International evaluation of the Danish marine models

Performed by the Panel of international experts

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1. Introduction

This report presents a scientific review of the Danish management approach regarding coastal waters in relation to the implementation of the European Water Framework Directive (WFD) in Denmark. The parties to the Agreement on Food and Agriculture Package (22 December 2015) have decided to evaluate the modelling tools (pressure-impact models) used to calculate the mitigation demands for nitrogen (N) runoff from land in the Danish River Basin Management Plans. The results of the evaluation will be utilised towards the development and application of models in the 3rd generation water plans valid for 2021-2027.

Task description by the Ministry of Food and Agriculture

In agreement with the EU Water Framework Directive, Denmark has produced the River Basin Management Plans devising a strategy for improving and securing that coastal waters, lakes, streams and ground waters fulfil the demand for Good Ecological Status as stated in the directive. For Danish coastal waters, it has been estimated that reductions in N runoff from land are the primary concern if goals of Good Ecological Status in coastal waters are to be fulfilled. On this background, mitigation measures have been implemented in the 2015-2021 River Basin Management Plans to additionally reduce the N runoff to coastal waters, corresponding to roughly half the total estimated reduction needs.

The task of the evaluation panel is to perform a thorough evaluation of the marine modelling tools that form the basis for the mitigation demands for land-based nitrogen (N) runoff in the Danish River Basin Management Plans with regards to the importance of N as well as other relevant pressures such as phosphorous, fisheries etc. In particular, the evaluation panel has to:

i. Evaluate the use of models for determination of type-specific reference values (according to the Water Framework Directive, Annex 2) for the water quality element phytoplankton (chlorophyll).

ii. Evaluate the use of models to determine environmental targets (Maximum Allowable Inputs (MAI) of nitrogen)) and mitigation needs to achieve good environmental status and evaluate differences and similarities between the use of different methods and model types for coastal waters with different typology.

iii. Evaluate the estimated nitrogen target loads and mitigation needs in the Danish River Basin Management Plans and evaluate the method for determining the Danish proportion of total mitigation needs. How is the current environmental status in Danish coastal waters determined by N runoff from Danish land areas in relation to other pressures such as N released from sediments and N loads from catchments in neighbouring countries and airborne N deposition (the Danish share of the total mitigation needs related N)?

Further, the Panel is expected to address the technical questions and comments from the stakeholders.

Recruitment of experts

The Danish Ministry of Environment and Food has been responsible for the recruitment of an international panel of five experts to carry out the evaluation. The recruitment of experts has been conducted by a nomination process where the Danish Ministry of Environment and Food has requested water management authorities in other countries (Sweden, Finland, Poland, Germany, The Netherlands and England) and the European

Environment Agency, Joint Research Centre (JRC) and the European Commission (DG Environment) to nominate experts to conduct the evaluation. It has been stated in the request that the nominees should have expert knowledge in the following areas: marine ecology, marine ecosystem models, statistical methods and experience in marine water management in relation to the Water Framework Directive.

The request by the Ministry resulted in the nomination of 14 experts of which 9 experts subsequently indicated that they were interested in being part of an expert panel. Of these, the Ministry has selected the following five experts to conduct the evaluation:

- Professor Peter Herman, Deltares, Institute for applied research in the field of water and subsurface, the Netherlands.
- Professor Alice Newton, NILU – Norwegian Institute for Air Research
- Professor Gerald Schernewski, Leibniz Institute for Baltic Sea Research, Warnemunde
- Director Bo Gustafsson, Baltic Nest Institute (BNI), Stockholm University, Sweden
- Senior Researcher Olli Malve, Finnish Environment Institute SYKE
- Professor Peter Herman was chosen as chairman of the Panel

The five experts were chosen according to an assessment of their qualifications with regards to experience with and competences in the following fields of study: *marine ecology/coastal ecology*, c*oastal ecosystem modelling*, *use of statistics in environmental science* and *marine management experience related to the implementation of the Water Framework Directive*.

1.1 Aim and focus of the evaluation

This section presents the aim and focus of the evaluation according to the international panel (hereafter referred to as the Panel) and should therefore be seen as the Panel's further operationalisation of the task description in section 1.1.

The main aim of the evaluation

The main aim of the evaluation is to review whether the marine models – as presented in the Scientific Documentation Report and as commented by the researches and stakeholders – *provide solid and robust scientific evidence that the proposed reductions in land-based N runoff will be both necessary and sufficient to reach Good Ecological Status as defined in the Water Framework Directive*.

- By "solid", the Panel means well based in international scientific literature, well performed, credible
- By "robust", the Panel means not unduly dependent on arbitrary details, reliable with acceptable precision
- By "necessary", the Panel means that by doing less the goals would not be reached
- By "sufficient", the Panel means that by executing the plans, there is a high probability of reaching the goals

The evaluation concerns the modelling tools (pressure/impact models) forming the basis for the mitigation demands for land-based nitrogen (N) runoff in the Danish River Basin Management Plans. The evaluation results will enter into the calculation of N mitigation demands for coastal marine areas in the 3rd generation water plans valid in 2021-2027.

The evaluation will answer questions related to points (i)-(iii) in the task description above and is therefore focused on the scientific underpinning of the plans, in particular the modelling tools. The evaluation must take into account the internationally agreed goals of achieving Good Ecological Status in the Water Framework Directive. One that basis, the Panel has defined the aim and focus of the evaluation as stated in the Text Box shown above.

The scope of the evaluation does not include other models than the marine model and other environmental targets than those applying to coastal areas. The scope of the evaluation does not include the societal costs and benefits of the measures that would be needed to fulfil the environmental targets.

1.2 The basis and process of the evaluation

The basis for the evaluation

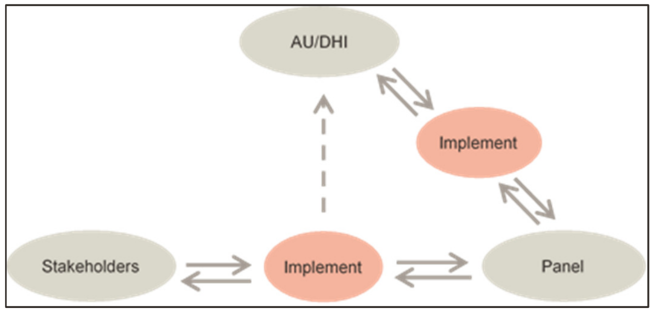
The basis on which the Panel has made the final evaluation consists of the following materials:

- The Scientific Documentation Report written by Aarhus University (DCE) and DHI in June 2017, which documents the model tools and calculated MAI that were developed for the Ministry over the period 2013-2015.
- Questions and comments from the stakeholders to the Scientific Documentation Report (see Annex 1 of the evaluation report)
- Answers from the researchers to questions and comments which were formulated by the Panel after the members of the Panel read and considered the report as well as the questions and comments from the stakeholders (see Annex 2a and 2b of the evaluation report).
- Answers from the Panel on how they took into account each of the technical questions and comments from the stakeholders (see Annex 3 of the evaluation report).
- Selected background materials cited by the researchers, the stakeholders and the Panel

The means for ensuring independence during the process

It is considered crucial that the evaluation of the Danish marine models be performed by independent scientists. In order to guarantee independence, it was decided that the Ministry of Environment and Food, the scientists from AU and DHI and the stakeholders should keep arm's length to the Panel throughout the process of the evaluation. Implement Consulting Group (Implement) was engaged by the Ministry to facilitate the process.

Figure 1. Communication model



As illustrated above, the communication model was designed to facilitate a dialogue that ensured arm's length between the involved parties and to promote a transparent flow of communication. Implement has been the link between the Panel, the stakeholders and the scientists. Besides facilitating the final writing workshop and the preparations leading up to that, the main role of Implement has therefore been to ensure timely communication and convey relevant material and information between the parties.

The evaluation process

The evaluation process started in June 2017. It resulted in an evaluation report on 19 September, which was finalised after a writing workshop in Helsingør which took place between 11-15 September. After the hearing process between 19. September and 2 October some minor corrections were made to the final report which was completed on 10 October.

The text and the activity plan below provide a more detailed overview of the evaluation process.

Initially, the stakeholders from Blåt Fremdriftsforum, the scientists from AU and DHI and the Panel were invited to participate in separate meetings where Implement explained the process of the evaluation. At the meetings, the activity plan and a communication model were presented to make sure that all parties were properly informed about the practical aspects, important deadlines and rules of communication. The process leading up to the final evaluation workshop was thereafter as follows with respect to each of the parties:

- The stakeholders received the Scientific Documentation Report written by the scientists from AU and DHI on 6 June and had until 4 July to formulate questions and comments to the report. The comments and questions had to be submitted in a table – made specifically for that purpose – that followed the structure of the report. A "hotline" for questions regarding the practical aspects of formulating and submitting the questions and comments was established by Implement. The stakeholders submitted their questions and comments on 4 July, and they were all forwarded by Implement to the Panel on 6 July. In order to sum up their main points of view in front of the Panel, the stakeholders were invited to participate in a physical meeting with the Panel hosted by the Ministry in Helsingør on 11 September during the writing workshop.
- The Panel received the Scientific Documentation Report at the same time as the stakeholders – on 6 June. Implement held a couple of status meetings with the Panel in June and the beginning of July when the comments and questions from the stakeholders were forwarded to the Panel on 6 July. Based on the reading of the Scientific Documentation Report and the questions and comments from the stakeholders, the Panel has jointly formulated questions to

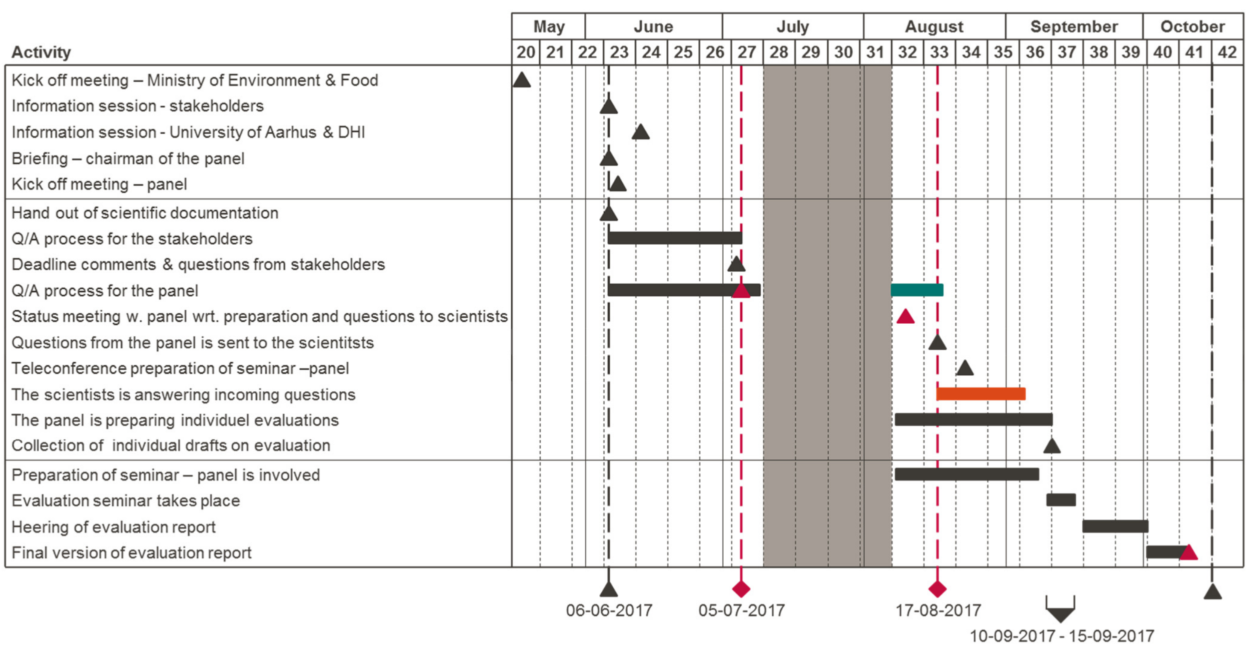
the Danish scientists that were forwarded by Implement on 15 August. The scientists from AU and DHI answered these questions on 4 September in order for the Panel to take it into account in the evaluation. Throughout August, Implement held some status meetings with the Panel to monitor the progression and to prepare the writing workshop in Helsingør.

