EUROPEAN COMMISSION
DIRECTORATE-GENERAL
Joint Research Centre

# Relationships between pressures, chemical status, and biological quality elements

# Analysis of the current knowledge gaps for the implementation of the Water Framework Directive

**Edited by**

**Anna-Stiina Heiskanen and Angelo G. Solimini**

2005 EUR 21497 EN

EUR 21497 EN

Printed in Italy

## SUMMARY

The general objective of the REBECCA project[1] is to provide relevant scientific support for the implementation of the Water Framework Directive (WFD). The two specific aims of the project are, firstly, to establish links between ecological status of surface waters and physico-chemical quality elements and pressures from different sources, and, secondly, to develop and validate tools that member states can use in the process of classification, in the design of their monitoring programs, and in the design of measures in accordance with the requirements of the WFD.

Historically, there has been great success in maintaining and improving the quality of surface waters by developing an understanding of the links between anthropogenic pressures (e.g. water abstraction, agriculture, and effluent discharges) and the chemical status of waters, although there remain many challenges in reliably designing and implementing the necessary programs of measures. Our present understanding of the link between chemical properties and ecological state, while good in some instances, is generally not adequate to support management intervention against ecological objectives.

In this report we review and identify information gaps in our knowledge on relations between pressures, chemical and ecological status for the major pressures types and biological quality elements. We also give an overview of the chemical parameters that are used to determine the ecological status of water body types and of the biological indicators currently applied and/or potentially applicable as classification parameters for inland and coastal waters.

This gap-analysis is needed to 1) identify the key areas of further work within the REBECCA project and 2) to identify the areas where further experimental or monitoring work would be needed (beyond the scope of REBECCA), due to lack of data or quantitative understanding of the functional relationships between chemical status and biological quality indicators. This report should help in focusing the on-going WFD intercalibration process in 2005-6. In particular it should provide insights on which biological and pressure parameters should be selected and which data there would be available to illustrate the degradation of the biological quality with respect of pressure gradients.

Regarding data availability to Rebecca project, major gaps were identified as follow:

- Data from the Mediterranean and alpine lake types
- Data from large part of the Mediterranean coastal & transitional types (with exception of Italy)
- Data on concentrations of toxic chemical substances in combination with biological quality element indicator data
- Macrophyte and benthic invertebrate data from many lake & coastal types
- Fish data from lakes (and transitional waters – not included in REBECCA)

Apart from these data gaps, there are common “knowledge gaps” in relation to tasks required by the WFD that exist for all surface water categories:

- Development of reference conditions
- Development of type-specific classifications
- Criteria for setting class boundaries
- Criteria for setting ecological thresholds

---

[1] http://www.environment.fi/syke/rebecca

- Supporting element classifications related to biological impact
- Relationships between nitrogen conditions and ecological responses
- Estimations of uncertainty in classifications
- Uncertainty in measured data
- Responses to combined pressures

Additionally, a summary of specific knowledge gaps for each combination of pressure and quality element is given in the following pages for inland and marine water ecosystems.

Lake ecosystems: summary of major knowledge gaps for pressure and quality elements.

| **Pressure** | **Element** | **Knowledge gap** |
|---|---|---|
| Eutrophication | *Phytoplankton* | Threshold concentrations for high/good and good/moderate boundaries<br>Taxonomic indicators for measuring impacts of nutrient pressures<br>Establishment of supporting physico-chemical conditions<br>Reference conditions in different lake types<br>Effect of seasonal variability on classification schemes<br>Ecological impact of nitrogen conditions |
| | *Macrophytes* | Relationships that distinguish effects of nutrients from effects of other variables<br>Type-specific reference conditions and classification schemes<br>Assessment of spatial and temporal variability |
| | *Phytobenthos* | Absence of classification schemes and quantitative relationships among phytobenthos and nutrients |
| | *Benthic invertebrates* | Relationships between level of oxygen depletion and taxonomic composition |
| | *Fish* | Standard fish indicators based for different lake types |
| | | Relationships between duration of minimum oxygen-concentrations and different fish indicators |
| Hydromorphological | *Macrophytes* | Confounding effects of site specific variability, including sediment and water quality |

Lake ecosystems: summary of major knowledge gaps for pressure and quality elements (continue from previous page).

| **Pressure** | **Element** | **Knowledge gap** |
|---|---|---|
| Euthrophication-hydromorphological | *Macrophytes* | Phytoplankton-macrophytes competition under fluctuating water level<br><br>How to account for temporal variability |
| Euthrophication-acidification | *Macrophytes* | Identification of quantitative indicators and classification schemes |
| Euthrophication-toxics | *Benthic invertebrates* | Identification of quantitative indicators and classification schemes |
| Acidification | *Phytoplankton, macrophytes, benthic invertebrates* | Effects of acidification needs to be separated by the effect of other co varying or interacting variables including nutrients and biotic interactions<br><br>Effect of acidification episodes |
| | *Fish* | Relationships among water parameters related to acidification and fish based indicators |

River ecosystems: summary of major knowledge gaps for pressure and quality elements.

| **Pressure** | **Element** | **Knowledge gap** |
|---|---|---|
| Eutrophication | *Phytoplankton* | Relationships among nutrient concentration and blooms |
| | *Macrophytes* | Response of macrophytes to nutrient in the sediments<br>Type specific applicability of empirical models<br>Effect of co varying variables<br>Development of quantitative indicators |
| | *Phytobenthos* | How to include filamentous algae in assessment schemes<br>Type specific applicability of empirical models<br>Effect of co varying variables<br>Development of quantitative indicators |
| Organic pollution | *Phytobenthos, macrophytes, benthic invertebrates* | Functional relationships among organic matter concentration and biota<br>Impact of different types of wastewater on biota<br>Response of biota around high/good and lower end boundaries<br>Effect of the interaction among organic pollution and discharge<br>Response of biota to urban and agricultural run offs |
| Acidification | *Phytoplankton, macrophytes, benthic invertebrates* | Effects of acidification needs to be separated by the effect of other co varying or interacting variables including nutrients and biotic interactions<br>Effect of acidification episodes |
| Hydromorphological | *Benthic invertebrates* | Effect of covaring variables |

River ecosystems: summary of major knowledge gaps for pressure and quality elements (continue from previous page).

| **Pressure** | **Element** | **Knowledge gap** |
|---|---|---|
| Combined pressures | *all* | Influence of spatial scales, including large (ecoregions), in predicting biological indicators outcome<br><br>Inter-regions comparative studies taking into account social and economic features |

Coastal ecosystems: summary of major knowledge gaps for pressure and quality elements

| **Pressure** | **Element** | **Knowledge gap** |
|---|---|---|
| Eutrophication | *Transparency* | Quantitative relationships among nutrient loading and transparency for certain types |
| | *Phytoplankton* | Spatial and temporal variability of nutrient limitation pattern<br><br>Taxonomic indicators for measuring the impact of loadings<br><br>Elucidating the role of nutrients and other variables in starting harmful algal blooms |
| | *Macrophytes* | Robust quantitative relationships among water quality and vegetation indicators<br><br>Quantify the effects of stochastic factors on macrophyte ecological attributes<br><br>Threshold concentrations for eutrophication |
| | *Benthic invertebrates* | Taxonomic indicators for measuring the impacts<br><br>Effects of sediment characteristics and organic matter content |

**Table of Contents**

**List of authors**

| | |
|---|---|
| 1. INTRODUCTION | Anna-Stiina Heiskanen, Angelo Solimini (JRC) |
| 2. LAKES | |
| 2.1. Eutrophication pressures | Laurence Carvalho, Lindsey Defew (CEH) |
| 2.1.1. Nutrient conditions | Laurence Carvalho, Lindsey Defew (CEH) |
| 2.1.2.Transparency conditions | Johanna Rissanen, Olli-Pekka Pietiläinen, Seppo Rekolainen (SYKE) |
| 2.1.3. Oxygen conditions | Orhan Ibram (DDNI) |
| 2.1.4. Phytoplankton | Laurence Carvalho (CEH) Johanna Rissanen, Olli-Pekka Pietiläinen, Seppo Rekolainen (SYKE) |
| 2.1.5. Macrophytes | Iain Gunn, Laurence Carvalho, Lindsey Defew (CEH) |
| 2.1.6. Phytobenthos | Laurence Carvalho (CEH) |
| 2.1.7. Benthic Invertebrates | Orhan Ibram (DDNI) |
| 2.1.8. Fish | Lindsey Defew (CEH), Ion Navodaru (DDNI) |
| 2.2. Hydro-morphological pressures | Seppo Hellsten, Juha Riihimäki |
| 2.2.1. Macrophytes | |
| 2.3. Combined pressures | |
| 2.3.1. Eutrophication – acidification – macrophytes | Lindsey Defew, Laurence Carvalho (CEH) |
| 2.3.2. Eutrophication – toxics - benthic invertebrates | Constanze O'Toole (TCD) |
| 2.3.3. Eutrophication – hydromorphological pressures - macrophytes | Seppo Hellsten, Juha Riihimäki (SYKE) |
| 3. ACIDIFICATION PRESSURES FOR LAKES AND RIVERS | Frode Kroglund (NIVA) |
| 3.1. Toxic components of acidification pressure | |
| 3.2. Phytoplankton | |
| 3.3. macrophytes | |
| 3.4. Benthic Invertebrates | |
| 3.5. Fish | |
| 4. RIVERS | |
| 4.1. Eutrophication pressures | Amelie Deflandre (CEH) |
| 4.1.1. Phytoplankton | |
| 4.1.2. Macrophytes | |
| 4.1.3. Phytobenthos | |
| 4.1.4. Benthic Invertebrates and fish | |
| 4.2. Organic pollution pressures | Jens Moller Andersen (NERI), Stefano Fazi (IRSA) |
| 4.2.1. Phytobenthos | |
| 4.2.2. Macrophytes | |
| 4.2.3. Benthic invertebrates and fish | |
| 4.3. Hydromorphological pressures – benthic invertebrates | Bente Clausen (NERI), Mike Dunbar (CEH) |
| 4.4 Combined pressures | Ana Garcia, Jean-Gabriel Wasson (CEMAGREF) |
| 5. COASTAL WATERS | |
| 5.1. Eutrophication pressures | Pirjo Kuuppo (SYKE) |

| | |
|---|---|
| 5.1.1.Transparency conditions | Gunni Ærtebjerg (NERI), Anouk Blauw (WL Delft) |
| 5.1.2. Phytoplankton | Pirjo Kuuppo (SYKE), Anna-Stiina Heiskanen (JRC) |
| 5.1.3. Macrophytes | Dorte Krause-Jensen (NERI), Saara Bäck (SYKE), Paul Erftemeijer (WL Delft) |
| 5.1.4. Benthic Invertebrates | Miguel Gaspar, Susana Carvalho (IPIMAR) |
| 6. TOXIC PRESSURES | Lena Blom, Eva Brorström-Lundén (IVL) |

**Authors' affiliations**


| | |
|---|---|
| Finnish Environment Institute (SYKE) | P.O. Box 140, FIN-00251 Helsinki Finland |
| European Commission Joint Research Centre (JRC) | Institute for Environment and Sustainability, Inland and Marine Waters Unit, TP290, 21020 Ispra, Italy |
| Norwegian Institute for Water Research (NIVA) | Brekkeveien<br>P.O.Box 173,<br>N-0411 Oslo, Norway |
| National Environmental Research Institute (NERI) | Vejlsøvej 25<br>DK 8600 Silkeborg, Denmark |
| DHI Water & Environment (DHI) | Agern Allé 5<br>DK-2970 Hørsholm<br>Denmark |
| WL Delft Hydraulics (WL Delft) | P.O. Box 177<br>NL-2600 MH Delft, The Netherlands |
| Centre for Ecology and Hydrology (CEH) | Bush Estate<br>Penicuik<br>EH26 0QB<br>Scotland |
| Centre National du Machinisme Agricole, du Gènie Rural, des Eaux et des Fōrets (CEMAGREF) | Groupement de Lyon, UR Biologie des Ecosystemes Aquatiques Laboratoire d'Hydroecologie Quantitative<br>3 bis quai Chauveau, BP 220,<br>69336 Lyon Cedex 09, France |
| Water Research Institute (IRSA) | 1, via Reno<br>00198 Rome, Italy |
| Swedish Environmental Research Institute (IVL) | Box 47086<br>SE-402 58 Gothenburg, Sweden |
| Trinity College Dublin (TCD) | Department of Zoology<br>Trinity College,<br>Dublin 2, Ireland |
| Instituto Nacional de Investigação Agrária e das Pescas (IPIMAR) | Avenida de Brasilia 1449-006,<br>Lisboa, Portugal |
| Danube Delta National Institute<br>(DDNI) | Babadag Street, 165<br>Tulcea , 8800<br>Romania |

# 1 INTRODUCTION

*Anna-Stiina Heiskanen and Angelo G. Solimini*

The general objective of the REBECCA project is to provide relevant scientific support for the implementation of the Water Framework Directive (WFD). The two specific aims of the project are, firstly, to establish links between ecological status of surface waters and physico-chemical quality elements and pressures from different sources, and, secondly, to develop and validate tools that member states can use in the process of classification, in the design of their monitoring programs, and in the design of measures in accordance with the requirements of the WFD.

In this report we review and identify information gaps in our knowledge on relations between pressures, chemical and ecological status for the major pressures types and biological quality elements. This included overview of the chemical parameters that are used to determine the ecological status of water body types and an overview of the biological indicators currently applied and potentially applicable as classification parameters for inland and coastal waters. The steps carried out for this task included 1) literature search (scientific & other publications), 2) overview of results from the relevant FP5 projects, and 3) national research projects, 4) overview of the knowledge gaps and information from CIS guidance documents, 5) compilation of the knowledge base of the project participants.

This document is highlighting the key points for focusing on key pressures (eutrophication, acidification, organic loading and toxic pressures), and the justifications for the selection of biological quality elements and parameters/ metrics for the further studies. The purpose of this report is to present the major gaps in the current knowledge with respect of functional relationships between key biological and pressure indicators and/ or chemical status indicators. This gap-analysis is needed to 1) identify the key areas of further work within the REBECCA, 2) identify the areas where further experimental or monitoring work would be needed due to lack of data or quantitative understanding of the functional relationships between chemical status and biological quality indicators (but which are beyond the scope, since only existing data will be used in REBECCA).

We have avoided extensive referencing to scientific literature in order to keep the report easily readable and allow quick screening of the major findings of this survey. Therefore only the most relevant references are included and listed in the bibliography. This document is not intended to be a scientific paper, but a working document, which the project partners and associated end-users within the WFD Common Implementation process (i.e. the members of the working group 2A on ecological status, and the national contact persons for the intercalibration of the ecological status classification protocols in EU and Candidate Countries) can use in the planning of their further work on data compilation and statistical analyses.

The current review is based on extensive literature search of several contributing partners (listed in the table below). The literature review produced a reference database with almost 7000 scientific references. Already this database is based on selective screening of the vast amount of references that were obtained by the project partners when querying various literature databases and the internet. Further screening of the key articles took place upon the review process, which all partners carried out while producing a condensed summary for each major pressure group and each relevant biological quality elements.

This report should help to focus the on-going work on planning of the WFD intercalibration process in 2005-6, especially on evaluation which biological and pressure parameters should be selected and which data there would be available to illustrate the degradation of the biological quality with respect of pressure gradients, as proposed to be examined by the

Geographical Intercalibration Groups (GIGs) [2] of the EU Member States and Candidate Countries. We hope that the work of the scientific partners in the REBECCA project would have a direct applicability in the current challenges in the implementation of the WFD, and therefore we feel that we have a unique opportunity to inject policy relevant scientific results in the on-going implementation process.

[2] See ECOSTAT 2004, *Guidance on the Intercalibration process*, for clarification of role and tasks of GIGs in the WFD intercalibration process. http: http://forum.europa.eu.int/Public/irc/env/wfd/library

# 2 LAKES

## 2.1 Eutrophication pressures

*Laurence Carvalho, Lindsey Defew*

Although the impact of nutrient pressures on biological quality is relatively well understood for lakes in qualitative terms, there has been very limited development of quantitative dose-response relationships, classification tools or models. The literature review identified a number of widely recognized, strong relationships between physico-chemical symptoms of eutrophication and associated biological responses. These can be split into two types of relationships: (1) primary responses (phytoplankton, phytobenthos and macrophytes) to nutrient state, and (2), secondary responses to primary production or production-related decreases in transparency and oxygen. The advantages and disadvantages associated with developing these elements as indicators of eutrophication pressures are briefly summarized below.

### *(1) Primary responses to nutrients*

- Phytoplankton abundance (chlorophyll a) – highly sensitive to nutrients and integrates the response to a range of possible limiting nutrients (P, N, Si). It is also a simple, practical measure of eutrophication impacts that is widely adopted across the EU Member States
- Phytoplankton composition - sensitive to nutrients and qualitatively well understood. Sediment records available for defining reference conditions. The response is not so well understood in quantitative ways and is complicated by the weather-driven physical structure of lakes
- Macrophyte abundance – sensitive to nutrients, but also to reduced light availability. Unimodal relationship due to nutrient stimulation of growth and reduced light availability in highly eutrophied lakes. Two stable states occur in eutrophic lakes (Scheffer et al. 1993), due to differences in food web structure: High macrophyte abundance, clear water, much zooplankton grazing, low fish predation on zooplankton representing a top-down controlled phytoplankton community, or alternatively high phytoplankton, low macrophyte system with turbid water, low zooplankton grazing and high fish predation on zooplankton, representing a bottom-up controlled system.
- Macrophyte composition - sensitive to nutrients and qualitatively well understood. Data-rich historical records available for defining reference conditions in some areas. The response to nutrients is, however, affected by light conditions, substrate quality and water level fluctuations (hydro-morphological pressures), and the combined effects of these factors with eutrophication is not well described in quantitative ways.
- Increased phytobenthos biomass

### *(2) Secondary responses to nutrients*

- Transparency - simple, practical measure of eutrophication impacts that is widely adopted across EC Member States, but affected by water colour and clay particles.
- Benthic invertebrate composition – oxygen depletion: sensitive to eutrophication pressures
- Fish kills caused by oxygen depletion

- Fish composition – changes caused by oxygen depletion: qualitatively well understood; different fish species have different sensitivity.

All these primary and secondary responses were recognized to be significant, but data are limited for some of these parameters compared with others, which puts practical limits on their selection as relationships to examine during this project. In particular, data on phytobenthos and nutrients, benthic invertebrates, fish and oxygen conditions in lakes are limited.

As the REBECCA project does not involve primary data collection, data gaps prevent analysis of some relationships in the project, although some relationships are being examined in projects in individual member states concurrently (e.g. phytobenthos-nutrients).

For these reasons, it is recommended that the following eutrophication-specific relationships in lakes are considered for further study in WP3 of the REBECCA Project (Table 2.1.1):

Table 2.1.1. Eutrophication specific relationships proposed to be studied for lakes in the REBECCA project.

| Pressure/ chemical status indicator | Biological response parameter |
|---|---|
| nutrient concentrations | phytoplankton abundance (biomass and chlorophyll a) |
| nutrient concentrations | phytoplankton composition |
| nutrient concentrations | macrophyte composition |
| nutrient concentrations | lower growth limit of macrophytes |
| nutrient concentrations | fish biomass |
| transparency | phytoplankton abundance (biomass and chlorophyll a) |
| transparency | lower growth limit of macrophytes |
| transparency | macrophyte composition |
| nutrient concentrations and transparency combined | macrophyte abundance |
| oxygen concentrations | fish composition (if data permits) |
| oxygen concentrations | benthic invertebrate composition (if data permits) |

In addition to the eutrophication-specific relationships, the response of a number of biological quality elements to four combinations of pressures should be examined (Table 2.1.2. and section 2.2):

Data is likely to be limited for examining dose-response relationships for combined pressures. It may, therefore, be more appropriate to understand how eutrophication

responses are modified when influenced by a second pressure through the use of generalized lake models, experimental data or long-term monitoring at case-study sites.

