental significance of the concentrations estimated for the pollutant or its breakdown products in the water body. In other cases, effect data and modelling of e.g. critical loads can also be used.

3a. Data on concentrations and loads
Obtaining data through monitoring and/or modelling.

3b. Comparing concentrations with limit/guideline values
Pollutants identified under Step 2 may be excluded where their concentrations are estimated to be lower than the most relevant critical value such as estimated LC50, NOEC, PNEC, EQS or model estimations for e.g. critical load.

Natural background concentrations of non-synthetic pollutants (mostly metals) may exceed EQS without them necessarily being considered relevant.

Potential bioaccumulations of the pollutant in sediment or biota should be considered.

4. Safety net
A safety net is needed to ensure that pollutants that may be environmentally significant are not incorrectly excluded from the list of specific pollutants during Step 3. For example, it should be considered;

- whether a number of small (individually minor) pollution sources may be expected to have a significant combined effect,
- whether there is a trend indicating the increasing importance of a pollutant, even though the limit value is not currently exceeded, and
- whether pollutants are present that have similar modes of toxic action and hence via additive or synergetic effects may cause significant impacts.

5. Final outcome
The final outcome is a list of specified pollutants that are relevant to a drainage basin or for specific water bodies within a drainage basin.

It is therefore the water authorities that select the relevant specific pollutants for each water body. Class boundaries should be established for these pollutants in accordance with Annex V of the WFD so that the status of the specific pollutants quality element can be established.

8.3 Establishing class boundaries

Class boundaries should be established for water, sediment or biota matrices depending on which of the matrices the most sensitive organism is exposed through. If ecotoxicological studies indicate that aquatic organisms are affected at the lowest concentrations of a pollutant, class boundaries should be established for water. If sediment-dwelling organisms are the most sensitive, the class boundaries should instead be established for sediment and if it is birds, mammals or humans who feed off the water environment (e.g. fish or crustaceans) and who, via secondary poisoning, react at the lowest concentrations, class boundaries should be established for biota.

The water authorities shall establish class boundaries between high and good and between good and moderate status in accordance with the normative definitions in Annex V Tables 1.2.1 - 1.2.2 in the WFD. How to set the boundary between good and moderate status is described in detail in Annex V, Section 1.2.6 of the WFD.

For help when establishing class boundaries, the water authorities can use the values that have already been established in accordance with the methodology described in Annex V of the WFD. As an example, there is the report entitled "Proposals for limit values for specific pollutants - support for the water authorities when classifying status and establishing environmental quality standards", in which the Swedish Chemicals Agency, on behalf of the Swedish EPA, has drawn up proposals for limit values which the water authorities can use as class boundaries for a number of chemical substances that are considered problematic in certain areas in Sweden.

8.4 Classification of status

When classifying the status of specific pollutants, the measured concentration in the water, sediment or biota in the water body of the substances identified as being discharged in significant quantities is compared to the class boundaries established by the water authority. The substance with the lowest status determines the total status for the specific pollutant quality element. "One out all out" is therefore the principle being used.

8.4.1 Non-synthetic pollutants

Regarding non-synthetic pollutants (mostly metals), tables 1.2.1 - 1.2.2 in Annex V of the WFD states that high status should correspond to undisturbed conditions, i.e. the natural background concentration in the water body. In this handbook, the background concentration is defined as the concentration found before industrialism had really started and before agriculture was rationalised and began using chemicals to a much greater extent. It is therefore not possible to simply use the concentration in a water body that currently has no direct discharges of the substance. Historical pollutants and contributions from diffuse sources, such as atmospheric deposition, should also be taken into consideration. The water authority makes an assessment of the natural background concentration for the water body based on all the information available. The class boundary between high and good status is set as the background concentration for the water body whilst the class boundary between good and moderate status is determined based on ecotoxicological data in accordance with the procedure laid down in Annex V, 1.2.6 of the WFD and is specified for the bioavailable concentration.

The measured filtered (0.45 µm filter) concentration is compared to the class boundaries. If any of the class boundaries are exceeded at this stage, a more detailed analysis should be performed to determine whether this is due to a significant environmental impact or whether the high concentration has natural causes. The analysis comprises:

1. Assessment of the background concentration

If the background concentration is high, the water authority should consider this and assess the risks for biological effects based on the local conditions. The natural level of most metals in water can be assessed with acceptable accuracy based on analyses from upstream points or nearby water areas that are undisturbed by local emissions and acidification. In the absence of such analysis values, standardised values for background concentrations can be used. There are OSPAR agreements with background values for metals in water, sediment and to a certain extent biota (OSPAR Agreement 2005-6 and OSPAR Agreement 1997-15).

2. Assessment of bioavailability
An analysed sample of the total filtered concentration of a metal tells us rather little about its biological effect. It is the bioavailable concentration that is significant as regards the magnitude of the impact the pollutant has on organisms. What proportion of the concentration is bioavailable depends on a number of different factors. It depends firstly on the type of discharge. If the discharge consists of metals in mineral form, only a small proportion is available compared to if the discharge consists directly of metal ions, which gives a very high bioavailability. The availability also depends on the chemical properties of the water. Based on the factors described, the water authority should make an assessment of the bioavailable concentration that can be compared to the class boundary. Models that calculate the bioavailable concentration based on total concentrations and determinants are currently being developed at the EU level but have yet to be sufficiently verified for Swedish conditions to be used straight away. It is possible to use these in combination with expert judgements, however.

8.4.2 Synthetic pollutants

Synthetic pollutants are substances that should not occur in the environment in undisturbed conditions. Regarding these substances, it is stated in tables 1.2.1 - 1.2.2 in Annex V of the WFD that high status should involve concentrations close to zero and at least lower than the detection limit when using the advanced analysis technique in operation. The class boundary between high and good status is hence consequently set to the detection limit. It is important, however, that the detection limit is defined for each relevant substance so that it is as low as possible in order to be measured using the current technology since different analysis methods can otherwise give rise to widely differing limits.

The class boundary between good and moderate status is determined based on ecotoxicological data in accordance with the procedure laid down in Annex V, Section 1.2. 6 in the WFD.

8.5 Comments

Class boundaries for pollutants should be calculated using the method described in Annex V, Section 1.2.6 of the WFD, i.e. the methods which EU Member States have agreed to use. This means that the established class boundaries are based on ecotoxicological effects studies on different trophy levels, and for humans or predators that feed off the water environment, and take the most sensitive organisms into consideration. These methods are not comprehensive and any additives or synergy effects are for example not taken into account even though shortcomings in the supporting data have been corrected with safety factors. Due to this, it cannot be guaranteed that effects on biota will not occur as a result of the exposure of hazardous substances despite no class boundaries being exceeded. Such effects should, however, be detected due to the fact that the biological quality elements must always be assessed. If the biology indicates an impact, the water body is classified as having moderate or worse status even if the physico-chemical status is good. The parameters currently assessed for the biological quality elements don't specifically indicate a toxic impact but do give a clearer response to nutrient stress or to hydromorphological impact. This will be developed in the future so that parameters are established which respond more clearly to a toxic impact.

In cases where class boundaries for a substance have been set for the water phase but measurement data is unavailable, data for the relevant substance in sediment or biota can be used to make an expert judgement of whether the class boundaries risk being exceeded or not. Conversion models can be used to estimate the equivalent of a sediment or biota concentration in water. Such a model is described in the report containing proposals for limit values from the Swedish Chemicals Agency. Furthermore, values for sediment that correspond to values for water have also been determined using the method described in Annex V, Section 1.2.6 of the WFD. These conversion models are rather unreliable and the results must be evaluated with an expert judgement. If a value is deemed to be close to a class boundary, this can be seen as an indication of a need for sampling in the water phase.

Annex C - Assessment criteria for hydromorphological quality elements

(This annex contains the text for all assessment criteria for hydromorphological quality elements and can be downloaded as a separate document from the Swedish EPA's website at www.naturvardsverket.se. The reason for this is so that the user can avoid having to download files that are very big and hence difficult to handle).

Contents

1 Introduction

The assessment criteria for hydromorphological quality elements in lakes and watercourses have been developed by SMHI (Swedish Meteorological and Hydrological Institute) and Jönköping County Administrative Board on behalf of the Swedish Environmental Protection Agency (Swedish EPA). All background reports on assessment criteria are presented online at www.naturvardsverket.se. There may be differences between what is contained in the background reports and in the handbook. The handbook is the most up-to-date publication and represents the Swedish EPA's position on the material.

This annex is one of three annexes to the Handbook for status, potential and quality requirements for lakes, watercourses, coastal and transitional waters. Please note that section numbering can recur in different annexes but a reference to a given section in the annex always refers to the relevant section of this annex.

1.1 Input quality elements and parameters

The hydromorphological quality elements are divided up into morphology, hydrological regime and continuity. In turn, the quality elements consist of one or more parameters (see Table 1.1).

The vast majority of the assessment criteria described in the handbook are regulated in Swedish EPA Regulations (NFS 2008:1) and General Guidelines on the Classification of and Environmental Quality Standards for Surface Water. The others can be used as support parameters in an in-depth assessment when there is sufficient supporting data and when the need arises.

As a result of insufficient knowledge, or the fact that it has not been considered relevant to Swedish conditions, assessment criteria have not been developed for all the parameters (normative definitions in accordance with Annex V of the WFD (2000/60/EC). Regarding hydromorphological quality elements in coastal and transitional waters, Chapter 6 only provides a short summary of feasible assessment bases that can be used as an aid when classifying status and potential.

The assessment criteria are presented in the form of tables with a five-point assessment scale, where class 1 indicates the least impact and class 5 the most. For each impact class, the status which a specific impact on hydromorphology is deemed to correspond to is also given. Considering the hydromorphological assessment criteria, the classification "moderate", "significant" or "heavy" impact, i.e. classes 3-5, means that the bodies of water which have been assigned this classification may be heavily modified. The higher the class, the stronger the indication that the body of water may be heavily modified.

Regarding the hydromorphological assessment criteria, no types have been established and in the handbook, object-specific tools are instead described to establish reference values and status classes. The only exception consists of an in-depth assessment criterion for the number of flow peaks, where a division has been made for southern and northern Sweden and the size of the river basin district.

Table 1.1. Summary of parameters for all hydromorphological quality elements for which assessment criteria have been developed. Parameters in italics cannot be found in the regulations but can be used as an aid to classification.

Lakes	**Quality elements**	**Parameters**
Hydromorphological elements	Continuity	Presence of man-made migration obstacles
	Hydrological regime	Prescribed regulation amplitude
		Impact on water level changes
	Morphological conditions	Land use in the vicinity
		Land use in sub-basin district
		Dead wood (number of pieces of wood)
		Modified littoral zone
		Number of ditches per km
Watercourses		
Hydromorphological elements	Continuity	Presence of artificial migration obstacles
		Degree of fragmentation
		Barrier effect
	Hydrological regime	Impact of flow regulation on the watercourse: - degree of regulation - modified mean high water, MHQ - reduced mean low water, MLQ
		Number of flow peaks per year
		Variation coefficient for daily flow
	Morphological conditions	Degree of straightening / canalisation
		Proportion of length cleared
		Number of road crossings per km
		Land use in the local environment
		Land use in sub-basin
		Number of ditches per km
		Dead wood (number of pieces of wood)
Coastal and transitional waters		
Hydromorphological elements	*Not available*	

1.2 Areas of use

Hydromorphology need only be assessed when the biological quality elements are classified as high status or maximum potential. In this way, the hydromorphological assessment criteria should work as a support to the biological assessment criteria when the ecological status is to be classified as good or high or when the ecological potential is to be classified as good or maximum. For the final ecological classification, therefore, the only relevant information is whether a body of water indicates high or good morphological status. Just as for other quality elements, the principle of "one out all out" applies even when the hydromorphological quality elements are to be combined/co-weighted (Figure 1.1.).

To obtain an accurate picture of the hydromorphological status, a five-point assessment scale has been developed. Using this five-point scale, the hydromorphological assessment criteria should also be possible to use to make a general assessment of the impact in an area. In other words, it will be possible to use the assessment criteria as a basis for an expert judgement of the status (provided that there is deemed to be a clear link between hydromorphological impact and effects on the biological quality elements) and in order to register improvements/deteriorations. The assessment criteria may possibly also be used as an aid in the work to assess whether a water body might potentially be declared artificial or heavily modified. The starting-point for assessing whether a water body is artificial or heavily modified is linked to the physical impact from certain types of specifically stated, public utility undertakings. If the physical impact is so extensive as to require hydromorphological remediation measures in order to achieve good status, but such measures would have a significantly negative effect on the undertakings, there may be grounds for declaring a water body to be heavily modified. If the ecological status is deemed to be high or good, there are on the other hand no grounds for identifying the water body as heavily modified. The process of declaring a body of water as heavily modified or artificial is dealt with in its entirety in the Swedish EPA's forthcoming guide on artificial and heavily modified water bodies.

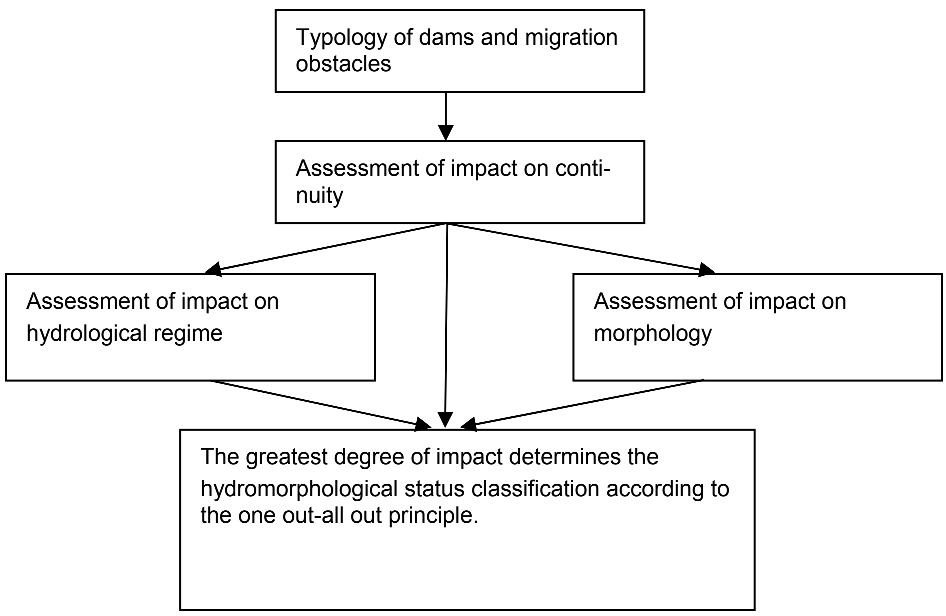

Figure 1.1. Draft working procedure when assessing the impact on hydromorphology in lakes and watercourses within a river basin. See also Section 5.1 for more information about the draft working procedure.

1.3 Data management tools

Data for the assessment of hydromorphological impact is best processed in digital map layers. ArcGIS and mainly ArcMap have been used to manage data during the development and testing of these assessment criteria. Since it is a question of dealing with background assessment data for many geographically defined objects, it is practical to work with the map layers in geodatabases, which allow tables with attributes to be used in Access. Classification results can be efficiently updated if attribute tables for map layers with water bodies are linked to tables in the geodatabase. Input data to the map analyses can originate from different types of databases or data tables. The results can also be processed in computation or database software after the map work.

When handling large volumes of data, it is also important that data tables with input data to be analysed have well-defined column content. Column headings should preferably not have more than 10 characters and text cells a maximum of 50 characters. Free text is to be avoided as this type of information is unusable in rational analysis work and occupies unnecessary space in data tables. Explanations of table content are given in separate documents for metadata or on separate tabs in Excel. Empty numerical cells are given the value 0 when imported into the data management software so it is practical to input an "unreasonable" figure such as -999 for missing observations in order to be able to distinguish missing observations from 0 in the database.

In tables with positioned point information, the point's position shall be specified using coordinates in accordance with RT90 2.5 Gon West or the coordinate systems that are relevant for national users. In Sweden, Lantmäteriet (The National Land Survey) began converting from RT90 to a globally adapted coordinate system SWEREF99. If the data table content represents properties or assessment results for a line or a surface area in a map layer, the data table shall contain the map object's unique ID so that data can be linked to the object in the map software.

In order to make it easier to classify many objects at the same time, the following *digital map layers* are recommended:

- Points for dams or other migration obstacles for fish
- Lines for the water system's flowpaths
- Polygons for water surfaces
- Water bodies
- Watersheds for river basins

2 Continuity

2.1 Introduction

Classification of continuity covers the changes in a water body that may affect the dispersion and free passage of animals, plants and nutrients. Impaired continuity, i.e. links between stretches of watercourse, in the form of dams, weirs or incorrectly sited culverts, constitute a major problem for the ability of various organisms to migrate and spread. The effect of fragmentation, including isolated fish populations as a consequence, entails a major risk of the genetic impoverishment of the confined populations and a risk of genetic drift. Fragmentation also means that it becomes impossible for fish to migrate from watercourses out to sea and lakes to mature, and to return to spawn. Assessment criteria based on the presence of man-made migration barriers use fish as indicator organisms. Salmon and trout have been chosen as the primary indicator species because their populations are most frequently adversely affected by migration barriers and because there is a good knowledge basis about the need and capacity of these species to migrate.

2.2 Input parameters

The quality element Continuity is divided into two sub-quality elements (see Table 2.1). In the sub-quality element Continuity in lakes, the only parameter included is man-made migration barriers. This parameter is regulated in NFS 2008:1.

See REG Annex 3, Section 1

Three parameters are included in the sub-quality element Continuity in watercourses: presence of man-made migration barriers, Degree of fragmentation and Barrier effect. All these parameters are regulated in NFS 2008:1.

See REG Annex 3, Section 2

Degree of fragmentation and Barrier effect must be combined or co-weighted into a common value. The final classification is obtained by comparing the combined value for Degree of fragmentation and Barrier effect with the value that is classified for the parameter Presence of man-made migration barriers. If these values show different levels of impact, it is the value that indicates the greatest anthropogenic disturbance that determines the result (see also Section 5.2, Co-weighting of continuity). This is done in accordance with the "one out – all-out" principle which in this case is allowed to apply at parameter level.