- The scientists from AU and DHI worked out the Scientific Documentation Report that was distributed by Implement to the stakeholders and the Panel. The scientists received the comments and questions by the stakeholders on 6 July as an orientation. As stated above, the scientists received questions from the Panel and replied to these on 4 September. During the writing workshop from 11-15 September, the scientists have answered a limited amount of additional questions from the Panel.
- The Ministry of Environment and Food was not directly involved in the evaluation process due to the principle of arm's length. Implement has occasionally informed the Ministry about the progress in the evaluation, and the parties had a dialogue about the practicalities of the writing seminar in Helsingør. Representatives from the Ministry were present at the meeting with the stakeholders and the Panel in Helsingør on 11 September.

The writing of the evaluation report was thereafter carried out by the Panel and facilitated by Implement in a final writing workshop in Helsingør between 11-15 September. The evaluation report was edited and submitted for hearing on 19 September.

The hearing of the evaluation report among stakeholders from Blåt Fremdriftsforum and the scientists from AU and DHI took place between 19 September and 2 October.

Figure 2. Activity plan of the evaluation process



After the hearing process, the evaluation report will be published by the Ministry of Environment and Food along with annexes containing the hearing comments and answers by the Panel. The activity plan above illustrates the entire process of the evaluation.

1.3 Content and structure of the evaluation report

The evaluation report is divided into a number of themes which the evaluation panel found to be the most important in order to cover the topics in the terms of reference and pursue the aim of the evaluation. According to the Panel, the main themes are those covered in Chapters 2-9 in the evaluation report which has the following structure:

- Introduction (Chapter 1)
- Compliance with the Water Framework Directive (Chapter 2)
- Coastal water typology (Chapter 3)
- The use of seagrass and Kd as environmental indicators (Chapter 4)
- Emphasis on nitrogen versus phosphorus (Chapter 5)
- Statistical modelling (Chapter 6)
- Mechanistic modelling (Chapter 7)
- Calculation procedures to estimate Maximum Allowable Inputs from model results (Chapter 8)
- Evaluation of Maximum Allowable Input results (Chapter 9)
- Overall assessment and conclusions (Chapter 10)
- Recommendations for going further (chapter 11)

By going through the most important themes and discussing the main problems within each theme, the review by the Panel focuses on whether these problems have been adequately solved in the Scientific Documentation Report – rather than going through the details in the report chapter by chapter.

This means that the review text by the Panel mainly concentrates on investigating possible weaknesses in the overall modelling approach followed by the researchers from Aarhus University (DCE) and DHI. However, the review contains conclusions with respect to both the strengths and weaknesses of the approach, and critical remarks should be viewed in the context of the overall assessment as presented in Chapter 10.

After the thematic chapters, the evaluation contains an overall assessment of the marine modelling approach and report. The final assessment will provide an answer to the central question on whether the modelling approach and report provides solid and robust scientific evidence that the proposed reductions in land-based N runoff will be both necessary and sufficient to reach Good Ecological Status as defined in the Water Framework Directive. Moreover, the assessment will answer other related questions to cover the terms of reference.

Finally, recommendations are given as to how the Danish marine models might be improved in the future. Focus is on improvement that can be made within a reasonable time frame and without investing excessive resources.

1.4 Future development following the evaluation

Once the researchers have made the adjustments to the modelling, they are encouraged to publish their work in peer-reviewed journals to showcase Danish leadership in this field.

The work of the researchers has been performed over decades and several administrations. This “organic” process has given rise to numerous interactions between the scientists and authorities. In order to avoid confusion and misunderstandings, terms of reference, the scope of the missions set by the Ministry and agreements on choices, e.g. indicators to be used, should be well-defined. This can be important for the further political process, but has not been subject to examination by the Panel.

The Panel hopes that the attention given to the views of the stakeholders and the responses of the researchers during the scientific scrutiny of the Scientific Documentation Report will help to build trust between the parties and contribute to a successful outcome.

2. Compliance with the Water Framework Directive

This chapter examines whether the Scientific Documentation Report complies with Directive 2000/60/EC, commonly referred to as the Water Framework Directive (WFD). It also addresses some of the concerns and questions of the stakeholders. In this chapter, we focus on the following general questions relating to the compliance with the Water Framework Directive:

- Is the procedure for setting the type-specific reference condition WFD compliant?
- Is the choice of indicators selected WFD compliant?
- Have the indicators been intercalibrated?
- Has the “one-out, all-out” principle been respected?

The questions are the basis for the subsections of the chapter. In addition, the chapter devotes attention to the question whether all relevant stressors have sufficiently been taken into account.

The community policy on water was adopted by the European Parliament and Council on 23 October 2000 as an integrated Community Directive 2000/60/EC, commonly referred to as the Water Framework Directive (WFD). It was published in the Official Journal (OJ L 327) on 22 December 2000 and was also adopted by member states (MS). In Denmark, it was adopted as national legislation in 2003.

Article 1 states: “The purpose of this Directive is to establish a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater”. Preamble 26 of the WFD states that “member states should aim to achieve the objective of at least good water status by defining and implementing the necessary measures within integrated programmes of measures, taking into account existing community requirements”. Article 4 introduces the concept of the River Basin Management Plans (RBMP) as fundamental to “making operational the programmes of measures”, and these are detailed in article 13. RBMP are a single system of water management by river basin, which are the natural geographical and hydrological units, instead of according to administrative or political boundaries.

The report “Development of models and methods to support the establishment of the Danish River Management Plans”, which we refer to as the Scientific Documentation Report, contributes to the implementation of the WFD to maintain or achieve Good Ecological Status in Danish Coastal Waters (CW). Therefore, an evaluation of the WFD compliance of the methodology and results is valuable and important.

2.1 Reference conditions and boundary setting

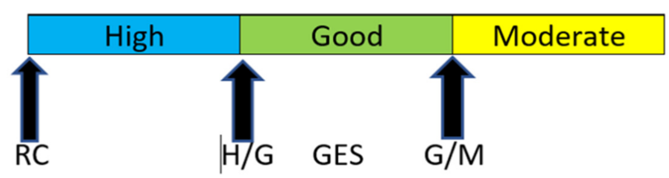
Annex 2 of the WFD (section 1.1) addresses the characterisation of surface water body types. First, the water bodies must be placed in one of the surface water categories: rivers, lakes, transitional waters or coastal waters. Another possible category is artificial and heavily modified bodies of water.

The WFD then specifies that the water bodies are to be differentiated according to their type. Denmark merged transitional waters with coastal waters, therefore a Fish BQE is not necessary. Denmark has 119 marine water bodies[1]. These are categorised into six open water body types and 12 estuarine water body types, all included in coastal waters, according to a report by Dahl et al (2005). In Chapter 3, we discuss the further implications and consequences of this typology. We note that in their answers to the Panel, the researchers state that there is a project proposal on an “update of the typology applied towards the RBMP 2021-2027”.

Annex 2 of the WFD (section 1.3) specifies the procedure for the “establishment of type-specific reference conditions for surface water body types”. Type-specific reference conditions (RC) may be either spatially based or based on modelling or may be derived using a combination of these methods. Where it is not possible to use these methods, member states may use expert judgement to establish such conditions. The Danish approach relies on modelling and a 1900 baseline, since there are no pristine systems that can be used as a reference. This approach is appropriate, WFD compliant and better than only using expert judgement.

Good Ecological Status (GES) falls between the “High/Good” boundary and the “Good-Moderate” status. The relationship between reference condition, the boundaries and GES is shown in Figure 3. The setting of the reference conditions and the boundaries, especially the G/M boundary, is important. This determines whether management measures are necessary. Classification below the G/M boundary requires management measures to be adopted.

Figure 3. Relationship between reference condition, the boundaries and GES



Target values must fall in the green (GES) range.

2.2 Choice of indicators

Annex 5 of the WFD specifies the quality elements (QE) for the classification of ecological status of coastal waters (1.1.4). Good Ecological Status is an assessment based on a combination of biological quality elements (e.g. phytoplankton, other aquatic flora and benthic invertebrates); Hydro-morphological elements (e.g. structure and substrate of the seabed, tidal regime); chemical and physico-chemical elements (e.g. transparency, oxygenation conditions, nutrient conditions).

The indicators chosen in the Danish RBMP report are Chlorophyll a, Kd and a benthic index as well as some secondary indicators in statistical modelling approaches (see Chapter 4 of this evaluation report). Denmark has not adopted transitional waters as a separate category, and therefore there is no need to include a fish biological quality element. Chlorophyll a is a proxy for phytoplankton biomass and has been intercalibrated (see 2.3 below). Kd is a measure of attenuation, hence an indirect measure of growth conditions for benthic plants and algae. Thus, it is not a direct indicator of aquatic flora (eelgrass), but rather a light control on the distribution of eelgrass. Furthermore, Kd is not independent of Chlorophyll a, since phytoplankton cells contribute to light attenuation and a loss in transparency. Kd has not been

[1] Scientific Documentation Report, section 3.1, p. 14, Bek.nr. 837 2016

intercalibrated, although eelgrass depth limit, for which it is a proxy, has (see 2.3 below). The Danish Benthic index addresses the benthic invertebrates' biological quality element. Chlorophyll a was the indicator chosen for the intercalibration of the WFD, which Denmark participated in. The marine models under evaluation only considered indicators for the physico-chemical elements' oxygenation condition and nutrient condition in the statistical modelling.

- Most of the calculations in the modelling are based on only Chlorophyll a and Kd to derive the targets for N
- Furthermore, the choice of Kd as an indicator for submerged aquatic vegetation (eelgrass) may be insufficient (Chapter 4)
- The inclusion of other indicators throughout the process and modelling (oxygenation condition and nutrient limitation) are also discussed in Chapter 4

2.3 Intercalibration

A Common Implementation Strategy (CIS) has been in operation since 2001, bringing together national experts, stakeholders and the Commission involved in the implementation of the WFD. During the process, a series of Guidance Documents and CIS Thematic Information Sheets were produced. These are not legally binding but give mainly technical advice about the implementation process.

The WFD requires the national classifications of Good Ecological Status to be harmonised through an intercalibration exercise, Birk et al (2013). This is to avoid adjacent water bodies being classified in a different way. Intercalibration was carried out by member states that share typologies and transboundary water bodies. In the case of Denmark, the shared water types were NEA 1/26C: NEA 8B and BC 6. These are explained in Table 1.