The latest CIS guidance on ecological classification (ECOSTAT 2003) recommends a "one-out, all-out" rule on the level of different quality elements (i.e. phytoplankton, macrophytes, benthic invertebrates, etc.) for determining the ecological status of individual sites. It is proposed to combine different parameters responding to the same pressure, within the same quality element using either averaging or some multimetric approach. Several biological parameters (such as biomass, species composition, frequency of blooms) indicative of the effects of a particular type of pressure (such as eutrophication), but representing one quality element (such as phytoplankton), may be identified and grouped (e.g. by averaging of using multi-metric methods). The ecological status of the site would then be indicated by the results indicative for the status of the biological quality element (BQEs) that indicates the greatest impact (see Figure 3, ECOSTAT 2003)). The REBECCA Project should, however, examine a number of relationships for each pressure, to decide objectively which should be selected, or how to combine those parameters to produce the most robust classification.

Table 2.1.2. Responses of biological quality elements for combination of pressures proposed to be studied for lakes in the REBECCA project.

| Pressure 1 | Pressure 2 | Biological response parameter |
|---|---|---|
| Eutrophication | water level fluctuations | macrophyte abundance and composition |
| Eutrophication | retention time | phytoplankton abundance (chlorophyll a) |
| Eutrophication | toxic pollutants | benthic invertebrates |
| Eutrophication | Acidification | Macrophyte abundance and composition |

Nutrients tend to be a secondary and indirect driver of species composition but are clearly a significant driver of productivity. One aim for the REBECCA project, therefore, will be to examine whether the relationship between nutrients and phytoplankton abundance provides a more robust biological measure of the impact of eutrophication pressures than the relationship between nutrients and phytoplankton composition.

Data gaps may exist for particular lake types. Coverage within project partners involved in WP3 is particularly limited for Mediterranean and alpine lake types. Developing dose-response relationships for these will rely on data being provided from elsewhere (e.g. Central, Baltic and Mediterranean Geographical Intercalibration Groups, ECOFRAME Project for very shallow lakes, EMERGE Project for alpine lakes).

### 2.1.1 Nutrient conditions

*Laurence Carvalho, Lindsey Defew*

In most freshwaters, phosphorus has for many years been believed to be the principal resource limiting the productivity of the system. As such, an OECD (1982) classification scheme relating total phosphorus concentrations to chlorophyll *a* and transparency (Secchi depth) was developed and has become widely accepted (Table 2.1.3).

This classification is based on data from a large number of lake types from lake regions across the globe. Lake depth, color and altitude/latitude (geology/temperature) will affect how effectively nutrients are transformed into phytoplankton biomass; different lakes will, therefore, show variable sensitivities to nutrient conditions.

Some of these issues are being overcome through the development of WFD-compliant state-changed phosphorus classification schemes, based on deviation from a site-specific reference phosphorus concentration derived using empirical models (e.g. Morpho-edaphic Index; Vighi and Chiaudani 1985) or palaeolimnological approaches.

Table 2.1.3: OECD classification scheme for lake trophic status (based on OECD 1982)

| | Annual mean Total Phosphorus (µg/l) | Annual mean Chlorophyll a (µg/l) | Annual maximum Chlorophyll a (µg/l) | Annual mean Secchi depth (m) |
|---|---|---|---|---|
| Ultra-oligotrophic | <4 | <1 | <2.5 | >12 |
| Oligotrophic | <10 | <2.5 | <8 | >6 |
| Mesotrophic | 10-35 | 2.5-8 | 8-25 | 6-3 |
| Eutrophic | 35-100 | 8-25 | 25-75 | 3-1.5 |
| Hyper-eutrophic | >100 | >25 | >75 | <1.5 |

The use of orthophosphate-phosphorus ($PO_4$-P) and soluble reactive phosphorus (SRP) concentrations for the assessment of eutrophication is problematic. Low concentrations do not necessarily indicate low trophic status, as phosphorus may be bound to biomass. Sampling and storage of samples before analysis can also significantly alter concentrations.

It has been recognized for some time that some lakes may be limited by nitrogen (Sakamoto 1966). The capacity of some cyanobacteria (blue-green algae) to fix atmospheric nitrogen was, however, viewed as a mechanism that would overcome any incipient or transient nitrogen-limitation (Schindler 1974). For this reason, there are no well-established lake classification schemes relating nitrogen conditions to ecological impacts. While it is almost certainly true that phosphorus is the resource limiting many freshwater systems, there is growing evidence that this paradigm is not correct for all systems, with both nutrient-poor upland lakes and phosphorus-enriched lowland lakes liable to nitrogen limitation for at least parts of the year.

The relationships between nitrogen conditions and ecological responses are a major knowledge gap that needs further consideration.

### 2.1.2 Transparency conditions

*Johanna Rissanen, Olli-Pekka Pietiläinen, Seppo Rekolainen*

Transparency or clarity of water is the depth that light penetrates into the water. There are three factors that influence transparency, the water itself, coloring matter in the water, and turbidity associated with particles in the water. The amount of turbidity in particular is heavily influenced by a range of variables, natural and anthropogenic, and can vary both temporally and spatially within a given water body. Suspended organic and inorganic matter, such as silt, clay, carbonate particles, fine organic particulate matter, plankton and other small organisms contribute to turbidity (Wetzel and Likens 1991). Transparency and turbidity are inversely related, with an increase in turbidity resulting from human activity, always resulting in a decrease in transparency. Transparency is a major factor controlling photosynthesis potential of phytoplankton and macrophytes in the water column. Besides many ecological effects, reduced transparency has a clear social impact through impairment in physical appearance and recreational suitability of surface waters.

### *WFD –associated characteristics*

The WFD stipulates that in the high ecological status the conditions for physico-chemical quality elements (including transparency) do not show signs of anthropogenic disturbance and remain within the range normally associated with undisturbed conditions. In the good status, physico-chemical parameters (including transparency) do not reach levels outside the range established so as to ensure the functioning of the ecosystem and the achievement of the values specified for the biological quality elements.

According to these normative definitions of the WFD, transparency can be used only as a supporting element for assessing ecological quality in surface waters, although in some surface water body types it may be indicative for the level of eutrophication.

## 2.1.3 Oxygen conditions

*Orhan Ibram*

Eutrophication is associated with an increased mineral nutrient supply, mainly nitrogen and phosphorus, which usually results in an increase in primary production (e.g. Sas 1989). This eventually leads to an increase in sedimentation of settling dead plankton (e.g. Baines and Pace 1994), which results in associated increases in decomposition by micro-organisms. Increased decomposition is accompanied by enhanced oxygen consumption by the micro-organisms, which can result in oxygen depletion, particularly in the bottom sediments and profundal zone of deep lakes. This indirect effect of eutrophication on oxygen conditions can lead to significant changes in the structure of oxygen dependent communities, either as a change in dominant species with mild enrichment or severe shifts in species composition or fish kills with more severe enrichment. The most sensitive communities to changes in oxygen conditions are animal communities, and therefore, only the relationships between benthic invertebrates and fish with oxygen conditions are reviewed in the following sections.

## 2.1.4 Phytoplankton

*Laurence Carvalho, Johanna Rissanen, Olli-Pekka Pietiläinen, Seppo Rekolainen*

The phytoplankton community is widely considered the first biological community to respond to eutrophication pressures and is the most direct indicator of all the Biological

Quality Elements (BQEs) of the state of a lake's environment in terms of nutrient concentrations in the water column. There are numerous socio-economic problems associated with increases in phytoplankton abundance with eutrophication, particularly with increasing frequencies and intensities of toxic cyanobacteria blooms. These include detrimental effects on drinking water quality, filtration costs for water supply (industrial and domestic), water-based activities and conservation status (sensitive pelagic fish species, such as salmonids and coregonids).

In some contexts, however, increasing phytoplankton abundance can be considered as a positive feature, for example, in increasing fisheries productivity. In summary, the phytoplankton community probably represents the most sensitive indicator of the environmental, social and economic impacts associated with eutrophication pressures.

Transparency/ Secchi depth is widely used as an indirect or surrogate estimate of the amount of phytoplankton or chlorophyll *a* and, thus, as an indicator of the eutrophication state of surface waters. Non-algal turbidity and color (mainly dissolved organic matter humic compounds and suspended solids) can also significantly attenuate light penetration into the water and affect the Secchi depth vs. phytoplankton/ chlorophyll *a* relationship.

### *WFD –associated characteristics*

WFD (Annex V, Section 1.2.2) outlines three phytoplankton-related quality elements and parameters that need to be considered in the assessment of ecological status of lakes:

- Phytoplankton composition and abundance
- Phytoplankton biomass and its effect on transparency conditions
- Planktonic bloom frequency and intensity

The normative definitions of the WFD associate declining ecological quality with increasing phytoplankton abundance and biomass due to accelerated growth of algae, possibly having an effect on other biological quality elements and physico-chemical quality of water, as well as with more frequent and intense phytoplankton blooms. These three criteria are considered further in the context of their specific use as metrics of eutrophication pressures.

Transparency is mentioned in Annex V of the WFD as a general physico-chemical factor supporting the biological elements. In the normative definitions of ecological status classifications for the biological elements (Annex V, 1.2.2) it is stated that, under high status conditions, the average phytoplankton biomass is consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions.

### *Knowledge gaps*

There are a number of gaps in our scientific knowledge that need further research for developing quantitative metrics associating phytoplankton with nutrients:

- Reference chlorophyll concentrations need to be identified for different lake types using empirical and/or mechanistic models.
- Threshold chlorophyll concentrations for high/good and good/moderate boundaries need to be identified – with associated confidence limits.
- Empirical and predictive models need to be developed to relate phytoplankton composition to nutrient conditions. Limited information on reference values for phytoplankton composition may be obtained from lake sediment records, although

this will be restricted to the diatom community. The ratio of negative to positive indicator species or higher taxonomic units should be developed and the use of functional groups should be explored to reduce uncertainty in classifications.

- The effect of season/weather-driven dynamics and sampling frequency on the confidence in phytoplankton classifications needs to be quantified for different lake types.
- Supporting physico-chemical conditions need to be established (particularly for phosphorus, nitrogen, transparency and oxygen conditions).
- Relationships between nutrients and transparency, and also between transparency and biological indicators have not been analyzed separately for different lake types the relationships between transparency and biological indicators, except chlorophyll *a*, have not been extensively analyzed, the role of non-algae light attenuation is not very well known, which is particularly important in certain lake types (humic and turbid waters).

### *2.1.5* Macrophytes

*Iain Gunn, Laurence Carvalho, Lindsey Defew*

The macrophyte community is generally regarded as a key indicator of the ecological status of lakes as macrophytes provide habitat for many other aquatic biota (e.g. fish, zooplankton, macro-invertebrates, wetland birds) to feed, seek refuge, or breed. Macrophytes are relatively long-lived organisms (months to years), compared with phytoplankton and invertebrates, which, because of their very limited motility, are intrinsically linked to the prevailing environmental conditions in both the surrounding lake water and sediments, through their roots and leaves.

Individual species are sensitive to physical and chemical changes in these media and hence make good indicators of both current environmental conditions and longer-term environmental changes. Macrophytes, in terms of assessing eutrophication pressures, can indicate enhanced nutrient concentrations through the direct effects on species growth (biomass) and through indirect effects on species composition, such as decreased transparency associated with nutrient-related increases in phytoplankton and epiphyton.

Transparency is a key physico-chemical factor controlling the distribution and abundance of submerged macrophytes in lakes. Changes occurring in the intensity and quality of light as it passes down through aquatic vegetation itself may exert a crucial effect on the development and structure of aquatic macrophyte communities, as well as on the photosynthetic efficiency and productivity of the vegetation (Sculthorpe 1967).

Aquatic macrophytes are influenced by the transmission of light (the photosynthetically active radiation (PAR) through water, which, in turn, impacts photosynthetic processes. In turbid conditions suspended particles scatter and absorb light resulting in attenuation of PAR in the water column. High turbidity may cause a change macrophyte communities from submerged to floating leaved or emergent vegetation (Hough and Forwall 1988).

Little is known of the influence of underwater light intensity and quality on the distribution of life forms and species of aquatic macrophytes. It may be that some plants may tolerate, or even prefer sustained low intensities and/or deficiencies in certain wavelengths.

#### *WFD –associated characteristics*

WFD (Annex V, Section 1.2.2) outlines two macrophyte-related quality elements that need to be considered in the assessment of ecological status of lakes:

- macrophyte community composition
- macrophyte abundance

The normative definitions in the WFD associate declining ecological quality with changes in the abundance and composition of macrophytic taxa, and indication of accelerated plant growth due to undesirable changes in the balance of organisms or physicochemical quality of water.

Transparency is included as a supporting physico-chemical element for macrophyte communities.

### *Knowledge gaps*

There are a number of gaps in our scientific knowledge of macrophyte-nutrient relationships that need further research:

- More studies on temporal and spatial variability of macrophytes communities are needed to improve classification schemes.
- Unclear what extent changes in macrophyte taxa are directly driven by nutrients, rather than by co-varying chemical, physical or biological parameters.
- It is unclear, particularly for shallow lakes, how appropriate linear approaches are for modelling temporal change at specific sites.
- Development of type-specific reference conditions for nutrients and macrophyte communities.
- Need a much greater geographical expansion of lake surveys, which provide detailed data on aquatic macrophyte populations and high quality nutrient chemistry to allow the assessment of nutrient effects within a particular lake type and the statistical importance of co-variables.
- Where high quality models exist for macrophyte assemblages at specific sites, attempts should be made to relate these to historical nutrient concentrations.

Whilst there has been some work done on surrogates for impacts of transparency/ turbidity, such as depth of the lower growth limit for macrophytes, currently there is not enough information to establish the responses of macrophyte species and communities to different levels of turbidity by which to establish reference conditions. Thus, there are a number of gaps in our scientific knowledge of macrophyte-transparency relationships that need further research:

- Establishment of transparency, turbidity, and colour measures for reference sites and key macrophyte species
- Reaction of macrophyte communities to changes in PAR.

## 2.1.6 Phytobenthos

*Laurence Carvalho*

The phytobenthos is an important component of primary production in lakes, although its contribution to overall primary production decreases with increasing lake depth. In deep lakes, the phytobenthos is, however, still an important component of littoral food webs, and will influence the structure of littoral macrophyte, invertebrate and fish communities.

Despite its potential significance, the phytobenthos has received relatively little attention in terms of its use as an indicator of lake quality. The fact, however, that the phytobenthos does respond to both water column nutrient concentrations and habitat quality, is accessible from the lake shore, and is less dynamic than the phytoplankton community has led to increasing interest in its use as a monitoring tool for lakes (US EPA 1998).

### *WFD –associated characteristics*

WFD (Annex V, Section 1.2.2) outlines two phytobenthos -related quality elements that need to be considered in the assessment of ecological status of lakes:

Phytobenthos composition and abundance

The normative definitions in the WFD associate declining ecological quality with increase of bacterial coats and tufts, which adversely affect phytobenthic communities.

### *Knowledge gaps*

- More work is needed to develop quantitative relationships between the phytobenthos community and nutrient conditions.
- No lake classification schemes based on phytobenthos composition or abundance have been fully developed.
- No models have been established associating phytobenthos composition or abundance in lakes to specific hydro-morphological or physico-chemical parameters.

## 2.1.7 Benthic Invertebrates

*Orhan Ibram*

The zoobenthos community are frequently used as environmental indicators of biological integrity because they are found in most aquatic habitats. They are a diverse and generally abundant group with a wide range of environmental tolerances and preferences. They therefore serve as a useful tool for detecting long-term environmental perturbation (Cairns and Pratt 1993) resulting from point and non-point sources of pollution. Because they play an essential role in key processes of lakes (food chains, productivity, nutrient cycling and decomposition) (Reice and Wohlenberg 1993), in principle, any environmental changes in lakes, for example in nutrient concentrations, should be reflected by changes in the structure of the benthic invertebrate community (Carvalho et al. 2002).

Profundal macroinvertebrates form an important link between detrital deposits and higher trophic levels in aquatic food webs (Brinkhurst 1974). Hypolimnion oxygen content, food quality and quantity, and water temperature are the main factors influencing the presence and biomass of benthic invertebrate species in lakes and reservoirs. The larvae of the midge genus *Chironomus* are common in the profundal zones of lakes and reservoirs around the world (Armitage et al. 1995), and the capacity of these insects to live at very low oxygen concentrations are well known.

Benthic communities respond differently to oxygen conditions because of many other factors influencing this parameter. Multiple regression analyses of reservoir data of Spanish lakes and reservoirs showed that *Chironomus* density decreased with depth and temperature and increased with alkalinity (Real 2000). Food availability has been postulated as a key factor for benthic communities, and the coupling of pelagic and benthic

production has been demonstrated in lake Esrom (Jonasson 1996) and lake Erken (Goedkoop and Johnson 1996). However in many freshwater bodies oxygen content has proved to be the limiting factor for the benthos, especially when the anoxic period lasts more than 4 months (Heinis and Davids 1993).

Zoobenthos community assessments are performed with the intention of detecting impairment in an aquatic ecosystem. Despite the linkage between bioassessment and water quality, there are surprisingly few examples of bioassessment used explicitly to support the development of numerical water-quality criteria (Ryan and Curtis 2003). One of the primary reasons for this is that traditional bioassessments are intentionally developed to capture the effect of a wide range of stressors to biological integrity. This lack of specificity results in ambiguity about the potential cause of impairment and, consequently, the levels of a stressor that may result in a threshold response.

Classification and monitoring schemes have focused on the profundal fauna of deep stratified lakes where environmental conditions are more uniform compared with shallow lakes or littoral areas of deeper lakes.

No classification scheme directly relates changes in benthic community structure to dissolved oxygen (DO) alone, for a number of reasons:

- Because of seasonal and diurnal variations in oxygen concentrations a large amount of measurements are needed with which to relate DO to benthic communities.
- The research for identifying minimum oxygen concentration is limited and further research is needed in this direction.

### *WFD –associated characteristics*

WFD (Annex V, Section 1.2.2) outlines three zoobenthos -related quality elements and parameters that need to be considered in the assessment of ecological status of lakes:

- Composition and abundance of benthic invertebrate fauna.
- Ratio of disturbance sensitive taxa to insensitive invertebrate taxa;
- Diversity of invertebrate communities

The normative definitions of the WFD associate declining ecological quality with decreasing diversity and increasing dominance of a few disturbance insensitive taxa. To comply with ‘good ecological status’ the composition and abundance of invertebrate taxa, the ratio of disturbance sensitive taxa to insensitive taxa and levels of diversity should show only slight signs of alteration compared to the type-specific communities.

### *Knowledge gaps*

Pressure-specific classification schemes based on zoobenthos communities

Classification schemes based on littoral or shallow lake zoobenthos communities.

Targeted research is needed to identify minimum oxygen concentrations necessary to support characteristic benthic invertebrate communities of various lake types in order to assign values to high and good status classes.

## 2.1.8 Fish

*Lindsey Defew, Ion Navodaru*

Associating fish population health and water quality conditions has a long been a tradition. Due to their complex requirement, fish are sensitive indicators for habitat quality at various spatial scales. As consumers and/or top predators, they subsume the trophic conditions across the food chain. They also provide detailed information on the respective trophic level. Fish caught by the fisheries industry and during sport fishing have long been used as indicators of water quality. Fisheries are important producers of food and income; inland fish are often used for human consumption, depending on the countries tradition. Recreational and subsistence fisheries are widespread, but less frequently monitored by statistics. Commercial fisheries are also locally important, with export and trade markets existing for high value fish.

Dissolved oxygen (DO) is a defining and limiting parameter for fish, affecting survival, growth, spawning, swimming performance, larval development and migration behavior behavior (Sams and Conover 1969, Doudoroff and Shumway 1970). Since oxygen is vital to other aquatic biota, including invertebrate food for fish, the species composition and biomass of fish communities can also be affected indirectly by oxygen availability. The composition of the fish community at a specific site is the result of various factors including environmental factors such as temperature, oxygen, flow and nutrients and the anthropogenic stocking and removal of fish.