Table 2.1. Parameters for the classification of continuity.

Assessment criterion (parameter)	Supporting data	Watercourses	Lakes
Presence of man-made migration barriers	Dam Register, biotope-mapping	X	X
Degree of fragmentation	Dam Register, biotope-mapping	X	
Barrier effect	Dam Register, biotope-mapping	X	

2.3 Presence of man-made migration barriers in lakes and watercourses

The presence of man-made and natural barriers limits the migration of non-stationary fish.

2.3.1 Requirements for supporting data

See REG Annex 3, Section 1.1.1

The Swedish Dam Register, which is managed by SMHI (Swedish Meteorological and Hydrological Institute), or background data from mapping carried out in accordance with the manual for 'biotope–mapping – watercourses', or another method that gives equivalent results, must be used when classifying the parameter Man-made migration barriers. The information must show geographical position and should be linked with a map layer showing the flow paths in the water system, e.g. the Swedish Watercourse Register SMHI 2006.

2.3.2 Classification of status

Man-made and natural migration barriers for fish are located and mapped in the watercourse system. Classification based on the presence of man-made migration barriers functions best if water bodies are limited to those where there is a man-made or natural migration barrier.

The presence of natural migration barriers is used as supporting data to map the natural migration sections in all linked watercourses in a river basin. The naturally linked migration routes are identified within each river basin and all water bodies within each *passable reference area* are given a unique code. If the river basin contains no natural migration barriers or large lakes that constitute important breeding grounds for migrant fish, all water bodies all the way down to the river mouth are naturally passable. Vänern and Vättern constitute examples of lakes that may be considered as breeding grounds for migratory fish (e.g. lake salmon).

Continuity within a migration area is undisturbed if there are no man-made migration barriers. All water bodies, lakes and watercourses within such an area are assigned high hydromorphological status, Class 1, as regards continuity.

Man-made migration barriers also have an adverse effect on organisms that migrate short distances. The impact on a water body is regarded as corresponding to Class 3, moderate status, if the migration barrier adjoins the water body. A water body is also accorded moderate hydromorphological status, Class 3, as regards continuity, if a man-made migration barrier lies in, or directly downstream from it. Class 3 status is also assigned to lakes which, because of man-made migration barriers, have lost a large proportion of the natural migration routes upstream and downstream and which in addition have one or more man-made migration barriers downstream.

Water bodies that lie upstream of a natural migration barrier are not disturbed by man-made migration barriers downstream from the natural migration barrier. In a natural state, fish migrating upstream would not in any case have had the possibility of reaching water upstream of the natural barrier. If there are no man-made migration barriers upstream of the natural barrier, all water bodies upstream of the natural barrier are assigned high hydromorphological status, Class 1, as regards continuity. This was the outcome of a test classification of the upper reaches of the River Vindelälven's main channel, which found that only a small proportion of anadromous fish in the natural state would have migrated to the upper reaches of the river. The man-made migration barrier in Stornorrfors was therefore assessed to have a negligible effect on the water bodies in the upper reaches of the river. The results of test classifications are shown in Figures 2.1 and 2.2.

If a man-made migration barrier is co-located with a natural migration barrier and has the same barrier effect, the man-made migration barrier should not be counted as disturbing the continuity.

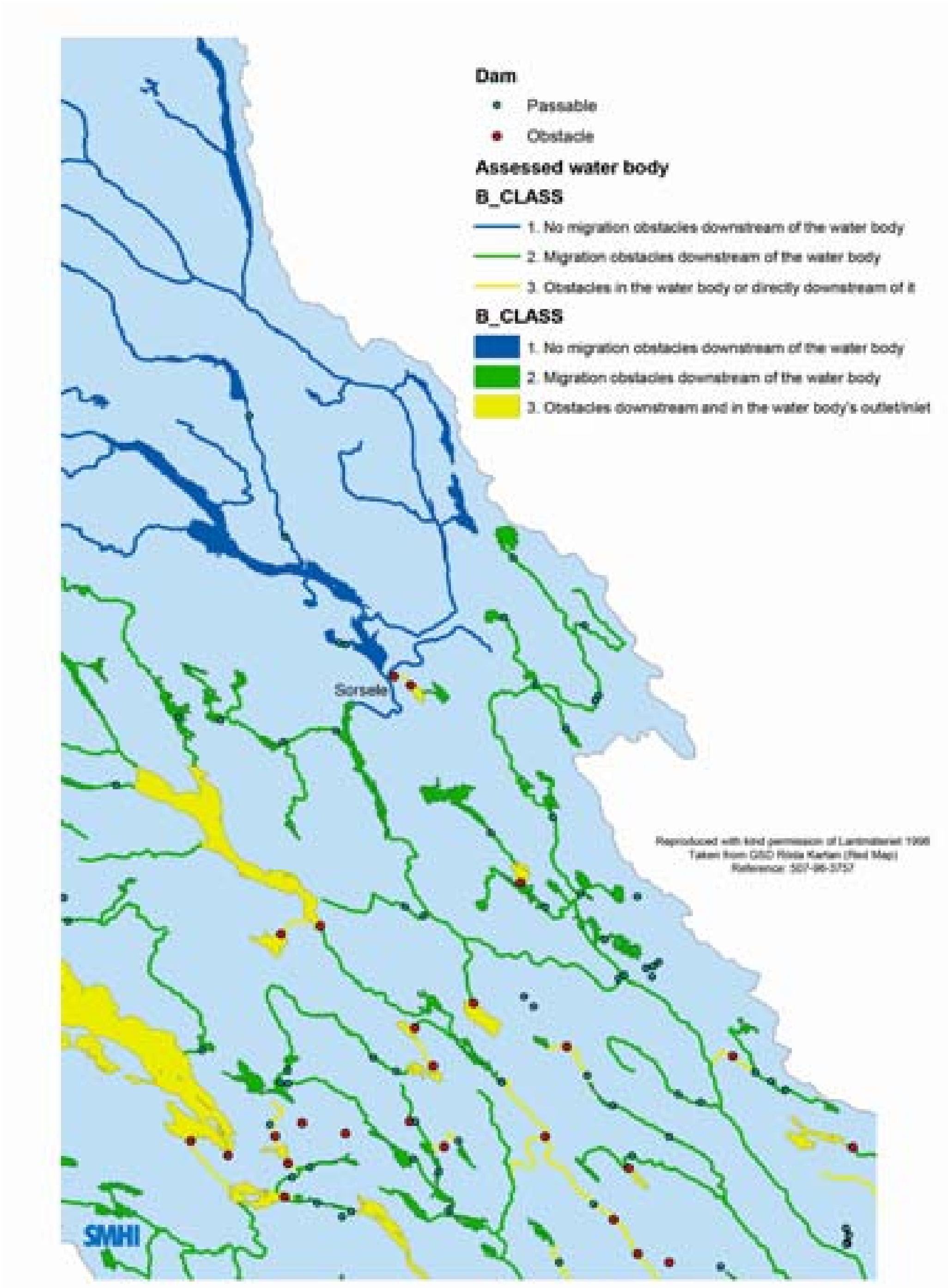

Figure 2.1. Examples of status classification of watercourses and lakes in the central reaches of the River Umeälven's basin, based on the presence of migration barriers. The classification was carried out on those watercourses that were reported to the EU in March 2005. Only these water bodies are shown on the map. Passable dams do not affect the classification of a water body. In this test, it has been assumed that the natural migration routes for the anadromous salmon in Vindelälven comprise all water bodies downstream of Sorsele. It has therefore been assessed that the migration barrier in Stornorrfors has no impact on Vindelälven at Sorsele or upstream of Sorsele. Source: SMHI.

Figure 2.2. Example of classification of the effect on watercourses and lakes in the lower reaches of the River Umeälven's basin, based on the presence of migration barriers. The classification was carried out on those watercourses that were reported to the EU in March 2005. Only these water bodies are shown on the map. Passable dams do not affect the classification of a water body. If a man-made barrier is in the middle of a defined water body, the whole water body is assigned Class 3. That applies for example to the water body where Stornorrfors lies, which should be divided so that the stretch of water downstream of Stornorrfors is assigned Class 1 and the stretch upstream is assigned Class 3. Source: SMHI.

Water bodies downstream of man-made migration barriers where there is also active water regulation must also be classified on the basis of assessment criteria for the impact of flow regulation on the hydrological regime (see Chapter 3).

Defined migration barriers should constitute boundaries between water bodies if the classification system is to function efficiently. This system can even be used if a migration barrier lies in the middle of a defined water body. There is then the same impact, Class 3, in the whole of the water body even if the fish can migrate all the way up to the migration barrier in the water body. Normally, the majority of migration barriers and dams should constitute boundaries between water bodies. Most permanent migration barriers usually function in such a way that they dam the water flow, which means that the water upstream or downstream of the dam differs in character because of the physical boundary. Conversely, if a migration barrier consists of a grille in the watercourse, it does not constitute the same clear physical boundary as compared with a dam construction. Shorter #running lengths with the majority of dams along the stretch do not need to be divided up into several watercourses if the fragments of watercourse between the dams are of minor importance.

What are to be regarded as less significant units depends on how the water bodies will be delimited in practice. In Water Quality Management Ordinance (2004:660), a surface water body is defined as “a delimited and significant body of surface water...” A number of migration barriers that lie close to one another can in this classification system be regarded as a single migration barrier and the whole stretch with a number of adjacent migration barriers can be considered as one water body. Such a water body might, for example, be assigned bad status after classification of the morphological quality elements, whereas it would have been assigned only moderate status after classification of continuity.

The parameter Presence of man-made migration barriers is given as Class 1 and 2 for the number of migration barriers downstream of the water body and as Class 3 for the number of migration barriers downstream and in, or adjoining, the water body.

See REG Annex 3, Section 1.1.2

2.3.2.1 IDENTIFICATION OF ECOLOGICALLY RELEVANT MIGRATION BARRIERS

An interruption in the river's continuity or an ecologically relevant migration barrier can, for the parameter Man-made migration barriers, consist of different types of dams and migration barriers for fish. Migration barriers can be divided into two different classes, depending on how they function and depending on whether they are man-made or natural:

1. Impassable migration barriers, which no fish, or only an insignificant number of fish, can actively pass.
2. Partial migration barriers, which at least some individuals of some fish species can actively pass, at least in some hydrological conditions.

These two types of migration barriers can be either man-made or natural and an effort should be made to identify them when making an inventory of migration barriers in a water system. In assessing the barrier's ecological relevance, it is of greatest importance to quantify how large a proportion of the migrating individuals of a fish species cannot pass in relation to the number that would have migrated if the barrier had not been constructed. However, extensive investigations may be needed to obtain this information. There may, for example, be salmon ladders which make a migration barrier only partial, but where the actual migration is only a fraction of what would have been the case in a natural state. It is then a partial migration barrier that is assessed to have significant ecological effect, for example, Stornorrfors in the Umeälven river.

It can be provisionally assumed that the dams that have not been recorded as having been demolished are ecologically relevant migration barriers. Dams that have salmon ladders, or where another migration route passes them, are assumed not to be ecologically relevant migration barriers unless there is information that the alternative migration route functions badly for some ecologically important species. In inventories, it is often stated that a barrier is partial or definitive for different fish species. The barriers that have been recorded as partial or impassable for mature salmon trout are assessed as ecologically relevant.

2.3.3 Class boundaries

The boundaries for the classification of the presence of man-made migration barriers are shown in Table 2.2.

See REG Annex 3, Section 1.1.3

Table 2.2. Classification based on the presence of man-made barriers between the water body and where it runs out into the sea, in major lakes or watercourses with fish that are naturally migratory if the barriers to migration are removed. The parameter does not include classes for poor and bad status.

Status	Class	Impact	Presence of man-made migration barriers
High	1	No impact	No migration barriers in or downstream of the water body
Good	2	Minor impact	Migration barrier downstream but not in or adjoining the water body
Moderate	3	Moderate impact	Migration barrier downstream but not in, or adjoining, the water body

2.4 Degree of fragmentation in the watercourse

The degree of fragmentation describes the extent to which migration is limited by man-made impassable migration barriers in a watercourse.

2.4.1 Requirements for supporting data

Classification of the degree of fragmentation must be based on survey material prepared in accordance with the manual on 'biotope-mapping - watercourses' or another method that gives equivalent results.

See REG Annex 3, Section 2.2.1

2.4.2 Classification of status

The degree of fragmentation is calculated according to the following the formula:

See REG Annex 3, Section 2.2.2

Degree of fragmentation = (1- (the longest stretch without impassable man-made migration barriers (km)/total length of the watercourse (km))) * 100.

When calculating the degree of fragmentation, no account is taken of natural barriers since these are not a measure of impact.

2.4.3 Class boundaries

The boundaries for the classification of the degree of fragmentation are to some extent taken from the boundaries stated in System Aqua.

See REG Annex 3, Section 2.2.3

When classifying the degree of fragmentation for a water body, a tributary is a watercourse constituting a water body that runs into, or is in direct upstream connection to, the water body that is to be classified. If the tributaries have not been mapped as regards migration barriers, a water body can at most be assigned good status. This presupposes that there are no migration barriers in the water body.

The boundaries for the classification of the degree of fragmentation are shown in Table 2.3.

Table 2.3. Classification of degree of fragmentation.

Status	Class	Impact	Degree of fragmentation
High	1	No impact	No migration barriers in the main channel or tributaries
Good	2	Minor impact	Presence of migration barriers in tributaries
Moderate	3	Moderate impact	Degree of fragmentation ≤25 %
Poor	4	Significant impact	Degree of fragmentation >25-50 %
Bad	5	Heavy impact	Degree of fragmentation >50 %

2.5 Barrier effect in watercourses

Barrier effect describes the distance to the nearest upstream or downstream man-made impassable migration barrier for a stretch of watercourse and is thus a measure of how large a part of the water body is closed off from migration because of a man-made migration barrier.

2.5.1 Requirements for supporting data

Classification of the degree of fragmentation must be based on survey material prepared in accordance with the manual on 'biotope-mapping - watercourses' or another method that gives equivalent results.

See REG Annex 3, Section 2.3.1

2.5.2 Classification of status

The degree of fragmentation is calculated according to the following the formula:

See REG Annex 3, Section 2.3.2

Barrier effect = (1- (Stretch to the first migration barrier/total length of the water-course))*100

In calculating the degree of fragmentation no account is thus taken of natural barriers since these are not a measure of impact.

2.5.3 Class boundaries

The class boundaries for the classification of the barrier effect are the same as those given in System Aqua 2004 and are shown in Table 2.4.

See REG Annex 3, Section 2.3.3

Table 2.4. Classification of barrier effect.

Status	Class	Impact	Barrier effect
High	1	No impact	No migration barriers
Good	2	Minor impact	Barrier effect ≤ 25 %
Moderate	3	Moderate impact	Barrier effect >25-50 %
	4	Significant impact	Barrier effect >50-75 %
Bad	5	Heavy impact	Barrier effect > 75 %

Background reports for section on continuity:

SMHI, 2007. Förslag till bedömningsgrunder för kontinuitet och hydrologisk regim, version oktober 2007 [Proposal for assessment criteria for continuity and hydrological regime, October 2007.

Jönköping County Administrative Board 2006. Bedömningsgrunder for hydromorfologi [Assessment criteria for hydromorphology]. Communication 2006:20.

3 Hydrological regime

3.1 Introduction

The classification of hydrological regime covers the changes a watercourse exhibits due to impact on quantity and dynamics in water flows.

It is mostly the watercourse downstream of the regulation site that is classified along with the lake or reservoir upstream of the dam construction that is affected by the regulation. To obtain a better basis for classifications in the entire river basin, flow statistics and assessments of the regulation impact from SMHI can be used. In 2007, SMHI completed and made available a summary of flow statistics for medium-sized and large watercourses in Sweden. The summary also included an assessment of the impact of regulation on the water-flow.

3.2 Input parameters

As regards hydrological regime, two classification levels are described in this handbook - a basic level and an in-depth detailed level. The basic level includes the parameters Prescribed regulation amplitude and Impact of flow regulation on the watercourse. The in-depth level includes Impact on water level changes, Number of flow peaks per year and Variation coefficient for 24-hour flow.

Regarding the quality element Hydrological regime in lakes, only the parameter Prescribed regulation amplitude is regulated in NFS 2008:1.

For the quality element Hydrological regime in watercourses, the parameter Impact of flow regulation on the watercourse, consisting of the sub-parameters Degree of regulation, Modified MHQ and Reduced MLQ, is regulated in NFS 2008:1, of which the sub-parameter Reduced MLQ is regulated in the form of a general guideline (GG) to NFS 2008:1.

Table 3.1. Parameters for classifying hydrological regime.

Assessment criterion (parameter)	Supporting data:	Watercourses	Lakes
Prescribed regulation amplitude	The maximum permitted regulation amplitude from water court judgements or other decisions on level regulation permits.		X
Impact on water level changes	24-hour series with water level, W, for regulated and unregulated state.		X
Impact of flow regulation on the watercourse: - degree of regulation - modified mean high water, MHQ - reduced mean low water, MLQ	Data on degree of regulation can be obtained from the regulation companies and in working material compiled by SMHI on behalf of the Swedish EPA[1]. Data on MLQ (mean low water) and MHQ (mean high water) was entered into the SVAR database for regulated and unregulated conditions concerning a few sites in large and medium-sized watercourses, where SMHI has deemed the calculation of data appropriate.	X	
Number of flow peaks per year	24-hour series with water level for regulated and unregulated state.	X	
Variation coefficient for 24-hour flows	24-hour series with water flow for regulated and unregulated state.	X	

The parameters used to classify Impact on the hydrological regime (Table 3.1) give a classification result for the site where the water flow is measured or estimated and primarily for the water body that lies downstream of the site. In order to able to rationally convert the results to the water bodies in a river basin, each site with classification results must be plotted with geographical coordinates. The site should also be linked to the #line object in the watercourse's hydrographical network[2] to which the classification most directly applies The result can then be rationally converted from the site with the water flow data or water flow statistics to the flow lines and then to the water body for the lake or watercourse in which the flow line with the impact indication lies.