Table 1. Common typologies intercalibrated for Chlorophyll a with Germany and Sweden

Code	Water type	Shared with	Intercalibration
NEA 1/26C	The North-East Atlantic, enclosed seas, exposed or sheltered, partly stratified	DE	Intercalibrated
NEA 8B	The North-East Atlantic Kattegat coastal waters	SE	Intercalibrated
BC 6	Baltic Coast (SW)	SE	Intercalibrated

As mentioned in 2.2, Chlorophyll a was the indicator chosen for the intercalibration of the WFD, which Denmark participated in. The overall status with respect to intercalibration of the indicators used in the Danish marine models is as follows:

- Chlorophyll a has been successfully intercalibrated with SE and DE
- Kd has not been intercalibrated (as confirmed by the researchers from Aarhus University (DCE) and DHI and the European Commission's Joint Research Centre).
- Eelgrass depth limit has been intercalibrated

2.4 “One out, all out”

The WFD preamble 11 specifies that it is “based on the precautionary principle and on the principles that preventive action should be taken”. The “one-out, all-out” principle is a key principle that reflects the WFD integrated approach for the protection of water resources and associated aquatic ecosystems. Quality elements comprised in the definition of ecological status provide a holistic picture of the health of the aquatic environment. The overall status would only be “good” if all the elements comprised are at least considered “good”. This ensures that all pressures capable of degrading the water status are addressed and are a guarantee of the environmental integrity of the objectives of the directive.

Progress achieved towards “good” status of water bodies can be reported using indicators at individual quality element level. However, this does not preclude the “one-out, all-out” principle. The WFD will be reviewed by 2019, taking into account the results of the second RBMPs. The proper implementation of the Nitrates Directive, which is a basic measure under the WFD, is necessary for the achievement of the WFD objectives. However, in many cases, this will not be sufficient, and additional measures will have to be taken by member states to ensure that the WFD objectives are reached.

Based on the “one-out, all-out” principle, indicators for different quality elements should be considered individually. If one is classified as below the G/M boundary, then management measures must be applied. This was not applied in the Scientific Documentation Report as confirmed by the researchers in their answers to the panel questions. The different methods of aggregation and their implications in both the WFD and MSFD are discussed in Borja et al (2014). We further discuss the “one-out, all-out” principle in relation to indicators in Chapter 4 and in relation to the calculation procedures in Chapter 8.

2.5 Other stressors on the ecosystem

In his classic paper on eutrophication problems, Cloern (2001) describes how the vision on eutrophication problems has evolved from viewing nutrient enrichment as a single isolated issue, towards a vision that emphasises the interactions between multiple stressors, the physics and hydrography of the systems, and eutrophication. He makes a plea for integrated models and tools that describe how nutrient enrichment modulates the response of ecosystems to other stressors, such as chemical pollution, introduction of invasive species, habitat modifications, fishing pressure and others, in the physical setting of a water body. The different stressors should not be viewed as additive factors with – from a management perspective – the option to choose reduction of any of these stressors to obtain a similar percentage of improvement in the ecosystem response. Restoring physical habitat quality, as an example, will have very little effect if eutrophication leads to oxygen problems, low transparency of the water or low phytoplankton quality due to the interactions of the causal factor “physical structure” with eutrophication. Reversely, remediation of eutrophication problems may not suffice to improve ecological quality, if additional action on other stressors is needed.

In many questions and comments of the stakeholders, reference was made to the report by Andersen et al (2017) that lists many stressors on the marine ecosystem and, using a particular weighting, concludes on an overall percentage of stress due to nutrient loading. One could try and argue that this provides evidence that similar improvements of ecological status could be obtained by working on other stressors than nutrient loading, but in so doing would miss the essential point that the effect of the different stressors is not additive and that the final ecosystem response is modulated by the interaction between the stressors, not their individual additive effect. The Panel endorses the fundamental view on interaction between stressors, and on the key role of nutrient loading and eutrophication in modulating the ecological response of Danish coastal waters, that is expressed in the Scientific Documentation Report and in the models (especially the mechanistic models) underlying the analyses.

This fundamental view on the importance of water quality as the main modulator in promoting Good Ecological Status is fully in line with the Water Framework Directive implementation and with the use of intercalibrated indicators such as Chlorophyll a and measures of chemical pollution as the prime measures of ecological status. Inclusion in the WFD was based on extensive and in-depth reviews of ample scientific evidence.

Other legal instruments, e.g. the Marine Strategy Framework Directive, take a broader view and also include more explicitly other stressors such as invasive species, shipping, fishing and physical modification. The Panel is convinced that these aspects fully merit inclusion in a holistic view on restoring Good Ecological Status but in no way decrease the importance that has to be attached to controlling nutrient loadings as a necessary condition for restoration of Good Ecological Status.

3. Coastal water typology

The Scientific Documentation Report uses a modified Danish coastal water typology as the basis to calculate reference conditions and targets for coastal waters as well as Maximum Allowable Inputs (MAI). The typology is a crucial element for all following steps. Therefore, in this chapter, the Panel evaluates its suitability, analyses its shortcomings and provides suggestions.

3.1 Basic idea behind typology

Annex 2 of the WFD gives instructions on how typology should be carried out and lists the obligatory and optional factors that can be used (see Chapter 2 of this evaluation). Most European Union member states applied the most specific system B. In this approach, the physical and chemical factors that determine the characteristics of coastal and transitional waters are latitude, longitude, tidal range and salinity as obligatory factors. Optional factors are current velocity, wave exposure, mean water temperature, mixing characteristics, turbidity, retention time, mean substratum composition and water temperature range.

The Common Implementation Strategy (see Chapter 2) for the WFD (2000/60/EC): "Guidance Document No 5" on "Transitional and Coastal Waters – Typology, Reference Conditions and Classification Systems" provides a detailed guideline for carrying out a characterisation of all water bodies, referred to as typology. The aim is to produce a simple physical typology that is both ecologically relevant and practical to implement. It aims at linking similar water bodies under one type to enable the establishment of type-specific reference conditions. Guidance Document No 5 suggests ranges for several factors that could be used in the typology.

Once the water has been characterised as transitional (TW) or coastal (CW), a typology for each is developed by the member states. Denmark, like Germany, chose to include estuarine waters within coastal waters because all the parameters, except depth, are the same. Whether a typology separates transitional and coastal waters or combines both under coastal waters does not make a difference for the calculation of reference conditions, targets and Maximum Allowable Inputs (MAI). It does not necessarily affect the number of types nor the number of water bodies in a country. Because of the narrow guidelines, most countries considered the typology development as a largely technical task.

3.2 The Danish typology

According to the Common Implementation Strategy for the WFD (2000/60/EC), Dahl et al (2005) divided the Danish coastal waters into 15 different types: 5 open water types and 10 estuary types. Transitional waters were included in the typology. Dahl et al (2005) state that "the large number of types reflects the strong salinity gradient present in the Danish coastal waters, but also that the physical factors that are relevant for defining a type, vary greatly among the Danish estuaries". The national typologies in the Baltic Region show many similarities, and in several cases coastal and transitional waters were merged into one system to reduce complexity. In the Scientific Documentation Report, the Danish typology is further simplified and types are merged. The aim of this simplification is that less Chlorophyll a reference and threshold values for a Good Ecological Status (target values that fall between the High-Good and Good-Moderate boundaries) have to be defined.

The Common Implementation Strategy for the WFD (2000/60/EC) reminds member states that, when developing a typology, they should keep the major objective of the Directive in mind, namely to establish a framework for the protection of both water quality and water resources preventing further deterioration and protecting and enhancing ecosystems. It is pointed out that typology is a tool to assist this process, and it is recognised “that a simple typology system needs to be complemented by more complex reference conditions that cover ranges of biological conditions” (p. 28). It means that every country has the freedom to adjust the typology to its own needs and to refine it to the required degree.

3.3 Suitability of the Danish typology

The major question is whether the typology in the Scientific Documentation Report is sufficiently detailed to allow the definition of reliable reference and target values for Chlorophyll a and the other indicators in all coastal waters. Reliable means that these values well reflect the ecological conditions and properties of all coastal waters. This is a precondition for defining target values that allow to derive to derive reliable MAI for each water body. The general impression is that the typology allows the derivation of suitable target values and MAI for water bodies in the sea with strong water mixing. An indication is that the inter-calibrated values for Chlorophyll a with Germany and Sweden for the sea and outer coastal waters are well in agreement with the results of the Scientific Documentation Report (Schernewski et al, 2015). In general, the comparable German Chlorophyll a target values for the open sea are slightly lower, but would allow a cross-border harmonisation.

Many fjords and coastal bays share a similar Chlorophyll a target concentration of 3.6 mg/m^3, namely Norsminde Fjord, Mariager Fjord (outer), Nissum Bredning, Randers Fjord (outer), Horsens Fjord, Kolding Fjord, Vejle Fjord, Odense Fjord, Nyborg Fjord, Kerteminde Fjord, Holckenhavn Fjord, Bredningen, Emtekær Nor, Nærå Strand, Nakkebølle Fjord, Dalby Bugt, Karrebæk Fjord and Roskilde Fjord.

The Scientific Documentation Report and the additional data tables provided by the authors of the report show that water bodies with diverse properties are represented by only one target value. The typology is too simplified to reflect the specific characteristics of the individual fjordic water bodies. The consequence is a large and not sufficiently justified variation in the required load reduction for each water body. In the understanding of the Panel, the Danish typology does not sufficiently reflect the individual properties of the many Danish fjords and inner coastal waters. The solution could be either to subdivide the typology for these systems, taking into account especially water exchange rate and fresh water discharge, or to develop individual Chlorophyll a target values for every single water body. The statistical modelling, especially when carried out across water bodies, could be an excellent basis for this.

3.4 Suitability of the Danish monitoring programme

A precondition for a refined typology for fjords and inner coastal waters is the existence of a suitable and comprehensive monitoring programme. The present Danish national monitoring programme includes more than 90 stations along the coast and in the sea. It is very comprehensive and seems to be well-adjusted to the WFD requirements. Altogether, 119 water bodies are separated in Denmark. In some cases, fjord systems are divided into two or more water bodies and are represented only by one monitoring station. Examples are Mariager Fjord, Randers Fjord, Vejle Fjord and Flensborg Fjord. It means that practically every water body or spatially linked group of water bodies (like a fjord) are represented by one monitoring station. This is important, because only the existence of a monitoring station and regular data collection allows assessing whether the target is reached or not.

3.5 Suggestions towards a modified approach

Such a comprehensive monitoring programme not only allows a refinement of the typology, but would allow the definition of individual Chlorophyll a reference and target values for every water body, respectively some spatially linked group of water bodies. We strongly suggest considering this approach, especially when the aim is to calculate as precise and water-body specific MAI as possible. Denmark is one of the few countries in Europe, where the necessary data, expertise and models are available for such a comprehensive approach. In detail, it has to be assessed if additional monitoring stations, temporary data collection at some locations or complementation of the monitoring programme with remote sensing might be necessary. Neither the application of the meta-modelling nor monitoring of the success of the proposed measures are possible without a minimal set of follow-up actions in the field.

3.6 The look abroad

Similar discussions took place in Germany as well, and the results are reflected in a national report and an international publication (Schernewski et al, 2015). Germany carried out a comprehensive revision of all German Baltic reference and target values for nutrients and Chlorophyll a. The discussion process within the accompanying official national working group came to the conclusion that especially the different estuaries and lagoons have so specific properties and behaviours, and that type-specific Chlorophyll a and nutrient reference and target values would be too general. As a consequence, specific Chlorophyll a and nutrient reference and target values were developed for every single water body, resulting in 35 major Chlorophyll a reference and target values for the German Baltic waters alone.

4. The use of seagrass and Kd as environmental indicators

The use of Chlorophyll a as an indicator for phytoplankton is widespread and accepted in the WFD. The indicator is intercalibrated between Denmark and neighbouring countries and is fully endorsed by the Panel. In this chapter, we will discuss the appropriateness of other indicators. We discuss the use of Kd as an indicator for aquatic macrophytes and angiosperms, and subsequently devote a discussion to the indicators for hypoxia and nutrient limitation, used in the statistical modelling approach. The main questions are whether these indicators are appropriate for demonstrating important ecological quality aspects, whether they can be related to nutrient inputs, and whether they should be maintained in the scientific modelling approaches.