### *WFD –associated characteristics*

WFD (Annex V, Section 1.2.2) outlines three fish-related quality elements that need to be considered in the assessment of ecological status of lakes:

- fish composition
- fish abundance
- fish age structure

The normative definitions in the WFD associate declining ecological quality with: 1) changes in species composition and abundance; 2) changes in type sensitive species; and 3) changes in the age structure shows signs of disturbance as failure of reproduction or development of a particular species.

### *Knowledge gaps*

- Fish indicators for different lake types are lacking.
- There are no specific lake classification schemes based on different fish-based metrics and oxygen as a direct parameter, since assessing status through measurements of oxygen conditions is complicated by the fact that DO concentrations fluctuate at annual, seasonal and diurnal time scales. It is difficult to define a single DO concentration for a water body or section that will be true both spatially and temporally.
- Targeted research is needed to identify minimum oxygen concentrations necessary to support characteristic fish communities of various lake types in order to assign values to high and good status classes. For fish, it may be necessary to identify site-specific values, related to the characteristic fish assemblages of different lakes.
- Continuous oxygen monitoring in a number of lake sites should be considered in order to provide empirical data on minimum oxygen conditions and to provide information to establish optimal sampling strategies (periods / frequencies) for fish – oxygen relationships.

## 2.2 Hydro-morphological pressures

*Seppo Hellsten, Juha Riihimäki*

Hydromorphological quality elements for lakes are described as hydrological regime and morphological conditions in the WFD. High hydromorphological status refers totally or nearly totally undisturbed conditions, but in good and moderate status the values are only consistent with the achievement of the values specified above for the biological quality elements.

Hydrological regime and morphological conditions are divided in following quality elements:

- Hydrological regime: The quantity and dynamics of flow, level, residence time, and the resultant connection to groundwater.
- Morphological conditions: Lake depth variation, quantity and structure of the substrate, and both the structure and condition of the lakeshore zone.

WFD (Art. 4(3)) allows Member States to identify surface water bodies, which have been hydromorphologically altered by human activity as “heavily modified” under specific circumstances. If the specified uses of such water bodies (i.e. navigation, hydropower, water supply or flood defense) or the “wider environment” would be significantly affected by the hydromorphological changes (restoration measures) required to achieve good ecological status and if no other, technically feasible and cost-effective, better environmental options exist, then these water bodies can be designated as “heavily modified” and good ecological potential is set as an environmental objective. Heavily modifies waters are thoroughly studied in EU CIS working group for heavily modified waters, which has produced a guidance document[3] with a tool box and separate book summarizing several case studies (Kampa & Hansen 2004).

Hydrological regime in lakes is disturbed by human activities if lakes are used for water storage for hydropower generation, irrigation, flood defense regulation, or if they are used for navigation, and in some cases even for recreation. Hydropower effects are typical in northern and high altitude lakes, which usually are not impacted by other pressures, whereas e.g. pressures due to regulation for navigation and recreation purposes are often affecting lowlands lakes situated at densely populated areas.

Morphological alterations such as dams and weirs affect continuity of rivers situated downstream from lake. Especially flood protection constructions and drainage of flood plains have created embankments, which can significantly change morphology of lowland lakes. Generally large scale morphological alterations are more common in small lakes surrounded by agricultural areas and population centres.

Morphological and hydrological alterations mainly affect the uppermost littoral zone, although changes in retention time can indirectly change also the trophic status of a whole lake. Overall decrease of water level can significantly increase on resuspension of bottom sediments. It is difficult to separate the effects of hydrological regime and morphological conditions from each other.

### 2.2.1 Macrophytes

[3] WFD CIS Guidance Document No 4. Identification and Designation of Heavily Modified and Artificial Water Bodies Produced by Working Group 2.2 – HMWB. Available at: http://forum.europa.eu.int/Public/irc/env/wfd/library

*Seppo Hellsten, Juha Riihimäki*

As a consequence of littoral oriented changes, littoral organisms are most sensitive quality elements with respect to the existing physical alterations. Littoral benthic invertebrate fauna is the most relevant groups for the assessment of hydropower generation impacts in regulated lakes. However, there is very little information and data available of the composition and abundance of littoral fauna from lakes. There are more data of data available on aquatic macrophytes which are also highly sensitive for hydromorphological alterations. Thus this quality element would be more suitable for further investigations.

Macrophytes, growing at the littoral zone, are one of the key indicators of the impacts of hydro-morphological changes, such as water level fluctuations in lakes. Helophytes are growing in the uppermost littoral zone, whereas *Isoetids, Elodeids* and *Charids* are occupying deeper parts of the littoral zones of lakes. Even small changes in water level fluctuations may effect distribution and elevation of zones. Morphological changes caused by dredging or embankments have a profound impact on the composition of littoral vegetation.

### *Knowledge gaps*

- Hydromorphological background information is largely lacking. Morphological structure of the shoreline is not known very well and effects of these changes are unclear. Water level fluctuation is followed in many reservoirs, but natural lakes are not usually followed.
- The littoral zone is a highly variable environment and therefore the relationship between hydromorphological factors and aquatic macrophytes are often site-specific.
- Macrophyte species richness depends on water quality and sediment properties – this makes it difficult to distinguish separate impact relationships between macrophytes and hydromorphological-factors. Nutrient enrichment may also compensate degradation caused by fluctuating water level.

## 2.3 Combined pressures

### 2.3.1 Eutrophication – acidification – macrophytes

*Lindsey Defew, Laurence Carvalho*

The pressures of eutrophication and acidification combined (primarily through long-range transported N-compounds) are likely to be most significant in predominantly rain-fed, oligotrophic, low alkalinity lakes. Aquatic macrophytes are intrinsically linked to lake water and sediments through their roots and leaves and individual species are sensitive to physical and chemical changes in these media. Although these lakes are strictly P-limited for phytoplankton, the macrophytes get P through the sediments, and can thus often increase their growth due to the increased N-input from the catchment.

The isoetid aquatic vegetation plays a major role in the ecological functioning of low alkalinity lake ecosystems (Murphy 2002). Changes from isoetid macrophyte assemblages to more acid-tolerant species such as *Juncus bulbosus* and *Sphagnum* spp have been well documented in West European lakes as a consequence of the combination of acidification and eutrophication from enhanced atmospheric N deposition (Arts 2002).

Methods for assessing changes in macrophyte community in response to eutrophication and acidification pressures combined could include large spatial surveys, palaeoecological studies, and monitoring following experimental liming and re-acidification studies.

The pressures of eutrophication and acidification combined are most likely to be significant in predominantly rain-fed, oligotrophic, low alkalinity lakes. These lakes are poor in (calcium) bicarbonate and nutrients (Roelofs et al. 2002). Acidification and eutrophication allow changes in N, P and C budgets, with particularly apparent impacts on macrophyte community composition. Nutrient-poor, low alkalinity lakes can have highly diverse plant communities, in which isoetid species dominate (*Lobelia dortmanna*, *Littorella uniflora* and *Isoetes* spp.).

Over the 20th century, however, isoetid vegetation in lowland lakes has declined greatly and become threatened in many European countries and North America (Smolders et al. 2002), with particularly devastating declines in the Netherlands, northern Germany and Belgium (Arts 2002). This provides important warnings for other countries/regions, not least in the context of the EU Habitats Directive, and highlights the need to focus on macrophyte responses to combined eutrophication-acidification pressures.

The relationship between macrophytes and acidity in lakes has received relatively little attention in the scientific literature despite macrophyte assemblages showing clear changes over a broad gradient. These changes can often be related to acidity and other parameters, including nutrient availability. There have, however, been few attempts to develop relevant classifications for these two pressures acting together. For a better understanding of the successional differences within low alkalinity macrophyte assemblages, further research and cooperation is needed, in which vegetation, physico-chemical and atmospheric deposition data are integrated (Arts 2002).

*Knowledge gaps*

- Relationship between macrophytes and acidity in lakes
- Combined impacts of nutrients and acidification on macrophytes

### 2.3.2 Eutrophication – toxics – benthic invertebrates

*Constanze O'Toole*

The investigation of the coupled effect of eutrophication and contamination is at a very early stage with limited knowledge. The relationship has not yet been quantified and no specific benthic invertebrate indicators have been selected for lakes.

Benthic invertebrates form an important element in the biological diversity of lakes and are a crucial food resource, especially for fish. They provide essential links in the food chain and are necessary for the natural energy and nutrient flow through the system. The various benthic organisms have differing sensitivities to environmental stressors and are well suited for the use as bioindicators for water quality monitoring. Traditionally, the responses have been studied separately without a combined approach. However, there is evidence that the trophic state may influence the bioavailability and cycling of contaminants and the level of contamination may influence the primary production and indirectly the status of eutrophication (Skei et al. 2000).

The environmental problems caused by eutrophication and toxic substances are mostly investigated separately without an attempt to investigate their joint effect. Interactions

between these two pressure types are not sufficiently investigated, since studies indicate that the two impacts can have interactive processes. Those interactions may lead to responses in the biota that cannot be predicted from each pressure alone (Skei et. al., 2000).

*Knowledge gaps*

- Effects of the interactions between nutrients and toxic substances on biota
- Effects of the interactions between metals and other toxic substances and nutrient loading on biota.
- The impact of eutrophication on the interfaces between air/water and water/sediments, critical routes for the organic and inorganic pollutant input and cycling, need to be investigated.
- Furthermore, the effect of benthic fauna activity needs to be better understood to quantify the flux of both contaminants and nutrients in and out of the sediment (Skei et al., 2000).
- The combined effect of eutrophication and toxicants on abundance, structure, diversity and sensitivity of the benthic invertebrate communities in lakes needs to be further investigated and quantified to establish their indicator value and a suitable classification system. A few ecological models (e.g. AQUATOX, IFEM and CATS-5) that simulate the combined fate and effects of nutrients and certain contaminants exist and can be applied to the benthic fauna in lakes.
- No classification scheme for the combined effects of eutrophication and contaminants in lakes is available.

### 2.3.3 Eutrophication – hydromorphological pressures - macrophytes

*Seppo Hellsten, Juha Riihimäki*

Eutrophication and hydromorphological pressures are often impacting lowlands lakes and reservoirs, which are used intensively. Vegetation in the shallow turbid lakes is sensitive for water level alterations, as consequence of the potential increase of sediment resuspension. Also changes in the residence time can rapidly change nutrient concentrations in water bodies. According to the WFD the hydromorphological quality elements of lakes are hydrological regime and morphological conditions. High hydromorphological status refers totally or nearly totally undisturbed conditions, but in good and moderate status the values are only consistent with the achievement of the values specified above for the biological quality elements.

Macrophytes are sensitive to changes in light climate as discussed in the chapter 2.1.5. of this report. Therefore water level alterations influence directly resuspension of bottom sediments and occurrence of different macrophyte species resulting in harmful effects on the littoral ecosystems. The general chemical quality of water bodies is influenced by the variations in retention time, which has thus an indirect effect on phytoplankton growth and macrophytes. Impacts of the combined effects of hydromorphological and eutrophication pressures can be managed by manipulating water level and retention time, although intensive use of shore areas can make such manipulations difficult.

*Knowledge gaps*

- Ecological quality indicators that integrate seasonal and annual variability of macrophyte community dynamics
- Interactions and competition between phytoplankton and macrophytes in eutrophic waters impacted by hydromorphological changes.
- Nutrient enrichment may also compensate degradation caused by fluctuating water level. On the other hand highly modified barren littoral zone is often more sensitive for eutrophication compared to vegetated well-developed littoral zone.
- How to distinguish the habitat level effects from lake wide effects?

## 3 ACIDIFICATION PRESSURES FOR LAKES AND RIVERS

*Frode Kroglund*

Acidification implies a reduction in pH. The causes can be local (due to natural and human activities) or due to long transported pollution (acid rain). To interpret or define a water body status correctly, lowered pH due to anthropogenic causes must be separated from natural causes. As both can result in the same “pH-reduction”, possibly having the same effects on the biological community, a given pH level can be an indicator of impaired ecological status in one water body, while the same pH-level indicates normal conditions in another.

Acidification can be continuous or episodic. Episodes, due to their short duration, can cause changes in the biological community that are not detected in any traditional water chemistry monitoring program. The toxic components in water acidified by acid rain are $H^+$ and aluminum (Gensemer and Playle, 1999). Toxicity is modified by water calcium, ionic strength, organic content and temperature. Species changes are in part due to effects caused by the chemical changes, in part due to biological interactions. E.g., the presence or absence of fish can be more important in controlling the presence or absence of fish sensitive species than the changes in water quality (Appelberg *et al* 1993). Changes in the top-predator community (fish) will effect the species composition and abundance of benthic invertebrates, which will affect further phytoplankton composition. Cascading effects in the lower levels of the food web can be related to the composition of the fish community, but also to the acidification status. Furthermore, different fish species and their life stages have varying sensitivity. Strain dependent sensitivity is described for several fish species. The interaction between chemical changes and biological responses are relatively well described, but the sum of interacting components makes the development of simple dose response relationships more uncertain.

Acidification is currently on the decline, but even provided the Gothenburg protocol being implemented; acidification will continue to affect sensitive areas for decades to come (Stoddard et al., 1999). There is further a major time delay between reduced deposition, water quality recovery and biological restoration, where chemical restoration must precede biological recovery. The rate of biological recovery depends on whether there are refuges within the watershed, and on dispersal rates and species interactions (see articles on recovery within AMBIO, 32, 2003). Recovery is unlikely for several fish species without reintroduction due to migratory barriers that fish cannot pass. As presence or absence of specific fish species has a profound effect on the whole ecosystem structure, the reference conditions are also related to the composition and abundance of the fish community.

The future impacts of acidification on biological elements are uncertain due to confounding climatic effects. Possible temperature increase may impact weathering rates (and improve water quality) while increased frequency and severity of sea salt episodes will influence metal mobilization (resulting in deteriorating water quality). However, after a sea salt episode, water quality can also improve due to addition of bases. Present levels of atmospheric nitrogen deposition will also counteract recovery.

Chemical criteria have been defined to protect biota against the adverse effects of acidification. These criteria are based on pH, aluminum and calcium, or on calculated variables such as calcium to aluminum ratios and acid neutralizing capacity (ANC). Of these, ANC is the most appropriate variable for clear water systems, where there are close relationships between biological status and dose (Lien et al., 1996; Juggins 2001). ANC can be used as a proxy of the concentrations of the true toxic elements, H+ and aluminum. However, this relationship is influenced for instance by concentrations of organic matter. As organic matter concentrations vary in the freshwater systems, criteria for concentration limits established using on data from one region do not necessarily apply in other regions. Chemical data with good or satisfactory quality is not available for all biological sampling

sites. This hampers the establishment of dose-response models, while the model construction has to be limited to the few elements for which good data is available.

Acidification affects both rivers and lakes in the same way. The main differences are related to the duration of episodes and to habitat heterogeneity, and to the sensitivity of the species present. Due to the unstable nature of aluminum, the exposure concentrations experienced by organisms in a river (downstream acid tributaries) can be higher than the actual concentrations determined in water samples. Due to the longer retention time in lakes, the form of aluminum is more stable, which reduces the analytical uncertainty.

1) Primary responses to acid rain

- Fish biomass reduction – sensitive to toxicants. Data from several studies are available. Simple practical measure of impact that is widely adopted, but on a limited number of lakes and rivers.
- Fish species composition – sensitive to the toxicants. Simple practical measure of impact that is widely adopted in national monitoring programs in all acidified regions. Data is generated by fish sampling and/or by using interview methods.
- Benthic fauna composition and abundance – sensitive to toxicants, data from several national studies. Data quality variable in different countries.
- Phytoplankton composition and abundance - sensitive to toxicants, data from several studies available. Paleo-biological historic reconstruction of pH has been performed in a number of lakes.
- Macrophyte composition and abundance - sensitive to toxicants, state of data more variable. Several studies have been performed.

2) Secondary responses to acid rain

- Benthic fauna composition – related to fish presence/absence. Effects caused by a change from vertebrate to invertebrate top predators.
- Macrophytes composition and abundance – effects related to the effects acidification has on N and P, CO2, water transparency, heat content, and effects on other chemical properties of the water.
- Changes to benthic community caused by changes in macrophytes and/or phytoplankton abundance and composition.

All of these primary and secondary responses are recognized as significant. Few studies include all data elements and many studies lack corresponding water chemistry. This imposes practical limitations on their usefulness in establishing the response indicators that are to be generated within the REBECCA project. Although all possible dose-response relationships are of interest, fish is proposed to be chosen to be the quality element in focus due to the large volume of data available, where the relationship between physico-chemical properties of water and different fish indicators is well established. Many of the relationships have been described earlier, but these studies have often neglected the role organic matter plays on water quality. In water containing organic acids, fish can be exposed to H+ as the main toxicant or to $H^+$ and aluminum in combination. Based on this, the following relationships between fish parameters and acidification are proposed to be analyzed in more detail within the REBECCA project.

Table 3.1.1. Acidification specific relationships with respect to fish to be studied in the REBECCA project

| | |
|---|---|
| pH | Effects related to elevated $H^+$-alone |
| Aluminum | Effects caused by aluminum and $H^+$ in combination |
| Organic matter | Effects organic matter has on water quality |
| Base cations | Effects base cation depletion has on water quality |
| Nitrogen | Effects N-saturation has on aluminum mobilization and ANC |
| ANC (acid neutralizing capacity) | Internationally favored index; but should be refined taking organic matter and effects of future nitrogen saturation into consideration. Effects of base cation depletion need to be evaluated. |

## 3.1 Toxic components of acidification pressure

Annex V of the WFD outlines several physico-chemical elements to be considered in classification. The biological quality elements are related to physico-chemical values of nutrient concentrations, pH and acid neutralizing capacity, which have a high relevance for acidification. If changes in these chemical quality elements show signs of anthropogenic activity, ecological status is downgraded. It is a challenge to separate natural acidic water bodies from water bodies acidified due to acid rain.

### *pH*

pH was regarded as the main toxic component up to 1980. Many of the biological elements that are affected by acidification are closely related to changes in $H^+$. This is in part due to the close relationship between pH and aluminum (Gensemer and Playle, 1999). Several response indexes based on pH have been developed. In naturally acidic waters, pH is low due to organic acids and the bioavailable metal concentration is generally low. In order to establish reference conditions for water body types, it is necessary to separate what effects are related to low pH alone and what affects are related to elevated $H^+$ and metal concentration in combination.

### *Aluminum*

Although aluminum is one of the most common minerals in soils, the presence of aluminum in water is generally low. Anthropogenic acidification will generally mobilize metals, especially aluminum on its toxic or bioavailable form. In stable conditions the concentration of bioavailable forms of Al are related to pH. In rivers, aluminum can be on non-stable forms having enhanced toxicity (Gensemer and Playle, 1999). Fish are mainly impacted by aluminum, and not by pH. Aluminum also have impacts on benthic invertebrate, macrophytes and phytoplankton communities, by influencing various physiological functions and growth (see reviews by Gensemer and Playle, 1999; Sparling and Lowe, 1996). As pH and aluminum concentrations are correlated, it is difficult, but still possible, to separate their effects. This distinction is important in order to discriminate water bodies where pH is low due to acid rain from cases where pH is low due to organic acids.

Aluminum is also toxic in transitional waters, within the salinity range from 0.5 to 15 ppt.

*Nitrogen*

The nitrogen component of acid rain has resulted in elevated concentrations of nitrogen in runoff from areas where nitrogen saturation is exceeded. In areas where critical load is exceeded, high nitrogen levels will lead into increase of the concentrations of $H^+$ and aluminum, thus increasing water toxicity. Increased nitrogen loading enhances plant growth in nitrogen-limited freshwaters and transitional (coastal) waters.

*ANC*

Changes in ANC (acid neutralizing capacity) mirror biotic responses. ANC can be calculated using several methods. ANC is often calculated as the sum of base cations minus the sum of strong anions, where the difference between those two is explained by variation in alkalinity, $H^+$, aluminum and organic anions. There is a strong relationship between ANC and aluminum. This relationship is affected by dissolve organic carbon (DOC) and nitrogen. Water toxicity relating to aluminum is further modified by cation concentrations.