[1] Olsson, H. 2005. Analyser av flödesserier och regleringsamplitud för utformning av bedömningsgrunder för hydromorfologiska kvalitetsfaktorer [Analyses of flow series and regulation amplitude to develop assessment criteria for hydromorphological quality elements]. Redovisning av ett uppdrag från Naturvårdsverket [Report of an assignment from the Swedish EPA]. SMHI ref no. 2004/1036/1933.

[2] SMHI, 2006. Svenskt Vattendragsregister [Swedish Watercourse Register]. SMHI Hydrologi nr 102.

3.3 Prescribed regulation amplitude

The presence of a regulation or waterworks dam in a lake's outlet means that the lake is disturbed by the regulation. How much the lake is disturbed by the regulation is primarily determined using data on prescribed or registered maximum regulation amplitude. Prescribed regulation amplitude has been chosen as an assessment criterion because it is the easiest parameter to obtain data on. Section 3.3.4 describes the methodology and classes to be applied should an in-depth classification of the impact be necessary. In-depth classifications require more input data and more work needs to be done to create a basis for the classification. An in-depth classification provides information about how the regulation is actually done whilst Prescribed regulation amplitude only gives details about the maximum regulation amplitude that is permitted in accordance with water judgements or other regulation permit decisions.

3.3.1 Requirements for supporting data

The dam register contains data on regulation amplitude. SMHI has tabulated preliminary data on prescribed regulation amplitude for 563 lakes, of which 554 are linked to a working version of the map layer for lakes in the SVAR database (Swedish Water Archive at SMHI). Figure 3.1 shows the classification results for these lakes in accordance with Table 3.2.

The prescribed regulation amplitude stipulated in a water judgement or a permit for water undertakings shall be used as a minimum when classifying the parameter Prescribed regulation amplitude.

See REG Annex 3, Section 3.1.1

3.3.2 Classification of status

Class A1 is allocated to lakes whose water level is not actively regulated. For example, lakes that have no regulation or waterworks dam at a site that constitutes an outlet from the lake are deemed not directly regulated and are assigned Class A1. The presence of regulation or waterworks dams indicates that some type of active water level regulation is taking place. The SMHI dam register contains a column called NV class. If the NV class is 1 or 2, the dam has been classed as a regulation or waterworks dam. In this case, water level variations in the regulated lake are probably not natural and the lake can therefore not be assigned Class A1.

There may be lakes with a prescribed regulation amplitude in Class A2 and perhaps even in Class A3, which, in an unregulated state, have a higher amplitude than that which has been prescribed. A lake which, because of regulation, has a more even level than its natural level can therefore, using these simple criteria, be assessed as slightly disturbed by regulation and possibly having the right conditions for good ecological status. Here, the in-depth classification in Section 3.3.4 can be applied.

In heavily regulated watercourses, the amplitude in lakes that have no dams, but that lie immediately downstream of regulation reservoirs, may be affected by the flow regulation. Even this type of lake should be assessed using the in-depth classification in Section 3.3.4.

The effect of permanent damming, where the flow is not regulated, is assessed with regard to continuity and morphological impact respectively. A permanent dam has primarily a permanent effect on the water level and this type of change is dealt with under the quality element Morphology in lakes, see Section 4.6.

See REG Annex 3, Section 3.1.2

Prescribed regulation amplitude for lakes is classified using the information given in water judgements or permits for water undertakings.

3.3.3 Class boundaries

See REG Annex 3, Section 3.1.3

The class boundaries for Prescribed regulation amplitude are given in Table 3.2. The class boundaries have not been tested against ecological effects in different types of lakes but have been proposed with the help of results from a few studied lakes.[3].

Table 3.2. The assessment criteria for the impact of regulation amplitude on water level variation in lakes.

Prescribed regulation amplitude			
Status	**A-Class**	**Impact on water level**	**Maximum permitted regulation amplitude**
High	A 1	No regulation or water-works dam	No active regulation taking place
Good	A 2	Slight	< 1 metre
Moderate	A 3	Moderate	1 – 2.99 metre
Poor status	A 4	Significant	3 – 9.99 metre
Bad status	A 5	Heavy	≥ 10

3.3.4 Impact on water level changes: In-depth Classification based on regulation amplitude

The maximum regulation amplitude prescribed for a regulated lake does not indicate how the level regulation has actually been performed. A better assessment of the impact of regulation of the water level in a lake can be made if water level series for regulated and unregulated conditions are compared to each other. An example of this type of background data is given in Figure 3.2, as the mean annual variation for the water level in Lake Siljan before and after regulation. If this is not available, natural water level series can be reconstructed using data on catchment and discharge.

[3] Marttunen, M., Hellsten, S., Glover, B., Tarvainen, A., Klintwall, L., Olsson, H. & Pedersen, T.S., 2006. Heavily regulated lakes and the European Water Framework Directive – Comparisons from Finland, Norway, Sweden, Scotland and Austria. E-Water. Official Publication of the European Water Association (EWA), 2006/5.

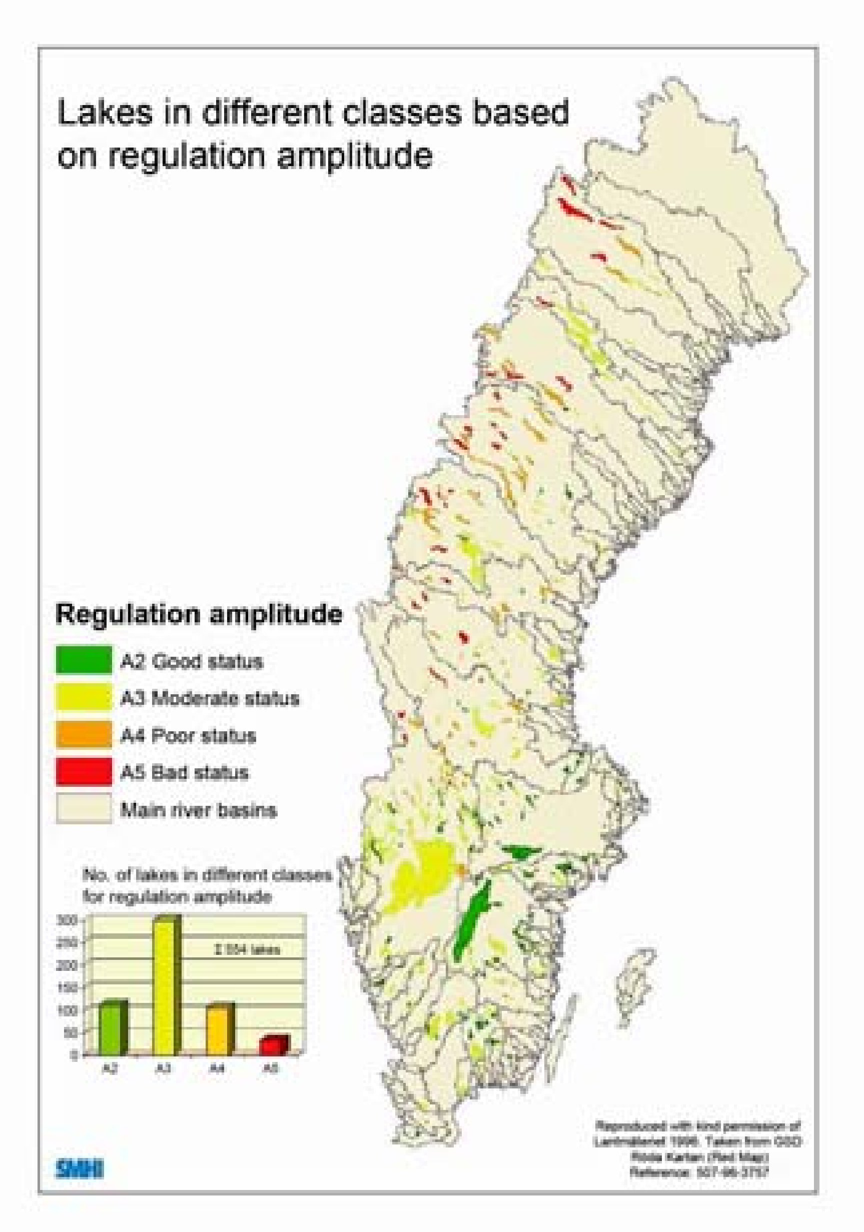

Figure 3.1. Classification of lakes by regulation amplitude in accordance with assessment criteria in Table 3.2. SMHI has a data table with data from 554 lakes on prescribed regulation amplitude that could be linked to the surface layer for lakes in the SVAR database[4]. Source: SMHI.

[4] Olsson, H. & Lundholm, K., 2007. Förslag till bedömningsgrunder för kontinuitet och hydrologisk regim, version oktober 2007 [Proposal for assessment criteria for continuity and hydrological regime, October 2007 version].

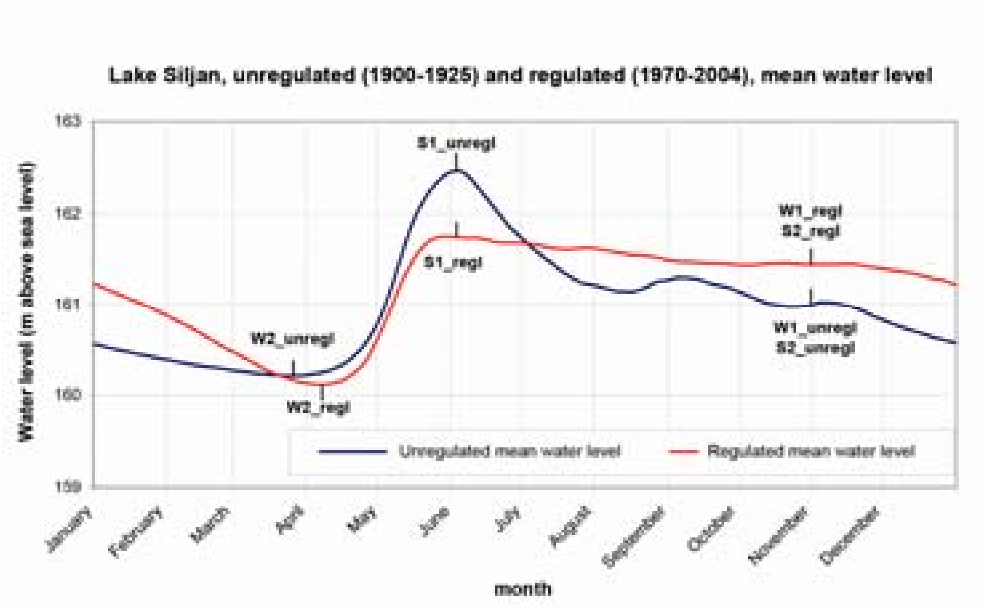

Figure 3.2. Mean water level in Lake Siljan before regulation, blue line, and after regulation, red line. S1 and S2 indicate the period for calculating modified water level during the summer and autumn. W1 and W2 indicate the period for calculating modified water level during the winter. Source: SMHI.

A model called REGCEL has been developed in Finland.[5] and can be used to analyse the impact on water levels in lakes. Using the Finnish model, over 50 parameters can be calculated, all of which are relevant for the ecological effects of water level regulation. More data in addition to time series is needed to calculate many of these parameters. Finnish researchers are in the process of developing software for the application of REGCEL. An Excel application can be made available via contacts at http://toolbox.watersketch.net. This assessment criterion has so far utilised two parameters in REGCEL and modified them slightly. Classifications are done using the following two parameters, which are relatively easy to calculate.

1. Change in water level during the winter: The difference between the lake's mean water level on 1 November and the lowest mean water level reached during ice cover is calculated for regulated conditions (W1_regl - W2_regl in Figure 3.2) and for unregulated conditions (W1_unregl - W2_unregl in Figure 3.2).

2. Change in water level during the summer and autumn: The difference between the mean water level at the time of the highest natural water level after the spring thaw and the mean water level on 1 November is calculated for regulated conditions (S1_regl - S2_regl in Figure 3.2) and for unregulated conditions (S1_unregl - S2_unregl in Figure 3.2).

[5] Hellsten, S., Marttunen, M., Visuri, M., Keto, A., Partanen, S. & Järvinen, E.A., 2002. Indicators of sustainable water level regulation in Northern River Basins: a case study from the River Paatsajoki water system in Northern Lapland. arch. Hydrobiol. Suppl. 141/3-4: 353-370.

Please note that for the regulated lake, the change in water level is calculated during the summer from the time for the highest mean water level in unregulated conditions.

To classify the impact, the deviation between natural drops in water level and regulated drops in water level is calculated for the two parameters Winter change, and Summer/Autumn change:

- Difference (Δ) between natural and regulated change in mean water level during the winter in accordance with Point 1 above.
- Difference (Δ) between natural and regulated change in mean water level during the summer and autumn in accordance with Point 2 above.

Using this calculation method, the value for classification of impact due to the change in the mean water level during the winter will be negative for reservoirs where the water is retained in the summer for abstraction during the winter. In the Lake Siljan example, the difference between the natural and regulated drop in the mean water level will be -0.5 in the winter, Table 3.3.

Regulation that causes the mean water level to increase or retain a higher than natural level during the summer gives a positive figure as a basis for classifying the impact of regulation on the water level change during the summer and autumn. In the Lake Siljan example, the difference between the natural and regulated drop in mean water level during the summer will be 1.2 metres, Table 3.3.

The changes in mean water level during the winter and summer/autumn respectively have been calculated for Lake Vänern, Lake Siljan, Lake Akkajaure, Lake Oulujärvi and Lake Kemijärvi[6]. Data for the Finnish lakes Oulujärvi and Kemijärvi has been calculated from graphs in Marttunen et al (2006)[7]. There are only a few easily accessible time series with water levels for regulated and unregulated conditions.

The computed results (See Table 3.3) show that the mean water level in Akkajaure dropped 11 metres more when regulated than in its natural state. The mean water level was higher in the regulation reservoir than in the natural lake. The rise in mean water level is a pre-condition for being able to store large volumes of water for abstraction during the winter. Lake Siljan and Lake Vänern are however not typical regulation reservoirs. They are regulated to benefit shipping and other operations. The drawdown during the winter in these lakes is relatively small and during the summer and autumn, the level is kept slightly higher than the natura

[6] Olsson, H. & Lundholm, K., 2007. Förslag till bedömningsgrunder för kontinuitet och hydrologisk regim, version oktober 2007 [Proposal for assessment criteria for continuity and hydrological regime, October 2007 version].

[7] Marttunen, M., Hellsten, S., Glover, B., Tarvainen, A., Klintwall, L., Olsson, H. & Pedersen, T.S., 2006. Heavily regulated lakes and the European Water Framework Directive – Comparisons from Finland, Norway, Sweden, Scotland and Austria. E-Water. Official Publication of the European Water Association (EWA), 2006/5.

level. The natural drawdown in the summer is counteracted in lakes that work as regulation reservoirs.

Table 3.3. Classification values for differences between regulated and natural changes in mean water levels Δ during winter = natural change in mean water level during the winter - change during the winter in a regulated lake. Δ summer and autumn = natural change in mean water level during summer and autumn - change in mean water level in a regulated lake during summer and autumn.

Difference between regulated and natural mean water level (metres)		
Lake	**Δ during winter**	**Δ summer and autumn**
Akkajaure	-11	11.4
Oulujärvi	-1.1	1.1
Kemijärvi	-5.9	2.3
Siljan	-0.5	1.2
Vänern	0	0.2

Table 3.4 gives the assessment criterion for the parameter Water level change. The assessment criterion is based on the information in Table 3.3, which is not enough data to make a classification using statistical distribution. The distribution is therefore an expert judgement within the framework given by the values for the five lakes. The distribution should therefore be seen as a preliminary proposal for testing other objects. In Finland, REGCEL has been further developed during the spring of 2007, but only a Finnish version of the Excel application is currently available. The Finnish proposals for assessment criteria assign class five to changes of more than three metres during the winter. Regulation amplitude has a connection between topography and the reservoir's hypsographs. As a result, Finnish regulation reservoirs generally have less regulation amplitude than Swedish ones. It is also likely therefore that a greater proportion of shorelines dry out if the water level drops 3 metres in a Finnish reservoir compared to a Swedish reservoir.

Table 3.4. Assessment criterion for change in mean water level in lakes based on information about regulation in five lakes. Impact classification based on differences in metres between natural and regulated changes in water level during the winter and summer/autumn respectively in accordance with Table 3.3.

Status	**Class**	**Impact winter**	**Impact summer/autumn**	**Impact on the water level change in lakes**
High status	N 1	No active regulation taking place	No active regulation taking place	No regulation or waterworks dam
Good status	N 2	Δ 0 – -1 m	Δ 0 – 0.5 m	Minor change
Moderate status	N 3	Δ -1 – -3 m	Δ 0.5 – 2 m	Moderate change
Poor status	N 4	Δ -3 – -6 m	Δ 2 – 5 m	Significant change
Bad status	N 5	Δ≤ -6 m	Δ 5 – 0.5 m	Heavy change

Even if the assessment scale has not been tested against biological effects, the mean levels developed in accordance with Figure 3.2 represent a good basis for an expert judgement.

3.4 Impact of flow regulation on watercourses

The classification of the impact of flow regulation is divided into the three sub-parameters:

- degree of regulation
- modified mean high water (MHQ)
- reduced mean low water (MLQ)

The classification can be supplemented by an in-depth classification according to the proposals given in Section 3.4.7. The in-depth classification provides a better classification basis and should primarily be made for water bodies that have been assigned different classifications using the sub-parameters in Table 3.5. Application of DHRAM (Dundee Hydrological Regime Assessment Method) should also be considered an in-depth classification.

3.4.1 Requirements for supporting data

Information about the presence of active flow regulation, degree of regulation and flow statistics are used to classify the impact of flow regulation.

Information about what requirements there are for supporting data for each sub-parameter respectively can be found in Sections 3.4.4, 3.4.5 and 3.4.6.

3.4.2 Classification of status

It is appropriate to use Degree of regulation to make a general classification of the impact on the natural flow regime. It can be applied to assess the impact of impoundments on the flow regime of all sizes within the river basin.