4.1 Kd as an indicator for the biological element "benthic vegetation, macroalgae and angiosperms"

One of the three main indicators used in the WFD as measures of Good Ecological Status is the condition for aquatic macrophytes and angiosperms. In most Danish estuarine and marine waters, this concerns eelgrass (*Zostera marina*), even though this is not the only species of angiosperm that occurs. Pondweed (*Potamogeton* species) and *Ruppia* may cover extensive parts of some systems and should be taken into account as "angiosperm vegetation". However, in systems where this occurs (e.g. Odense Fjord), it is still eelgrass that dominates the deeper (>1.5 m) parts and thus remains the most critical indicator. The Panel has no complete overview of the situation in all the different water bodies, but stresses the generality of the required "angiosperm vegetation" indicator, so that it may occasionally differ from the single "*Zostera* maximum depth" indicator, at least in principle.

In the scientific documentation report and the underlying model work, water transparency, expressed as the light extinction coefficient Kd (m^{-1}) is used as a proxy for the depth limit for eelgrass. This is based on solid scientific evidence that eelgrass needs a light intensity at the bottom of between 10-20% of the incident light. The choice of 14% is based on this literature, and on area-specific experiments for Danish waters, and is well justified. However, the Panel points out that, due to the non-linear interaction between light intensity and Kd and in the presence of temporal variability in Kd, the average light intensity reaching the bottom in a water system at a particular depth may differ significantly from the light intensity calculated at this depth using the average Kd.[2]

As clearly stated in the report, water transparency is a necessary but insufficient condition for seagrass to re-establish in these estuarine systems. Recent studies of seagrass reproduction, as well as adult seagrass survival, have pointed to factors such as disturbance by floating algae, resuspension of fine material, disturbance by lugworms, herbicides and others to influence recovery (Flindt et al, 2016; Kuusemäe et al, 2016; Canal-Verges et al, 2016). It is likely that the presence of eelgrass itself plays a role in these circumstances, not only as a seed source but also by collecting and fixing fine sediment material. It has been shown in general (van der Heide et al, 2011) and in a specific restoration case in North America (Orth et al, 2012) that this may lead to alternative stable states and strong non-linear behaviour: once extensive seagrass

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[2] As an example, if Kd has a lognormal distribution with a logarithmic mean of -1.0 (mean approx. 0.4 m^{-1}), and a logarithmic standard deviation of 0.5, the depth was on average 14% of incident light reaches the bottom is 6.15 m, whereas the depth limit calculated from the average Kd is 4.85 m. Note that the difference depends on the statistical distribution of Kd in the field, which the Panel cannot evaluate.

meadows are present, they contribute to keeping the water clear and extend their range to deeper waters, but in the absence of meadows the water remains turbid and prevents the development of meadows. As a consequence, restoration of seagrass meadows (i.e. transition from unvegetated to vegetated state) may require more stringent conditions to be fulfilled than what is needed to maintain an existing vegetation. These more stringent conditions may relate to lower nutrient loadings, but may also relate to the exclusion of other disturbing factors. Therefore, it is unlikely that Kd as a sole indicator covers the entire range of conditions needed for eelgrass restoration, but it is even more unlikely that restoration will succeed without at least restoring Kd to the levels needed for the Good-Moderate boundary conditions.

The time course of Kd in the water bodies studied by statistical modelling is shown in the Annexes to this evaluation report. In most cases, it is very difficult or impossible to detect a significant downward trend in the values. Even though a significant correlation of summer (June to August) averages of Kd with N load is reported for 16 out of 22 stations [p 94], the slopes of these relations are very low [p 94], and no material changes in yearly averages are observed over time despite changes in N loading. Similarly, in the mechanistic modelling, slopes for change of Kd as a function of N load are usually small, and the model is not able to reproduce the reference (observed around 1900) Kd values by modelling reference loads of 1900. The total range in Kd across all systems at reference loadings is approximately 0.15 m^{-1}, whereas the total range in the observations is approximately 0.35 m^{-1}. The model is calibrated to reproduce the mean reasonably, but is not able to capture the full (temporal and cross-system) variation very well.

Based on a statistical analysis of all Danish systems since the 1980s, Riemann et al (2016) report a significant increase in Secchi depth between 1980 and 1990, when many systems made the transition from hypertrophic to eutrophic state, but absence of any systematic trend afterwards. In addition, they remark that Secchi disc depth data even overestimate the effect on Kd, because of a shift from scattering to absorption as the main light extinction mechanism between 1980 and 2010. In summary, none of the within-system statistical analyses or models seem to be able to demonstrate a strong dependence of Kd on nutrient loading in the period 1990-2013.

However, when viewed across systems, the data shown in annex B of the Scientific Documentation Report for Chlorophyll a and Kd in the systems studied with the statistical modelling strongly suggest a close correlation between average Chlorophyll a concentration and average Kd over the study period (see Figure 4 in Chapter 8). It is likely that a common cause – most probably the relative influence of the freshwater end member in the water of the estuary – determines both. However, within each of the systems, we do not observe a correlated evolution in time of the two indicators over the 1990-2012 period. A further interesting observation is that the targets for Kd and Chlorophyll a, as derived in the Scientific Documentation report, show exactly the same correlation but in a narrower range of both indicators. As these reference values represent historic conditions, the suggestion is that on a centennial time scale, Kd and Chlorophyll a covary in time. Therefore, it is possible that Kd does respond to nutrient loading, but with significant delay and only on long time scales.

From these considerations, the Panel concludes that both indicators represent eutrophication effects, but that the estimation of the effect of nutrient reduction on Chlorophyll a is more reliable than the estimation of this effect on Kd.

The Scientific Documentation Report suggests that other causes, in particular the influx of dissolved and particulate organic matter from freshwater, as well as long-term storage of fine and fluffy sediment material, influences the transparency of the water. Trends in benthic filter feeders (that have decreased significantly in biomass between 1990 and

2017 – see Riemann et al, 2016) may also be a causal factor. It can be assumed that filter feeders decrease in biomass as a consequence of the decrease in phytoplankton primary production, which may result in less filtration and fixation of fine particles in the sediment. It is difficult to evaluate each of these hypotheses, but the very good correlation between average chlorophyll and average Kd across systems suggests that the influx of some substance with the freshwater, that may be higher nowadays than in 1900, plays a dominant role. In their answer to the questions of the Panel, the researchers have thoroughly analysed and dismissed the possibility that herbicides play a significant role in this freshwater influence. The most probable hypothesis is that the influx of coloured organic substances has increased between 1900 and now.

The consequences of incorporating Kd as an important indicator for water quality, in the absence of strong slopes between nutrient loading and Kd, are different for the mechanistic modelling and the statistical modelling exercises. The mechanistic modelling estimates which part of the distance between target and status can be bridged by reducing Danish land-based N sources. It corrects for this fraction in the calculation of the effort required. The Panel finds this approach appropriate and does not think it leads to unjustifiable overestimation of efforts needed.

The statistical modelling approach does not follow the same reasoning as the mechanistic modelling. For some water bodies, N load reductions of well above 100% are calculated to be needed in order to bring Kd down to target levels. This is of course physically impossible. The problem is solved by “translating” the required very high efforts into realisable efforts [25% when the calculation is 25%-100%, 50% for calculation 100%-200%, 75% for calculation >200%). Despite questions to the researchers, the Panel has not been able to discover the logic behind this translation. The researchers argue that this is basically expert judgement and further argue that 25% is the order of magnitude of interannual variation of the N load, therefore when an effort is estimated to be “large”, it should be above this level but not too much. In the opinion of the Panel, the “translation” introduces an unnecessary element of arbitrariness into the whole procedure that is in contrast with the general evidence-based approach and that therefore exposes the entire procedure to unproductive criticism. The Panel furthermore observes that the situation here is analogous to the situation treated in the chapter on mechanistic modelling, where very often the target value cannot entirely be reached with reduction of Danish land-based N. Therefore, the Panel suggests harmonising the approach across the two modelling lines and adopting the approach of the mechanistic modelling also in the statistical modelling.

In further work, the Panel recommends reviewing the approach for this WFD indicator by starting from the basic observation that not Kd, but survival and restoration of aquatic angiosperm vegetation is the real criterion. In some systems, this criterion may actually be fulfilled by other species than eelgrass (e.g. *Ruppia* or *Potamogeton* species), in which case the criterion could also be considered as generally fulfilled. However, in most cases, eelgrass will be the species of interest. As mentioned above, recent modelling work of Kuusemäe et al (2016) and Flindt et al (2016) has taken a more comprehensive view on restoration of eelgrass, and the influence of nutrient loading on the process. This work is actually built into the mechanistic models used in the present study, but the results have not been directly used in order to estimate the influence of nutrient reduction on seagrass restoration. The Panel proposes to make better use of these models, probably after more extensive validation, to more directly estimate the effect of nutrient reductions on seagrass development possibilities.

In view of the apparent difficulties in estimating the effect of nutrient reductions on Kd at short time scales, the insufficiency of Kd as a representation of all factors needed for restoration of seagrass, and the high correlation between Kd and Chlorophyll a both in status and targets at longer time scales, the Panel suggests to relatively downweigh the importance of Kd in the final calculations of reductions needed. It further recommends pursuing studies attempting to estimate conditions for seagrass restoration based on already developed more comprehensive models. In the absence of the latter, and given the correlation between Kd and Chlorophyll a, the Panel is of the opinion that adherence to the “one-out, all-out” principle with respect to Kd and Chlorophyll a, is not imperative. A weighted average of reduction needs for both indicators might be preferable.

4.2 Other indicators used in the statistical modelling

In contrast to the mechanistic modelling, the statistical modelling bases its conclusions on three other indicators: (1) the occurrence of hypoxia, (2) ecological signs of hypoxia from nutrients and (3) chlorophyll and (4) the number of days of N limitation of phytoplankton growth. Indicator (2) and (3) are given half weight as they estimate one element together. Compared to Chlorophyll a and Kd, the combination (2) - (3) and the indicators (1) and (4) are given half the weight.

The Panel is surprised by the inclusion of these indicators in only one line of modelling, as it could also have been done in the mechanistic modelling. The latter contains all the variables needed to estimate hypoxic/anoxic conditions as well as direct estimates of nutrient limitation of phytoplankton growth. The asymmetric situation leads to a decrease of comparability of the two models and decreases credibility of the procedure of averaging both approaches, e.g. in meta-modelling. The Panel further notes that in meta-modelling based on the statistical approach, the ancillary indicators are sometimes included and sometimes not, depending on data availability.

With respect to the occurrence of hypoxia, the researchers note in the Scientific Documentation Report that:

“There is direct evidence for a relationship between nutrient loadings and oxygen concentrations in bottom water (Markager et al, 2006) and the size of hypoxic/anoxic areas (Scavia et al, 2003; Christensen et al submitted). However, these relationships are complicated by a considerable time lag and a high sensitivity to climate variables like water temperature and wind stress.”

With respect to the number of days with nutrient limitation, figure 8.7 of the Scientific Documentation Report shows a direct correlation with Chlorophyll a concentrations, but with considerable scatter (considering the log scale of the y-axis). Two questions can thus be posed: Do the additional indicators measure a significantly different indicator compared to Chlorophyll a and Kd, and can the effects of nutrient reduction on both indicators be estimated reliably?