The relationship between ANC and biological quality elements is based on empirical data and verified by laboratory experiments (Baker *et al.* 1987; Reckhow *et al*. 1987, Ormerod 1993, Lien *et al*. 1996; Henriksen *et al*. 1999; Bulger et al 1993). Response curves can be generated using logistic regression, since most organisms show graded responses (Juggins *et al*. 2001). Recent studies carried out in the UK, have resulted in construction of response surfaces for diatoms, macrophytes, invertebrates and fish.

## 3.2 Phytoplankton

There is a considerable literature which indicates that acidity has an important influence on phytoplankton species composition in soft water lakes, although specific mechanisms are difficult to ascertain (Sparling and Lowe, 1996; Gensemer and Playle, 1999; Lindstrøm et al., 2004). Species composition, biomass and productivity are related to pH, alkalinity, Si, DIC (dissolved inorganic carbon) and ANC. While aluminum affects both nutrient and Si availability, it has also direct toxic effects by reducing survival and growth and impacting cell morphology of some species. Acidic lakes tend to contain smaller number of species and in many cases there is a change in the composition of the dominant taxa with increasing acidity. Acidification also affects species composition indirectly through DIC and water transparency. Some species are favored by acidification due to a reduced competition by acid-sensitive taxa. Phytoplankton assemblage is also sensitive to changes in biotic food web interactions.

*Knowledge gaps*

There are a number of gaps in our understanding of relationships between physico-chemical properties of water and phytoplankton indicators that needs to be elaborated to develop quantitative metrics:

- The effects of $H^+$ needs to be separated from aluminum.
- Impacts of the indirect effects of acidification via changes of DIC and Si-concentrations.

- Impacts of the indirect effects of acidification via nutrient recycling and nutrient sources for algal growth.
- Impacts of the temporal variability of acidification episodes.
- Separating impacts of abiotic from biotic interactions on phytoplankton.

## 3.3 Macrophytes

Macrophyte communities provide habitats for other aquatic biota (e.g. fish, macro-invertebrates). Changes in macrophyte species composition and abundance will affect the presence/ absence of other organisms. Aquatic macrophytes have been show to be sensitive to acidification, in addition to other physico-chemical properties of water. In Europe, effects are frequently described qualitatively, but few studies have focused on quantitative changes (Sparling and Lowe, 1996; Gensemer and Playle, 1999; Lindstrøm et al., 2004). At present it was been shown that macrophyte distribution is more strongly related to DIC than to other chemical parameters. Carbon is essential to all vegetation as construction material. When a water-body becomes acidified, the form of inorganic carbon (carbon dioxide, bicarbonate, organic carbon) will cause the main impact on the species composition and diversity (Arts 2002; Madsen et al 2002). Some species appear to favor elevated nitrogen deposition.

A recent work has summarized the tolerance and sensitivity of macrophytes to acidification. Vegetation was divided in four categories: not sensitive, weakly sensitive, moderately and strongly sensitive. An index of acid sensitivity, based on the content of acid-sensitive species within a sample, was developed for phytoplankton and benthic algae (Lindstrøm *et al* 2004).

*Knowledge gaps*

There are a number of gaps in our understanding of relationships between physico-chemical properties of water and macrophyte indicators that needs to be elaborated to develop quantitative metrics:

- The effects of $H^+$ needs to be separated from aluminum.
- Impacts of the indirect effects of acidification via changes of DIC and Si-concentrations.
- Impacts of the indirect effects of acidification via nutrient recycling and nutrient sources for macrophyte growth.
- Impacts of the temporal variability of acidification episodes.
- Separating impacts of abiotic from biotic interactions on macrophytes.

## 3.4 Benthic Invertebrates

Benthic invertebrates are commonly used as environmental indicators of biological integrity (Larsen et al 1996; Gensemer and Playle, 1999). Several species are very sensitive to acidification, where changes in species composition and abundance are used to monitor acidification and recovery from acidification.

Impaired water quality can have a direct affect on species composition and abundance, where other species are more affected by changes in predator-prey relationships, nutrient

cycling and to changes in the macrophyte community. Many species are sensitive for changes in the fish community than to changes in water quality.

*Knowledge gaps*

There are a number of gaps in our understanding of relationships between water physico-chemical properties and macrophyte status that needs to be elaborated to develop quantitative metrics:

- The effects of $H^+$ needs to be separated from aluminum.
- Impacts of the temporal variability of acidification episodes.
- Separating impacts of abiotic from biotic interactions on benthic invertebrates.

## 3.5 Fish

Fish is a sensitive indicator that responds semi-quickly to changes in water quality. Fish generally raises a large public interest and has a high socio-economic value. Acidification effects on fish populations are well documented from all areas impacted by acidification.

Different species and life stages have different sensitivity. Acidification affects population dynamics of fish at several levels. Moderate acidification can affect growth, development and recruitment. As acidification becomes more severe, survival is reduced. Traditionally fish kills are associated with respiratory and ionoregulatory malfunction. For anadromous species (species migrating between fresh and salt water), fish populations can also extinct if their seawater performance is reduced. Fish seawater performance is extremely sensitive to all metals, and will impact survival much earlier than in freshwaters. Fish populations are generally not impacted by food availability. Bottom up relationships have therefore minor importance (Rosseland and Staurnes, 1994; Sparling and Lowe, 1996; Gensemer and Playle, 1999).

Aluminum is the prime factor affecting survival, growth, reproduction, and behavior of fish (Gensemer and Playle, 1999). Since aluminum is not vital to any aquatic biota, there is no specific defense mechanism against this metal. The toxic properties of aluminum are manifested through accumulation onto the fish gills. There are close relationships between acidification pressure and aluminum concentrations in fish gills. Fish status is, however, better described by ANC or similar proxy variables. In the current acid deposition rates, ANC is correlated to both H+ and aluminum. Several confounding factors have been identified potentially influencing the future values of ANC under reduced acid rain conditions.

*Knowledge gaps*

There are a number of gaps in our understanding in the relationships between water quality (defined as ANC, pH, aluminum, DOC, base cations etc.) and fish status that needs to be elaborated to develop quantitative metrics for fish, particularly when acid rain deposition is reduced:

- The effects of H+ needs to be separated from aluminum.
- Acidification effects to age structure, growth and reproduction need to be separated from natural variability.
- Organic material affects the relationship between toxicants and fish status. Current agreed ANC limits in water will not protect fish if there is more than 4 mg DOC.

- How will the relationships between pH, aluminum and cation concentrations (ANC) change with reduced acid deposition?
- How will reduced retention of NOx affect the ANC/ aluminum relationship?
- Impact of concurrent nitrogen loading, temperature increase and sea salt episodes on recovery from acidification?
- Episodes can have biological importance that are not easily detected.

# 4 RIVERS

The ecological consequences of different pressures on river ecosystems differ depending of the type of pressure. The REBECCA approach regarding rivers is to describe and establish relations between each type of pressure, the relevant supporting parameters and the primary biological quality parameters of the Water Framework Directive likely to respond to the pressure as indicated in the table below.

| **Type of pressure** | **Supporting parameters** | **Primary biological response parameters** |
|---|---|---|
| Physical modification | Hydromorphology parameters | All (attached algae, phytoplankton, macrophytes, invertebrates, fish) |
| Nutrient enrichment | Nutrient concentrations | Attached algae, phytoplankton, macrophytes |
| Organic enrichment | BOD, TOC etc. | Invertebrates, attached algae |
| Toxic substances and acidification<br><br>(see Chapter 3 and 6) | Concentrations of toxic compounds, pH, alkalinity | All |
| Combined effects | Concentrations of substances<br><br>Hydromorphological parameters | All |

## 4.1 Eutrophication pressures

*Amelie Deflandre*

Levels of autotrophs (phytoplankton, macrophytes and phytobenthos) in rivers are responsive to changes in inorganic nutrient concentrations, especially at low nutrient levels. For all these biological elements, increased nutrient concentrations can potentially lead to an overall increase in biomass and the competitive exclusion of nutrient-sensitive, slow-growing species if no other factor intervenes. Large-scale phytoplankton growth is limited to rivers with a long residence time, macrophytes and phytobenthos to areas where light penetration reaches the bed of the river.

Numerous studies have been undertaken to examine the relationships between macrophytes and benthic diatoms and nutrient status in rivers. However, we now need to explore the possibilities to generalize the results of these different studies across Europe. Thus it is necessary to develop the relationships between nutrient loadings and levels and the ecological status of rivers based on periphyton, macrophytes and phytoplankton quality elements and parameters in rivers. Further understanding of the role of sediments for the nutrient dynamics of macrophytes and phytobenthos is needed as well as a better understanding of the relative importance of nitrogen and phosphorus.

Recent studies indicate that, in addition to phosphorus, also nitrogen may be limiting plant production and biomass in rivers (e.g. Leland 1995, Snyder et al. 2002). Potential nutrient limitation experienced by the algal communities can be determined using the ratio of soluble N/P (Mainstone and Parr 2002), while enrichment experiments (Bothwell 1988, Johnston et al. 1990, Perrin et al. 1987), measure of phosphatase activity (Whitton and Kelly 1995) or specific growth rates (Biggs 1990) provide information of the actual nutrient limitation experienced by plant communities.

Benthic invertebrates and fish fauna are only indirectly responding to inorganic nutrient changes, through change of habitat and food resources and therefore analysis for REBECCA will focus primarily on the linkages between autotrophs and nutrients.

The overall major requirements in relation to the implementation of the Water Framework Directive is to develop relationships that indicate to which extent nutrient concentrations must be reduced in different rivers to achieve a good ecological status assessed from the occurrence of macrophytes, periphyton and/or phytoplankton in the river.

### 4.1.1 Phytoplankton

The phytoplankton community in rivers can be changed as a result of increasing the concentrations of phosphorus and/or nitrogen in the river water. Enrichment in N and P can increase the phytoplankton biomass (Vanni and Findlay 1990). Increases in N and P are thought to increase bloom frequency and intensity. Species have an optimal N:P ratio (Hecky and Kilham 1988). Phytoplankton is more closely related to inorganic nutrients in spring.

Phytoplankton generally is more significant in deep, slow-flowing rivers, where periphyton and macrophytes cannot develop and where the residence time is sufficiently long to allow development of phytoplankton.

Excessive algal growth can affect the use of water for water supply. Substantial algal growth can lead to haloform production after chlorination, deoxygenation in transfer pipelines, algal penetration into potable supply and taste and odour problems (Jones 1984). Blooms may also be toxic to fish and macroinvertebrates and shade macrophytes and periphyton.

#### *WFD –associated characteristics*

Annex V of the WFD outlines three phytoplankton-related quality elements that need consideration in rivers:

- Taxonomic composition
- Phytoplankton abundance and its effect on transparency
- Planktonic bloom frequency and intensity

#### *Knowledge gaps*

Relationships between nutrient concentrations and phytoplankton development in rivers are not well known. Nutrient concentration may not be the only responsible for triggering blooms. Therefore the respective effects of N and P, and their ratios and different forms on phytoplankton biomass and assemblages need further investigation in large rivers.

### 4.1.2 Macrophytes

Macrophytes are also sensitive to inorganic nitrogen and phosphorus concentrations in the water column. They are able to take up nutrients directly from the water column into their shoots They are widely present in shallow rivers where light reaches the riverbed. In nutrient-limited reaches, an increase in inorganic nutrients (P and N) may lead to an increased growth of macrophytes, an increase in biomass and a shift from slow-growing, nutrient-sensitive species to fast-growing, nutrient-tolerant species. In the lowland river systems, that are naturally nutrient rich, inorganic nitrogen and phosphorus are not always limiting macrophyte growth. Rooted macrophytes can also uptake inorganic nutrients from riverbed sediments in varying proportions. The ratio between shoot and roots biomass may reflect nutrient availability. However, physical characteristics as nature of riverbed and flow velocity can significantly affect the repartition of macrophytes.

#### *WFD –associated characteristics*

Annex V of the WFD outlines two macrophytic-related quality elements that need consideration:

- Taxonomic composition
- Macrophytic abundance

#### *Knowledge gaps*

- Macrophyte responses related to sediment nutrients are not yet known. The response of macrophytes to nutrient enrichment or stripping of the water column may be affected by storage of nutrients in riverbed sediments, potentially causing lags in macrophyte response to reduced nutrient inputs.
- Empirical models have been developed to separate species and community assemblages of macrophytes. However, these models need to be tested for their applicability for different European river types.
- The combined effects of eutrophication and other types of pressures (e.g. weed cutting, dredging, physical stream channel modifications) on river macrophytes need to be elucidated.
- The existing relationships between macrophyte indices and inorganic nutrients are mostly qualitative. Quantitative relations need to be established for development of classification tools for WFD implementation.

### 4.1.3 Phytobenthos

Like other autotrophs, phytobenthos development depends on availability of inorganic nutrients. An increase in inorganic nutrient (P and N) availability often leads to increased growth of phytobenthos and an increase in biomass (Snyder et al. 2002, Welch et al. 1992), with a shift from slow-growing, nutrient-sensitive species to fast-growing, nutrient-tolerant species (Biggs 1990).

Phytobenthos may develop on the sediments, rock bed or macrophytes, depending on taxa and local conditions. Some taxa are directly in contact with riverbed sediments (where nutrient internalization occurs) and therefore may derive nutrients directly from sediment pore water. The thick mat structure and the sediment source of nutrients can therefore create local nutrient conditions within the phytobenthos mat, which are different from the water column.

*WFD –associated characteristics*

Annex V of the WFD outlines two phytobenthos-related quality elements that need consideration:

- Taxonomic composition
- Phytobenthos abundance

*Knowledge gaps*

- Testing of the sensitivity of other phytobenthos indicator groups than benthic diatoms, and consequent development of empirical relations between nutrient pressures indicators such as green filamentous algae, are needed.
- Quantitative relationships between phytobenthos community structure parameters and nutrients are lacking.
- Several empirical models of phytobenthos community indicators have been developed in restricted areas and often outside Europe. Applicability of these models to European river types within larger geographical regions (ecoregions) needs to be tested.
- Combined effects of nutrients (nitrogen and phosphorus) vs. organic matter and nitrogen vs. phosphorus are lacking.

### 4.1.4 Benthic Invertebrates and fish

Benthic invertebrates and fish fauna are present in most watercourses. The relatively long life span makes them good integrators of multiple pressures over several months (macroinvertebates) to several years (fish). Except for toxic impacts at high concentrations of ammonia, benthic invertebrates and fish communities are not directly dependent on phosphorus and nitrogen concentrations in water.

Eutrophication may influence benthic invertebrates and fish through food web (algae and invertebrates) and habitats (macrophytes protecting fish and invertebrates from predation). An increase in inorganic nutrients is thought to increase fish species richness (except for already eutrophic rivers) and biomass. For the worst cases of eutrophication, internalization of the excess of autotrophic biomass will create an increased biological oxygen demand that can deplete the slow-flowing parts of the river in oxygen and reach levels lethal for benthic invertebrates and fish. In this case, it affects total biomass and the age structure of fish communities.

River invertebrates and fish communities as quality indicators to assess nutrient impacts and needs for interventions are of minor relevance for REBECCA compared to macrophytes, periphyton and phytoplankton, as they are not directly impacted by the nutrient level.

## 4.2 Organic pollution pressures

It is well known that changes in the quantity and quality of organic matter in rivers affect the river biota (phytobenthos, macrophytes, benthic invertebrates and fishes). Relations between biological oxygen demand (BOD) levels in rivers and saprobic indices are well established for rivers polluted by urban wastewater. However, changes in biota caused by degradable organic matter also occur at low increases in the concentration of organic matter and without a substantial oxygen deficit in the river water. Mathematical models to predict concentrations of DO, BOD, ammonium and other chemical constituents in rivers as functions of the discharge of pollutants and river characteristics are well developed, but they usually require a very substantial amounts of input data. Most of the literature screened concerned benthic invertebrates, whereas less information is available on other biological elements.

Knowledge gaps identified regard the response of biological elements to variability in the quality of organic matter, the response to variability through time of the input, and the combined effect of organic matter and other stressors.

The major pollution sources are organic matter in domestic wastewater, in industrial wastewater (from food processing and from other industries processing degradable organic matter e.g. pulp and paper industry), in agricultural discharges from livestock production and in discharges from fish farming. Also occasional discharges such as storm-water run-off from urban areas can be an important pollution source.

The specific impacts mainly depend on two factors: the resulting increase in the concentration of organic matter in the receiving water and the quality of the organic matter discharged. Further, the ecological damage depends on the hydromorphological characteristics of the river and on the other pressures on the river ecosystem.

Biological wastewater treatment has dramatically reduced the wastewater pollution impact in many European rivers. The organic loading from a wastewater discharge is often reduced by a factor of 10 when an effective biological treatment is established, but at the same time the quality of the organic matter discharged has changed and therefore also the type of ecosystem impact.

The implementation of the WFD focuses on impacts on biota. In practice quality criteria are expressed as a biotic index characterizing the biological status of the water body. There is a long tradition for application of such indices in Europe, but not much experience for using quantitative descriptions of relations between the index values and the concentrations of pollutants.

The degree of impact on the river biota has been established from studies at different levels or organic pollution. This led to the development of the “Saprobic System” by Kolkwitz and Marsson (1902). According to the Saprobic System water bodies are characterised by the degree of heterotrophy (ratio between heterotrophic and autotrophic processes in the water body). Relations between BOD levels in rivers and saprobic indices are well established for rivers polluted by urban wastewater, especially for wastewater, that is not biologically treated, and for increases in BOD levels of more than about 1 mg $l^{-1}$. As the natural background level of BOD in rivers is also around 1 mg $l^{-1}$, minor changes in BOD level caused by wastewater discharges are likely to affect a river ecosystem.

A determination of the saprobic value at a river station is based on a sampling and identification of species of fauna and flora at the station and a comparison with the Saprobic characteristics established for each species (see Liebmann 1951, Sladecek 1973). The Saprobic System is widely used in Central Europe. No single species is representative of a single saprobic zone. Rather its distribution will follow a normal curve that stretches over several zones. The shape and area of this distribution curve defines the

saprobic „valency“ of the species and the position of the apex is its saprobic value. Various saprobic values have been published for a very large part of the European river biota.

During the last decades several EU countries have introduced other types of quality indices frequently based on invertebrates and on the difference in impacts on sensitive and on tolerant species of pressures considered. An example is the RIVPACS (River InVertebrate Prediction And Classification System) system developed in UK. RIVAPACS gives site-specific predictions of the macroinvertebrate fauna based on environmental features, and sets a target of the fauna to be expected in the absence of environmental stress (e.g. pollution, or habitat degradation). Comparison of this target with the observed fauna at the site forms the basis of the biological assessment.

### 4.2.1 Phytobenthos

Benthic diatoms are widely used as water quality indicators. Occurrence, species composition and biomass are usually related to the level of inorganic nutrients (see chapter 4.1). Organic pollution also affects the occurrence of benthic diatoms, and for many species saprobic values can be found in the literature (Liebmann 1951). Fjerdingstad (1964) describes a system for the estimation of the pollution levels in streams based on the communities of benthic phyto-microorganisms.

Eichenberger and Wuhrmann (1966) studied the effects of addition of sewage to experimental channels in Switzerland and found significant impacts on microbenthos with organic enrichments corresponding to a few mg BOD/l. Both organic matter and nutrients were considered to be responsible for the impacts. Other component of the phytobenthos of relevance for the organic pollution includes macroalgae and sewage fungi. The latter are well known indicators of pollution of streams with easily degradable organic matter.

### 4.2.2 Macrophytes

Very little knowledge seems to be available on the impacts of organic pollution on river macrophytes, at least no scientific papers were found describing the effects. Relations to nutrients are more likely, especially if the level of nutrient enrichment is not very strong (see chapter 4.1.).

However, it can be anticipated that with high levels of organic pollution also macrophytes would affected. Two factors are likely to be important: 1) Deposits of organic matter and low redox potentials in the sediments will prevent the occurrence of many macrophytes and increased turbidity of river water, and 2) increased growth of attached micro-organisms will reduce the light availability for the macrophytes and therefore reduce their abundance/ coverage.