The degree of regulation shows how much water can be stored or impounded upstream of a site in a watercourse in relation to annual flow volumes at the site. The degree of regulation does not show how the flow regulation has been implemented.

The indicators Modified MHQ and Reduced MLQ can give further indication of the impact on the flow, e.g. short-time regulation, which is not seen in the degree of regulation. If the degree of regulation indicates high status and the change in MHQ or a reduced MLQ indicates moderate status, the most serious impact indication, i.e. moderate status, is valid.

Detailed information about how each sub-parameter shall be classified is given in Sections 3.4.4, 3.4.5, and 3.4.6.

3.4.2.1 PRESENCE OF ACTIVE REGULATION

Class F1 in Table 3.5, no flow regulation, occurs in river basins where there are no regulation dams. These river basins are identified using information about the presence of dams and their purpose. There is probably no active flow regulation in watercourses that have no regulation or waterworks dams.

The assessments of regulation impact that can be found in water flow statistics compiled by SMHI can also be used to identify regulated and unregulated watercourses respectively.

3.4.3 Class boundaries

The class boundaries for Degree of regulation have been determined relatively subjectively. In previous projects developing supporting data for the preliminary classification of heavily modified water bodies, a 20% degree of regulation has been applied, but there is no biological background data that supports this choice. It should instead be considered as an expert proposal that should be tested. In the working material to which SMHI has had access, a few different alternatives for class boundaries for Degree of regulation and Modified MHQ have been compared. The proposed class boundaries give the same classification results at most of the sites where both parameters were available.

Regarding Reduced MLQ, the scaling is based on data from about 40 sites, approximately 10 of which have a 100% reduction in MLQ. The reduction in MLQ can also be negative depending on the regulation strategy. This type of impact is intended to be captured by the Modified MHQ parameter.

The three sub-parameters are classified using the class boundaries given in Table 3.5. Explanations of the names in Table 3.5 and descriptions of statistics and methodology are given in Sections 3.4.4, 3.4.5 and 3.4.6.

See REG Annex 3, Section 4.1.3

Table 3.5. Class boundaries for Degree of regulation, Modified mean high water (MHQ) and Reduced mean low water (MLQ). The change in MHQ can either be positive or negative depending on the regulation strategy. The percentage classification scale has the same numerical values on the negative and the positive side.

IMPACT OF FLOW REGULATION ON WATERCOURSES					
Status	Class	Impacts	Degree of regulation	Change in MHQ (%)	Reduced MLQ (%)
High status	F 1	No regulation impact	0	0	0
Good status	F 2	Minor regulation impact	>0 – 9.99	-4.99 – +4.99	>0 – 9.99
Moderate status	F 3	Moderate regulation impact	10 – 19.99	-5 – - 9.99 +5 – +9.99	10 – 29.99
Poor status	F 4	Significant regulation impact	20 – 49.99	-10 – -49.99 +10 – +49.99	30 – 79.99
Bad status	F 5	Heavy regulation impact	≥50	≤ -50 ≥ +50	80 – 100

3.4.4 The sub-parameter Degree of regulation

3.4.4.1 REQUIREMENTS FOR SUPPORTING DATA

Supporting data and results already calculated by SMHI or water regulation companies should be used for classification. Alternatively, modelled or calculated mean water values from other representative flow series can be used. At least 10-year-long flow series with 24-hour observations and calculating reservoir volumes shall be used as a basis for classifying the degree of regulation.

See GG to Annex 3 Section 4.1.1

See REG Annex 3, Section 4.1.1

3.4.4.2 CLASSIFICATION OF STATUS

The degree of regulation at a certain site in a watercourse system is the relationship between the total stored volume upstream of the site and the annual mean water flow for the site. The degree of regulation can be used as an early indicator of how significant the impact of regulation is at that particular site along the watercourse.

There are differences in the natural flow regime between northern and southern Sweden which can justify different classification scales for the degree of regulation. For example, a higher degree of regulation can be acceptable for a specific impact in southern Sweden compared to northern Sweden. Uncertainties as to the ecological effects of a certain degree of regulation mean that the development of a common classification scale for the whole country, as given in Table 3.5, is still considered to be justified.

The degree of regulation is calculated using the formula:
RG = 100 * STORE/ QV

See REG Annex 3, Section 4.1.2

where RG is the degree of regulation in %
STORE is the sum of all stored volumes (m^3) upstream, and
QV is the annual flow volume (m^3).

Data on degree of regulation and stored volumes can be obtained from the water regulation companies, power companies and SMHI, but the information has not been compiled in a national database. Even though it might be difficult to derive data on stored volume to calculate the degree of regulation, estimates of the stored volume can be made by multiplying the surface area of the reservoir by the regulation amplitude. This leads to a certain overestimation of the stored volume, however.

The annual flow volumes can be calculated using the flow statistics compiled by SMHI. Computed degrees of regulation will be entered into the SMHI database of water flow statistics for large and medium-sized watercourses in Sweden.

3.4.5 The sub-parameter Modified mean high water (MHQ)

See GG to Annex 3 Section 4.1.1

See REG Annex 3, Section 4.1.1

3.4.5.1 REQUIREMENTS FOR SUPPORTING DATA

Supporting data and existing calculation results from SMHI should be used to classify mean high water (MHQ). Modelled or calculated series from other representative flow series can also be used as an alternative. At least 10-year-long time series with 24-hour observations shall be used as a basis for the classification of mean high water.

3.4.5.2 CLASSIFICATION OF STATUS

MHQ is the mean value of each year's highest 24-hour water level over a number of consecutive years. MHQ is affected both in watercourses downstream of the regulation reservoir and downstream of sites that are regulated in some other way. MHQ is included in the national set of flow statistics compiled by SMHI (see Section 3.1). The proposed classification boundaries in Table 3.5 are based on the division shown in Figure 3.3 and on a comparison with the degree of regulation for sites where data on the degree of regulation was also available. The classification is therefore based primarily on impact indicators. The ecological effects of the proposed impact classes have not been tested.

The sub-parameter used to classify the impact on MHQ is the percentage deviation from natural MHQ:

See REG Annex 3, Section 4.1.2

Modified MHQ (5) = 100 * ((MHQN - MHQR)/MHQN)

MHQN is the mean high water (m3/s) in unregulated conditions and
MHQR is the mean high water (m3/s) in regulated conditions.

A flow regulation often leads to a reduction in MHQ but there are examples of regulation strategies that lead to higher than natural MHQ. These examples are most palpable in southern Sweden downstream of large lakes. Both types of impact have been incorporated into the same sub-parameter according to Table 3.5. The percentage class boundaries for reduced MHQN are positive figures and the percentage class boundaries for increased MHQN are negative figures.

Figure 3.3 shows how many sites there are in each classification if the calculated percentage change in MHQ is 5 %. At most of the sites, the regulation has reduced MHQ by 5-40 %. The data does not consist of statistically independent observations since there are several sites in the same watercourse where the same regulation can affect more than one site in a similar way. In Sweden, there are many more watercourses where regulation has changed MHQ by 0 - 10 %.

There are 18 sites where MHQ has changed by more than (- 30 %), i.e. where the regulation has increased MHQ by more than 30 % and these are all in southern Sweden. The sites are downstream of the lakes Mälaren, Hjälmaren, Vänern, Vättern and Bolmen.

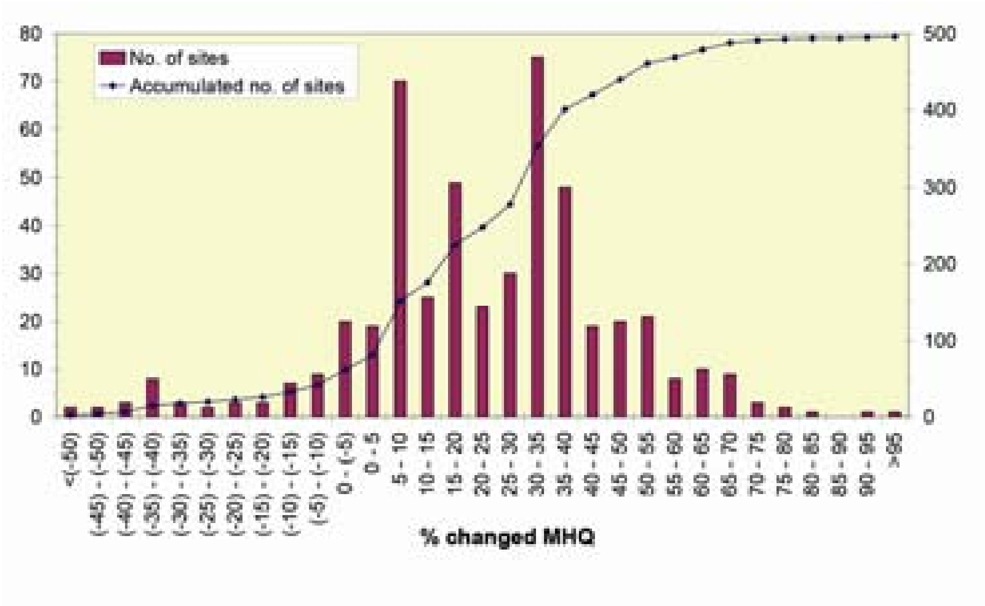

Figure 3.3. Number of sites in the classification if the calculated percentage deviation between natural MHQ and regulated MHQ is 5 %. The data covers 496 sites that are affected by regulation to a greater or lesser extent. There are many more regulated watercourses in Sweden where the change in MHQ is 0 -10 % in addition to those represented in the background data. Source: SMHI

Change in MHQ due to regulation of a watercourse has been a criterion for characterising different types of water flow series in a compilation of water flow statistics completed at SMHI in 2007. The compilation covers a large number of sites in large and medium-sized watercourses in Sweden. The classification is based on analyses of changed MHQ according to the same principles outlined above. It is only a three-point scale but the classification is still useful. Assessments have been produced, even for watercourses where the prerequisites for compiling statistics or flow data are poor, for both regulated and unregulated conditions. Since there are assessments for many sites in Sweden, the results can be used to interpolate the degree of impact at several different places along the watercourse.

In order to prevent natural flow variations from affecting the results, it is recommended that time series of at least 20 years be used if this is possible. The use of shorter time series than 10 years can have a major effect on the reliability of the results.

3.4.6 The sub-parameter Reduced mean low water (MLQ)

3.4.6.1 REQUIREMENTS FOR SUPPORTING DATA

Data from measurement series should be used as far as possible in order to calculate mean low water (MLQ). Modelled flow series can be used as an alternative. Time series of at least 10 years with 24-hour observations should be used as supporting data for the classification.

See GG to Annex 3 Section 4.1

Downstream of a dam, a regulation may lead to a reduction or total drain-off of the water flow. Less permanent reductions in the flow should be included when classifying the impact of the flow regime using the parameter Number of flow peaks per year. The impact during shorter time periods may give impact readings that are not indicated by the parameters proposed above.

Hydraulic analysis and morphological input data are actually required to assess the effect in ecosystems as a result of heavily reduced flows. In other words, it is difficult to define a quantitative and simple method for assessing the effect of reduced flow. One parameter that may work on large watercourses in proposed in Table 3.5. This proposal is based on how much the low water has been reduced. The impact status of reduced water flow can only be assessed on watercourses that don't dry out naturally.

MLQ is the mean value of each year's lowest 24-hour water flow over a period of consecutive years.

See GG to Annex 3 Section 4.1

Changed MLQ (5) = 100 * ((MLQN-MLQR) / MLQN)

MLQN is the mean low water (m3/s) in unregulated conditions and
MLQR is the mean low water (m3/s) in regulated conditions.

Data on MLQ (mean low water) is available in the SVAR database for regulated and unregulated conditions concerning a few sites in large and medium-sized watercourses, where SMHI has deemed the calculation of data appropriate. MLQ for unregulated conditions can be calculated from modelled flow series but this presupposes, among other things, that discharge curves for unregulated conditions already exist or are established. MLQ in regulated watercourses must be calculated from measured flow series or using models that have detailed information about how the watercourse is regulated.

3.4.7 In-depth classification of the impact of flow regulation on a watercourse

3.4.7.1 DHRAM

DHRAM (Dundee Hydrological Regime Assessment Method) is an assessment model that bases an impact classification on the co-weighting of the outcomes from 32 different flow indicators. Long flow series with 24-hour values for unregulated and regulated flows are needed to calculate the flow indicators. The 32 indicators are divided into five groups:

1. Mean flows for each calendar month (12 indicators).
2. Minimum and maximum flows with 5 different time durations (10 indicators).
3. Day for maximum and for minimum flow according to the Julian calendar (2 indicators).

4. Annual number and duration in number of days for high and low flows respectively (4 indicators).
5. Mean increase and mean reduction in flow and number of flow increases per year (3 indicators).

   An application for implementing DHRAM is currently being developed by Finnish researchers. This application will be made available at http://toolbox.watersketch.net/.

3.4.7.2 NUMBER OF FLOW PEAKS IN SOUTHERN SWEDEN. IN-DEPTH CLASSIFICATION BASED ON SHORT-TERM REGULATION

Further assessment of the impact of regulation should be made using an indicator for short-term regulation especially for sites classed as F2 and F3 in Table 3.5. This indicator is made up of the number of flow peaks occurring per year. A short-term regulated flow has more flow peaks than a natural flow. A similar indicator is included in DHRAM; indicator group 5.

Different classifications are proposed for this parameter to be applied in southern and northern Sweden respectively. The boundary between northern and southern Sweden when applying this parameter goes close to Limes Norrlandicus or the border than separates northern and southern Sweden as regards the ecoregions developed in order to classify lakes and watercourses into different types. It is uncertain as to which scale is the best one to use for sites located close to this border. Both scales should be tested in doubtful cases.

The scale is different for different sizes of river basin. In the background data used to develop the proposal, there are major differences in the number of flow peaks between regulated and unregulated flows in large river basins in northern Sweden (Norrland) compared to small river basins in southern Sweden. The background data for southern Sweden was worse than the data available for northern Sweden.

For sites in large river basins, those that are > 1 000 km^2 in southern Sweden and > 2 000 km^2 in the north, it is possible to classify the regulation impact based solely on the values for the number of flow peaks per year for the regulated flow series. It makes things easier if we only need to have access to one flow series for regulated conditions and don't need to develop an unregulated flow series as a reference.

For basins that are < 1 000 km^2 in southern Sweden and < 2 000 km^2 in the north, it is however appropriate to assess how much the water regulation has changed the number of flow peaks per year compared to unregulated conditions. The proposed assessment criterion is based on the ratio of the number of peaks per year for regulated flow divided by the number of peaks per year for unregulated flow. An unregulated flow series can sometimes be found for time periods prior to the construction of regulation dams. A natural flow series can however also be reconstructed using natural discharge curves and runoff series for lakes. It can also be calculated using existing or new HBV models where regulation strategies are not simulated in the model.

The assessment criterion for flow peaks is presented in Table 3.6 for southern Sweden and Table 3.7 for northern Sweden. The tables contain preliminary figures that should be tested. Should the assessment value for basins < 1 000 (km^2) and < 2 000 (km^2) be < 1, the flow has more peaks naturally than when it is regulated. This can occur in some areas where there is also a certain amount of uncertainty regarding input data to reconstructed flow series, especially for larger basins with several dams upstream. If this does occur, it is a question of contacting the relevant expert, who can examine the area in question more closely.

Table 3.6. Assessment criteria for Number of peaks per year as an indication of modified hydrological regime in southern Sweden. RB in the table = size of river basin. The parameter does not include classes for high or bad status.

Number of flow peaks per year in southern Sweden				
Status	**Class**	**Impact**	**RB <1 000 km^2, peaks regulated/unregulated**	**RB ≥1 000 km^2, peaks/year**
Good status	TS 2	Minor impact	1 – 1.05	< 27
Moderate status	TS 3	Moderate impact	1.05 – 1.25	27 – 40
Poor status	TS 4	Significant impact	≥ 1.25	≥ 40

Table 3.7. Classes for Number of peaks per year as an indication of modified hydrological regime in northern Sweden. RB in the table = size of river basin. The parameter does not include classes for high or bad status.

Number of flow peaks per year in northern Sweden						
Status	**Class**	**Impact**	**RB <1 000 km^2, peaks regulated/unregulated**	**RB 2 000-4 000 km^2, number of peaks/year**	**RB 4 000-10 000 km^2, number of peaks/year**	**RB ≥10 000 km^2, number of peaks/year**
Good status	TN 2	Minor impact	1 – 1.05	< 26	< 34	< 42
Moderate status	TN 3	Moderate impact	1.05 –1.25	26 – 42	34 – 48	42 – 54
Poor status	TN 4	Significant impact	≥ 1.25	≥ 42	≥ 48	≥ 54

If the classification of flow peaks reveals a greater impact than other parameters for the classification of the impact of flow regime, it is the greatest impact indicator that is valid for the overall classification. The number of flow peaks shall be calculated from a long series of 3-day (72-hour) mean values. When calculating the number of flow peaks, the number of turning points in the curve of 24-hour values shall be counted and then divided by two. An example of a flow curve for calculating the number of flow peaks is shown in Figure 3.4.

More information on the number of flow peaks per year as an assessment parameter can be found in two earlier investigations[8] [9].