The Panel is of the opinion that both criteria lead to doubt about the usefulness of these indicators. It is clear that phytoplankton production and biomass is related to the amount of organic matter sinking to the bottom and fuelling oxygen consumption. This could be a reason to include the spring in the Chlorophyll a indicator, but anyhow a correlation between Chlorophyll a and the probability of occurrence of hypoxia can be expected. Note, moreover, that excessive summer Chlorophyll a is the ecological sign of hypoxia used. The high dependence of occurrence of hypoxia on weather conditions induces considerable variability that obscures effects of nutrient reductions on the indicators. There is essentially only one (on/off) observation per year. In addition to these difficulties, a rather arbitrary look-up table approach has to be used in order to estimate the required nutrient reduction for improvement in the hypoxia indicators.

For the indicator “days with nutrient limitation”, it can be expected that nutrient reduction, if effective on Chlorophyll a at all, can only be effective through the increase of the time duration of nutrient limitation. It is hard to see how the response of this indicator could differ from the response of Chlorophyll a concentration. The latter, however, can be measured more easily and more reliably. We note that there is considerable disagreement in the literature on the correct value of Km, the Monod limitation parameter, and that it differs considerably between different phytoplankton species and groups. Also for this indicator, a look-up table has to be used to estimate the required load reductions.

In summary, even though the ancillary indicators aim at describing important ecological phenomena, it is not easy to translate them into required load reductions (expert judgment and look-up tables are needed) and their added value compared to Chlorophyll a and Kd is limited. Therefore, the Panel is of the opinion that these indicators do not bring a substantial improvement of the approach. The Panel recommends using the mechanistic models to better study how the important phenomenon of oxygen depletion can be linked directly to required nutrient reductions before using it in practice to estimate required nutrient reduction. If, based on these studies, it can be decided to use these additional indicators, they should be introduced in both statistical and mechanistic modelling approaches for consistency of the approach.

5. Emphasis on nitrogen versus phosphorus

In this chapter, it is evaluated to what extent the Scientific Documentation Report *a priori* focused exclusively on nitrogen (N) reductions as measures to reach Good Ecological Status, or if evidence is given that rules out positive effects from phosphorus (P) load reductions. We thus address the question whether management options were unnecessarily limited by focus on reduction of the yearly N load only.

5.1 Phosphorus limitation

The nutrient emissions from large point sources were dramatically reduced already during the 1980s, causing a large and relatively sudden decrease in the P loads. After that first effort, focus has been directed to mitigation of N loads, primarily through measures in agriculture. This has resulted in a smoother, but substantial, decline in the nutrient inputs (mostly N inputs) thereafter. Currently, the overall inputs of N and P are roughly about 4.2 and 3.4 times higher, respectively, than estimated reference inputs for the year 1900 (Riemann et al, 2016). This indicates that the N/P ratio of nutrient inputs is not exceptionally deviating from the historic inputs.

Previous studies have shown substantial and significant reductions of primary production (e.g. Timmermann et al, 2014) and Chlorophyll a concentrations (e.g., Riemann et al, 2016) in response to the early P load reductions. Thus, there is no doubt that reduction of P loads can, in principle, lead to improvement of water quality in terms of WFD indicators. However, it is uncertain to what degree these historical responses are transferrable to present-day conditions, because emissions from point sources did not have the annual cycle of the diffuse sources and, moreover, they were observed in generally hypertrophic situations that are not comparable to the present state.

Traditionally, marine coastal waters have been regarded as N limited, but in the past decades, scientists have become increasingly aware of complicated co-limitation patterns and intricate nutrient dynamics. Processes such as N fixation and sediment P release can modify long-term response compared to the direct response of phytoplankton to nutrient additions on short time scales. A number of studies from Danish waters confirm that N is in general limiting algal production during summer time, and P is often limiting in spring, but there are seasonal and spatial variations of nutrient limitation. These field studies suggest that at least in a number of systems, regulation of annual primary production by P load reduction could be feasible.

5.2 Treatment of nitrogen and phosphorus in the Scientific Documentation Report

There are several elements that have contributed to the large emphasis on N load reduction in the Scientific Documentation Report. In particular, we discuss the nature of the indicators used, the selection of the study period, the procedures of the statistical modelling and the characteristics of the mechanistic model.

The basis for all calculations are the indicators Chlorophyll a and Kd during summer. This has potential implications for the exclusive focus on nitrogen load reductions. Summer phytoplankton in most Danish water bodies is predominantly nitrogen limited. The choice of summer Chlorophyll a as an indicator may have focused the attention primarily on processes that are dominant in summer and on nitrogen loads as a primary factor responsible for eutrophication. This is also pointed out in the Scientific Documentation Report, where it is suggested that developing new indicators focusing on other parts of the season would give a more diverse focus on both nitrogen and phosphorus.

In general, the Panel is of the opinion that the selection of indicators only representing summer conditions could be too restrictive. In waters with some degree of stratification, the spring bloom has the highest contribution to export production, fuelling the organic matter on the sediment and largely determining the oxygen demand in the rest of the season that could lead to P release from the sediments. The Scientific Documentation Report suggests that limitation of the spring bloom by P occurs in a number of water bodies, thus suggesting that the effectiveness of P load reduction on an indicator representing the full growing season could be significant.

Another factor potentially excluding possible influence of P in the analysis is the selected period for the statistical model (1990-2013). This period excludes most of the period of major development of efficient sewage treatment in the 1980s that caused a major decrease in point source P loads. For most water bodies, the P load trends that are now dominated by diffuse sources are less significant than N load trends, and thereby it is naturally more difficult to find significant effects. However, the Panel endorses the choice of period, because the seasonality and mechanisms of P limitation in current situations may differ from the historical, point-source dominated situation, as argued above.

In the statistical modelling approach, the variable selection procedure may have masked the potential role of phosphorus load reduction. There is a bias in the selection of variables towards regressions with N. This occurs first through the automated variable selection process. Whenever N load is selected as the dominant variable, possible P dependence is disregarded because P load is no longer considered as a secondary independent variable. If, on the other hand, P is selected as the dominant controlling variable, that regression model is not used. Thus, potential influence of P load reductions, or combinations of N and P load reductions, are not investigated further.

The mechanistic models include all relevant processes for modelling effects from both N and P and combinations of them both. However, major focus in the formulations of scenarios is on N, and the few scenarios, including also P reductions, are not detailed and perhaps not optimal for exploring the influence from P load reduction. In addition, we observe that for a significant portion of the water bodies, the models seem to overestimate P concentrations during summer. This can have eliminated the potential impact from P load reductions on the indicators.

5.3 Possible implications for management

Based on the different factors leading to a focus on N load reduction, the Panel concludes that the study does not demonstrate significant contributions from P loads on the summer indicators, but the evidence is not strong enough to exclude that P reductions or combined N and P reductions could be effective in reducing year-averaged chlorophyll levels as well as sediment oxygen demand.

Keeping the option for combined N/P reduction open may have significant management implications in regions where very large N load reductions are demanded. Focused studies resulting in an envelope of combinations of Maximum Allowable Inputs of N and P would probably lead to greater flexibility and more cost-efficient nutrient reduction management in these areas. In making this recommendation, the Panel acknowledges that great efforts have already been made to reduce the P load from urban waste waters, and that little gain has to be expected from intensifying those efforts, following the law of diminishing returns. However, any innovative approach to reducing remaining P loads, including the P load from agriculture, could significantly enlarge the portfolio of potential measures. The Panel recommends using basin load models in combination with the mechanistic models used in the Scientific Documentation Report to investigate these possibilities.

5.4 Seasonality

The exclusive focus on summer indicators in combination with water bodies with short residence times implies a direct link between summer loads and the indicator. Typical residence times in Danish estuaries are short in many cases, ranging from a few days to about 3 months (Rasmussen and Josefsson, 2002). Even if the indicators would include the spring phytoplankton bloom, regulation by N loads would mostly focus on the summer period in water bodies where P limits the spring bloom. There seems to be a possibility to regulate Good Ecological Status by focusing on the summer loads, rather than on the yearly integrated loads. The Panel recognises that the problem is complicated by N retention in the system in the form of organic N stocks accumulating over the season and even years, so that the calculation is not straightforward. Moreover, spatial displacement of problems to other systems as a consequence of flushing winter nutrient loads has to be taken into account. Even so, the Panel estimates that the modelling tools developed, especially the mechanistic modelling, are able to investigate scenarios with seasonal regulation of the N (and P) input into the system. Therefore, nutrient load management could be focused on optimising the effect in the coastal estuaries. The Panel does not have a complete overview of the potential, in agricultural practice, to focus in particular on summer N load. However, it recommends exploring the possibilities to do so and use the mechanistic models to estimate how this would affect the GES indicators.

6. Statistical modelling

In this chapter, the Panel reviews and sums up the objectives and basic setup of statistical modelling and their usage in defining Maximum Allowable Inputs (MAI). The main issues are averaging of statistical and mechanistic models, the analysis of within and cross-system variability in Chlorophyll a and Kd responses, collinearity of phosphorus (P) and nitrogen (N) loading, filtering out the effect of flushing and the uncertainty and resulting risk of over- and under-dimensioning of MAI.

6.1 Setup

The statistical model approach as presented in the Scientific Documentation Report aims at demonstrating the dependence of the indicators Chlorophyll a, Kd, hypoxia, anoxia, number of days with nutrient limitation on the N and P loading of the system as well as on some other physical and chemical characteristics of the system. The statistical models (there is one model per sufficiently monitored water body) also estimate how concentrations of Total N and Total P depend on the nutrient loadings and physico-chemical characteristics. These latter analyses are informative on the functioning of the systems but are not really used any further in the overall modelling procedure.

A few basic choices for the setup of the statistical modelling have been made at the start of the study. The most important choices were:

- Restrict the database analysed by the statistical modelling to the period 1991-2012. This implies that the major decrease in P input, as well as the ecological consequences of this decrease, in general are not part of the analysed database.
- Restrict the construction of statistical models to those systems where sufficient data are available. What is “sufficient” is always open to discussion, but the Panel is of the opinion that the choices are reasonable and have been well justified.
- Use annual averages of nutrient loads, concentrations and other variables as the basis for modelling.
- Construct one statistical model per water body without cross-system model building.
- Perform a variable selection method for significant independent variables, where (due to collinearity problems) only one type of nutrient loading (either N or P) was entered into the set of independent variables.

The most important results of the statistical models are the slopes of the relation between N loads and the indicators Chlorophyll a and Kd. These slopes are only determined, if N load was selected as the most important independent variable and thus entered into the statistical model. When this was not the case, substitute solutions have been used. The statistical model that was used to estimate the slopes (Partial Least Squares) is different from the model used to select the variables (Multiple Linear Regression).

The construction and use of the statistical model are well explained in the Scientific Documentation Report. Measures of goodness-of-fit are given at different stages in the description. No formal uncertainty analysis of the model as a whole, nor variance estimation of the estimated parameters (in particular the N load – indicator slopes) have been given.

6.2 Panel evaluation of basic model setup

The Panel distinguishes three major uses of the results of the statistical modelling:

- Estimation of the relation between nutrient loading and indicators at relatively long time scales (5-10 years), as a basis for estimation of reference conditions of Chlorophyll a, and of the effectiveness of load reductions in reaching the target conditions.
- Provide insight into the water body characteristics that explain the differences between water bodies in status or slopes.
- Provide an independent, evidence-based check on the accuracy of the mechanistic modelling approach.

The Panel remarks that the statistical models are not needed to ascertain that nutrients, both N and P, are important for phytoplankton. This point was also made by the researchers, stressing that the body of scientific evidence showing these relations is massive.