### 4.2.3 Benthic invertebrates and fish

Carbon, energy, and inorganic nutrients budgets of many streams and rivers are dominated by processes associated with large pools of detritus organic matter both in the dissolved and particulate forms (Webster and Meyer 1997); heterotrophic microbes, mainly fungi and bacteria, mediate these processes (Meyer 1994, Findlay et al. 2002).

Through the detritus-based food web, organic matter and microbes become an important food source for a wide range of macroinvertebrates, whose abundance often correlates with

the OM standing stocks (Findlay et al. 2002). Macroinvertebrate activity, vice versa, speed up the organic matter decomposition rates (Cummins 1973, Fazi and Rossi 2000). Organic matter indirect effects, mediated by microbial oxygen consumption, also affect the macroinvertebrate community structure.

Both these direct and indirect impacts of organic matter make the analysis of macroinvertebrate community structure one of the most relevant indicators of organic matter pollution in rivers and streams.

### *Knowledge gaps (all biological elements combined)*

- Functional relationships between concentrations of organic matter and most biotic indices are not established.
- There is a lack of knowledge on the impact of the wastewater quality on biota. This would be important to establish outlet criteria for the river ecological quality objectives.
- There is a lack of knowledge on the responses of biotic indicators to organic pollution, especially at the lower end of the pollution range (i.e. around the border between high and good ecological status).
- More knowledge on the responses of benthic and phytobenthos to organic pollution is needed.
- Practically no quantitative information on biological impacts of organic matter loading combined with variability in flow regime (low vs. fast flowing river regimes) was found in the literature.
- Very limited knowledge is available on the impacts of urban storm-water, agricultural run-off, or intermittent industrial discharges on river biota
- Organic pollution in relation to other pressures. There is little general information on the combined impacts of organic matter, nutrient enrichment, hydromorphological impacts and toxic compounds on river biota.

## 4.3 Hydromorphological pressures - Benthic invertebrates

*Bente Clausen, Mike Dunbar*

Hydromorphological pressures include all changes caused by human influences to either the flow regime or the morphology of the stream that affect the stream biota. The most important hydromorphological pressures are:

- building of dams for hydropower, water supply or other purposes
- canalisation and/or dredging of rivers or streams to improve drainage or navigation;
- weed cutting to improve drainage;
- abstraction of water directly from the stream or from groundwater for water supply or irrigation or by diversion, e.g. for hydropower or irrigation;

The LIFE index (Lotic Invertebrate index for Flow Evaluation, Extence et al. 1999) was formulated in UK to test whether it is possible to link changes in benthic invertebrate community structure with indices of historical river flow at a gauge close to the sample site. The LIFE index can be calculated from species or family-level bio-monitoring data. It is important to note that the index is expected to be sensitive to natural and artificial flow changes, it thus allows an extrinsic hypothesis to be tested.

LIFE is currently being used in England and Wales as part of the implementation of Catchment Abstraction Management Strategies and the Water Framework Directive. It is also being tested in the STAR project. A similarly constructed index (MFR – mean flow rank) is currently being developed by the Environment Agency of England and Wales to relate flows to macrophyte communities, but there is no published information yet available.

*Knowledge gaps*

- No methods have been developed to distinguish combined effects of different pressures. Often a change in biology is associated with several hydromorphological changes and there are no methods to predict the effect of each individual hydromorphological factor (e.g., hydrological and morphological).
- The known relations between hydromorphological changes in rivers and the resulting impacts on biota are not sufficient to cover WFD needs.
- There is a lack of understanding of the role of macrophytes for structure and physical properties of the habitat. Small weedy lowland streams are often impacted by several concurrent pressures, because there the water demand is usually high.
- Quick methods of assessing impact of the critical hydrological pressures and their effects on the biota are needed.
- Biological indicators or metrics that reflect the degree of hydromorphological pressure, including the lateral connectivity of streams. The LIFE index is an invertebrate indicator that varies with hydromorphology, but no appropriate indicators or metrics have been developed for other trophic levels.

## 4.4 Combined pressures

*Ana Garcia, Jean-Gabriel Wasson*

In most cases aquatic ecosystems are impacted by multitude of pressures which act concurrently. Consequently managers have to identify priorities for the actions to be taken depending on the importance and the possibilities to diminish the pressures. Moreover, different pressures are not spatially homogenous, and thus decision makers have to decide which areas are in priority. Finally, various human activities generate different kind of pressures. Therefore integrated policies to restore river ecosystems must be tailored to diminish pressures from different socio-economic sectors, such as agriculture, industry, urban areas, etc.

Therefore political decisions require spatially extensive understanding of the scales and magnitudes of the different problems. A successful implementation of the WFD will require a common framework and policies for large territories and the development of such a framework requires scientific support and tools.

However, from the scientific point of view, the problem is complex, for two reasons:

1 - Many pressures are difficult to identify and measure, especially when large areas are in consideration. In most cases, there is not the same level of information available on the magnitude and intensity of the pressures, such as inputs of chemical substances from the diffuse or point sources or on hydro-morphological alterations, at the local than on the basin scale. For this reason, it is often necessary to consider the d*riving forces of the pressures* to get spatially homogeneous information. For example, it is very difficult to determine the amount of pesticides and nutrients discharged in a large basin, the sediment yield due to erosion, the degree of river canalization etc., but in many cases there is access to the

information on the percentage of land cover dedicated to various agricultural activities that generate different pressures.

2 - In contexts of large spatial areas, the key issue is the influence of scales when studying ecological processes. One important problem is the possibility of up scaling ecological relationships: how a model developed at a local scale can be safely extrapolated to larger areas? Another problem is related to the interactions between the natural characteristics that determine the sensitivity of river ecosystems and anthropogenic alterations: has the same pressure similar impact in different regions?

To address these questions, it is necessary to study pressure-impact relationships in geographically homogeneous areas, and to identify the relative influence of human and natural drivers on ecosystem responses. The appropriate scale is to work at the level of "hydro-ecoregions", which are considered to be homogeneous with respect of natural factors controlling aquatic ecosystems.

There are few studies dealing with this topic, and from those it is difficult to separate the impact of combined pressures on different biological communities. The impact of combined pressures is generally assessed as the response of fish or macroinvertebrates, and sometimes algal, communities. Further the approaches for the quantification of the status of physical habitat are variable.

### *Knowledge gaps*

Despite the strong need, there are not much research carried out to establish clear functional relations across larger spatial scales. For example, we found a disagreement in the different papers reviewed, especially concerning the influence of spatial scale to predict changes in the biological indicator values: some papers indicate landscape-level factors to be the only variables to predict biological indicators, while in other papers relationships beyond the local scale measurements have not been found. Is the reason for this discrepancy due to the different human and natural contexts considered, or is it due to the different methods used?

Existing studies have been mostly focused in one basin, watershed or ecoregion only. We have found a lack of comparative inter-regional studies. However, to support decision-making in the larger river basin scale in the variable regions of the Europe, inter-regional approach would be very useful to identify 1) issues comparable even if regions are spatially separated, and 2) issues that are different for various regions and should be managed differently even within a same country. Such approach would set a coherent natural framework for assessment and management of river systems.

So far the pressures-impact relationships have not often been studied using ecoregional scale. Thus it is still an open question, whether the ecoregional approach will improve the predictive capacity of pressure-impact models. Natural constraints probably determine both the ecosystems sensitivity and the applied land use practices typical for each region, thus leading to different regional responses. The issue of interactions between natural variability and human impacts on regional scale is very complex and requires further investigations.

Based on the results of the literature review, the relative importance and the order of the different driving forces impacting stream ecosystems are still uncertain. When comparing the different forms of land use, some authors found that agriculture caused the most detrimental impacts, while in some cases strongest impacts on river ecosystems were found in the urbanized areas. This could be due to the differences in the methodology applied (bioindicators, methods, models). However, on the other hand there may be hierarchical relations between these driving forces, which vary depending on the ecoregion or on the level and type of human impacts.

The natural sensitivity of the river ecosystems may influence the degree of impairment. The socio-economic context determines the intensity of the pressures due to different land use practices, as well as the policies adopted to alleviate the impacts. The key issue and the result of this analysis is that the relationships between driving forces and pressures must be better understood to introduce appropriate treatment practices for different systems. For example, the same agricultural practice and land use may have different effects in terms of pollution, sediment yield etc. on river ecosystems, depending on their intensiveness or spatial extensivity. In the same way, two urban areas of the same size may not have the same impacts due to different wastewater treatment practices, which may depend on the local, regional or national socio-economical context. Thus the treatment practices may increase or reduce the intensity of the pressures. However, it is not easy to define relevant indicators of treatment practices.

It needs to be understood better, if different spatial scales require development of different functional relationships, for example the conditions at the riparian corridors vs. basin-level conditions. It is difficult to distinguish between the influence of local riparian corridor, upstream riparian corridor, basin or sub-basin conditions and land use when studying causes for ecological impairment at a single site. Again, the papers reviewed in this analysis present different results concerning the importance of the different spatial scales. The restoration of the river corridors is often seen as a key action to improve ecological status, but the actual “buffering capacity” of riparian zones needs to be properly evaluated. This is also the case in the evaluation of the possible existence of thresholds in the relations between driving forces and ecological status. Some managers are questioning, if a good ecological status is attainable in the heavily urbanized basins, or in the intensively cultivated areas. Finding and proposing solutions for such questions will be crucial for the implementation of the WFD. Some of the models analyzed indicate the existence of possible threshold levels, but the underlying scientific reasoning on the causal reasons is still not clear.

In general, it seems that the current knowledge is not sufficient to give clear advice to decision makers how to deal with complex impacts of combined pressures across different spatial and temporal scales. Such approaches are still novel and thus there is a lack of conceptual models. Large-scale analysis comparing relationships through different scales, countries, and ecoregions are still necessary.

# 5 COASTAL WATERS

## 5.1 Eutrophication pressures

*Pirjo Kuuppo*

Coastal ecosystems receive nutrients either directly from the sources on the coastal line, rivers that bring nutrients from their catchments, and from the atmosphere. The increased nutrient loading from anthropogenic sources has caused eutrophication of coastal ecosystems, the symptoms of which are excess accumulation of phytoplankton biomass, depletion of oxygen in bottom waters, increased frequency of noxious algal blooms, increased turbidity, deterioration of coastal food webs and reduction of biodiversity.

Many coastal and estuarine ecosystems receive high nutrient loading today. Estimates of the increase of phosphorus and nitrogen loading to estuarine systems range between 2-6-fold and 1.5-14-fold compared to the turn of the 20$^{th}$ century (Conley 1999,Cloern 2001). Atmospheric loading of nitrogen to the north Atlantic area has increased 5-10 -fold since beginning of industrialisation (Paerl and Whitall 1999), and atmospheric deposition of nitrogen may represent 10-50 % of the anthropogenic N flux to the water surfaces (Pryor and Barthelmie 2000). Nutrient loading from diffuse sources (direct runoff from land, fish farming, etc.) to the coastal waters is more difficult to estimate.

The extent to which nutrient loads is reflected in coastal ecosystems depend largely on their physical characteristics. While regions of vertical stratification, restricted water exchange and long residence time, with low tide and low mixing accumulate more nutrients and are thus more in risk to eutrophication, nutrients received in upwelling areas, open coastal areas with high tide or currents are rapidly diluted and transported to the open sea.

As nitrate concentrations increase due to anthropogenic loading, lotic systems are moving towards not only higher N:P ratios, but also lower Si:N ratios, and coastal areas are likely to have both P and Si limitation in northern Europe (Jickells 1998, Turner et al. 2003). Also other human activities, such as damming of rivers, leads to increasing silicate retention and smaller silicon load to the seas (Humborg et al. 2000). During the last decades, N: P ratios have increased dramatically in the Dutch coastal waters (de Jonge et al. 2002), Danish coastal areas (Jørgensen 1996), and the Black Sea (Shtereva et al. 1999) as a result of increase in N-loads and decrease in P-loads.

When considering eutrophication in coastal areas, not only ambient nutrients concentrations but also nutrient ratios are more relevant . The ratio of total N and total P ratio is not, however, a good indicator, because total nutrient concentrations include a large fraction of dissolved organic nutrient that cannot be utilized by phytoplankton. The ratio of dissolved inorganic nitrogen and phosphorus, DIN:DIP can be used as an indicative number of potential nutrient limitation, while experimental studies (e.g. nutrient addition bioassays, mesocosm experiments) reveal the responses of phytoplankton communities on nutrients.

Phytoplankton growth is generally considered to be limited by one of the major nutrients, (N, P), in addition to diatoms which are dependent of silica (Si). While phosphorus is regarded as the main limiting nutrient in freshwaters, marine open waters are primarily nitrogen-limited. In estuaries, coastal transient areas and river plumes, the limitation pattern can shift from P to N limitation towards the open sea, as well as shift between N and P limitation seasonally (review in (Conley 1999).

It has recently been shown, that iron (Fe) can be limiting phytoplankton growth also in eutrophic coastal waters (Zhang 2000), and not only in open oceans. Heavily nitrogen-

impacted estuaries and coastal waters are moving from nitrogen limitation towards secondary nutrient (P, Si, Fe) limitation (Paerl 1998).

### 5.1.1 Transparency conditions

*Gunni Ærtebjerg, Anouk Blauw*

Transparency of the water column in coastal waters indirectly reflects the nutrient loading/nutrient status. Increased nutrient loading often lead to increases in phytoplankton biomass in the water column, which in turn decreases the transparency. Other sources such as discharges of suspended solids or dissolved coloured substances, e.g. untreated sewage or industrial effluents, may locally directly affect the transparency of the water.

Changes in transparency due to changes in nutrient loading will affect the depth of the euphotic zone with sufficient light for primary production, and thus the depth limits of macrophytes, e.g. sea grasses and macroalgae. Different species have different light requirements and changed transparency therefore also affects dominance patterns of the vegetation. Besides changing the depth distribution of macrophytes, lowered transparency might influence the recreational value and the tourist industry negatively.

Comparison of long-term Secchi depths measurements during the period 1919-1939 with those from the period 1969-1991 in the Baltic Proper indicated that Secchi-depth had decreased about 0.05 m $y^{-1}$ (Sandén and Håkansson 1996). Likewise, in the northern Adriatic water transparency has decreased since 1960 due to increased phytoplankton biomass related to nutrient enrichment via the Po river discharge (Justic et al. 1995). Eutrophication related decreases in water clarity have also been reported from the eastern Adriatic. In the Dutch Wadden Sea (the Marsdiep) Cadee and Hegeman (2002) found a decrease in the annual mean Secchi depth from 1.20 m to 1.0 m from the early 1970's to 2000. Also improvements of transparency due to local reductions of nutrients and suspended matter discharges from sewage and industry have been reported from several coastal areas.

There is substantial evidence that the serious decline of seagrasses (both *Zostera marina* and *Zostera noltii*) in Dutch waters is partly related to a decreasing water transparency and eutrophication (Philippart 1994). Several other factors, however, such as significant changes in the tidal range, currents, exposure and salinity due to the construction of dikes and dams in Dutch estuaries, destructive fishing techniques (such as bottom trawling and cockle fisheries), as well as the occurrence of an epidemic disease (wasting disease) have contributed to the overall decline.

#### *WFD –associated characteristics*

Transparency is not specifically mentioned as a quality element in the WFD. However, transparency is linked to normative definition (Annex V, sections 1.2.3. and 1.2.4.) of the phytoplankton quality element so that in high status phytoplankton should have a biomass, which does not significantly alter the type specific transparency conditions. Therefore, transparency is an important supporting element for assessing the state and development of the ecological quality of a water body.

*Knowledge gaps*

In order to use transparency as a quality element for establishing classification schemes for eutrophication in coastal waters, it is necessary to know the quantitative relation between transparency and nutrient loading to the system. This relation is most often not yet established. However, in some areas it has been possible to establish this quantitative relationship, either through direct correlation, or use of more or less complex models. For instance in the Danish estuaries significant relationship between total nitrogen concentration and the chlorophyll a concentration as well as the Secchi depth has been established (Nielsen et al. 2002). The next step would be to establish the reference conditions, which in most cases is not available, and therefore must be determined from models.

Unsolved problems in using transparency as a quality element relate to both methodological issues and to a weak response to nutrients in many systems. Transparency can directly be measured as attenuation of light through the water column using light meters or alternatively using a Secchi Disc. However, Secchi depth is only an approximate evaluation of the transparency of water, and it is used primarily for its simplicity. Secchi depth (SD) is related to the attenuation coefficient (Kd) by Kd = a/SD, where a varies between 1 and 2, with the lowest values in turbid coastal waters (Holmes 1970, Kirk 1983, Buiteveld 1995).

In some cases the transparency is not or only weakly coupled to the nutrient loading/nutrient status of the system. In naturally high turbid waters as in some tidal areas, the anthropogenic discharges of nutrients may have only little or no effects on the transparency, e.g. if the phytoplankton production is light limited and not nutrient limited, or the irradiance attenuation is far dominated by resuspended sediments. Also in shallow non-stratified areas with high density of suspension feeding bottom fauna, the transparency can be more dependent on the density of e.g. mussels than the nutrient loading.

### 5.1.2 Phytoplankton

*Pirjo Kuuppo, Anna-Stiina Heiskanen*

Phytoplanktonic primary producers are the first organisms to respond to elevated nutrient concentrations in their environment. Most phytoplankton species, with an obvious exception of picoplanktonic algae (that are ubiquitous in aquatic systems) respond positively and predictable to nutrient enrichment in all European coastal areas (Olsen et al. 2001).

High phytoplankton biomass results in increased amount of organic matter to be degraded after sedimentation by bacteria, meso- and macrofauna, which may lead to hypoxia or anoxia of bottom waters. Long-lasting eutrophication causes recurrent or even permanent oxygen deficit on bottoms, leading to self-fertilization in coastal areas, in estuaries and enclosed seas, such as the Baltic Sea. High biomass of phytoplankton increases the turbidity, which decreases the recreational value of coastal waters.

Although direct causal connection between eutrophication and increased frequency of harmful algal blooms has not been proven, it is generally believed that their frequencies have increased worldwide due to the increased nutrient input (Hallegraeff 1993, Anderson et al. 1993). The toxic blooms cause considerable economical harm on the global scale, and also in the European coastal zones (Cloern 2001). Moreover, the toxic bloom events constitute a serious health risk for humans in terms of the toxins produced by the algae that are concentrated in shellfish cultivated for human consumption.

### *WFD –associated characteristics*

The WFD outlines the biological quality elements related to phytoplankton in coastal waters in the Annex V. These are:

- Phytoplankton composition and abundance of phytoplankton taxa
- Average phytoplankton biomass and water transparency
- Frequency and intensity of phytoplankton blooms

According to the WFD (Annex V), declining ecological quality of coastal waters is characterised by slight ('good status') or moderate ('moderate status') disturbance in the composition of phytoplankton abundance and taxa, slight or moderate changes in the biomass compared to the high status, and slight or moderate increase in the frequency and duration of phytoplankton blooms.

### *Knowledge gaps*

Even though coastal eutrophication is a severe problem in many European coastal areas and estuaries, research focus in Europe has been imbalanced and fragmented, with most work done in the North Sea and the eastern Mediterranean, whilst publications from the Atlantic Coast of southern Europe and Scandinavia are scarce. As a whole, the knowledge of eutrophication processes and impacts on coastal ecosystems is 20 years behind that of freshwaters (Vidal et al. 1999).

Gaps can be identified e.g. in nutrient limitation patterns in different areas of Europe, variations in nutrient limitation along spatial and temporal scales, and nutrient limitation of potential harmful or bloom-forming phytoplankton species. Furthermore, comparisons between the ecosystems in European coastal areas are scarce.

It is not sufficiently demonstrated that some functional phytoplankton groups (such as flagellates or nitrogen fixing cyanobacteria) gain advantage on changing nutrient conditions due to increased eutrophication. There are potential indications that such changes may have taken place in many coastal areas, but the various national databases containing long-term phytoplankton data have not been sufficiently explored to provide statistical evidence for such structural changes in the phytoplankton communities. Such aggregated information of the taxonomic composition of phytoplankton community in addition to data on biomass levels, combined with techniques to detect changes in bloom frequency, may potentially provide a robust tool for classification and assessment of coastal waters as required by the WFD.