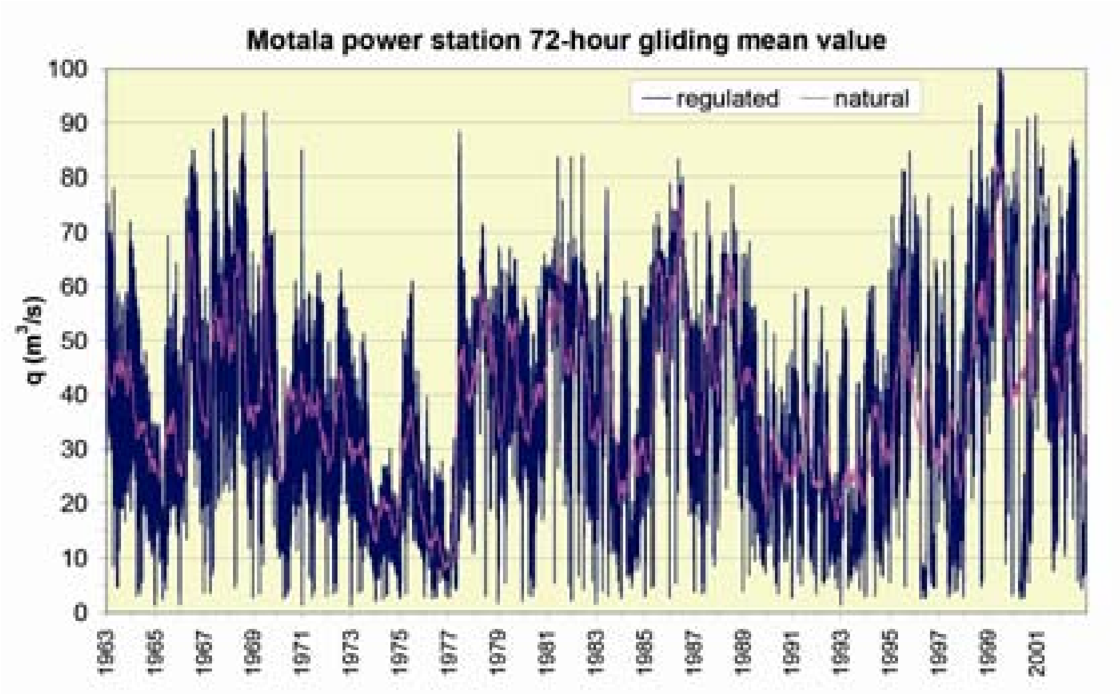

Figure 3.4. Examples from Lake Vättern's outlet of flow series to calculate the parameter Number of flow peaks per year. The blue (dark) curve is the regulated flow and the red (light) curve is the natural reconstructed flow series. The natural series has 8 peaks per year whereas the regulated series has 58 peaks per year. The river basin is 6 378 km^2. Source: SMHI.

3.4.7.2.1 Supporting data:

Supporting data: 24-hour series with water flow for regulated and unregulated state. A number of series can be found in a database at SMHI. Water flow series can be calculated using a set of models and appropriate input data, e.g. lake volumes and discharge curves for regulated and unregulated states.

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[8] Jutman, T. & Olsson, H. 2003. Förslag till analyser för att utforma bedömningsgrunder för hydromofologisk kvalitetsklassning av vattenförekomster i sjöar och vattendrag [Proposal for analyses to design assessment criteria for hydromorphological quality classes of water bodies in lakes and watercourses]. Redovisning av ett uppdrag från Naturvårdsverket [Report of an assignment from the Swedish EPA]. SMHI ref no 2002/1797/1933.

[9] Olsson, H. 2005. Analyser av flödesserier och regleringsamplitud för utformning av bedömningsgrunder för hydromorfologiska kvalitetsfaktorer [Analyses of flow series and regulation amplitude for the design of assessment criteria for hydromorphological quality elements]. Redovisning av ett uppdrag från Naturvårdsverket [Report of an assignment from the Swedish EPA]. SMHI ref no. 2004/1036/1933.

3.4.7.3 NUMBER OF FLOW PEAKS IN NORTHERN SWEDEN. IN-DEPTH CLASSIFICATION BASED ON THE REGULATION IMPACT IN NORTHERN SWEDISH WATER REGIMES

The variation coefficient calculated from 24-hour flow values can be used as a supplement to the classification of the regulation impact on watercourses with northern Swedish flow regimes. This parameter works to classify impact when the spring flood is reduced and stored. The proposed scale in Table 3.8 depends on the size of the river basin.

Variation coefficient = standard deviation/mean value. More information on the variation coefficient as an assessment parameter can be found in two earlier investigations (see footnotes 5 and 6).

Table 3.8. Classes for variation coefficients for 24-hour flows as an indicator of modified hydrological regime. RB in the table = size of river basin. The parameter does not include classes for high or bad status.

Variation coefficient for 24-hour flow						
Status	**Class**	**Impact**	**RB <2000 km^2**	**RB 2000 – 4000 km^2**	**RB 4000 - 10000 km^2**	**RB ≥10000 km^2**
Good status	V 2	Slight impact	> 1.2	> 1.0	> 0.8	> 0.7
Moderate status	V 3	Moderate impact	1.0 – 1.2	0.8 – 1.0	0.6 – 0.8	0.5 – 0.7
Poor	V 4	Significant impact	< 1.0	< 0.8	< 0.6	< 0.5

If the classification of variation coefficients for 24-hour flow indicates a greater impact than the other parameters used to classify impact on flow regimes, it is the greatest impact indicator that is valid for the overall classification.

Figure 3.5 shows an example of how the variation coefficient decreases when regulation reduces the high flow that is natural in connection with the spring thaw.

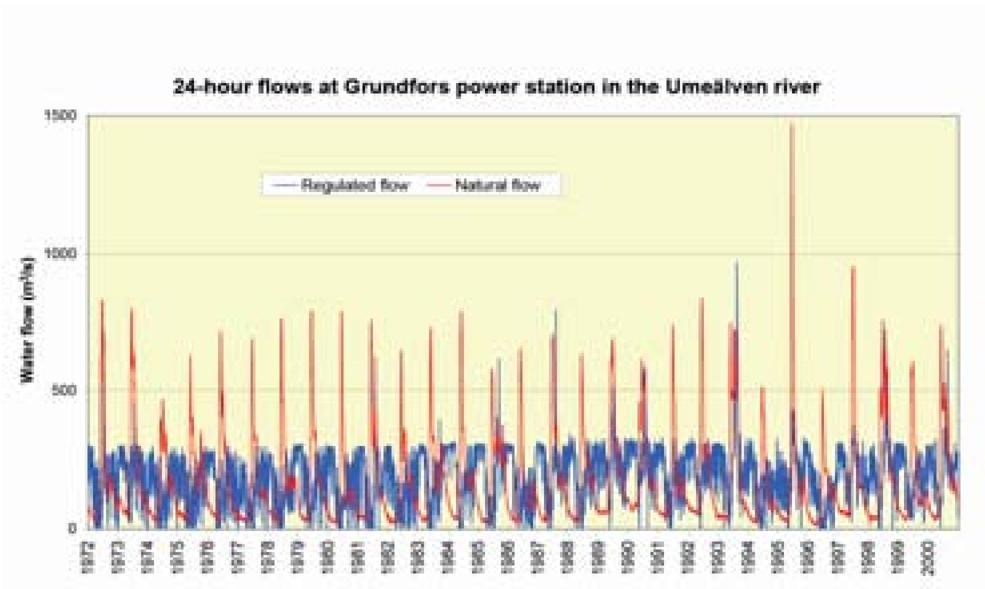

Figure 3.5. 24-hour series for regulated and unregulated flows respectively at Grundfors hydropower station in the Umeälven river, which has a river basin of 7 763 km^2. The natural unregulated series is calculated using measurement data and can be found in a database at SMHI. The variation coefficient is 0.97 for the natural series and 0.51 for the regulated series. The impact assessment is based on the variation coefficient for the regulated series and indicates significant impact, class 4, according to Table 3.8. Source: SMHI.

3.4.7.3.1 Background material:

Background material: 24-hour series with water flow for regulated and unregulated state. Water flow series can be calculated using a set of models and appropriate input data, e.g. lake volumes and discharge curves for regulated and unregulated states. In the assessment criteria, we have not specified requirements as to how the reference series of natural flows shall be produced. It is obviously not a problem if the natural time series have been measured during a period prior to regulation. In that case, there are only requirements for 24-hour values and time series length, at least 5-10 years (see next paragraph) If the natural series have however been reconstructed from measurement data or have been modelled, the quality of the information on the volume of the reservoir and knowledge about the natural discharge from the unregulated lake becomes crucial for the results.

24-hour water flow series in regulated and unregulated conditions make up the basic classification background data needed to assess the impact on the flow regime. This data is needed to calculate flow peaks, variation coefficients, indicators in DHRAM, MQ, MHQ and MLQ. The MQ, MHQ and MLQ already calculated by SMHI for large and medium-sized Swedish watercourses can be used if the regulation in these watercourses has not been modified over the last five years.

Background reports to the section on hydrological regime:

SMHI, 2007. Förslag till bedömningsgrunder för kontinuitet och hydrologisk regim, version oktober 2007 [Proposal for assessment criteria for continuity and hydrological regime, October 2007 version].

4 Morphological conditions

4.1 Introduction

The classification of morphological impact covers the changes a water body has undergone as a result of e.g. road construction, old timber floating routes, forestry and agriculture as well as buildings and constructions of various kinds.

Regarding morphology, a system with two classification levels is described in this handbook (Table 4.1). The basic level consists of Level 2 and is a classification level where the background data that forms the basis of the classification is deemed to be available in the form of digital map layers, historical maps and official documentation. Level 1 is characterised by background data from field inventories where more detailed information about a water body is collected and stored. The reason why the system is divided into two levels is the considerable variation among different regions in Sweden as regards background data about their water bodies. The assessment criterion must be designed in accordance with a system that minimises the risk of making a tougher impact classification for well-mapped areas than areas with poorer information about the impact status. In order for it to be possible to make a fair comparison between different parts of the country and to ensure all water bodies can be classified, a basic classification level (Level 2) is needed where background data must be available to all instances that perform the impact classification. When classifying a water body, the optimum level (Level 1) is therefore applied using all the parameters, where this is possible and where data is available. In areas where such background data is not available, water bodies are classified based on background data in accordance with Level 2.

Table 4.1. Background data for the classification of anthropogenic impact. Elements in italics are not included in any of the assessment criteria.

Level 1		
Bodies of water with background data from:	Information about:	Type of impact
Biotope mapping Equivalent field inventories Electric fishing protocols	Route Form Retaining ditches Bank stabilisation Dead wood Soil type in adjacent areas *Bottom substrate* *Bottom structure* *Structure of the riparian/shoreline zone*	Straightening/canalisation Clearing Ditching Water abstraction Bank stabilisations
Level 2		
Bodies of water with background data from:	Information about:	Type of impact
GIS Land cover data Cadastral map Timber floating documentation Ditching documentation Ordnance survey maps	Overhead road crossings Adjacent land use Land use in the sub-basin Route Connectivity Ditches	Road network Forestry Agriculture Water abstraction

4.2 Input parameters

Morphology is divided into the two quality elements “Morphology in lakes” and “Morphology in watercourses” (see Table 4.2). The quality element “Morphology in lakes” includes the parameters “Adjacent land use”, “Land use in the sub-basin”, “Modified littoral zone”, “Number of ditches per km” and “Dead wood”. All these parameters are stipulated in Regulations (REG) NFS 2008:1, of which “Dead wood” is stipulated in the form of a general guideline (GG).

See REG Annex 3, Section 6.5

See GG to Annex 3 Section 6.4.11

The quality element of “Morphology in watercourses” includes the parameters “Adjacent land use”, “Land use in the sub-basin”, “Number of ditches per km”, “Degree of straightening/canalisation”, “Number of overhead road crossing per km of watercourse”, “Dead wood” and “Proportion of length cleared”. All these parameters are stipulated in Regulations (REG) NFS 2008:1. “Dead wood” is stipulated as a general guideline (GG).

Table 4.2. Parameters for classifying morphological impact.

Assessment criterion (parameter)	Supporting material:	Watercourses	Lakes
Adjacent land use	Land cover data, cadastral map	X	X
Land use in the sub-basin	Land cover data, cadastral map	X	X
Dead wood (number of pieces of wood)	Biotope mapping, inventory	X	X
Modified littoral zone	SMHI, water judgements, biotope mapping		X
Number of ditches per km	Cadastral map, ditching documentation, biotope mapping	X	X
Degree of straightening/canalisation	Biotope mapping, maps, GIS, Historical documents	X	
Proportion of length cleared	Biotope mapping	X	
Number of road crossings per km	Cadastral map	X	

4.3 Adjacent land use

Adjacent land use is divided up in accordance with System Aqua into three groups of heavily disturbed types: Clear-cut areas, farmland (including pastureland) and built/developed areas (including quarries). The consequences of these adjacent artificial land types are that the protective forest edge disappears and with it screening, insolation protection, biota drop-off and supply of dead wood. Furthermore, nutrient leakage to the water from the surrounding land increases. The lack of vegetation on the surrounding land can also increase soil erosion in certain topographical conditions resulting in more transport of humus and fine-particle matter into the water body. Studies of forest streams have indicated that the immediate vicinity (0-5 m) was more closely connected to the status of the watercourse than the less immediate vicinity (5-30 m from the stream) and the rest of the catchment area respectively.[10].

4.3.1 Requirements for supporting data

See REG Annex 3, Section 5.2.1

See GG to Annex 3 Section 5.2.1

Information from field controls or map analyses shall be used when classifying. When classification is based on field controls, supporting data from biotope mapping or equivalent field inventories should be used. When classification is based on

[10] Markusson, K. 1998. Omgivande skog och skogsbrukets betydelse för fiskfaunan i små [The impact of the surrounding forest and forestry operations on fish fauna in small forest streams] National Board of Forestry, Report: 8, p 35

map analysis, supporting material from land cover data and the cadastral map should be used.

4.3.2 Classification of status

If the adjacent land areas have been mapped as part of biotope mapping or a similar field survey, the results of this are used when classifying status. Analysis and classification of adjacent land use in accordance with Level 2 is performed as follows:

- A watercourse theme digitalised in a scale of 1:10 000 or the cadastral map's lakes and watercourses are used as a basic map in GIS.
- A 2-pixel (50 m) buffer is then put around the lakes/on each side of the watercourses using information from land cover data.
- The areas are extrapolated using data on new clear-cut areas from the National Board of Forestry's GIS system[11] and buffered (20 m) house constructions from the cadastral map.

See REG Annex 3, Section 5.2.1

Figure 4.1 below is an example of a digital map image after extrapolation.

[11] The National Board of Forestry's GIS system is called "Kotten" (Pine-cone), the GIS system developed by the Board to check how Sweden's forests are managed, using data on felling operations, replanting, etc.

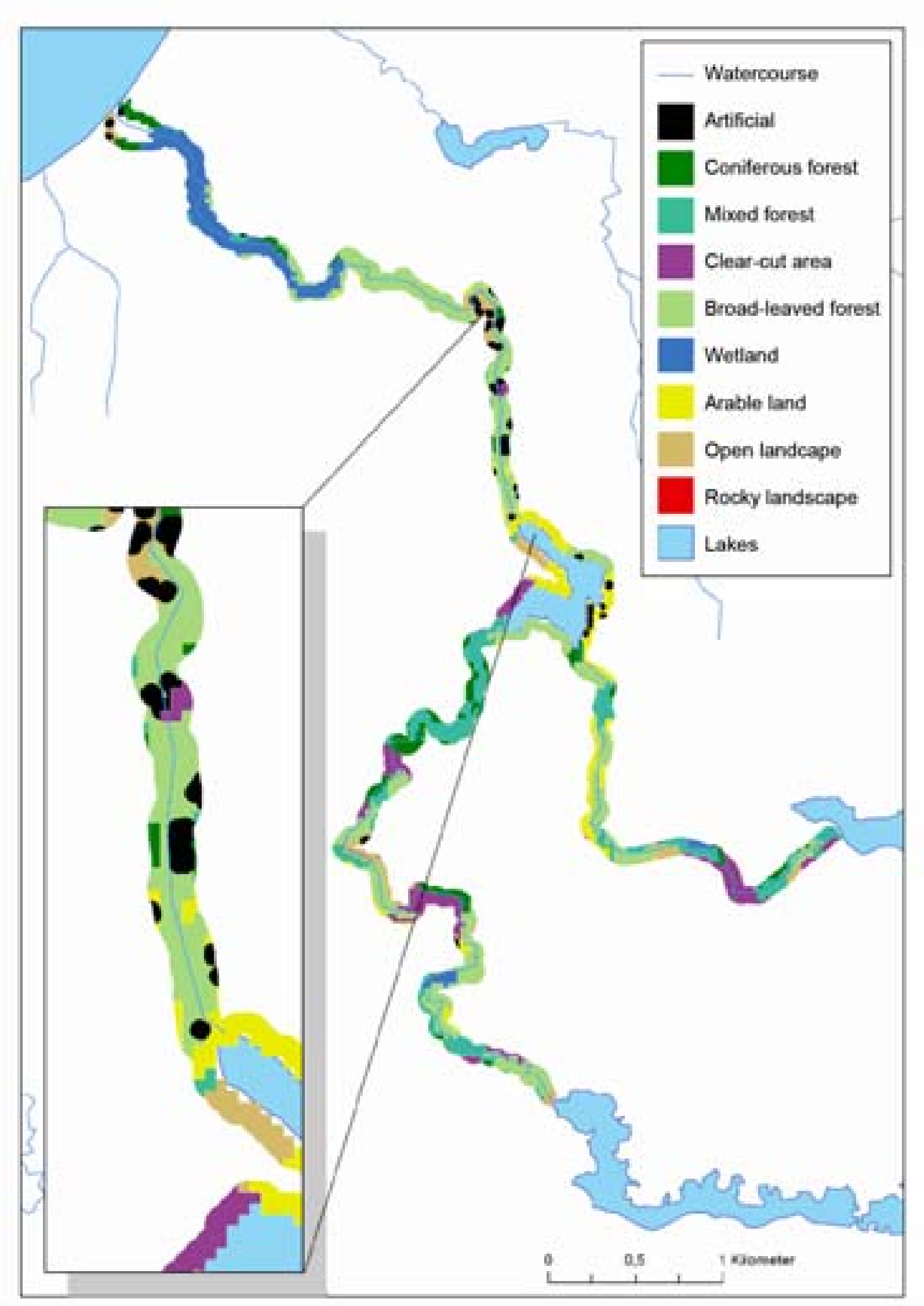

Figure 4.1. Watercourse with 2-pixel buffer (50 m) on either side. The colours are land cover data with supplementary information on artificial land from the economic map and the National Board of Forestry's GIS system.

The percentage occurrence of the three artificial land-types clear-cut area, farmland and built/developed land is calculated in relation to other adjacent land-types. In accordance with Biotope Mapping- Watercourses[12], land is counted as clear-cut area until the new forest has reached an average height of 1.3 metres. Information about the condition of adjacent land also emerges when mapping the biotopes of watercourses. If this inventory is up-to-date, the results can be used to assess impact.