In contrast to the researchers, however, the Panel questions if the step of variable selection was needed at all. It involves mixing of two methods (MLR and PLS). It also leads to the suggestion that in some systems, N load was not involved at all in determining Chlorophyll a and Kd. In systems where N load was selected as the most important determining factor, possible secondary effects of P load cannot be shown and are obliterated. The most important consequence of this option, however, is that it may lead to biased estimates of the slopes and MAI. If, in a particular water body, the slope is very small (close to zero), it is very likely that N load will not be selected as the most important independent variable in the variable selection procedure. Subsequently, for this system, the slope will be estimated as the average type-specific slope, almost inevitably leading to a higher slope than shown by the data. This will then lead to a lower reference and target value for the system than the one suggested by the data. As these reference values will enter into a type-specific averaging afterwards, the final consequences of these choices become difficult to assess, but likely affect the targets for all systems in the type.

Moreover, the Panel is of the opinion that there is no real reason for estimating the short-term response of the indicators on year-to-year variations in nutrient loads, with or without time lags of a few months. Both nutrient loads and concentrations of nutrients and chlorophyll are known to vary considerably with freshwater discharge, which is variable from year to year. Short-term (i.e. year-to-year) responses of indicators to short-term variations in nutrient loads will not necessarily be the same as the decadal-scale responses that the study really wants to estimate. For instance, high discharge will not only increase the total load of nutrients to a system, but simultaneously also decrease the freshwater residence time and thus the ability of the ecosystem to take up and use these nutrients. This may contrast with a decadal-scale increase in nutrient load, where clearer and possibly also different ecological responses might be expected.

Therefore, the Panel is of the opinion that a clearer focus on the long-term slopes and the cross-system variability is needed. Through the use of mixed or Bayesian hierarchical models, short-term and long-term variations can be separated and collinearity between variables can be built in as part of the model (Malve & Qian, 2006).

Danish water systems differ in a number of morphological and hydrographical characteristics, leading to a diversity of systems that is not very well captured by the few types used in the typology (see Chapter 3 in this evaluation report). However, there are a few characteristics that presumably dominate the differences in nutrient, chlorophyll and Kd status between systems. The relative influence of freshwater in the water, dependent on discharge rates, flushing rates and exchange rates with the coastal system, will most probably be a key parameter. Nutrient concentrations in seawater are relatively stable and do not differ very much between the reference conditions and now. In contrast, nutrient concentrations in freshwater are much higher and are obviously much more directly influenced by nutrient loads. As a consequence, it may be expected that much of the variation in status and slopes between systems may be explained with a cross-system statistical model as suggested above. The main purpose of this setup is to improve within-system estimates of slopes with information coming from similar systems elsewhere, and to improve meta-modelling applications. It should lead to a model that estimates the slopes (which are the results of primary importance) based on independent variables summarising the water body characteristics, while simultaneously estimating (and evaluating) system-specific deviations. Such an approach could constitute an improvement with respect to the current within-system modelling approach.

In order for the statistical model to provide an independent, evidence-based check on the results of the mechanistic modelling, two requirements must be fulfilled. First, the procedures of the statistical and mechanistic modelling should not be unduly mixed at early stages (see comments in Chapter 8 in this evaluation report). Second, the statistical model should contain a formal estimation of variances of the estimated parameters. Statistical modelling techniques have much better formal methods to estimate uncertainty than mechanistic models, and this opportunity should be taken in order to better formalise both uncertainty resulting from modelling and from data uncertainty. For this evaluation to be effective, the setup of a single cross-system statistical model is better suited than the current set of separate within-system models.

6.3 Panel evaluation of statistical model results

Even though the simultaneous development of two model lines, statistical and mechanistic, may seem redundant at first sight, the Panel endorses the continuation of this approach. The richness of the Danish database is an internationally exceptional asset that provides the opportunity of an evidence-based check on mechanistic model outcomes. This asset should be used, and the two modelling lines are a very good way to do so. However, the Panel recommends strengthening this aspect, e.g. by keeping the two model lines more separate and independent throughout the modelling procedure, so that the check becomes clearer and more explicit in the final stages of result interpretation. Furthermore, as specified above, the Panel is of the opinion that a cross-system approach in the statistical modelling would strengthen the possibilities of obtaining insight into possible causes for model divergence and would assist better in choosing final management strategies based on the model comparison. In addition, a formal uncertainty analysis of the statistical model would contribute to this goal.

With respect to the present outcomes of the statistical modelling, the Panel sees reasons to suspect bias in estimated slopes and reference values due to the variable selection procedure, as specified above. The Panel suspects that the slope estimates, being a mixture of short-term and long-term ecological responses, might be biased as estimators of the long-term response. However, the Panel does not consider these remarks as a reason to entirely dismiss the statistical model results as unreliable. The mentioned discrepancies are probably minor in comparison with the overall range of the results and in comparison with the inevitable variability in the observations. The within-systems PLS regression approach used is robust and not expected to be overly influenced by the mixture of short and long time scales. The variable selection procedures have led to the replacement of slopes with type-averaged slopes, but mostly in types with small slopes. Nonetheless, there is enough reason to improve the statistical model and the slope estimates that follow from it.

7. Mechanistic modelling

The mechanistic models are evaluated in this chapter with respect to included processes and technical implementation, performance and the different scenarios that are used.

7.1 The models

The mechanistic modelling is based on the DHI systems MIKE 3 combined with ECOLAB. Four models are set up: a large-scale model encompassing the whole Baltic Sea up to Skagerrak (IDW model) and three models of specific estuaries; Limfjorden, Roskilde Fjord and Odense Fjord (estuary models). In all, 45 of the 119 Danish water bodies are covered by the mechanistic models. The IDW and estuary models differ in some specific ways, adapting them to the circumstances. However, the three implementations of the estuary model are identical in terms of processes, but needed somewhat different calibration.

The pelagic dynamics in both models follow classic NPZ concepts similar to other models, and the bacterial loop is not explicitly resolved. An addition to many other similar models is that internal nutrient pools are explicitly modelled using the Droop equations (Droop, 1968). Both models also feature explicit benthic vegetation state-variables, but not benthic fauna.

The estuary models are quite comprehensive in terms of processes, including sophisticated representation of benthic vegetation and elaborate description of resuspension coupled to dynamic wave-shear processes from the hydrodynamic model. Spatial sediment characteristics are taken into account both for sediment-water interaction and as controlling the benthic vegetation.

Specifically, the IDW includes three autotrophic groups to take into account the seasonal succession and nitrogen fixation typical for the open sea areas of the Baltic Sea. Further, the representation of the sediments does not include explicit representation of inorganic particles and instead an empirical direct relationship between shear stress and turbidity is used. Simplification of the sediment module was necessary because of lack of detailed information from the wider area and because of computational constraints.

In many biogeochemical models, Chlorophyll a is estimated from the autotroph biomass in retrospect using a specific ratio. The models used in the Scientific Documentation Report are more advanced in this aspect in that Chlorophyll a is dynamically calculated based on fitness of the autotrophs and light conditions. In the IDW model, where there are three autotrophic functional groups, the weighted average contributions from all groups are taken into account in calculating the production and removal of Chlorophyll a.

The water transparency, Kd, is computed from a relationship that includes Chlorophyll a concentration, detritus carbon, (coloured) dissolved organic carbon and inorganic matter. All these components are explicitly modelled, although the inorganic matter representation in the IDW is a less complicated empirical relationship than in the estuary model.

In summary, the models are quite comprehensive and include all processes that we think are relevant for the problem at hand. The MIKE system, with its sub-components, is a mature system, although it is not so frequently used by research scientists, and, therefore, there are not that many peer-reviewed articles with applications as there are for some open access model systems. Despite this, we have no reason to question the model system capabilities.

7.2 Model setup, calibration and validation

All model setups have high resolution, both in horizontal and vertical. The IDW resolution is sufficient to resolve the internal physical dynamics, both of the narrow straits and geostrophically balanced Kattegat-Skagerrak front. The computing cost of the high resolution and high degree of complexity is significant, leading to a trade-off in the execution of calibration and experiment simulations. All relevant forcing functions are taken into account in a sensible way. The time period of simulation was 2002-2011. A critical part of the riverine inputs is the division of whatever carbon data available into the different categories of organic carbon in the model, especially the CDOC (coloured dissolved organic carbon) that influences Kd and the refractory and labile fractions of organic nutrients. This has been handled to the extent possible according to the Scientific Documentation Report.

At least the hydrodynamics of the IDW model have been used previously and were set up as a part of the EIA for the Fehmarnbelt fixed link project. The models for Odense Fjord and Roskilde Fjord are applied to vegetation modelling applications in Kuusemäe et al (2016) and Flindt et al (2016). Only the Limfjord model is newly developed. Thus, there is some history behind three out of four implementations.

All four model implementations are calibrated independently. That resulted in somewhat different parameter setting, also of the structurally identical estuary models. According to the researchers, there are only about 10 parameters that differ, and all of these are within the sediment module. The actual calibration procedure is not described in detail, but for the three models that have a past history, it can be expected that this has been an iterative process over some time.

7.3 Validation

The hydrodynamics are evaluated quantitatively with respect to salinity and temperature. Salinity is important since it indicates whether circulation is correct and gives the right mixing between the riverine water and open sea water in the estuaries of different sizes. Temperature is of less importance for the circulation, but of imperative importance for the biogeochemical processes. The quantitative comparison shows that the model results are well within the criteria. Upon request from the Panel, the researchers supplied direct time-series comparisons between observations and model results for all four models, and inspection of these shows excellent agreement between model and data for both salinity and temperature. The Panel is convinced that the models give a quite accurate representation of the physical processes.

The validation of the biogeochemical models is done primarily through comparison with observations of Chlorophyll a, Kd and nutrient concentrations. To simplify presentation of the validation of the biogeochemical processes in the models, results are aggregated per water type and month. This presentation may hinder interpretation of the magnitude of the difference between modelled and observed annual cycles. Quantitative skill assessment was performed by computing a cost function (measuring mean deviation scaled by variation of the variable) and correlation from simultaneous model results and observations. Upon request from the Panel, the researchers also supplied example time-series of concurrent observations and model results from selected locations for the standard measured variables.

The model separates well the differences in Chlorophyll a, Kd and nutrients between the water types, and mean values are well captured for all variables.

The seasonal cycle of Chlorophyll a is well captured, although levels are somewhat low during late spring – early summer in type 2 and 3 water bodies, and the autumn bloom seems to be underestimated in type 1 and 2 water bodies. The seasonal cycle of Kd is quite weak in especially type 1 and 2 water bodies, so it is difficult to value the accuracy from the seasonal averages in these water bodies. The tendency for all types 1-3 is, however, that Kd in summer is less than during winter, indicating some influence from an early spring bloom, but probably more from winter river runoff and turbidity from resuspended material. The time-series plots of Kd supplied by the researcher confirm the complications. The two open sea stations show seemingly random variations in time of observed Kd due to short-term variability, and no visual seasonal cycle or trend can be identified. There is no annual cycle (and only small variation) to be seen in the time-series supplied for Odense Fjord and central Limfjorden, neither in observations nor in model results. In Roskilde Fjord, there is significant variation in Kd with the seasons, but from visual inspection the pattern is irregular and not very well captured by the model, and it is not obvious what causes the variations. The time-series with clear seasonal cycle are from the inner part of Skive Fjord, and here the model accurately simulates the low Kd in winter time and high Kd in summer time.

The seasonal TN is modelled accurately for all water types. Winter DIN is somewhat overestimated in type 1 and 2 water bodies, and DIN is somewhat overestimated in late spring – early summer in type 3 waters. The overestimate of winter DIN in type 1 waters is confirmed for the time-series examples supplied by the researchers. However, overall, the model performs well on the nitrogen cycles.