Despite the accumulating evidence that toxic or otherwise harmful algal blooms have increased their frequency and duration in anthropogenic ally eutrophied coastal waters, little is known about the processes that lead to their initiation, e.g. what is actually the role of nutrients in comparison to other factors, such as light and mixing. Moreover, there is no experimental/ statistical relationship yet established between nutrient status or loading and the frequency of phytoplankton blooms.

Whereas in lake ecosystems the Vollenweider model of the relationship of phosphorus loading and phytoplankton biomass (chlorophyll-a) is applicable for 75 % of lakes, it is not possible to develop a straightforward application of nutrient loading models for the coastal/estuarine ecosystems. This is mainly because coastal areas and estuaries respond to increased nutrient loading in different ways, depending on the intrinsic buffering mechanisms of the ecosystem. However, some preliminary work on development of Vollenweider-type models for coastal and estuarine waters have been already carried out, with promising results (Meeuwig et al. 2000, Nielsen et al. 2002). Nutrient loading models combined with appropriate phytoplankton status indicators (as suggested above) are

required for development of river basin management plans and programs of measures based on reliable assessment of land-based nutrient loading impacts on coastal & estuarine waters.

### 5.1.3 Macrophytes

*Dorte Krause-Jensen, Saara Bäck, Paul Erftemeijer*

Benthic macrophytes have a lifetime of months to years and most species grow at the same spot attached to bottom substratum. Their growth pattern therefore reflects a time-integrated response to physical and chemical characteristics of the habitat. Seagrasses and different algal species or functional algal groups have different nutrient demands and changes in nutrient concentration therefore affect dominance patterns of the benthic vegetation as well as dominance patterns between planktonic algae and benthic vegetation (Sand-Jensen and Borum 1991, Duarte 1995, Pedersen 1995).

Planktonic and opportunistic algae (mostly filamentous species) are generally favoured by high nutrient concentrations and tend to oust seagrasses and perennial algae in eutrophic areas. Their increased biomass shades the perennial vegetation and limits its depth distribution, thereby further accelerating the decline of the perennial vegetation (Duarte 1995). Other physico-chemical variables like salinity, exposure level and substrate composition as well as biological factors like grazing or contagious diseases may also affect growth- and loss processes and may blur the response of the vegetation to nutrient loading.

Changed dominance patterns of the coastal primary producers from benthic macrophytes to planktonic algae or from long-lived seagrasses and macroalgae towards opportunistic algae, as a consequence of increased nutrient concentrations and reduced water transparency, may affect the macrophyte community functional attributes. Moreover, opportunistic algae grow and decompose faster than perennial species and may thereby generate a temporal imbalance between oxygen production and consumption increasing the likelihood of anoxia. Blooms of filamentous algae also reduce the recreational value of coastal waters as they accumulate in the water column and on the beach and anoxic events associated with macroalgal blooms have negative effects on the ecosystem (benthic invertebrates, fish kills, smell etc., Norkko & Bonsdorff 1996).

#### *WFD –associated characteristics*

Annex V of the WFD outlines two macrophyte-related quality element indicators that need consideration:

- Composition of aquatic flora
- Abundance of aquatic flora

According to the WFD (Annex V), a high ecological status is defined as: all disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present, and the levels of macroalgal cover and angiosperm abundance are consistent with undisturbed conditions. At good ecological status, most disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present, and the levels of macroalgal cover and angiosperm abundance show slight signs of disturbance. In moderate status: A moderate number of the disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are absent. Macroalgal cover and angiosperm abundance is moderately disturbed and may be such as to result in an undesirable disturbance to the balance of organisms present in the water body.

*Knowledge gaps*

There is a general need to develop more and better quantitative relations between vegetation indicators and water quality. The current quantitative relations (such as eelgrass depth limit vs. nutrient loading) have only low predictive power and do not provide sufficient foundation for assessing water quality or predicting future status of marine benthic vegetation under contrasting water quality regimes.

In the Mediterranean Sea, the balance between recruitment and mortality rates of *Posidonia oceanica* is a useful indicator for predicting the future status of the slow-growing *Posidonia* meadows, though quantitative relations to water quality have not been established yet.

Quantitative effects of stochastic factors like storms, high temperature or anoxic events on the extent and duration of decline of macrophyte abundance and biomass are difficult to predict, and are not well described in most existing models.

There is a need for identifying of possible threshold levels of eutrophication where decrease of macrophytes is accelerated due to e.g. shading, sediment resuspension, or cascading effects through the food web interactions, or where recruitment/ improvement of the macrophyte communities is delayed or prevented.

Effects of reductions in nutrient load on vegetation indicators. Many long-term data sets have documented the progressive deterioration of benthic vegetation along with increased nutrient load but there is limited knowledge on the opposite process. It would be useful to carefully analyse case studies where nutrient loads have been reduced in order to identify time-scales and pathways of recovery.

### 5.1.4 Benthic invertebrates

*Miguel Gaspar, Susana Carvalho*

The scientific community generally supports the use of benthic macrofaunal communities in the assessment of biological effects of human activities in marine and estuarine environments (Long and Chapman 1985; Radenac et al. 1997; Savage et al. 2001). This widespread support results from some characteristics of benthic animals that can be summarised in:

- are in contact with sea bed sediments (that are recognised as a sink for many contaminants);
- have an important role in cycling nutrients and contaminants;
- are relatively sedentary and long-lived (though providing a measure of the contaminant effects integrated over time);
- are sensitive to anthropogenic activities and respond in a rather predictable way;
- are commercially valuable resources or are relevant food sources for economically or recreationally important species (Rees et al. 1990; Dauer 1993).

Therefore, especially concerning the latter characteristic, the depletion of benthic communities following sediment contamination, dredging or bottom trawling fisheries may represent an important socio-economic problem since they support directly or indirectly several human activities (Rhoads et al. 1978; Chapman et al. 1987; Rees et al. 1990; Kaiser et al. 2000).

### *WFD –associated characteristics*

Annex V of the WFD outline two benthic invertebrate fauna-related elements that need consideration:

- benthic invertebrate composition
- benthic invertebrate abundance

The WFD normative definition indicates that declining ecological quality is associated with differences in composition and abundance of invertebrate taxa from the types-specific communities as well as with a decrease in the level of diversity of invertebrate taxa.

### *Knowledge gaps*

Several works have been focused on the adequate taxonomic resolution to assess environmental disturbances (Warwick 1988; Somerfield and Clarke 1995; Olsgar et al. 1997). According to these authors, for soft bottom macrobenthos, disturbance effects are often detectable at higher taxonomic levels, such as family level, when using multivariate methods. Warwick (1993) also suggested that in environmental impact studies, it would be more important to analyse a larger number of stations/replicates at a higher taxonomic level than a smaller number of stations at the species level.

There is frequently a positive correlation obtained between taxonomic groupings and ecological functionality (Warwick 1993). Several works also assessed macrobenthic impacts due to different sources of disturbance using higher taxonomic levels (Bascom 1982; Berge 1990; Savage et al. 2001; Gaspar et al. 2003).

Many of the techniques used in benthic ecology are complementary, and should be used in combination of others (Elliot 1994). Independently of the methods and taxonomic resolution applied, the study of macrobenthic communities must be complemented by the study of sediment characteristics, such as grain-size analysis and organic matter content.

# 6 TOXIC PRESSURES

*Lena Blom, Eva Brorström-Lundén*

The main sources of toxic substances to water bodies in Europe may be categorised as agriculture, sewage treatment plants, urban runoff, industry, lake/river sediment and landfills. Typically, each of the sources has some particular substances that may affect the water environment because of their toxicity.

The European Community Water Framework Directive, WFD (2000/60/EC) requires the identification of pressures to which water bodies are subject, in particular related to the toxic substances listed in the Annex VIII of the Directive. Further, the member states are required to assess the likelihood, which surface water bodies would fail to meet the environmental quality standards. The WFD defines environmental quality standards as concentrations of pollutants in water, sediment or biota that should not be exceeded in order to protect human health and the environment.

The classification of the ecological status of water bodies, according to the Annex V of the WFD, includes consideration of the concentrations of specific pollutants. For the water body to have a high ecological status, synthetic pollutants should have concentrations close to zero, and non-synthetic pollutants should have concentrations close to the background levels. For the good ecological status, concentrations below the environmental quality standard are required.

The European Commission develops proposals for quality standards applicable to priority substances. For specific pollutants that are not in the priority substance list, the member states should set environmental quality standards (EQS) for water, sediment or biota. When possible, both acute and chronic toxicity data should be used as a basis of the decision. The basic groups of biological organisms to be considered are algae and/or macrophytes, the planktonic crustacean Daphnia spp. (or respective representative organisms for saline waters), and fish.

Depending on the amount and type of toxicity data available, different safety factors are used in the setting of the environmental quality standard as outlined in the Technical Guidance Document on risk assessment (ECB 2003). Data on persistence and bioaccumulation should be taken into consideration, if available.

## 6.1 Overview of WFD priority substances levels in aquatic matrixes

The WFD has identified 32 priority substances (Annex X, decision 2455/2001/EC). The substances were selected on the basis of their risk to the aquatic environment or to human health via the aquatic environment. Effects and biological relations, biological and toxicological effects of the 32 prioritized substances are here investigated. The list of priority substances is presented in Table 6.1.1. together with some information about its uses or emission sources. The present knowledge concerning the environmental occurrence of these substances is scattered, and differs widely between different substances.

Table 6.1.1. Major uses and selected emissions sources and the legal status of WFD priority substances (Sternbeck and Brorström-Lundén 2003).

| Name | Uses or emission sources |
|---|---|
| Polycyclic aromatic hydrocarbons, PAH | Incomplete combustion |
| Anthracene | Incomplete combustion |
| Naphthalene | Incomplete combustion |
| Trichlorobenzene | Industrial chemical |
| Pentachlorobenzene | Unknown uses |
| Hexachlorobenzene, HCB | Biocide; unintended formation |
| Alachlor | Pesticide |
| Atrazine | Pesticide |
| Chlorfenvinphos | Pesticide |
| Diuron | Pesticide |
| Endosulfan | Pesticide |
| Hexachlorocyclohexanes, HCHs | Pesticide |
| (Lindane, gamma-HCH) | Pesticide |
| Isoproturon | Pesticide |
| Chlorpyrifos | Pesticide |
| Pentachlorophenol, PCP | Pesticide |
| Simazine | Pesticide |
| Trifluralin | Pesticide |
| Polybrominated diphenyl ethers, PBDE ## 47, 99, 100 | flame retardant |
| Nonylphenol, 4-para-nonylphenol | Industrial chemical; forms through degradation of NP-ethoxylates |
| Octylphenol, 4-tert-octylphenol | Industrial chemical; forms through degradation of OP-ethoxylates |
| Hexachlorobutadiene, HCBD | Industrial chemical; by-product from chlorinated solvent production |
| C10-13-chloroalkanes, SCCA | Lubricant; cutting fluid |
| Di(2-ethylhexyl)-phthalate, DEHP | Plasticiser |
| Benzene | incomplete combustion; component i petroleum products |
| Trichloromethane (chloroform) | Solvent |
| 1,2-Dichloroethane | Solvent |
| Dichloromethane | solvent in medical industry |
| Tributyltin, TBT | Antifoulant; preservative; stabiliser in plastics; |
| Lead, Pb | Numerous |
| Mercury, Hg | Numerous |
| Nickel, Ni | Numerous |
| Cadmium, Cd | Numerous |

Most of the studies found in the literature report the concentrations of some substances only. A study by IVL considered the levels of all 32 priority substances (Sternbeck and Brorström-Lundén 2003).

The available data in this review are from scientific literature and one Swedish database and covers the areas mostly from Northern Europe. Other national data have so far been difficult to access. In summary, concentrations of priority substances found in the available literature sources divided in water phase, sediments and biota are presented in Table 6.1.2. The data available in water are scarce.

In general, the detectable levels of the 32 priority substances in the water phase are rare, while for metals they are detected more easily. Only in few cases, levels of some organic substances, like polycyclic aromatic hydrocarbons (PAHs), atrazine, diuron, lindane (gamma-HCH), isoproturon, pentachlorophenol (PCP), simazine, polybrominated diphenyl ethers (PBDE ## 47, 99, 100), hexachlorobutadiene (HCBD), benzene and tributyltin (TBT), are reported to be detectable.

For sediments there is more measurements performed for the 32 priority substances in collected data. The grade of detection for the 32 priority substances in sediments is also higher in sediments than water. All the metals are generally detected, and the organic substances are also often found in detectable levels in the sediments. .

In biota all priority substances were reported to be measured in the data of the reviewed sources. In the investigated studies the following toxic substances were found in detectable levels: PAHs, naphthalene, trichlorobenzene, pentachlorobenzene, hexachlorobenzene (HCB), HCHs, gamma-HCH, isoproturon, PCP, PBDEs, HCBD, DEHPs, benzene, trichloroethane, dichloromethane, TBT and all the priority metals.

More concentration data should be available especially in databases on the WFD 32 priority substances, but unfortunately this data was not accessible for us at this point. Within the European Commission the AMPS-group (analysis and monitoring of priority substances) (European Commission, 2004) is working in the Quality Standards setting and guidance on aspects of *e.g.* monitoring for waters, sediment and biota. Their findings should further be used within REBECCA. The purpose of water monitoring is to establish compliance with Environmental Quality Standards (EQS). For comparable monitoring data among member states the same target determination in the same matrix need to be addressed.

Table 6.1.2. Concentrations of the WFD priority substances in the water, sediment and biological matrixes found in the data sources available (mostly from Europe). Data sources are background for all except for the ones marked point * and urban * identified specifically from point and urban sources (n.d. corresponds to substances analyzed, but with values below the detection limit).

| | Water | | Sediment | | Biota | |
|---|---|---|---|---|---|---|
| | min | max | min | max | min | max |
| Name | ng $l^{-1}$ | ng $l^{-1}$ | ng $g^{-1}$ dw | ng $g^{-1}$ dw | ng $g^{-1}$ dw/lipid | ng $g^{-1}$ dw/lipid |
| PAHs | 0.13 | 8500 | 100 | 36000 | 9 | 250 |
| Anthracene | n.d. | | 2 | 210 | 2 | 10 |
| Naphthalene | n.d. | | 2 | 150 | 0.01 | 79 |
| Trichlorobenzene | n.d. | | n.d. | 6 | n.d. | 16 |
| Pentachlorobenzene | n.d. | | 4 | 10 | 0.003 | 16 |
| Hexachlorobenzene | n.d. | | n.d. | 3 | 0.001 | 41 |
| Alachlor | n.d. | | <9 | <13 | n.d. | <50 |
| Atrazine | n.d. | 0.5 | <1 | <2 | n.d. | <50 |
| Chlorfenvinphos | n.d. | | <6 | 236 | n.d. | <50 |
| Diuron | 400 | 600 | n.d. | 30000 | n.d. | <10 |
| Endosulfan | n.d. | | n.d. | 10000 | n.d. | <50 |
| HCHs | n.d. | | n.d. | 12.5 | 5 | 830 |
| (Lindane, gamma-HCH) | n.d. | 70 | 0.06 | 140 | 3.5 | 481 |
| Isoproturon | n.d. | 500 | n.d. | 30 | n.d. | <50 |
| Chlorpyrifos | n.d. | | <1 | <2 | n.d. | |
| PCP urban * | | | 0.2 | 14 | | |
| PCP point * | 5 | 950 | 2 | 644 | 0.19 | 330 |
| PCP | 5 | 3300 | 0.02 | 210 | 20 | 1100 |
| Simazine | n.d. | 200 | <1 | 23 | n.d. | <50 |
| Trifluralin | n.d. | | <3 | <6 | n.d. | |
| PBDE (## 47, 99, 100) | 0.010 | 0.090 | 0.01 | 2.7 | 0.l4 | 30 |
| Nonylphenol | 180000 | 80 | 30 | 5300 | <10 | <60 |
| Octylphenol | n.d. | | n.d. | 140 | n.d. | <10 |
| HCBD point * | n.d. | 950 | | | | 36 |
| HCBD | 0.27 | 3.2 | <1 | 430000 | 0.06 | <50 |
| $C_{10-13}$-chloroalkanes | n.d. | | 0.12 | 3300 | n.d. | |
| DEHP | n.d. | | 25 | 35000 | 350 | 4500 |
| Benzene | 0.05 | 0.3 | <570 | <17000 | <10 | <340 |
| Trichloromethane | | | <1 | <28 | <50 | 260 |
| 1,2-Dichloroethane | | | <100 | <2800 | n.d. | <50 |
| Dichloromethane | | | n.d. | <57000 | 1100 | 3400 |
| TBT | 0.1 | 4.7 | 3.6 | 5800 | 0.3 | 360 |
| Pb | 3 | 15000 | 30000 | 430000 | 2 | 74800 |
| Hg | 0.1 | 9 | 10 | 3300 | 4 | 1050 |
| Ni | 10 | 47600 | n.d. | 63000 | 4 | 3180 |
| Cd | 1 | 1070 | 200 | 7000 | 7 | 66500 |

## 6.2 Overview of the toxic effects of priority substances on biological quality elements

A compilation of data on toxic effects of the priority set substances on organisms representing different trophic levels of aquatic ecosystems (biological quality elements as required for the classification in the WFD) was carried out. The review was based on sources available through data/ literature search and (mostly) from the ECOTOX database of the US EPA (http://www.epa.gov/ecotox/). Furthermore, a number of reports on risk evaluation of the substances were reviewed.

We are aware that this information is not complete and also that data is incomplete due to technical impairments (literature sources may report only parameters in the upper effect scale, whereas levels at the lower scale would be more relevant). Table 6.2.1. shows a summary of the detected impacts of a set of priority substances on groups of biological organism representing different trophic levels of aquatic ecosystems.

All the 32 priority substances are classified within the reversed proposal for a list of priority substances in the context of the water framework directive (COMMPS procedure) (COMMPS, 1999) .

Table 6.2.1. Summary of observed toxic effects of a priority set of substances on biological organisms belonging to the major trophic groups and biological quality elements as required by WFD (cells high-lighted with green indicate observed toxic effects). Phytop.= phytoplankton; Macroph.= macrophytes, Phytobts= phytopbenthos and periphyton; Zoobts= benthic invertebrates and zooplankton (see next page).

| Toxic Substance | Phytop | Macroph. | Phytobts | Zoobts |
|---|---|---|---|---|
| Alachlor | ■ | ■ | | ■ |
| Anthracene | ■ | ■ | | ■ |
| Atrazine | ■ | ■ | ■ | ■ |
| Benzene | ■ | | | ■ |
| Brominated diphenylethers | ■ | | | ■ |
| Cd and its compounds | ■ | ■ | ■ | ■ |
| Chloroalkanes, C10-13- | | | | |
| Chlorfenvinphos | ■ | | | ■ |
| Chlorpyrifos | ■ | ■ | | ■ |
| Dichloroethane, 1,2- | ■ | | | ■ |
| Dichloromethane | ■ | | | ■ |
| Di-(2-ethylhexyl)-phthalate (DEHP) | ■ | | | ■ |
| Diuron | ■ | ■ | | ■ |
| Endosulfan | ■ | | | ■ |
| Endosulfan, alpha- | ■ | | | ■ |
| Hexachlorobenzene | ■ | | | ■ |
| Hexachlorobutadiene | ■ | | | ■ |
| Hexachlorocyclohexane | | | | ■ |
| Lindane, γ-hexachlorocyclohexane | | | ■ | ■ |
| Isoproturon | ■ | ■ | ■ | ■ |
| Pb and its compounds | ■ | | | ■ |
| Hg and its compounds | ■ | | | ■ |
| Naphthalene | ■ | | | ■ |
| Nickel and its compounds | | | | ■ |
| Nonylphenols | ■ | ■ | ■ | ■ |
| Nonylphenol, 4- | ■ | | ■ | ■ |
| Octylphenols | | | | ■ |
| Octylphenol, 4-tert- | | | | ■ |
| Pentachlorobenzene | ■ | | | ■ |
| Pentachlorophenol | ■ | ■ | | ■ |
| Polycyclic aromatic hydrocarbons | | | | |
| Benzo(a)pyrene | ■ | | | ■ |
| Benzo(b)fluoroanthene | | | | ■ |
| Benzo(g,h,i)perylene | | | | ■ |
| Benzo(k)fluoroanthene | | | | ■ |
| Fluoroanthene | ■ | ■ | | ■ |
| Indeno(1,2,3-cd)pyrene | | | | |
| Simazine | ■ | ■ | ■ | ■ |
| Tributyltin compounds | ■ | ■ | | ■ |
| Tributyltin cation | | | | |
| Trichlorobenzenes (1,2,3-; 1,2,4- and 1,3,5-) | | | | ■ |
| Trichlorobenzene, 1,2,4- | ■ | | | ■ |
| Trichloromethane, chloroform | ■ | | | ■ |
| Trifluralin | ■ | ■ | | ■ |

*Impacts of seagrasses and macroalgae*

Existing knowledge on effects of toxic substances on seagrasses have recently been summarised by Short and WyllieEcheverria (1996), by Hemminga and Duarte (2000), and for Baltic biota by Szefer (2002). Only oil has been observed to have large-scale impacts on sea grasses. Various degrees of injury to sea grasses have been observed from plant mortality and declines in plant cover to virtually no losses at all. Although several studies have demonstrated the ability of many sea grass species to bioaccumulate heavy metals from contaminated water or sediment, there is no evidence as yet of adverse effects of heavy metal exposure on sea grasses. In contrast, herbicides have been reported to be particularly detrimental to mangroves and sea grasses in tropical marine ecosystems (Peters et al. 1997).