Adjacent land use is calculated as the proportion of disturbed land in relation to the total amount of land. Disturbed land includes:

See REG Annex 3, Section 5.2.2

- urban structure,
- industry,
- mining areas,
- quarries,
- building sites,
- farmland,
- pastureland and
- clear-cut areas.

4.3.3 Class boundaries

See REG Annex 3, Section 5.2.3

Class boundaries for the classification of Adjacent land use are given in Table 4.3

Table 4.3. Class boundaries for heavily disturbed land-types adjacent to the water body.

Status	Class	Impact	Adjacent land use
High	1	No impact	≤10 % adjacent land is artificial
Good	2	Minor impact	>10-20 % adjacent land is artificial
Moderate	3	Moderate impact	>20-40 % adjacent land is artificial
Poor	4	Significant impact	>40-60 % adjacent land is artificial
Bad	5	Heavy impact	>60 % adjacent land is artificial

[12] Halldén A, Liliegren Y. and Lagerkvist G. 2002. Biotopkartering - vattendrag [Biotope mapping – watercourses]. Metodik för kartering av biotoper i anslutning till vattendrag 2002 [Methodology for mapping biotopes adjacent to watercourses 2002]. Jönköping County Administrative Board, Communication 2002:55.

4.4 Land use in the sub-basin

Forestry and agricultural land and other disturbed land areas in a water body's sub-basin/s (sub-RB) can have an effect on watercourses and lakes as a result of nutrient leakage, increased turbidity and transport of metal pollutants to adjacent waters. It is therefore important to also include land-types in the sub-basin when classifying the status of a water body. Land-use in the landscape has been shown to have a clear connection with the status of watercourses.[13]. Previous analyses have indicated significantly divergent values of benthic macroinvertebrate indices where land in the river basin has consisted of a large proportion (25 %) of agricultural land [14].

4.4.1 Requirements for supporting data

See REG Annex 3, Section 5.3.1

See GG to Annex 3 Section 5.3.1

Information from field controls or map analyses shall be used when classifying. When classification is based on field controls, supporting data from biotope mapping or equivalent field inventories should be used. When classification is based on map analysis, supporting material from land cover data and the cadastral map should be used.

4.4.2 Classification of status

When classifying Land use in the sub-basin, the same map data as that used in the analysis of Adjacent Land use can be employed. Digitalised maps showing sub-basin boundaries from SMHI (sub-basin layer 2007) are used with land cover data put on top. Proportion of disturbed land; clear-cut areas, farmland and building sites/developed land are then calculated from the land cover data. The map is then extrapolated using information on new clear-cut areas from the National Board of Forestry's GIS system.

See REG Annex 3, Section 5.3.2

Land use in the sub-basin is calculated as a proportion of disturbed land in relation to the total amount of land. Disturbed land includes:

- urban structure,
- industry,
- mining areas,
- quarries,
- building sites,
- farmland,
- pastureland and
- clear-cut areas.

[13] Degerman, E. et al., 2006. Classification and assessment of degradation in European running waters. Fisheries management and ecology, In press.

[14] Sandin, L., 2003. Benthic macroinvertebrates in Swedish streams: community structure, taxon richness, and environmental relations. Ecography 26: 269-282.

4.4.3 Class boundaries

The class boundaries for Land use in the sub-basin follow the class boundaries according to System Aqua. An in-depth analysis from several regions should be performed in the future to increase the accuracy of the class boundaries.

Class boundaries for the classification of Land use in the water body's sub-basin are given in Table 4.4.

See REG Annex 3, Section 5.3.3

Table 4.4. Class boundaries for Land use in the sub-basin.

Status	Class	Impact	Land use in the sub-basin
High	1	No impact	≤10 % of the sub-basin consists of artificial land
Good	2	Minor impact	>10-20 % of the sub-basin consists of artificial land
Moderate	3	Moderate impact	>20-40 % of the sub-basin consists of artificial land
Poor	4	Significant impact	>40-60 % of the sub-basin consists of artificial land
Bad	5	Heavy impact	>60 % of the sub-basin consists of artificial land

4.5 Dead wood (number of pieces of wood)

Apart from good continuity, good spawning bottoms and considerable stand site variation, the abundance of salmon fish is also dependent on the production of invertebrates in and around the watercourse. This is made possible to a large extent by the occurrence of mainly dead wood in, above and adjacent to the water. Clear-cutting next to the watercourse resulting in lost biomass drop-off and fewer fallen trees reduces the possibilities for good living conditions for fish. An adequate buffer zone, left during logging or other type of land exploitation, offers good possibilities for fish in the form of nutrient drop-off, shading, reduced insolation, less disturbance and a rise in the future production of dead wood. Dead wood has a proven positive effect on both biological dynamics and biological diversity. The structure formed by wood in water also gives the watercourse the capacity to retain nutrients for longer and organic material is more efficiently converted in various biological processes. Furthermore, wood in water has a moderating effect on negative erosion processes. Direct effects of wood in water are the possibility it offers as stand sites for fish and protection against predators and currents. A rise in the number of trout in forest waters with an increased volume of wood has been seen in Swedish forest streams[15].

[15] Degerman, E., Magnusson, K. and Sers, B., 2005. Fisk i skogsbäckar [Fish in forest streams]. World Wide Fund for Nature (WWF), Levande skogsvatten [Flourishing forest waters, p 31.

The volume of dead wood is calculated as the number of pieces of wood (longer than 1 metre and more than 10 cm in diameter) per 100 metres. The volume of dead wood is derived during biotope mapping, electric fishing and similar field surveys. Adjacent land areas can, however, give an indication of the amount of dead wood. The connection between older, unharvested forest and the volume of dead wood in water has been proven,[16] which means that even if the actual volume of dead wood cannot be established, adjacent land made up of e.g. ancient coniferous forest is important since this land-type most probably has an impact on the volume of wood in the water. Clear links have been proven between the volume of dead wood and fish abundance (trout). An increase in dead wood up to 8-16 pieces (>1m long, >10cm in diameter) has led to a rise in the number of trout at such sites[17]. A further increase up to 25 pieces/100m could be ascertained in the same survey, though without significance.

4.5.1 Requirements of supporting data

See GG to Annex 3 Section 5

Information from field controls should be used when carrying out the classification. The field control should be performed in the form of biotope mapping data or similar field inventory.

4.5.2 Classification of status

See GG to Annex 3 Section 5

Data on the volume of dead wood in the water is compiled from water biotope mapping or similar field inventories. The classifications in biotope mapping protocol A is transformed using:

Class according to biotope mapping protocol A7. Dead wood	Class	Status
3 (> 25 logs)	1	High
2 (6-25 logs)	2	Good
1 (< 6 logs)	4	Poor
0 (0 logs)	5	Bad

The parameter Dead wood cannot be used in areas where the land is low-productive (“impediment land”), such as rocky or marshy ground, since the supply

See GG to Annex 3 Section 5

[16] Degerman, E., Halldén, A. and Törnblom, J., 2005. Död ved i vattendrag – Effekten av skogsålder och skyddszon på mängd död ved [Dead wood in watercourses - The effect of forest age and protection zones on the volume of dead wood]. Report:, Worldwide Fund for Nature (WWF), Levande skogsvatten [Flourishing forest waters], p 18

[17] Ibid.

of dead wood to the watercourse is, for obvious reasons, not as high as in forest land.

The volume of dead wood is calculated as the number of pieces of wood >1m long and > 10cm in diameter / 100 metres.

4.5.3 Class boundaries

Class boundaries for classifying dead wood are given in Table 4.5.

See GG to Annex 3 Section 5

Table 4.5. Class boundaries for the volume of dead wood in watercourses.

Status	Class	Impact	Dead wood (number of pieces of wood) >1m long and > 10cm in diameter / 100 metres.
High	1	No impact	>16 pieces
Good	2	Minor impact	>10-16 pieces
Moderate	3	Moderate impact	>6-10 pieces
Poor	4	Significant impact	≤6 pieces
Bad	5	Heavy impact	0 pieces

4.6 Modified littoral zone

Previous lowerings and raisings of the water level in lakes, often as a result of regulation when producing hydropower, have affected their littoral zones (shoreline zones), e.g. in the form of a modified composition of flora. Invertebrates living in the littoral zone and that constitute an important source of food for fish are also affected when the water is regulated, due to the changed water level. Water level changes in water bodies are therefore seen as having a significant impact on biology.

Lasting changes in the littoral zone can be caused by permanent increases and decreases in the water level, extensive dredging operations and diggings, the filling-in of shore areas and sounds, modified outlet, etc.

4.6.1 Requirements for supporting data

Registers of raised and lowered water bodies can be found at SMHI.[18] and in some cases in regional water judgements.

See REG Annex 3, Section 5.4.1

[18] SMHI, 1995. Sänkta och torrlagda sjöar [Lowered and drained lakes]. SMHI Hydrology: 62.

Information from field controls or map analyses shall be used when classifying. When classification is based on field controls, supporting data from biotope mapping or similar field inventories should be used. When classification is based on map analysis, supporting material from land cover data and the cadastral map should be used.

See GG to Annex 3 Section 5.4.1

4.6.2 Classification of status

The effect magnitude of an impact on the water body's littoral zone depends on a large number of factors, including the shoreline gradient, the type of lake bottom, surrounding environments and the climate. For this reason, a rough 3-point scale is presented here divided into high, moderate and bad status, for the parameter Modified littoral zone. Some of the limit values are taken from System Aqua.

To distinguish between different types of impact and at what time any interventions may have been performed, a time limit of 50 years and a difference in level of 1 meter have been used as benchmarks for the class boundaries. Modified littoral zone can either be calculated as the proportion of disturbed stretch/ the total stretch of shoreline*100, or as the changed water level in metres.

See REG Annex 3, Section 5.4.2

If, when classifying the parameter Modified littoral zone, it is ascertained that a water body has moderate or bad status, at the same time as all biological quality elements are classified as high status, a reasonability assessment can be performed by the water authority. In cases where this assessment indicates that the classification of the parameter is unreasonable, the previous classification can be ignored. The next step is to perform an expert judgement, which may result in a new classification of status or potential, either for the entire water body or for individual quality elements. In this context, it is important to consider whether parameters or quality elements that react slowly to environmental changes ("inert parameters") have had enough time to react to the impact factors in question.

See REG Chapter 2. Sections 8-9

In cases where the water level in a water body has been actively regulated, the parameter Modified littoral zone is not used. The reason for this is the difficulty in defining the littoral zone in such cases.

In the class boundaries for Modified littoral zone, there is no connection between time and alterations of water level in the requirements for Moderate status. The effect of this is hence that a water body that has been subject to water level changes several decades ago might risk being classified as moderate instead of high status. Because of the other conditions that would have to be fulfilled to have the water body classified as Moderate status and the fact that an expert judgement could alter a classification based on anthropogenic impact, the risk of an incorrect assessment can be prevented.

4.6.3 Class boundaries

Class boundaries for classifying Modified littoral zone are given in Table 4.6.

See REG Annex 3, Section 5.4.3

Table 4.6. Classification for Modified littoral zone in lakes. The parameter does not include classes for good and poor status.

Status	Class	Impact	Modified littoral zone
High	1	No impact	The water level has changed by <0.5m, or, interventions performed over the last 50 years have changed <10% of the shoreline, or, inventions performed more than 50 years ago have changed <25% of the shoreline.
Moderate	3	Moderate impact	The water level has changed by 0.5m to 1m, or, interventions performed over the last 50 years have changed 10-25% of the shoreline, or, inventions performed more than 50 years ago have changed 25-50% of the shoreline.
Bad	5	Heavy impact	The water level has changed by >1m, or, interventions performed over the last 50 years have changed >25% of the shoreline, or, inventions performed more than 50 years ago have changed >50% of the shoreline.

4.7 Number of ditches per km

The effect of outflowing ditches in water bodies can be in the form of e.g. increased sediment supply, especially in high water periods. The result of the increased concentration of fine particles has an effect in the form of embedded boulders and the silting-up of potential spawning and breeding grounds and greater water turbidity. Many ditches have been dug with the intention of draining the surrounding land in order to increase forest production or to create more agricultural land. Such ditching operations have led to serious hydrological effects particularly on smaller watercourses, such as increased desiccation and reduced water flow.

4.7.1 Requirements for supporting data

See REG Annex 3, Section 5.5.1

See GG to Annex 3 Section 5.5.1

Information from field controls or map analyses shall be used when classifying. When classification is based on field controls, supporting data from biotope mapping or equivalent field inventories should be used. When classification is based on map analysis, supporting material from land cover data, the cadastral map, timber floating documentation, ditching documentation and ordnance survey maps should be used.

4.7.2 Classification of status

See REG Annex 3, Section 5.5.2

Data on the number of ditches can be compiled with the help of both results from any water biotope mapping that has been performed in combination with the cadastral map. If there is no data from biotope mapping for the watercourse in question, only the cadastral map is used as background data. A ditch is defined as an artificial/excavated, often straight channel that flows out into the water body.

The number of ditches is calculated as the number of ditches per km of watercourse or lake shoreline.

4.7.3 Class boundaries

See REG Annex 3, Section 5.5.3

Class boundaries for the classification of the number of ditches per km are given in Table 4.7

Table 4.7. Class boundaries for Number of ditches per km

Status	Class	Impact	Number of ditches per km
	1	No impact	<1 ditch
Good	2	Minor impact	1-3 ditches
Moderate	3	Moderate impact	>3-5 ditches
Poor	4	Significant impact	>5-7 ditches
Bad	5	Heavy impact	>7 ditches

4.8 Degree of straightening/canalisation

Straightening/canalisation has mostly been caused by the drainage of agricultural land and older forestry operations in the form of timber floating. These interventions, whereby water furrows have been redug, have resulted in previously more or less winding or meandering watercourses losing the qualities that are significant for an undisturbed watercourse. Erosion of the outer edges of the furrow in its curves, which creates a protective overhang in the river bank for larger fish, disappears. The deposition of fine-particle matter changes and the current velocity is not reduced by the meandering, but instead, when straightened, gains a higher velocity resulting in substantially changed erosion patterns and washout effects. This also results in the loss of the continuous change in the water furrow which is important for most aquatic organisms in a naturally efficient watercourse.

4.8.1 Requirements for supporting data

Information from field controls or map analyses shall be used when classifying. When classification is based on field controls, supporting data from biotope mapping or equivalent field inventories should be used. When classification is based on map analysis, supporting material from the cadastral map, timber floating documentation, ditching documentation and ordnance survey maps should be used.

See REG Annex 3, Section 6.2.1

See GG to Annex 3 Section 6.2.1

4.8.2 Classification of status

The degree of straightening/canalisation can be derived from biotope mapping and similar field inventories or from map analyses. The best assessment is made if biotope mapping data is available to use in combination with map material. In cases where water bodies have been biotope-mapped and the emerging data enables a level 1 classification to be performed, the following applies for biotope mapping protocol A - Water biotopes; culverted, dam, impounded and clearing grade 3 are synonymous with straightening/canalisation. In biotope mapping, clearing grade 3 therefore corresponds to a straightening or redigging, whilst clearing grade 1 - 2 corresponds to clearing and is a separate parameter.

Historical maps and documents, e.g. any timber floating documentation, ditching documentation and ordnance survey maps, showing the original route of the watercourse, constitute an important basis for the classification of the degree of straightening. These can be used as a screening tool, to identify where investigative measures are to be implemented. The first measure should then be, for example, to perform biotope mapping to investigate and confirm any impact.

The degree of straightening/canalisation is calculated as the proportion of straightened or canalised stretch of watercourse in relation to its total length.

See REG Annex 3, Section 6.2.2

4.8.3 Class boundaries

Class boundaries for the classification of the degree of straightening/canalisation are given in Table 4.8.

See REG Annex 3, Section 6.2.3

Table 4.8. Class boundaries for the proportion of straightened or canalised stretch of the water-course's total length.

Status	Class	Impact	Degree of straightening/canalisation
High	1	No impact	No straightening
Good	2	Minor impact	≤10%
Moderate	3	Moderate impact	>10-40%
Poor	4	Significant impact	>40-70 %
Bad	5	Heavy impact	>70 %

4.9 Proportion of length cleared

The creation of floating channels for timber transport caused problems in shallow stretches and in stretches containing a lot of large boulders. These stretches have hence been particularly susceptible to clearing operations to clear away such elements either by blasting with explosives or by lifting boulders and stones away from the middle of the furrow and putting them along the edges and on the banks. This was done to obtain a more even depth and create a smooth, unobstacled floating channel. Interventions such as these have resulted in a loss of important stand sites for fish, residence sites for invertebrates and macrophytes and have changed the current conditions, making them more homogenous. This has also resulted in a reduction in heterogeneous turbulence which creates pools and oxygenates the water.

4.9.1 Requirements for supporting data

See REG Annex 3, Section 6.3.1

See GG to Annex 3 Section 6.3.1

Information from field controls or map analyses shall be used when classifying. When classification is based on field controls, supporting data from biotope mapping or equivalent field inventories should be used. When classification is based on map analysis, supporting material from the cadastral map, timber floating documentation, ditching documentation and ordnance survey maps should be used.

4.9.2 Classification of status

Information on clearing can be found in databases from any biotope mapping that has been performed and is a parameter for a level 1 classification. In the biotope mapping protocol, parameter A9, there is a 4-point scale for the degree of clearing. Regarding this assessment criterion, classes 2 and 3 are counted as cleared. This consequently means that a classification of the degree of clearing to 3 when biotope mapping is carried out is translated to both cleared and straightened in the assessment criterion. Assessment 0 or 1 is therefore not assessed as cleared. If data in the form of timber floating documentation or ditching documentation is available, it can also be used to give an indication of the degree of clearing and then formally becomes a level 2 classification.

Proportion of stretch cleared is calculated as the proportion of length cleared divided by the total length of the watercourse.

See REG Annex 3, Section 6.3.2

4.9.3 Class boundaries

See REG Annex 3, Section 6.3.3

Class boundaries for the classification of Proportion of length cleared are given in Table 4.9.