There seems to be a consistent overestimation of DIP in the summer in type 1 and 2 waters, although somewhat later in type 1 than in type 2 waters. From inspection of the time-series, it seems that the problem is larger in the IDW, smaller in the Limfjorden model, while in the Odense Fjord and Roskilde Fjord, the seasonal cycle is quite correct. Winter DIP and TP concentrations are accurately modelled for all water types.

The quantitative validation in terms of cost function and correlation confirms the qualitative validation discussed above. Overall, the model is low in bias (cost function) indicating that the levels are modelled accurately, with exception of DIP in type 3 and to some extent type 1 waters and Kd in the type 5 waters. However, correlation is absent for type 1 water bodies and weak for type 3 water bodies for Kd.

The comparison between modelled and observed primary production indicates that the model performs well in this respect.

7.4 Reference conditions simulation

A hindcast simulation representing conditions around 1900 was performed. Forcing in general was kept as for the 2002-2011 period, but loads and nutrient boundary conditions needed adjustments. Appropriate waterborne and airborne loads were obtained from existing well-established data sets, and boundary concentrations in Skagerrak were adjusted according to previously published methodology. To overcome the computational challenge of running the whole of the Baltic Sea to steady state, initial conditions were adjusted in the IDW model according to literature values. It is the Panel's opinion that the setup of the simulation of reference conditions with the mechanistic model is sound and based on current published scientific knowledge on the nutrient loads around 1900.

Reference Chlorophyll a concentrations for all water bodies were extracted as average of the last 5 years of the simulation. In a few cases, simulations were repeated in order to be sure that average conditions were in equilibrium with the reference loads. It is unclear whether Chlorophyll a concentrations were spatially averaged over the water bodies or not.

7.5 Scenarios and establishment of cause-effect relationships

A prerequisite in construction of load reduction scenarios is implementation of BSAP for other countries than Denmark. That implies major reductions of primarily phosphorus to Baltic Proper, Gulf of Finland and Gulf of Riga, but also nitrogen to Baltic Proper, Kattegat and Gulf of Finland. The response time to the load reductions to Baltic Proper and the Gulfs is very long. Estimations show that during the first decade after implementation, all changes are within natural variability, but significant reduction in winter nutrient (primarily phosphorus) concentrations will be seen between one and two decades after implementation (HELCOM, 2013). The response time-scale has been shown to vary between models (Eilola et al, 2011), but is long in all cases. This means that the influence from load reductions to the Baltic Proper and the Gulf is limited during the decade considered here. It could be noted that in the longer perspective, nutrient concentrations would continue to decrease, and according to the underlying calculations in the BSAP, load reductions to the Baltic Proper was a prerequisite for obtaining GES in the Danish straits.

Three nitrogen reduction scenarios for the Danish loads were constructed by reducing proportionally all waterborne loads by 15, 30 and 60%, respectively.

There is also a set of scenarios where the three nitrogen scenarios are combined with a spatially distributed phosphorus reduction scenario according to reductions specified by the Danish EPA. It is mentioned that no significant effect could be detected from the P load scenarios, but there is no further elaboration on these scenarios. If the distribution is such that most of the reduction occurs to relatively few water bodies, there could potentially be an effect in these that would not be seen overall.

Indicator values from model results are calculated as water body spatial means, and these are corrected to match the mean observational value at the measurement station.

The scenarios without phosphorus load reduction are used to estimate parameters for a simplified surrogate model, built on temporal averages over 2007-2011. The three scenarios are used to establish the linear response function. The extrapolated value at present day Danish loads will represent the indicator value, given only reductions by other countries. In the Scientific Documentation Report, also an average indicator value from the reference scenario is included to indicate how much higher the value will be because of higher loads from other countries. For most water bodies and indicators, the linear approximation is appropriate. It should be remembered though that only a proportion of the full effect from BSAP reductions has had time to develop in the scenarios, and one would expect that for open sea water bodies, especially in the south, water quality will continue to improve as time goes by.

There are implications from the approach of running scenarios with a constant proportional load decrease in the scenarios. Some water bodies will be subject to change due to load reductions to adjacent water bodies. Therefore, one cannot directly sub-divide MAI to individual water bodies, if there is a risk that reduction is necessary also in adjacent basins to obtain GES. To fully disentangle the individual contribution spatially between all water bodies, one would need to test sensitivity to load reductions to each individual water body by itself, and perhaps, if the effect is non-linear, even combinations of water bodies. This would be a major computational challenge, and the improvement in the results would most probably be minor. The reason for the latter is that the problem mostly applies to open sea water bodies that would in any case integrate the load reduction for a relatively large region, while enclosed water bodies still are mostly dominated by local reductions.

7.6 Conclusion on the mechanistic models

Having evaluated the mechanistic models, the Panel comes to the following conclusions:

- The models are clearly state-of-the-art, both in terms of numerical techniques and processes included. The quality of the results follows a high standard and is as good as, or better than, other similar coupled physical-biogeochemical model systems.
- The hydrodynamics seem to perform excellently.
- Levels of Chlorophyll a, Kd and nutrients are accurately modelled across water body types.
- The biogeochemistry seems to perform overall somewhat better for nitrogen than for phosphorus, although in the models for Roskilde Fjord and Odense Fjord, also phosphorus performs excellently. Weakest is the performance of nutrients in the IDW model, where relatively frequently DIP seems to be overestimated during summer or early autumn and nitrogen during winter. Observed short-term variability in Kd in open waters is such that it seems impossible to model.
- Long-term response to large changes in nutrient loads has not been validated.
- The nitrogen reduction scenarios are appropriately set up and relevant.
- The scenario for P reduction is not extensively described, and it cannot be judged whether it forms sufficient basis for exclusive focus on N.
- It should be noted that in a longer time perspective, >10-20 years, the effect from BSAP load reductions will influence the open sea water bodies, especially in the southern part of the region.
- It would be extremely valuable to extend the mechanistic modelling system to as many water bodies as possible.

8. Calculation procedures to estimate Maximum Allowable Inputs from model results

In this chapter, we discuss the general build-up of the procedure to estimate reference conditions, Good-Moderate boundary targets and the required N load reductions to reach the target conditions. These procedures are based on the statistical and mechanistic model results, but use and interpret them in a diversity of ways. In our discussion, we focus on how the different models interact and on the different steps taken to arrive at the final MAI per water body.

8.1 Steps in the calculation of targets and MAI

Despite the general logical nature of the procedure, and even though the Scientific Documentation Report gives extensive explanations of the detailed procedures followed, it is not easy to follow and weight the different steps used in deriving the Maximum Allowable Inputs (MAI) for the water bodies. The essential steps, as the Panel understands them, are summarised in the diagram shown in Table 2. The left column refers to the procedures followed in the statistical modelling, and the right column to the mechanistic modelling. Joint cells point to steps where both approaches are joined.

Table 2. Essential steps in the calculation procedure of targets and MAI in the Scientific Documentation. Steps with averaging have red boxes.

<table>
<tr><th>Statistical modelling</th><th>Mechanistic modelling</th></tr>
<tr><td>Estimate the slope of the Chlorophyll a/N load relation for those systems where N load was selected as a significant independent variable in the regressions. For 8 water bodies where this was not the case, the average type-specific slope was used.</td><td></td></tr>
<tr><td>Estimate the slope of the Kd/N load relationship, and substitute with average type-specific slopes where no significant relations could be found (6 water bodies).</td><td></td></tr>
<tr><td>Estimate 1900 reference Chlorophyll a levels, using 1900 N loads and the slopes as input.</td><td>Estimate 1900 reference Chlorophyll a levels, using a 1900 scenario with adjusted nutrient inputs (N, P), adjusted benthic stocks etc.</td></tr>
<tr><td>Do not estimate 1900 Kd from the models; use historic observations instead. Where no direct observations were available, use observations from nearby similar water bodies.</td><td>Use the same historic data on Kd as 1900 reference as the statistical modelling.</td></tr>
<tr><td colspan="2">Estimate Chlorophyll a reference levels per water body type by averaging the reference levels coming from the statistical and the mechanistic models of all water bodies in the type. Notes: For type 1, some subtypes are defined; a few systems have a status aparte.

The slopes are not averaged, but kept per water body and model type.</td></tr>
<tr><td colspan="2">The same procedure is NOT followed for Kd. Historic references are used in both approaches.</td></tr>
<tr><td>Estimate the required N load reduction to reach the target values for Chlorophyll a, Kd, hypoxia, anoxia, days with N limitation. Where logical inconsistencies may exist (reductions >100%), use a look-up table to substitute calculations. Unclear how this is done if</td><td>Estimate the required N load reduction to reach the target values for Chlorophyll a and Kd, taking into account the fraction due to Danish land-based sources. Based on scenarios with varying degree of</td></tr>
</table>

>100% is needed for Chlorophyll a.	overall reductions of N input.
Calculate the required load reduction as a weighted average of the results in previous step.	Calculate the required load reduction as a simple average of the results in previous step.
	Smooth the variability in required N load reduction by regional averaging. Unclear what was the basis for delineating the regions.
Meta-model systems without a model. IF status information is available, use type-averaged slopes for N Chla, N Kd and N other indicators (latter only if their status is known). Calculate weighted average required N reduction. ELSE use type-averaged required reduction.	Meta-model systems without a model. IF status information is available, use type-averaged slopes for N Chla, N Kd. Calculate average required N reduction. ELSE use regionally averaged required reduction.
Average required N reduction for meta-modelled systems across statistical and mechanistic approaches.	
IF mechanistic model exists for system: Drop information from statistical model and only use mechanistic model result. ELSE use statistical model result.	
Apply upstream-downstream rules.	
All done!	

8.2 Averaging and "ensemble modelling" aspects in the procedure

In this procedure, both modelling approaches are largely independent and focused on individual water bodies. However, four critical averaging steps intervene:

- The Chlorophyll a targets are averaged per type, over the statistical and mechanistic models. As far as the Panel understands it, this does not apply to the slopes, which remain water body-specific. In the Scientific Documentation Report, the averaging is justified as a means of reducing variability. As a consequence of the averaging, there is a possibility of incompatibility between slopes and targets: More than 100% of N load should be reduced. It remains unclear how this is solved. Presumably the reductions >100% enter the weighted average over indicators as such and are corrected by the averaging over indicators.

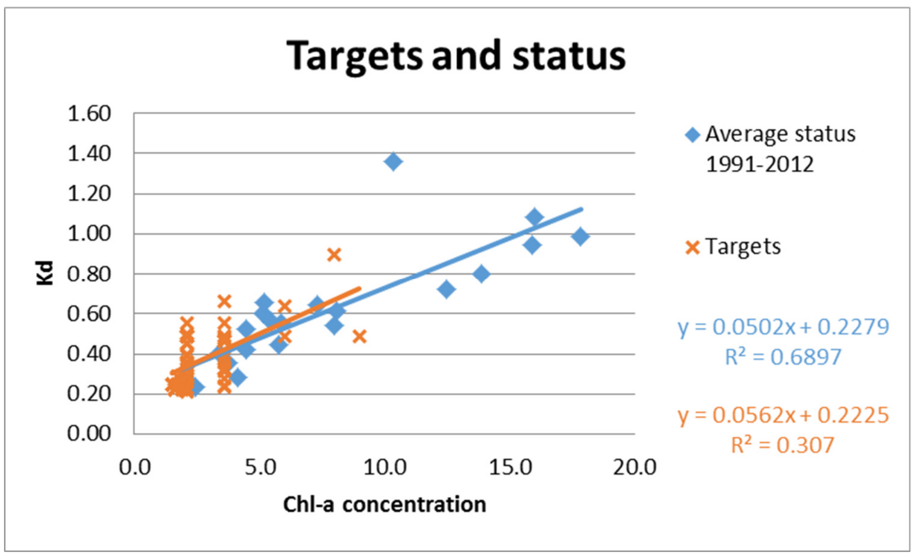
- The required load reduction for the different indicators (most importantly Chlorophyll a and Kd) is averaged quite early in the procedure. This averaging violates the "one-out-all-out" principle, as clearly stated by the researchers, but is justified in the Scientific Documentation Report based on arguments of reducing random variation in the required load reductions.