In plants and macrophytes chlorate is rendered toxic by conversion to chlorite via reduction of nitrate reduction. In the Baltic Sea reduced distribution and abundance of the brown macroalga Fucus vesiculosus in the proximity of pulp mills where chlorate is discharged to the surroundings has been demonstrated (Rosemarin et al. 1994). Baltic Sea algae are mostly nitrogen limited and chlorate can easily take a place of nitrate. Chlorate plus nitrate interact such that cells do not discriminate between the two. Results from a mesocosm test system indicated that the whole group of brown algae was sensitive to chlorate. Further there were no differences between annual and perennial brown algae. Metabolic of this group must be studied to understand the results. Why other algal groups did not react to moderate input of chlorate?

The literature includes many examples of analyses of e.g. heavy metal content in the tissue of benthic vegetation and macroalgae have often been suggested bioindicators of the presence of such substances as they are able to accumulate high concentrations of the substances in their tissue. This ability seems to imply that the species in question are tolerant to the given toxic substances. Measures of tissue concentrations of selected substances are therefore appropriate indicators of the presence of toxic substances in the habitat but inappropriate indicators of the ‘health’ of the benthic communities unless they can be related to an effect on e.g. distribution, abundance, composition or growth processes of the plants.

Most existing knowledge regards accumulation of toxic substances in algae and sea grasses rather than possible effects on the benthic vegetation. The high levels of accumulation suggest that some species are extremely tolerant to toxic substances, but information on tolerance limits and effect of toxic substances on benthic vegetation is lacking.

The summary of the observed trends of a few toxic substances concentrations in some marine indicator biota (i.e. Baltic herring, NE Atlantic cod and mussels and Mediterranean mussels; EEA 2003) indicates that there is very little information of the concentrations and trends of Lindane in marine biota, and also lack of data on the concentrations of DDT and PCBs in Mediterranean (Figure 6.2.1).

Summary of trends in concentrations in biota in Baltic Sea, the North East Atlantic Ocean, and Mediterranean sea — Table 1

| | Baltic Herring | NE Atlantic Cod | NE Atlantic Mussels | Mediterranean Mussels |
|---|---|---|---|---|
| Cadmium | 😐 | 😐 | 🙂 | 🙂 |
| Mercury | 😐 | 😐 | 🙂 | 🙂 |
| Lead | 😐 | 🙂 | 🙂 | 🙂 |
| DDT | 🙂 | 😐 | 🙂 | ni |
| PCBs | 🙂 | ☹ | 🙂 | ni |
| Lindane | ni | ni | ni | 🙂 |

🙂 inconsistent but decreasing trend;

😐 no trend;

☹ upward trend;

ni = no information

Muscle analysed in herring; liver analysed in cod except for mercury where muscle data was used.

Source: complied by ETC/WTR from OSPAR, Helcom and EEA Mediterranean member countries

**Figure 6.2.1.** Summary of the observed trends of a few toxic substances concentrations in some marine indicator biota (i.e. Baltic herring, NE Atlantic cod and mussels and Mediterranean mussels; EEA 2003).

## 6.3 Overview of current models for assessing toxic impacts

The potential use of modeling methods in relation to toxic substances and the WFD includes fugacity modeling and Quantitative Structure Activity Relationships (QSARs). The fugacity models can provide a link between pressures and concentrations of specific pollutants in the environment. Thus, they have a potential applicability in assessing the likelihood that surface water bodies will fail to meet the environmental quality standards and can be used to estimate critical target loads.

QSARs are theoretical models that can be used to predict the physicochemical, environmental and biological properties of molecules. The notion that there is a relation between chemical structure and biological activity is not new and QSARs have been developed for over a hundred years (Schultz et al. 2003). In the last 20 years there has been an enormous increase in the application of these models, due to the rapid increase in computing power and new methodology for molecular modeling and statistical analysis.

With respects of the WFD implementation requirements, the QSAR models can provide:

- estimates of acute and/or chronic toxicity of substances to different trophic levels in cases where experimental data is missing, which can be used when setting EQS.
- data on persistence and bioaccumulation in cases where experimental data is missing, which can be used when setting EQS.
- QSAR models are frequently used to estimate physical parameters needed for fugacity modelling, i.e. partition coefficients, persistence and other parameters, since experimental data of this type is often scarce.

Moreover, the fugacity models can provide a link between pressures and concentrations of specific pollutants in the environment. Thus, they have a potential applicability in assessing the likelihood that surface water bodies will fail to meet the environmental quality standards and can be used to estimate critical target loads.

The development of reliable QSARs for toxicity requires a large amount of data from toxicity test that have been performed on different substances using a standardized protocol. This prerequisite cannot be fulfilled for a large number of species. Indeed, the primary reason for using toxicity QSARs is the lack of available toxicity data! In chemical risk assessment, one or a few species from each different trophic level are used to represent the toxicity to all species in that level. Most regulatory protocols include acute lethality for fish and planktonic crustacean, Daphnia spp., and chronic or reproduction data for algae, fish and Daphnia spp. Currently, QSARs are available for most substance groups for acute toxic effects to organisms used in chemical risk assessment.

Deteriorated ecological status may be caused also by other substances than the priority substances. QSARs models are potentially more useful in cases of other toxic substances since experimental data on properties and effects of the priority substances are, in general, much better known.

Endocrine disrupting chemicals (EDCs) are an area of growing concern in chemical risk assessment. Most of the end-points related to endocrine disruption used in QSAR modeling are in vitro tests related to receptor binding or biological effects of receptor binding, e.g. gene activation. QSARs are considered important and potentially useful due to the complexity and cost of in vitro and in vivo tests for endocrine disruption.

The current status of QSARs for endocrine disruption (Schmieder et al. 2003) are that most models are developed for specific substance groups. Some models have been developed based on diverse sets of chemicals with the purpose of screening new chemicals for EDCs. Focus is on the human estrogen receptor (hER) with few applications to other human receptors and ERs from rodents and calf. No QSARs for endocrine disruption in aquatic organisms have been found.

Van Leeuwen et al at University of Utrecht (Van Leeuwen and Vanderzandt 1992) have used QSAR estimates of toxicity of narcotic chemicals to predict no-effect levels (NELs) at the ecosystem level by means of recently developed extrapolation methods. Equilibrium partitioning theory was used to derive NELs for aquatic sediments and internal toxicant concentrations for aquatic organisms. Calculations were carried out for 102 narcotic substances.

## 6.4 Summary of knowledge gaps and further research needs

With the evolution in computational methodology and capacity that have made 3D molecular calculations feasible on desktop computers, the fundamental challenge is to build the required toxic effect databases that can be used for modeling. It is a common understanding that a main factor limiting further QSAR development is the lack of data needed to build new models (Bradbury et al. 2003, ECETOC 2003, Russom et al. 2003). Important points to consider is uncertainty estimates in reference data and testing according to standardized protocols.

The availability of QSARs for chronic predicted no-effect concentrations (PNEC) is very limited. In the absence of measured PNECs, these are frequently calculated by using an assessment factor and predicted acute effect concentrations. In the risk assessment an

assessment factor of 1000 is used in the EU to reflect the large uncertainty in extrapolating acute effect data to no-effect concentrations. If data on chronic tests is available, lower assessment factors can be used (ECB 2003).

There is considerable incentive to develop QSARs for aquatic PNEC but the work in this field has only started. The ECETOC report (ECETOC 2003) discusses this issue extensively, and lists a number of possible advantages compared with current practices, and recommendations for further research. QSARs model development for PNEC is a priority for further research especially in relation to the applicability for the WFD.

There are gaps in QSAR toxicity model availability both with respect to chemical substance groups within some end-points and with respect to availability of models for certain end-points (Schultz et al. 2003). Few QSARs available are valid for non-organic chemicals (Comber et al. 2003) and also their validity for metallo-organic chemicals is limited.

Most of the QSAR work has been devoted to assess the toxicity on freshwater organisms. There are very few QSARs models for toxicity to estuarine and marine organisms and the quality of these models is generally lower than for freshwater organisms (Comber et al. 2003; ECETOC 2003). This also applies for sediment-dwelling organisms. The need for development of new QSARs for these organism groups is evident.

Approximately 70% of all industrial chemicals are considered to act acutely via non-polar or polar narcosis. In general, non-polar narcosis, based on log $K_{OW}$, can be modeled only for individual groups of chemicals. A remaining challenge is to model acute and chronic effects of chemicals whose toxicity is elicited through other mechanisms such as covalent binding, receptor interaction, etc. (Bradbury et al. 2003).

There are only few models to estimate chronic toxicity and their validation is insufficient. Currently, QSARs models cannot be used for estimation of toxicity to sediment-living organism or for the chronic toxic effects in the aquatic environment in general. A significant obstacle to further development is the lack of databases with reference toxicity data.

Further, there are no models available to illustrate the toxic effects on population levels in the aquatic ecosystems.

All methods discussed here focus on single chemicals and to not take into account additive, synergistic or antagonistic effects. This is a significant problem for practical use of QSARs for the application of WFD purposes. Mode of action-based QSARs do, however, open a possibility in this direction. Testing of this approach would be an interesting area for further research.

Mackay and collaborators (Mackay et al. 2003) suggested that the back-tracking of models is used for risk assessment of chemicals in cases where emissions are uncertain or unknown. The approach is also potentially useful in the context of the WFD. The load (emission) estimates are often very uncertain but if a PNEC (or similar) concentration from a QSAR model can be used as a starting point for reverse calculation by a fugacity model. The calculation (which will probably be of a trial-and-error type) produces an estimate of the critical load that would give an effect. It is then usually feasible to see if the actual loads are of this magnitude or much smaller. This enables investigation of which substances that can be responsible for an observed ecological effect even in the absence of analytical data (i.e. environmental concentrations) and how much the loads have to be decreased to lower the concentration of the substance below the PNEC:

Comber and collaborators (Comber et al. 2003) point out the need for improvement of existing models for soil-water partitioning and sediment-water partitioning. The most likely area to further increase the usefulness of QSARs for soil and sediment sorption is not through improved QSAR models for organic carbon normalized partition coefficients, $K_{OC}$,

but through improved understanding of the sorption process and inclusion of other sediment properties than organic carbon content.

Further, existing models do not cover sufficiently abiotic degradation processes (photolysis, hydrolysis) in the aquatic environments.

The QSARs models for biodegradation are primarily of classification type while the required input to fate models, e.g. EUSES, is often quantitative. Thus there is a need for new QSARs for biodegradation half-lives (Comber et al. 2003; ECETOC 2003). It should be noted, however, that several factors, e.g. the multitude of degradation mechanism and inaccurate reference data, could be expected to make QSAR modeling difficult for this end-point.

QSAR models can predict bioconcentration factors approximately. The need to account for metabolism is evident and it is a weak point of the existing models. There are no models available for bioaccumulation.

## 6.5 Recommendations for focus of the research efforts in REBECCA

Although the lack of toxicity models for marine, estuarine and sediment living organisms is evident and the development in this area would be very useful for the WFD implementation purposes, it is our opinion that an effort in this field within REBECCA would not give significant results. The lack of reference toxicity data is evident and the REBECCA project does not include experimental toxicity testing.

It is extremely difficult to have a complete ecological status overview, since there is a substantial lack of information and surveillance systems from many European countries. There also seems to be a substantial problem in communication in the EU across the national borders, since it is very difficult to get data from different on-going monitoring programs from countries outside REBECCA consortia. Most of the available international data is to be extracted from the scientific journals.

To measure all compounds in water is not always recommendable *e.g.* if a substance rather participates or partitions to sediment or bioaccumulates then it is higher concentrations in the sediment and/or in the biota. The properties of the particular substance must be accounted for in the monitoring procedures.

It is recommended that the research efforts within REBECCA should be focused on:

- Development of methods for estimating reliability of QSAR predictions. There are a large number of models that provide good predictions for large sets of substances but are not accepted or trusted. Development of prediction diagnostics for new and existing models can considerably increase the reliability and acceptance for such models and thus promote their use.
- QSAR development for PNEC. Practically all toxicity models are developed for lethal or effect concentrations, but actually the no-effect concentrations are relevant for the water quality objectives and ecological status assessment of the WFD. The small amount of work done in this field so far suggests the feasibility of this approach.
- Work towards "one database" for concentration data on surface waters within the member sates on the 32 priority substances accessible through the European Commission.
- The recommendations for analysis and monitoring of toxic substances within REBECCA should be based on the results of the European Commission AMPS-group.

- Within the recommendation for monitoring within WFD in compliance of EQS, measure where the substance of interest may be found ether it is water, sediment or biota.

## 7 BIBLIOGRAPHY CITED THROUGH THE TEXT


AMBIO, 32, 2003. Entire volume.

Anderson, D. M., P. M. Gilbert, and J. M. Burkholder. 1993. Harmful algal blooms and eutrophication: nutrient sources, composition and consequences. Estuaries:704-726.

Appelberg, M., Henrikson, B., Henrikson, L. and Svedäng, M., 1993. Biotic interactions within the littoral community of Swedish forest lakes during acidification. AMBIO 22: 290-297.

Armitage, P. D., P. S. Cranston, and L. C. V. Pinder, editors. 1995. The Chironomidae: Biology and Ecology of Non-Biting Midges. Chapman & Hall, London.

Arts, G. H. P. 2002. Deterioration of atlantic soft water macrophyte communities by acidification, eutrophication and alkalinisation. Aquatic Botany 73:373-393.

Axelman, J., D. Broman, and C. Näf. 2000. Vertical flux and particulate/water dynamics of polychlorinated biphenyls (PCBs) in the Open Baltic Sea. Ambio 29:210-216.

Baines, S. T., and M. Pace. 1994. Relationships between suspended particulate matter and sinking flux along a trophic gradient and implication for the fate of planktonic primary production. Canadian Journal of Fisheries and Aquatic Sciences 51:25-36.

Baker, J.P., Creager, C.S. and Christensen, S.W., 1987. Identification of Critical Values for Effects of Acidification on Fish Populations. USEPA Project E5-25, Washington DC, 155 pp.

Bascom, W. 1982. The effects of waste disposal on the coastal waters of southern California. Environnmental Science and Technology 16:226-236.

Berge, J. A. 1990. Macrofauna recolonization of subtidal sediments. Experimental studies on defaunated sediment contaminated with crude oil in two Norwegian fjords with unequal eutrophication status. I. Community responses. Marine ecology progress series 66:103-115.

Biggs, B. J. F. 1990. The use of relative specific growth rates of periphytic diatoms to assess enrichment of a stream. New Zealand Journal of Marine and Freshwater Research 24:9-18.

Bothwell, M. L. 1988. Growth rate responses of lotic periphytic diatoms to experimental phosphorus enrichment: the influence of temperature and light. Canadian Journal of Fisheries and Aquatic Sciences 45.

Bradbury, S. P., C. L. Russom, G. T. Ankley, T. W. Schultz, and J. D. Walker. 2003. Overview of data and conceptual approaches for derivation of quantitative structure-activity relationships for ecotoxicological effects of organic chemicals. Environmental Toxicology and Chemistry 22:1789-1798.

Brinkhurst, R. O. 1974. The benthos of lakes. St Martins Press, New York.

Buiteveld H. 1995. A model for calculation of diffuse light attenuation (PAR) and Secchi depth. Neth J Aquat. Ecol. 29: 55-65

Bulger, A.J., Lien, L., Cosby, B.J. and Henriksen, A., 1993. Brown trout (Salmo trutta) status and chemistry from the Norwegian 1000 lake survey – statistical analysis. Can.J Fish.Aquat Sci. 50:575-585.

Cadee, G. C., and J. Hegeman. 2002. Phytoplankton in the Marsdiep at the end of the 20th century; 30 years monitoring biomass, primary production, and Phaeocystis blooms. Journal of Sea Research 48:97-110.

Cairns, J. J., and J. R. Pratt. 1993. A history of biological monitoring using benthic Macroinvertobrates. Pages 10-27 *in* D. M. Rosenberg and V. H. Resh, editors. Freshwater Biomonitoring and Benthic Macroinvertobrates. Chapman & Hall, New York, NY.

Carvalho, L., H. Bennion, A. Darwell, I. Gunn, A. Lyle, D. Monteith, and M. Wade. 2002. Physico-chemical Conditions for Supporting Different Levels of Biological Quality for the Water Framework Directive for Freshwaters. Report to the Environment Agency.

Chapman, P. M., R. N. Dexter, and E. R. Long. 1987. Synoptic measures of sediment contamination, toxicity and infaunal community composition (the sediment quality triad) in San Francisco Bay. Marine ecology progress series 37:75-96.

Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine ecology progress series 210:223-253.

Comber, M. H. I., J. D. Walker, C. Watts, and J. Hermens. 2003. Quantitative structure-activity relationships for predicting potential ecological hazard of organic chemicals for use in regulatory risk assessments. Environmental Toxicology and Chemistry 22:1822-1828.

Conley, D. J. 1999. Biogeochemical nutrient cycles and nutrient management strategies. Hydrobiologia 410:87-96.

Constanza, R., R. d'Argue, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387:253-260.

Cummins, K. W. 1973. Trophic relations of aquatic insects. Annual Review of Entomology 18:183-206.

Dauer, D. M. 1993. Biological criteria, environmental health and estuarine macrobenthic community structure. Marine Pollution Bulletin 26:249-257.

de Jonge, V., M. Elliott, and E. Orive. 2002. Causes, historical development, effects and future challenges of a common environmental problem: eutrophication. Hydrobiologia 475-476:1-19.

Doudoroff, P., and D. L. Shumway. 1970. Dissolved oxygen requirements of freshwater fishes. FAO Fisheries Technical Paper No 86.

Duarte, C. M. 1995. Submerged Aquatic Vegetation in Relation to Different Nutrient Regimes. Ophelia 41:87-112.

ECB. 2003. Technical Guidance Document (TGD) on risk assessment of chemical substances following European regulations and directives. European Chemicals Bureau, Ispra.

ECETOC. 2003. (Q)SARs: Evaluation of the commercially available software for human health and environmeental endpoints with respect to chemical management applications. Technical report no 89, European Centre for Ecotoxicology and Toxicology of Chemicals, Brussels.