Table 4.9. Impact classification in the form of proportion of length cleared.

Status	Class	Impact	Proportion of length cleared
High	1	No impact	0 %
Good	2	Minor impact	≤10%
Moderate	3	Moderate impact	>10-25 %
Poor	4	Significant impact	>25-50 %
Bad	5	Heavy impact	>50 %

4.10 Number of overhead road crossings per km

One of the studies performed[19] indicates that a large proportion of the road culverts at these road crossings, in some areas up to 88 %, constitute migration barriers. These do not just affect fish but also other fauna such as invertebrates and otters. Other studies also indicate that the culvert itself, even though it does not constitute a migration barrier physically, can have a negative effect on migrating trout fry, since it has been observed that they are unwilling to pass through the culvert[20]. Overhead road crossings may also have a secondary effect in that the roadside clear-cut areas reduce insolation protection.

[19] Bergengren, J.,1999. Vandringshinder och spridningsbarriärer inventerade i 11 vattensystem i Västernorrland [An inventory of migration and dispersion barriers in 11 water systems in Västernorrland] Västernorrland County Administrative Board 1999:1

[20] Kemp, P.S., Gessel, M.H. and Williams, J.G., 2005. Seaward migrating subyearling chinook salmon avoid overhead cover. Journal of Fish Biology, 67, 1381-1391.

See REG Annex 3, Section 6.4.1

See GG to Annex 3 Section 6.4.1

4.10.1 Requirements for supporting data

Information from field controls or map analyses shall be used when classifying. When classification is based on field controls, supporting data from biotope mapping or equivalent field inventories should be used. When classification is based on map analysis, supporting material from land cover data and the cadastral map should be used.

See REG Annex 3, Section 6.4.2

4.10.2 Classification of status

Number of roads such as; public highways, private roads and forest roads that cross the watercourse can be identified using field inventories or map material (cadastral map) and the number of overhead crossings per km is calculated.

The number of roads is calculated as the number of road crossings per km of watercourse.

See REG Annex 3, Section 6.4.3

4.10.3 Class boundaries

Class boundaries when classifying the Number of overhead road crossings are given in Table 4.10.

Table 4.10. Classification of Number of overhead road crossings per km of watercourse.

Status	Class	Impact	Number of overhead road crossings per km
High	1	No impact	<1
Good	2	Minor impact	1-3
Moderate	3	Moderate impact	>3 – 6
Poor	4	Significant impact	>6 – 10
	5	Heavy impact	>10

5 Co-weighting of hydromorphological parameters and quality elements

5.1 Draft working method

This handbook proposes an approach whereby the impact on continuity in the form of the parameter "Presence of man-made migration barriers" is classified first. This is because an assessment of the impact on continuity requires all the migration barriers throughout the river basin to be surveyed. A dam may prevent salmon from migrating to their natural spawning grounds, which prior to the dam being built were a long way up in the water system. The continuity of a water body must therefore be described based on supporting knowledge from the entire water system and this knowledge can also be used when classifying hydrological regime and morphology.

The dams surveyed for the classification of continuity usually impact the hydrological regime upstream of the water body as well and in one or more water bodies downstream of the dam. The presence of dams and migration barriers is also an element that impacts the morphology of the water bodies. It is therefore rational to coordinate the compilation of assessment material and to utilise digital map analysis when classifying.

Assessing the impact of water regulation on hydrological regime also includes an assessment of how much the water level in the regulated lakes is affected and how much the water flow is affected downstream of the regulation dam. Here, we have the advantage of working with assessment material and assessment results in a map system where there is defined data on the regulating dams and about the regulations, as well as with data on flow statistics. This material makes it easier for us to assess how far downstream from a regulating dam or an assessment site a certain impact on hydrological regime might stretch.

A body of water classified in accordance with the proposed system will be assigned the hydromorphological status given by the heaviest impact according to one of the defined quality elements or parameters in the classification system. If, for example, the classification of continuity and hydrological regime gives good hydromorphological status and a classification of the morphology gives moderate status, the total classification will be moderate status.

5.2 Co-weighting of continuity

Regarding the quality element Continuity in watercourses, the Degree of fragmentation and Barrier effect parameters shall be co-weighted (combined) to produce a common value in accordance with Table 5.1. The co-weighting of the Degree of fragmentation and Barrier effect parameters shall be performed by multiplying their status class (assessed class) by the corresponding coefficient in accordance with Table 5.1. The totals are entered in the Total value column. A mean value of the total values is then calculated giving a total classification. The final classification based on the total classification can then be derived from Table 5.2.

Regarding the quality elements Continuity in watercourses, the final status classification is then derived by comparing the final classification for Degree of fragmentation and Barrier effect to the classification obtained from the parameter "Presence of man-made migration barriers". If these values indicate different impacts, the value indicating the greatest anthropogenic disturbance will be decisive. This happens in accordance with the "one out all out" principle, which in this case applies on the parameter level.

See REG Annex 3, Section 2.1

Regarding the quality element "Continuity in lakes", only the parameter "Presence of man-made migration barriers" is to be classified.

Table 5.1. Table for co-weighting the degree of fragmentation and barrier effect.

Assessment criterion (parameter)	Assessment level	Assessed class	Coefficient	Total value
Degree of fragmentation	1		2	
Barrier effect	1		2	
Total classification				

Table 5.2. Class boundary interval in accordance with the total classification in table 5.1.

Total classification interval	Class	Status
2.0 – 3.6	1	High
3.7 – 5.2	2	Good
5.3 – 6.8	3	Moderate
6.9 – 8.4	4	Poor
8.5 – 10	5	Bad

5.3 Co-weighting of hydrological regime

Hydrological regime is divided into two quality elements. The parameter "Prescribed regulation amplitude" is included in the quality element "Hydrological regime in lakes". The support parameter "Impact on water level changes" fulfils the function of an in-depth analysis and is recommended in cases where there is a need and where there is supporting data. This is then done within the framework of

See REG Annex 3, Section 3.1

a reasonability assessment/expert judgement (See Sections 4.1.1 and 4.4 of the main handbook).

The quality element "Hydrological regime in watercourses" consists of the parameter "Impact of flow regulation on watercourses" which in turn is divided into the sub-parameters "Degree of regulation", "Modified mean high water (MHQ)" and "Reduced mean low water (MLQ)". The status for the quality factor "Hydrological regime in watercourses" is determined in accordance with the "one out all out" principle. The two support parameters "Number of flow peaks per year" and "Modified variation coefficient" fulfil the function of an in-depth analysis and are recommended in cases where there is a need and where there is supporting data. This is then done within the framework of a reasonability assessment/expert judgement (See Sections 4.1.1 and 4.4 of the main handbook).

See REG Annex 3, Section 4.1

See GG to Annex 3 Section 4.1

Table 5.3. A summary of classes within the assessment criterion for Hydrological regime in lakes.

Assessment criterion (parameter)	High status	Good status	Moderate status	Poor status	Bad status
Prescribed regulation amplitude	A1	A2	A3	A4	A5
Impact on water level changes	N1	N2	N3	N4	N5

Table 5.4. A summary of classes within the assessment criterion for Hydrological regime in watercourses.

Assessment criterion (parameter)	High status	Good status	Moderate status	Poor status	Bad status
Impact of flow regulation on the watercourse					
Sub-parameter - Degree of regulation	F1	F2	F3	F4	F5
Sub-parameter - Modified MHQ	F1	F2	F3	F4	F5
Sub-parameter - Reduced MLQ	F1	F2	F3	F4	F5
Number of flow peaks per year		T2	T3	T4	
Variation coefficient for 24-hour flow		V2	V3	V4	

5.4 Co-weighting of morphology

The co-weighted classification for the quality elements "Morphology in lakes" and "Morphology in watercourses" is performed by multiplying the status class (assessed class) of the input classified parameters by the corresponding coefficient in Table 5.5. This computation results in a total value for each parameter respectively. For the parameters where it has been possible to specify a value (i.e. where there has been data available), a mean value of the total values is then calculated which in turn gives a total classification. The final classification based on the total classification can then be derived from Table 5.6.

Table 5.5. Table for calculating morphological impact.

Assessment criterion (parameter)	Assessment level	Assessed class	Coefficient	Total value
Land use in the vicinity	2		3	
Land use in the sub-basin	2		2	
Dead wood (number of pieces of wood)	1		3	
Modified littoral zone	2		2	
Number of ditches per km	2		2	
Degree of straightening/canalisation	2		4	
Proportion of length cleared	1		3	
Number of overhead road crossings per km	2		3	
Total classification				

Table 5.6. Class boundary interval in accordance with the total classification in table 5.5

Total classification interval	Class	Status
2.60 – 4.68	1	High
4.69 – 6.76	2	Good
6.77 – 8.84	3	Moderate
8.85 – 10.92	4	Poor
10.93 – 13.75	5	Bad

In individual cases, a situation may occur after co-weighting where most of the parameters indicate a higher status/potential in relation to one or a small number of other parameters. There is then a risk that information on separate though significant morphological encroachments in a water body is not reflected in the final classification. After co-weighting has been performed in accordance with Tables 5.5 and 5.6, the results are therefore checked and the following applies for the co-weighted classification of the Morphology quality element:

> A total status of higher than “Good” cannot be given if:
> - the classification of a parameter with coefficient 4 is higher than 2

or
- the classification of at least two parameters with coefficient 3 is higher than 2.

A total status of higher than “Moderate” cannot be given if:
- the classification of a parameter with coefficient 4 is higher than 3
- the classification of a parameter with coefficient 3 is higher than 4
or
- the classification of three parameters with coefficient 3 is higher than 3.

A total status of higher than “Poor” cannot be given if:
- the classification of a parameter with coefficient 4 is higher than 5
or
- the classification of all parameters with coefficient 3 is higher than 3.

Background reports to sections on morphology:

Jönköping County Administrative Board, 2006. Bedömningsgrunder for hydromorfologi [Assessment criteria for hydromorphology]. Communication No 2006:20.

6 Basis for assessment of hydromorphological conditions in coastal waters

6.1 Introduction

Within the framework of the work done by the Swedish EPA to develop assessment criteria, the scientific basis has not been deemed sufficient to develop national assessment criteria for coastal and transitional waters. The following section therefore only aims to give examples of how statistics on shoreline developments and on ports can be used as a basis for assessments of hydromorphological elements in coastal and transitional waters. The information can for example be used to identify where physical impact is considerably likely.

6.2 Assessment parameters

There are many parameters that can be used to assess the impact on coastal water areas. These can be divided into two main groups. The activities and properties present in:

1. shoreline areas
2. coastal waters

The shoreline parameters give an indirect indication as to whether the coastal water body has been physically impacted. Parameters for activities and phenomena in shoreline areas that can be used as indirect indicators of impact on the water body include buildings, population, urban areas, roads and other constructions. Shoreline areas refer here to phenomena that are less than 100 metres from the shoreline.

Parameters for activities and phenomena in coastal waters that can be used as a more direct indicator of impact on the water body include boat traffic, ports, dredging, shipping lanes, piers, jetties and other constructions.

Examples of where supporting data can be found for some of these parameters are given in Table 6.1. For a more detailed description of different impacts and their sources, please refer to Påverkansbedömning för ytvatten enligt EG:s Ramdirektiv för vatten [Impact assessment for surface water in accordance with the European Water Framework Directive][21].

[21] Wallin, Mats., Olsson Håkan., Zackrisson, Jessica (SMED), 2004: Påverkansbedömning för ytvatten enligt EG:s Ramdirektiv för vatten - tillgängliga metoder, verktyg och modeller samt utvecklingsmöjligheter för SMED&SLU [Impact assessment for surface water in accordance with the European Water Framework Directive - available methods, tools and models as well as development opportunities for SMED & SLU]. Final report, 18 February 2004.

Table 6.1. Examples of indicators of impact on coastal waters and information sources.

Impact	Sources/background data
Buildings	Lantmäteriet (National Land Survey)
Population	Statistics Sweden, Red Map
Urban areas	Statistics Sweden
Roads	Swedish Land Cover Data, red map
Constructions	County Administrative Board, Municipality, Statistics Sweden
Ports	Swedish Maritime Administration, Lantmäteriet, County Administrative Board
Dredging	County Administrative Board
Jetties	County Administrative Board

6.3 Number of shoreline buildings per kilometre of shoreline

Statistics Sweden has data on the number of buildings within 100m of lake and coastal shorelines and river banks. Prior to submitting the first report to the EU in 2005 in accordance with the WFD, Statistics Sweden used statistics from the Lantmäteriet 2003 Buildings Register and urban areas according to maps from 2000 in order to calculate the number of buildings that were close to the coastal water bodies listed in the SMHI register of marine areas from 2004. An improved basis for assessment was produced in 2005 when SMHI calculated the number of buildings per km of shoreline around the basins listed in the 2005 version of the SMHI marine area register.

First of all, regarding the 549 basins that were included in the data, the 10 percentile was calculated for basins with both the highest and the lowest total number of shoreline buildings respectively. The limit values for these were 0.5 and 10 buildings per kilometre of shoreline respectively. Between these class boundaries, a further three classes were created by dividing the difference between the class boundary values of 0.5 and 10 by three. The resulting assessment scale is illustrated in Table 6.2. The classification results are shown on a map in Figure 6.1.

Table 6.2. Limit values between classes for indications of physical impact using statistics on the number of buildings per km of shoreline and the classification of the 549 coastal water basins into the five different classes.

Shoreline buildings			
Impact	**Class**	**Number of buildings per km of shoreline**	**Number of basins per impact class**
Negligible impact	1	0 – 0.499	56
Minor impact	2	0.5 – 3.699	223
Moderate impact	3	3.7 – 6.666	145
Significant impact	4	7 – 9.999	68
Heavy impact	5	≥ 10	57

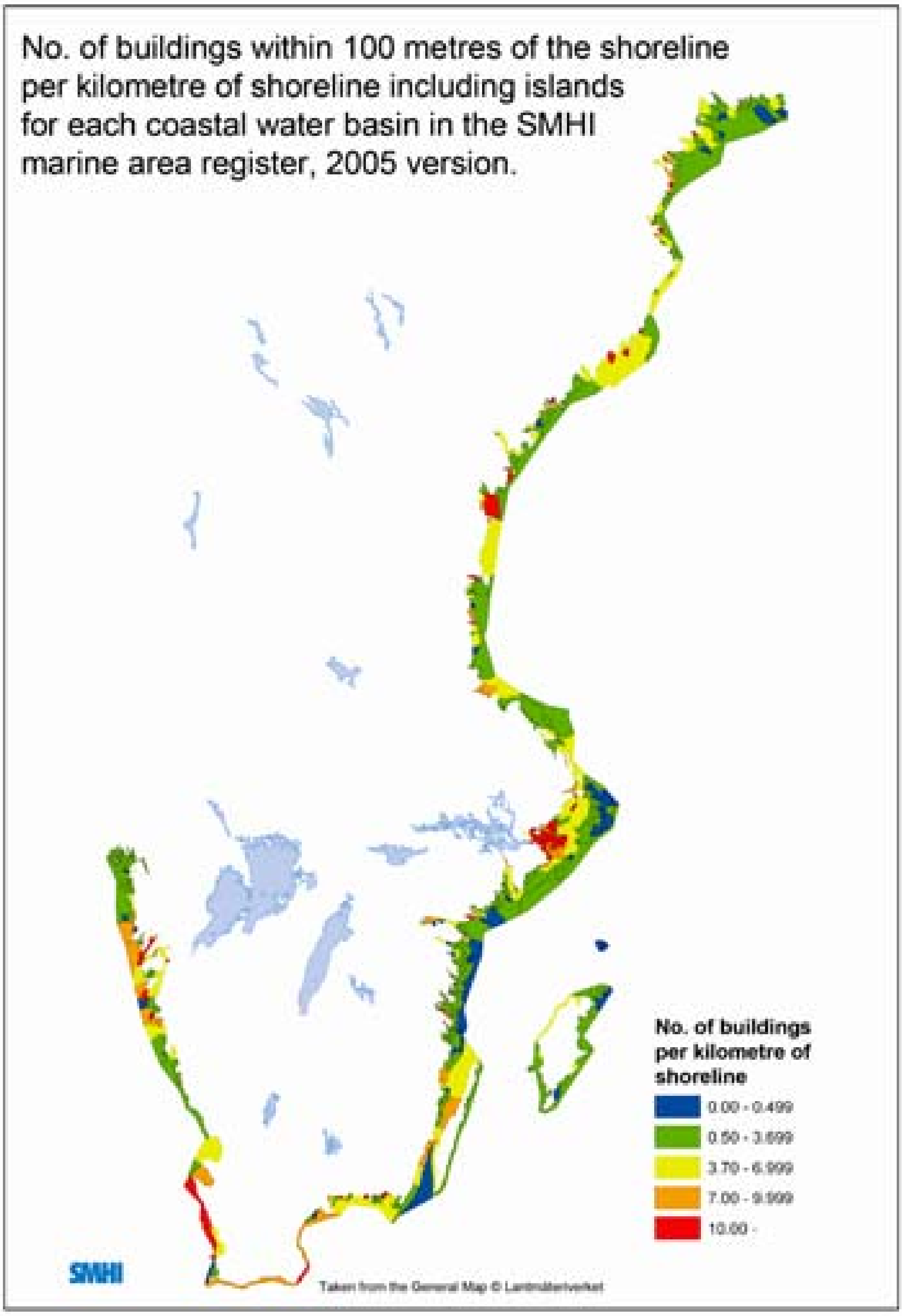

Figure 6.1. Results of the assessment scale based on the statistical distribution of the number of shoreline buildings per stretch of shoreline and coastal water basin. The classification has been done in accordance with the division presented in Table 6.2.

Of the 549 coastal water basins, one, the Södra Kalmarsund deep-sea water area has no islands or coastline according to this map information. Other coastal water basins have stretches of shoreline ranging from 1 to 646 km. There are three basins with stretches of shoreline longer than 500 km. The basin with 646 km of shoreline, the northern-most part of the Norra Bohuslän archipelago coastal water, is a very large basin with most of its shoreline situated in Norway. Since statistics produced by Statistics Sweden do not include buildings in Norway, the number of buildings per km of shoreline has been underestimated for this basin. In addition to the basin that has no shoreline, 20 basins have no registered shoreline buildings according to this background data.