ECOSTAT. 2003. Overall Approach to the Classification of Ecological Status and Ecological Potential. European Commission. Available at: http://forum.europa.eu.int/Public/irc/env/wfd/library

ECOSTAT 2004. *Guidance on the Intercalibration process*. European Commission. Available at: http://forum.europa.eu.int/Public/irc/env/wfd/library

EEA. 2003. Europe's water: An indicator-based assessment. European Environment Agency.

Eichenberger, E. and K. Wuhrmann 1966. Über Jahreszeitliche Veränderungen der Besiedlungsdichte in Modellfliessgewässer mit verschiedene Abwasserbelastung. Verhlungen of Internationale Vereinigung fur Limnologie 16:888-896.

Elliot, M. 1994. The analysis of macrobenthic community data. Marine Pollution Bulletin 28:62-64.

Henriksen, A., Fjeld, E. and Hesthagen, T., 1999. Critical load exceeding and damage to fish populations. Ambio 28:583-586, 1999.

Extence, C. A., D. M. Balbi, and R. P. Chadd. 1999. River flow indexing using British benthic macroinvertebrates: a framework for setting hydroecological objectives. Regulated Rivers Research and Management 15:543-574.

Fazi, S., and L. Rossi. 2000. Effects of macro-invertebrate on leaf detritus processing rate: a macrocosm experiment. Hydrobiologia 435:127-134.

Findlay, S., J. Tank, S. Dye, H. M. Valett, P. J. Mulholland, W. H. McDowell, S. L. Johnson, S. K. Hamilton, J. Edmonds, W. K. Dodds, and W. B. Bowden. 2002. A cross-system comparison of bacterial and fungal biomass in detritus pools of headwater streams. Microbial Ecology 43:55-66.

Fjerdingstad, E. 1964. Pollution of streams estimated by benthal phytomicroorganisms. I. A saprobic system based on communities of organisms and ecological factors. International Review ges. Hydrobiologie 49:63-131.

Gaspar, M. B., F. Leitao, M. N. Santos, L. Chicharo, M. A. Dias, A. Chicharo, and C. C. Monteiro. 2003. A comparison of direct macrofaunal mortality using three types of clam dredges. Ices Journal Of Marine Science 60.

Gensemer, R.W. and Playle, R.C., 1999. The bioavailability and toxicity of aluminum in aquatic environments. Critical Reviews In Environmental Science And Technology 29:315-450.

Goedkoop, W., and R. K. Johnson. 1996. Pelagic-benthic coupling:profundal benthic community response to spring diatom deposition in mesotrophic Lake Erken. Limnology and Oceanography 41:636-647.

Hallegraeff, G. 1993. A review of harmful algal blooms and their apparent global increase. Phycologia 32:79-99.

Hecky, R. E., and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichments. Limnology and Oceanography 33.

Heinis, F., and C. Davids. 1993. Factors governing spatial and temporal distribution of chironomid larvae in the Maarsseveen lakes with special emphasis on the role of oxygen conditions. Netherlands Journal of Aquatic Science 27:21-34.

Hemminga, M. A., and C. M. Duarte. 2000. Sea grass ecology.

Holmes RW. 1970. The Secchi disk in turbid coastal waters. Limnol. Oceanogr. 15: 688-694.

Hough, R. A., and M. D. Forwall. 1988. Interactions of inorganic carbon and light availability as controlling factors in aquatic macrophyte distribution and productivity. Limnology and Oceanography 33:1202-1208.

Humborg, C., D. J. Conley, L. Rahm, F. Wulff, A. Cociasu, and V. Ittekkot. 2000. Silicon retention in river basins: Far-reaching effects on biogeochemistry and aquatic food webs in coastal marine environments. Ambio 29:45-50.

Jickells, T. D. 1998. Nutrient biogeochemistry of the coastal zone. Science 281:217-222.

Johnston, N. T., C. J. Perrin, P. A. Slaney, and B. R. Ward. 1990. Increased Juvenile Salomnid Growth by Whole-River Fertilisation. Canadian Journal of Fisheries and Aquatic Sciences 47:862-872.

Jonasson, P. M. 1996. Limits for the life in the lake ecosystem. Verhandlungen der internationale Vereiningung fur Theoretische und Angewandte Limnologie 20:1-33.

Jones, F. H. 1984. The dynamics of suspended algal populations in the lower Wye catchment. Water Research 18:25-35.

Jørgensen, B. B. 1996. Case Study - Aarhus Bay. Pages 137-154 *in* B. B. Jørgensen and K. Richardson, editors. Eutrophication in Coastal Marine Ecosystems. American Geophysical Union, Washington, DC.

Juggins, S., 2001. The CLAM biological-chemical database: the development and application of biological models to predict taxon distribution from SSWC and MAGIC hydrochemical models. In Curtis, C. & Simpson, G. [ed.] Acidification of Fresh Waters: the role of nitrogen and the prospects for recovery. A report to the Department of the Environment Food and Rural Affairs. ECRC, University College London.

Justic, D., N. N. Rabalais, R. E. Turner, and Q. Dortch. 1995. Changes in nutrient structure of river-dominated coastal waters:stoichiometric nutrient balance and its consequences. Estuarine, Coastal and Shelf Science 40:339-356.

Kaiser, M. J., E. F. Spence, and P. J. Hart. 2000. Fishing-gear restrictions and conservation of benthic habitat complexity. Conservation Biology 14:1512-1525.

Kampa, E. and W. Hansen. 2004. Heavily modified water bodies, Synthesis of 34 Case Studies in Europe. Springer 321 p.

Kirk JTO. 1983. Light and Photosynthesis in Aquatic Ecosystems. Cambridge Univ. Press, 401 pp.

Kolkwitz, R., and M. Marsson. 1902. Grundsätze für die biologische Beurteilung des Wassers nach seine Flora und Fauna. Mitt. Prüfungsanst. Wasserversorg. Abwasserreinig 1:1-64.

Larsen, J., Birks, H.J.B., Raddum, G.G. & Fjellheim, A., 1996. Quantitative relationships of invertebrates to pH in Norwegian river systems. Hydrobiologia, 328: 57-74.

Leland, H. V. 1995. Distribution of phytobenthos in the Yakima river basin, Washington, in relation to geology, land-use and other environmental-factors. Canadian Journal of Fisheries and Aquatic Sciences 52:1108-1129.

Liebmann, H. 1951. Handbuch der Frischwasser und Abwasserbiologie, München.

Lien, L., Raddum, G.G., Fjellheim, A. and Henriksen, A., 1996. A critical limit for acid neutralizing capacity in Norwegian surface waters, based on new analyses of fish and invertebrate responses. Science of the Total Environment, 177: 173-193.

Long, E. R., and P. M. Chapman. 1985. A sediment quality triad: measures of sediment contamination, toxicity and infaunal community in Puget Sound. Marine Pollution Bulletin 16:405-415.

Lindstrøm E.A., Brettum, P., Johansen, S.W. and Mjelde, M., 2004. Freshwater vegetation in Norway. Critical limits to acidification – effects of liming. NIVA-report 4821: 132p.

Mackay, D., J. Hubbarde, and E. Webster. 2003. The role of QSARs and fate models in chemical hazard and risk assessment. Paper prepared for Quantitative Structure-Activity

Relationships (QSAR) Proceedings of the QSAR 2002 Conference, Ottawa May 2002. Qsar & Combinatorial Science 22:106-112.

Madsen, T.V., Olsen, B., Bagger, J., 2002. Carbon acquisition and carbon dynamics by aquatic isoetids. Aquatic Botany 73: 351-371.

Mainstone, C. P., and W. Parr. 2002. Phosphorus in rivers- ecological and management. Science of the Total Environment 282-283:25-47.

Meeuwig, J. J., P. Kauppila, and H. Pitkanen. 2000. Predicting coastal eutrophication in the Baltic: a limnological approach. Canadian Journal of Fisheries and Aquatic Sciences 57:844-855.

Meyer, J. L. 1994. The microbial loop in flowing waters. Microbial Ecology 28:195-199.

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

Nielsen, S. L., K. Sand-Jensen, J. Borum, and O. Geertz-Hansen. 2002. Phytoplankton, Nutrients, and Transparency in Danish Coastal Waters. Estuaries 25:930-937.

Norkko A & E Bonsdorff. 1996. Altered benthic prey-availability due to episodic oxygen deficiency caused by drifting algal mats. P.S.Z.N.I. Mar Ecol. 17: 355-372

OECD. 1982. Eutrophication of Water: Monitoring, Assessment and Control. Organisation for Economic Cooperation and Development, Paris.

Olsen, Y., H. Reinertsen, O. Vadstein, T. Andersen, I. Gismervik, C. Duarte, S. Agusti, H. Stibor, U. Sommer, R. Lignell, T. Tamminen, C. Lancelot, V. Rousseau, E. Hoell, and K. A. Sanderud. 2001. Comparative analysis of food webs based on flow networks: effects of nutrient supply on structure and function of coastal plankton communities. Continental Shelf Research 21:2043-2053.

Olsgar, F., P. J. Somerfield, and M. R. Carr. 1997. Relationships between taxonomic resolution, macrobenthic community patterns and disturbance. Marine ecology progress series 172:25-36.

Ormerod, S.J. (1993) Assessing critical levels of acid neutralizing capacity (ANC) for stream organisms using empirical data. In: (Eds. M. Hornung and R.A. Skeffington) Critical Loads: Concepts and Applications. London, HMSO: 115-118.

Paerl, H. W. 1998. Structure and function of anthropogenic ally altered microbial communities in coastal waters. Current Opinion in Microbiology 1:296-302.

Paerl, H. W., and D. R. Whitall. 1999. Derived-derived atmospheric nitrogen deposition, marine eutrophication and harmful algal bloom expansion: Is there a link? Ambio 28:307-311.

Pedersen, M. F. 1995. Nitrogen limitation of photosythesis and growth: comparison across plant communities in a Danish estuary (Roskilde Fjord). Ophelia 41.

Perrin, C. J., M. L. Bothwell, and P. A. Slaney. 1987. Experimental enrichment of a coastal stream in British Columbia: effects of organic and inorganic additions on autotrophic periphyton production. Canadian Journal of Fisheries and Aquatic Sciences 44:1247-1256.

Peters, E. C., N. J. Gassman, J. C. Firman, R. H. Richmond, and E. A. Power. 1997. Ecotoxicology of tropical marine ecosystems. Environmental Toxicology and Chemistry 16:12-40.

Philippart, C. J. M. 1994. Eutrophication as a possible cause of decline in the sea grass Zostera noltii of the Dutch Wadden sea. University of Nijmegen.

Pryor, S. C., and R. J. Barthelmie. 2000. Particle dry deposition to water surfaces: Processes and consequences. Marine Pollution Bulletin 41:220-231.

Radenac, G., P. Miramand, and J. Tardy. 1997. Search for impact of a dredged material disposal site on growth and metal contamination of Mytilus edulis (L.) in charente-Maritime (France). Marine Pollution Bulletin 34:721-729.

Real, M. 2000. Chironomus species (Diptera, Chironomidae) in the profundal benthos of Spanish reservoirs and lakes: factors affecting distribution patterns. Freshwater Biology 43:1-18.

Reckhow, K.H., Black, R.W., Stockton, T.B. Jr., Vogt, J.D. and Wood, J.G. (1987) Empirical models of fish response to lake acidification. Canadian Journal of Fisheries and Aquatic Science, 44: 1432-1442.

Rees, H. L., D. C. Moore, T. H. Pearson, M. Elliot, M. Servile, J. Pomfret, and D. Johnson. 1990. Procedures for the monitoring of marine benthic communities at UK sewage sludge disposal sites.

Reice, S. R., and M. Wohlenberg. 1993. Monitoring freshwater benthic macroinvertebrates and benthic processes: measures for assessment of ecosystem health. *in* V. H. R. D.M. Rosenberg, editor. Freshwater Biomonitoring And Benthic Macroinvertebrates. Chapman And Hall, New York.

Rhoads, D. C., P. L. McCall, and J. Y. Yingst. 1978. Disturbance and production on the estuarine sea floor. American Scientist 66:577-578.

Roelofs, J. G. M., E. Brouwer, and R. Bobbink. 2002. Restoration of aquatic macrophyte vegetation in acidified and eutrophicated shallow soft water wetlands in the Netherlands. Hydrobiologia 478:171-180.

Rosemarin, A., K.-J. Lehtinen, M. Notini, and J. Mattson. 1994. Effects of pulp mill chlorate on Baltic sea algae. Environmental Pollution 85:3-13.

Rosseland, B.O. and Staurnes, M., 1994. Physiological Mechanisms for Toxic Effects and Resistance to Acidic Water: An Ecophysiological and Ecotoxicological Approach. In: Acidification of Freshwater Ecosystems: Implications for the Future, edited by C. E. W. Steinberg and R.F. Wright. John Wiley & Sons Ltd.:227-246.

Russom, C. L., R. L. Breton, J. D. Walker, and S. P. Bradbury. 2003. An overview of the use of quantitative structure-activity relationships for ranking and prioritizing large chemical inventories for environmental risk assessments. Environmental Toxicology and Chemistry 22:1810-1821.

Ryan, S. K., and J. R. Curtis. 2003. Integrating bioassessment and ecological risk assessment: an approach to developing numerical water-quality criteria. Environmental Management 31:795-809.

Sakamoto, M. 1966. Primary Production by the Phytoplankton Community in Some Japanese Lakes and Its Dependence on Lake Depth. Archiv fur Hydrobiologie 62:1-28.

Sams, R. E., and K. R. Conover. 1969. Water quality and migration of fall salmon in the lower Willamotte river. Fish Commission of Oregon, Portland.

Sandén, P., and B. Håkansson. 1996. Long-term trends in Secchi depth in the Baltic Sea. Limnolory and Oceanography 41:346-351.

Sand-Jensen, K., and J. Borum. 1991. Interactions among phytoplankton, periphyton and macrophytes in temperate freshwaters and estuaries. Aquatic Botany 41:137-176.

Sas, H. 1989. Lake restoration by reduction of nutrient loading. Academia Verlag Richartz GmbH. St. Augustin.

Savage, C., J. G. Field, and R. Warwick. 2001. Comparative meta-analysis of the impact of offshore marine mining on macrobenthic communities versus organic pollution studies. Marine ecology progress series 221:265-275.

Scheffer, M., S.H. Hosper, M.L. Meijer, B. Moss & E. Jeppesen, 1993: Alternate equilibria in shallow lakes. Trends in Ecology and Evolution 8:275-279.

Schindler, D. W. 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. Science 184:897-899.

Schmieder, P. K., G. Ankley, O. Mekenyan, J. D. Walker, and S. Bradbury. 2003. Quantitative structure-activity relationship models for prediction of estrogen receptor binding affinity of structurally diverse chemicals. Environmental Toxicology and Chemistry 22:1844-1854.

Schultz, T. W., M. T. D. Cronin, J. D. Walker, and A. O. Aptula. 2003. Quantitative structure-activity relationships (QSARs) in toxicology: a historical perspective. Journal of Molecular Structure-Theochem 622:1-22.

Sculthorpe, C. D. 1967. The Biology of Aquatic Vascular Plants. Edward Arnold, London.

Short, F. T., and S. WyllieEcheverria. 1996. Natural and human-induced disturbance of sea grasses. Environmental Conservation 23:17-27.

Shtereva, G., S. Moncheva, V. Doncheva, O. Christova, and I. Shterev. 1999. Changes in chemical parameters in the Bulgarian Black Sea coastal area as an indication of the ecological state of the environment. Water Science and Technology 39:37-45.

Skei, J., P. Larsson, R. Rosenberg, P. Johnson, M. Olsson, and D. Broman. 2000. Eutrophication and contamination in aquatic systems. Ambio 29:184-194.

Sladecek, V. 1973. System of water qualification from the biological point of view. Archiv für Hydrobiologie 7: 1-218.

Smolders, A. J. P., E. C. H. E. T. Lucassen, and J. G. M. Roelofs. 2002. The isoetid environment: biogeochemistry and threats. Aquatic Botany 73:325–350.

Snyder, E. B., C. T. Robinson, G. W. Minshall, and S. R. Rushforth. 2002. Regional patterns in periphyton accrual an diatom assemblage structure in a heterogeneous nutrient landscape. Canadian Journal of Fisheries and Aquatic Sciences 59:564-577.

Somerfield, P. J., and K. R. Clarke. 1995. Taxonomic levels, in marine community studies, revisited. Marine ecology progress series 127:113-119.

Sparling, D.W. and Lowe, T.P., 1996. Environmental hazards of aluminum to plants, invertebrates, fish and wildlife. Rev.Environ. Contam. Toxicol. 145:1-127.

Sternbeck, J., and E. Brorström-Lundén, et al. 2003. WFD priority substances in sediments from Stockholm and the Svealand coastal region. IVLreport.

Stoddard, J.L., Jeffries, D.S., Lukewille, A., Clair, T.A., Dillon, P.J., Driscoll,C.T., Forsius, M., Johannessen, M., Kahl, J.S., Kellogg, J.H., Kemp, A., Mannio, J., Monteith, D.T., Murdoch, P.S., Patrick, S., Rebsdorf, A., Skjelkvale, B.L., Stainton, M.P., Traaen, T., van Dam, H., Webster, K.E., Wieting, J. & Wilander, A., 1999. Regional trends in aquatic recovery from acidification in North America and Europe. Nature, 401: 513-622.

Szefer, P. 2002. Metal pollutants and radio nuclides in the Baltic Sea - an overview. Oceanologia 44:129-178.

Turner, R. E., N. N. Nancy, D. Justic, and Q. Dortch. 2003. Future aquatic nutrient limitations. Marine Pollution Bulletin 46:1032-1034.

US EPA. 1998. Lake and Reservoir Bioassessment and Biocriteria Technical Guidance Document (EPA 841-B-98-007). Office of Water, Washington DC.

Van Leeuwen, C. J., and P. T. J. Vanderzandt, et al. 1992. Application of Qsars, Extrapolation and Equilibrium Partitioning in Aquatic Effects Assessment .1. Narcotic Industrial Pollutants. Environmental Toxicology and Chemistry 11:267-282.

Vanni, M. J., and D. L. Findlay. 1990. Trophic cascades and phytoplankton community structure. Ecology 71:921-937.

Vidal, M., C. M. Duarte, and M. C. Sanchez. 1999. Coastal Eutrophication Research in Europe: Progress and Imbalances. Marine Pollution Bulletin 38:851-854.

Vighi, M., and G. Chiaudani. 1985. A simple method to estimate lake phosphorus concentrations resulting from natural background loadings. Water Res. 19:987-991.

Warwick, R. M. 1988. The level of taxonomic discrimination required to detect pollution effects on marine benthic communities. Marine Pollution Bulletin 196:259-268.

Warwick, R. M. 1993. Environmental impact studies on marine communities: pragmatical considerations. Australian Journal Of Ecology 18:63-80.

Webster, J. R., and J. L. Meyer. 1997. Organic matter budget for streams: a synthesis. Journal of North American Benthological Society 16:141-161.

Weisberg SB, Ranasinghe JA, Dauer DM, Schaffner LC, Diaz RJ & JB Frithsen. 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. Estuaries 20**:**149-158

Welch, E. B., J. M. Quinn, and C. W. Hickel. 1992. Periphyton biomass related to point-source nutrient enrichment in seven New Zealand streams. Water Research 26:669-675.

Wetzel, R. G., and G. E. Likens. 1991. Limnological Analyses, 2nd. Ed. edition. Springer-Verlag.

Whitton, B. A., and M. G. Kelly. 1995. Use of Algae and Other Plants for Monitoring Rivers. Australian Journal Of Ecology 20:45-56.

Zhang, J. 2000. Evidence of trace metal limited photosynthesis in eutrophic estuarine and coastal waters. Limnology and Oceanography 45:1871-1878.

**Mission of the JRC**

The mission of the JRC is to provide customer-driven scientific and technical support for the conception, development, implementation and monitoring of EU policies. As a service of the European Commission, the JRC functions as a reference centre of science and technology for the Union. Close to the policy-making process, it serves the common interest of the Member States, while being independent of special interests, whether private of national.

EUROPEAN COMMISSION
DIRECTORATE-GENERAL
Joint Research Centre