6.4 Impact indicators for ports

Prior to submitting the first report to the EU in 2005 in accordance with the WFD, data on the number of port visits by vessels and on loaded and unloaded cargo, compiled by Statistics Sweden for 117 coastal ports, was used. The number of port visits, and volume of loaded and unloaded cargo (in tonnes) were used as indicators of physical impact on the coastal water basins on which the ports are situated. The statistics were for 2002 and covered vessels with a gross tonnage of 20 tonnes or more. Fishing ports and fishing vessels were therefore not included in the statistics.

Table 6.3 shows the class boundaries applied in the assessment that was conducted prior to submitting the report in 2005. The names of the classes differ slightly from the names used in the report. For example, in the report, classes 4 and5 were amalgamated and called "Significant impact".

Table 6.3. The limit values between the classes indicating the degree of physical impact using the statistics on loaded and unloaded cargo in ports, number of port visits by vessels and the number of classified coastal water basins in the five classes.

Impact	Cargo (tonnes)	Number of basins	Number of visits	Number of basins
1. Negligible impact	500 – 99 999	11	1 – 99	11
2. Minor impact	100 000 – 499 999	15	100 – 499	15
3. Moderate impact	500 000 – 999 999	14	500 – 1 499	22
4. Significant impact	1 000 000 – 4 999 999	19	1 500 – 4 999	13
5. Heavy impact	≥ 5 000 000	12	≥ 5 000	10

Statistics Sweden supplied statistics to SMHI in 7 size classes since it did not wish to divulge the absolute figure for any one particular port. When SMHI applied the figures, it was not possible for example to make classes 3 and 4 smaller since they were separate classes in the background data supplied by Statistics Sweden. There were also approximations when adding classes in cases where there was more than one port in a coastal water body. Each basin was therefore classified according to the biggest impact indication that existed for the area.

If the classifications according to Statistics Sweden are added together and there is more than one port per coastal water basin, the classification of basins will be as Table 6.3, which is shown on the maps in Figure 6.2 and Figure 6.3. The maps in Figure 6.4 and 6.5 also show points for the various ports that have provided the classifications of the larger coastal water areas. We can then see how the shoreline ports can give rise to impact in basins that extend to different distances out to sea, which depends on how they are defined.

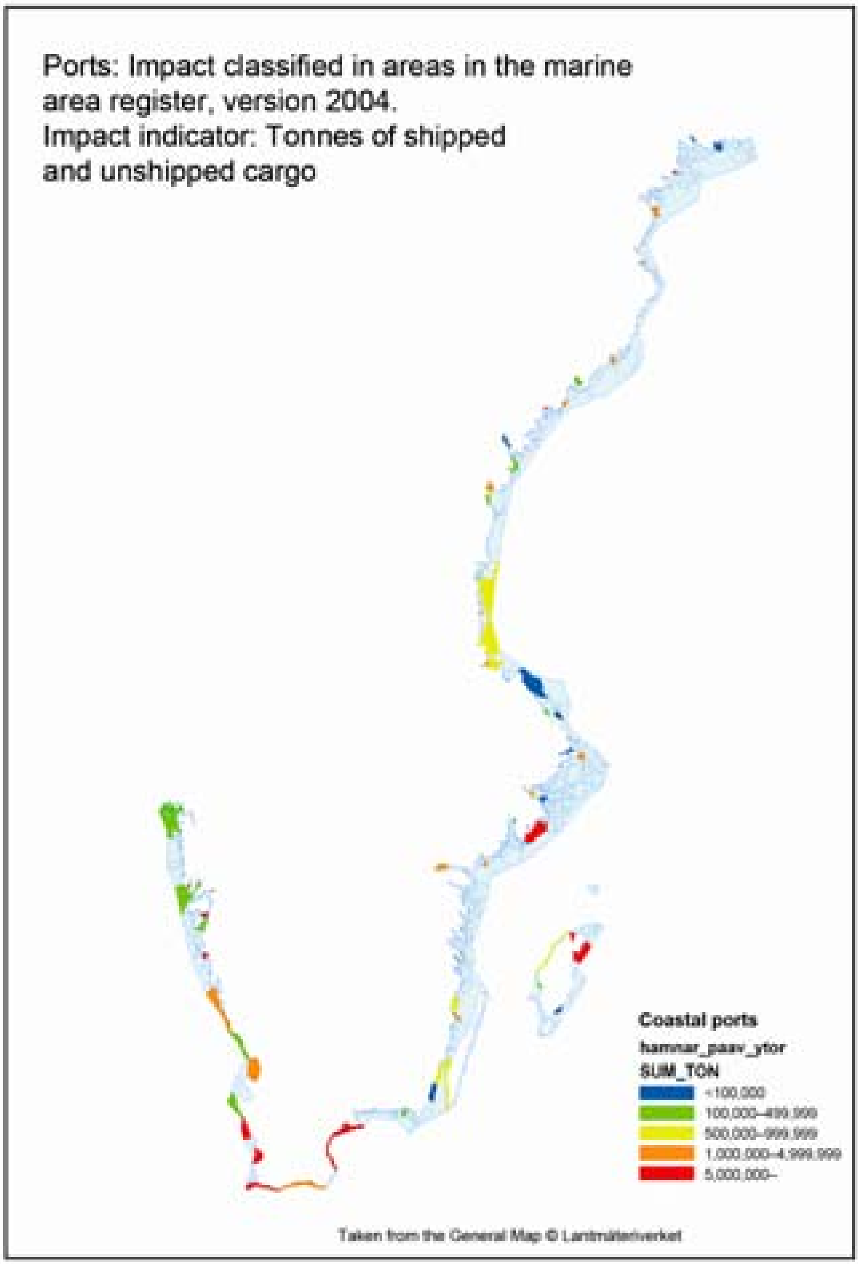

Figure 6.2. Classification of coastal water areas in the 2004 version of the SMHI marine areas register. The classification is based on statistics on the volume of loaded and unloaded cargo (in tonnes) in ports in accordance with Table 6.3. Source: SMHI

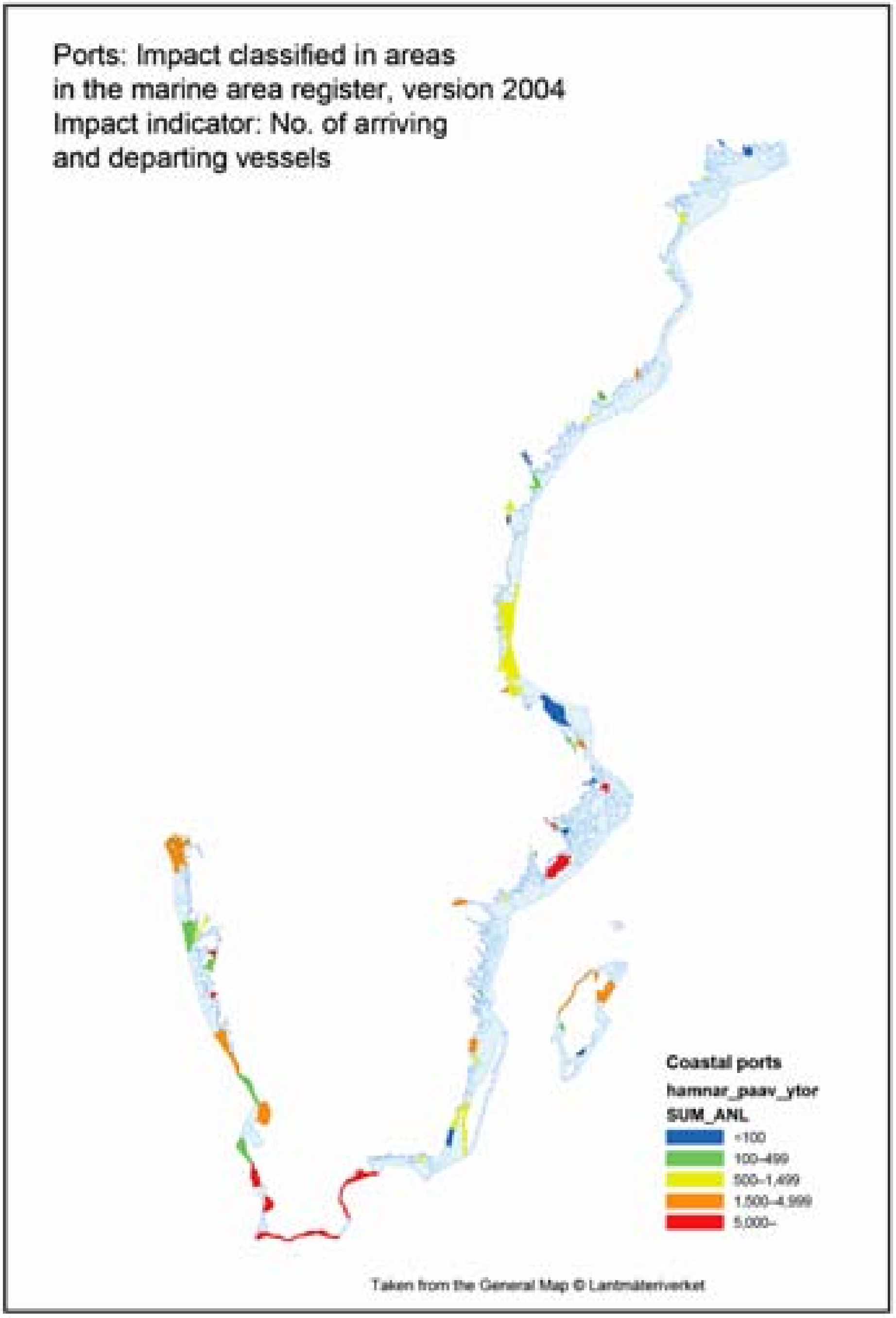

Figure 6.3. Classification of coastal water areas in the 2004 version of the SMHI marine areas register. The classification is based on statistics on the number of visits by vessels to ports in accordance with Table 6.3. Source: SMHI.

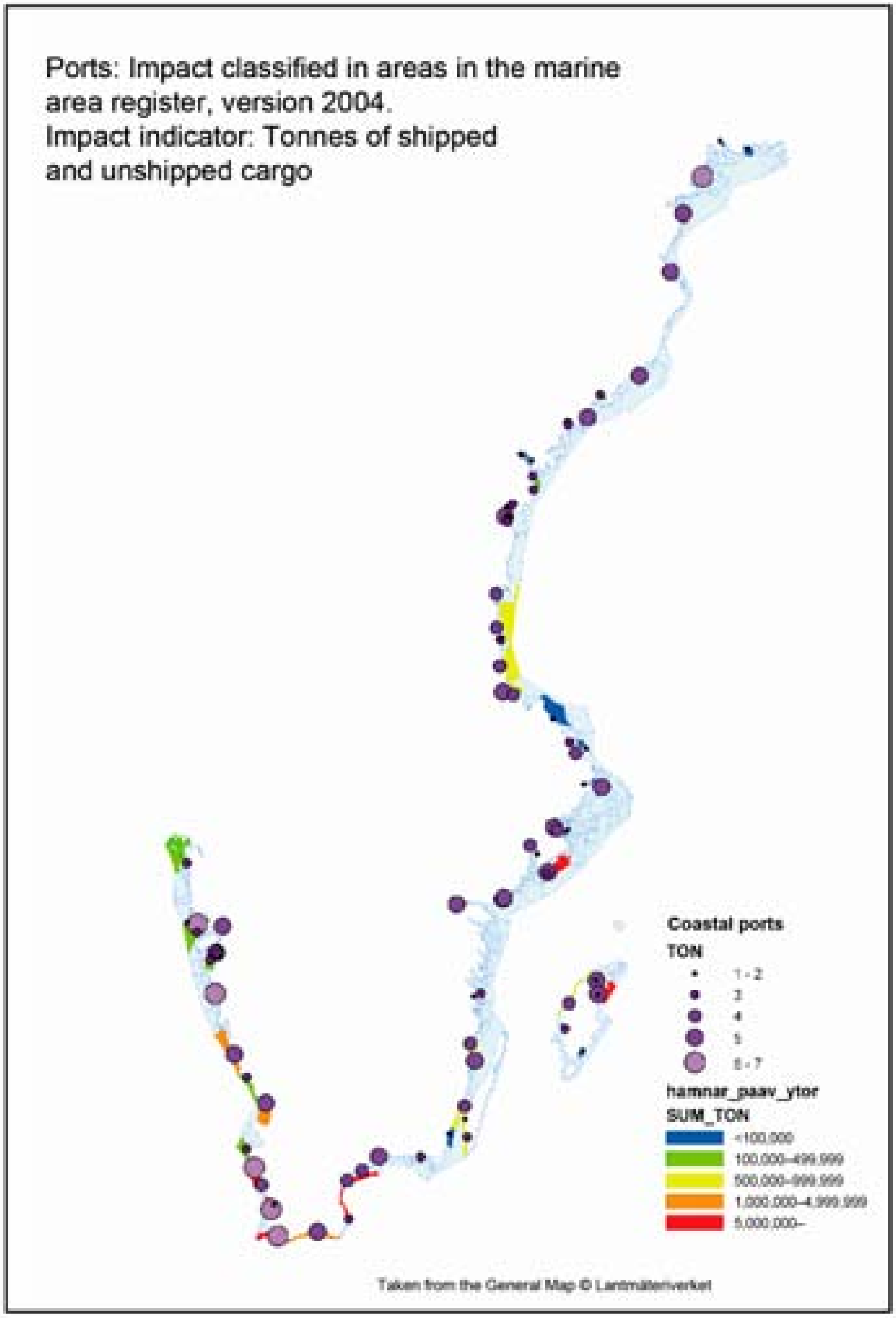

Figure 6.4. Classification of coastal water areas in the 2004 version of the SMHI marine areas register. The classification is based on statistics on the volume of loaded and unloaded cargo (in tonnes) in ports in accordance with Table 6.3. Ports that are associated with the water areas are represented by differently sized points according to classes for the number of tonnes of loaded and unloaded cargo, as defined by Statistics Sweden. Source: SMHI.

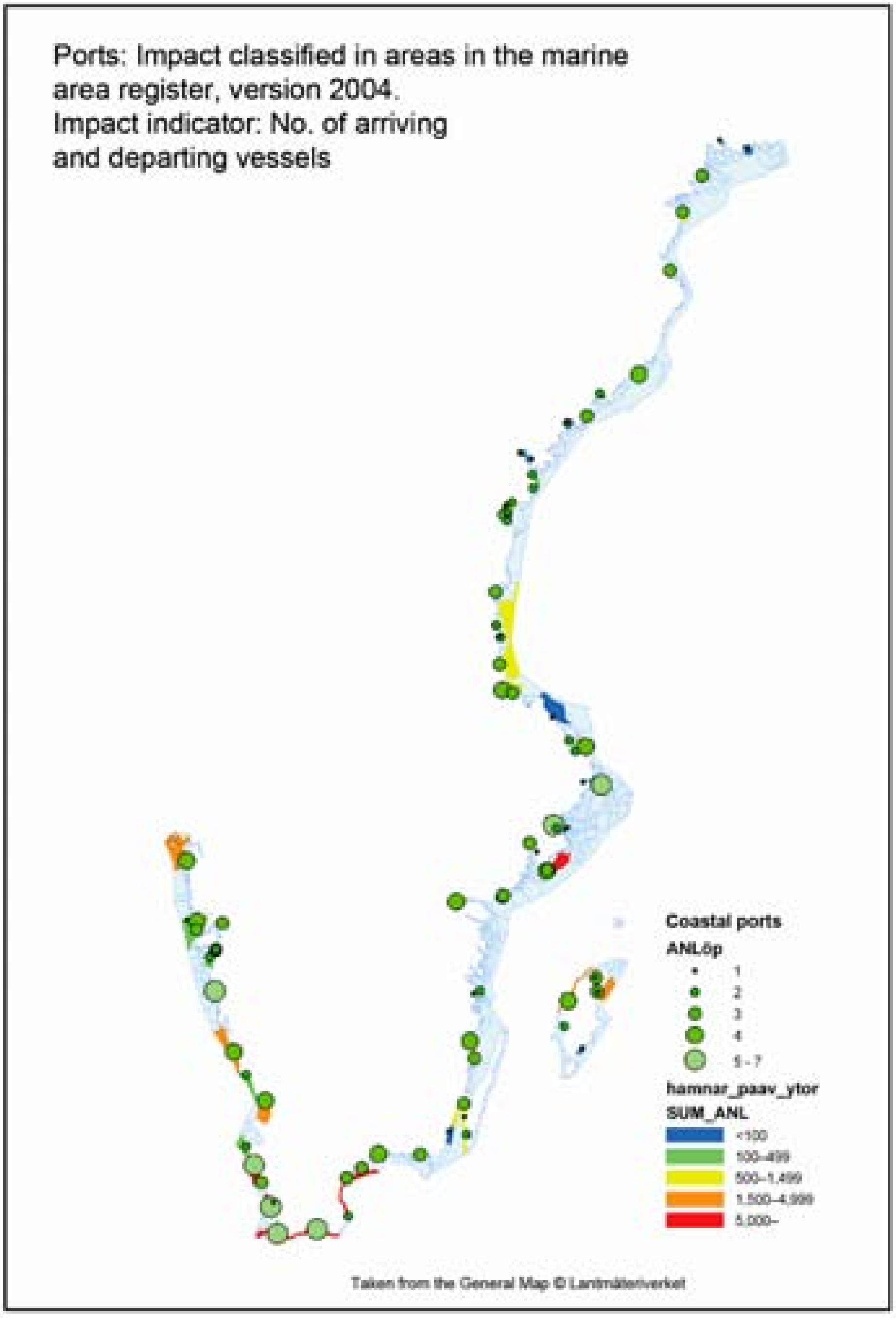

Figure 6.5. Classification of coastal water areas in the 2004 version of the SMHI marine areas register. The classification is based on statistics on the number of visits by vessels to ports in accordance with Table 6.3. Ports that are associated with the water areas are represented by differently sized points according to the classes for the number of visits by vessels defined by Statistics Sweden. Source: SMHI