- The mechanistic model results, in terms of required percentage load reduction, are spatially averaged before the final meta-modelling step is applied to systems without monitoring data. This step obscures differences between water systems in regions. In the Scientific Documentation Report, it is very shortly discussed, and justified based on observed variability between systems in the regions that is – without proper argumentation – attributed to variability in the data on system status. It is not clear to the Panel why this would be the justification for a relatively drastic step in reducing the spatial differentiation in the model results. The Panel feels that this is an important yet poorly argumented step in the entire procedure.

- For meta-modelled systems, statistical and mechanistic meta-model results are averaged. This is not the case, however, for individually modelled systems where the mechanistic model has prevalence in the cases where both models are available.

In the opinion of the Panel, the most problematic aspect of the procedure is the averaging of Chlorophyll a reference (and GM boundary target) values across model types and within water body types. As the types are relatively broad and contain water bodies with quite dissimilar characteristics, this loss of local detail can easily lead to situations where too much effort is spent in one system and too little in another. In that case, there will be both economic and ecological loss of efficiency. In this respect, it is informative to compare the targets used for Kd and for Chlorophyll a across all systems, as is shown in Figure 4.

Figure 4. Comparison of target values for Kd and Chlorophyll a across all water bodies. Target values are shown as orange crosses. For comparison, the average status values 1990-2012 are shown as blue diamonds. Regression lines of the two sets are remarkably similar.



The figure shows that the target values of Kd, based on historic observations, are quite variable within each of the water body types and largely overlapping between types. There is much more discrimination between water bodies based on Kd than based on the (uniformed) Chlorophyll a targets, implying that the latter are not optimised for the water bodies.

The averaging of the required load reductions across the indicators (primarily Chlorophyll a and Kd) early on in the procedure renders it impossible to judge whether, and how much, the final results in terms of MAI depend on this violation of the one-out-all-out principle. If, for reasons of compliance to the WFD procedures, it would be decided that this is unacceptable, the present results cannot be used to make the recalculation. Also, no elements are offered to evaluate the importance of this decision.

The adoption of common Chlorophyll a target values across the statistical and mechanistic models also leads to a loss of independence of the two modelling approaches. As a consequence, comparison of the required N load reduction obtained by the two different methods, as a check on the methodology, is not really possible anymore. The Panel is of the opinion that it would be better to keep both methods separated up to the last stage and then do an in-depth comparison, taking into account water body characteristics to explain or understand any discrepancies. Averaging two independent model results as “ensemble modelling” is an option that can be taken in order to reduce variation in results, but it does not necessarily lead to a better solution. If one of the methods is biased (e.g. is clearly unable to make reliable estimates in particular types of systems), averaging is a worse solution than dropping the bad prediction. The availability of extensive databases on almost all systems should allow the model comparison to be evidence-based (e.g. model comparisons with data can be made for systems where predictions differ substantially), so that well-justified choices can be made.

Very little justification is given for the choice to give prevalence to the mechanistic models where both models are available. Even if the choice could be well justified (which is questionable since an independent comparison is impossible), it contrasts with the meta-modelling approach where both are averaged. Consistency in the choice would improve the overall approach.

In the opinion of the Panel, the spatial averaging of the mechanistic model results is not necessary nor justified. There can be good reasons, from a management point of view, to smoothen required load reductions regionally so that no abrupt changes in requirements occur at too small scales where control would be virtually impossible. However, such decisions could better be made on the basis of a map showing the original model results, allowing one to judge whether a management problem is posed or not. As it is performed now, it remains unclear to what degree the spatial averaging leads to under- or overdoing in particular watersheds, leading to lack of transparency in the management rules.

Summarising, the Panel recommends postponing the averaging operations to the very last stages of the procedure. This will keep the two modelling approaches independent, it will allow estimating the consequences of violating the one-out-all-out principle and avoid confusion due to regional averaging. As a consequence, the effects of different modelling strategies and different indicators will remain clear in the different results on nutrient reduction. A close examination and comparison of these differences will allow making informed decisions on the choice of strategy.

8.3 Conceptual differences between modelling approaches

There are a few points where the statistical and mechanistic modelling approaches are not conceptually consistent. The most important point is that the statistical modelling takes into account additional indicators, while the mechanistic models (although capable of doing the same, as all relevant variables are calculated in the model) only focus on Chlorophyll a and Kd. Even if this would not lead to large differences (this cannot be controlled based on the report), the Panel feels that it leads to a different treatment of water bodies, depending on the model applied to it, that is difficult to justify. Based on the observation that the ancillary indicators are strongly correlated with Chlorophyll a and can hardly be justified as probing independent characteristics of the ecosystem (see Chapter 4), the Panel suggests dropping these ancillary indicators from the procedure. It will make the two modelling approaches more comparable without apparent loss of information on the ecosystem. The ancillary variables could better be used as corroborating evidence for the need to take action (or not) in the water bodies concerned and as documentation of the range of ecological results to be expected from sanitation of the nutrient input.

A second potential inconsistency between the two models is that the mechanistic modelling explicitly separates how much of the distance to target can be reached by reducing Danish land-based N loading alone, while such separation is not done for the statistical model. However, the latter bases its regression approach on Danish (in fact: local) land-based N loads, so that, implicitly, the reasoning is probably more similar than it may appear when written out. Overall, the consequences of this difference are extremely difficult to trace, but intuitively the Panel does not estimate this to be a major conceptual difficulty. It may, however, have consequences in practice. A comparison of independent model results of the two approaches would also inform better about this aspect.

8.4 Meta-modelling

With respect to meta-modelling, the Panel remarks that the coarseness of the typology also has potential impact on the meta-modelling. It is clear that for the meta-modelling, some knowledge on the (un-sampled or under-sampled) systems must be used in order to set the best targets and use the appropriate slopes. As the typology used is rather coarse, the current choices may not be optimal for these systems. The Panel sees a role for statistical modelling here, provided that the statistical modelling would also focus on understanding and modelling cross-system differences (in slopes and consequently in targets) as a function of hydrographic and morphological characteristics of the systems. Particularly the importance of freshwater influence in the systems and the flushing rates may be overarching determining characteristics. The Panel estimates that a regression-based approach could be better than a classification approach.

In the opinion of the Panel, the meta-modelling of the North Sea water bodies is less reliable than that of the other water bodies. For the North Sea coastal systems, the background modelling, which has focused on the Baltic systems instead, is not very strong, and the meta-modelling is based on daring extrapolations from systems with quite different ecological characteristics. The Panel recommends that more study is made of the North Sea estuaries in order to improve the estimates of the nutrient load reduction requirements, based on their very different physical and ecological characteristics as well as on the very different basis (OSPAR intercalibration) for the references and targets.

9. Evaluation of Maximum Allowable Inputs results

Maximum Allowable Inputs (MAI) define the annual load of nutrients, in this context of nitrogen, that are acceptable to keep a coastal water body in a Good Ecological Status according to the WFD or allow a water body to return to this status. Since nutrient load management is a complex task and nutrient load reductions are associated with high costs, reliable overall and water body-specific MAI are of outstanding importance. In this chapter, the Panel reflects to what extent the suggested MAI can be regarded as reliable enough to form the basis for policy and management actions.

9.1 The overall Danish MAI in an international framework

The nutrient reduction scheme of the HELCOM Baltic Sea Action Plan was revised in the 2013 HELCOM Ministerial Meeting, based on a new and more complete dataset as well as an improved modelling approach. The new MAI, compared to the reference inputs of 1997-2003 for the Baltic Sea sub-basins Kattegat and Danish straits, demands only a minor load reduction requirement of about 3%. In this revision, Denmark agreed to reduce N loads to the Baltic Sea (from both land and air) by 2,890 t/a and P loads by 38 t/a. The Scientific Documentation Report suggests low N load reductions (>10%) for Western Jutland and most parts of Zealand as well as Lolland and Falster (Figure 8.23, p. 127). This seems reasonable and is well in agreement with international requirements.

However, to meet the targets for a good status, the Scientific Documentation Report demands much higher load reduction, especially on Funen and Jutland. Here, Denmark faces a situation similar to the Baltic coastal waters of Germany. Especially the inner coastal waters, estuaries and bays in Germany require higher N load reduction than demanded in the HELCOM Baltic Sea Action Plan to reach the GES. According to the German plans, the N load from German Baltic river basins has to be reduced by 21,500 tTN/a, with an average maximum allowed total N concentration in rivers of 2.5 mg/l, resulting in an overall reduction of 34%.

For Denmark, depending on the model approach, an average overall reduction between 29% and 34% is suggested. There are many similarities with respect to geomorphology, land use pattern and intensity as well as population and state of sewage purification between the German and Danish Baltic catchments, and the coastal waters share many similarities too. Therefore, the very good agreement in the assumed relative reduction requirements between both countries indicates that the values meet the right order of magnitude and seem reasonable.

However, the reliability of water body-specific MAI depends on the approach for calculating reference conditions and subsequent target conditions, the typology and type-specific targets, the considered indicators, the applied weighting, the model and meta-model approach as well as the data processing and aggregation. The major question is if all these aspects are sufficiently taken into account and if the application has a sufficient quality to determine reliable water body-specific MAI and mitigation needs to achieve the GES in Danish coastal waters.

9.2 Historic conditions as basis for target setting

The process in the Scientific Documentation Report follows the implementation guidelines of the WFD. It means that it is based on historic reference conditions and assumes that these conditions can serve as a basis for the definition of present and future targets. The reference conditions describe the status of biological quality elements that would exist in a situation with no or very minor disturbance from human activities. Reference conditions are therefore not pristine conditions. The WFD allows different methods to calculate reference conditions. In countries with long monitoring data records and the availability of suitable models, historic conditions are usually used as reference state. Because of data availability, this period often refers to a period around 1900, being aware that this period not always reflects a state with very minor disturbance from human activities. Similar to Germany, the Scientific Documentation Report uses the years around 1900 as reference. The Panel finds this approach well justified and the data basis sufficient and suitable.

However, it is obvious that between 1900 and today, land-use pattern and population densities have changed and different regions in Denmark developed differently until today. Further, the year 1900 is well suitable to reflect a high ecological status in rural areas, while cities already at that time emitted significant amounts of untreated sewage and caused pollution in their surroundings beyond the thresholds for a high ecological status.

For the definition of reliable targets, the question is less how did it look like in 1900, but rather how would reference conditions in a region look like, assuming present land-use and population pattern. This means that targets and water body-specific MAI based on historic conditions around 1900 bear uncertainties and for some water bodies may require a deeper analysis. This is especially true for areas with known strong changes between 1900 and today. However, the Panel agrees that this approach is the best choice that still ensures full compliance with technical WFD implementation guidelines.

In Germany, the official national working group on targets and MAI discussed if reference conditions should be calculated for and translated into the present situation. The approach was to use combined river basin and marine models and present population density and land-use pattern as well as the historic specific emissions per hectare and capita to calculate resulting regionalised Chlorophyll a and nutrients concentrations. The idea was to use the values as reference conditions to account for the fact that different regions developed differently during the last 120 years and to be able to provide even more reliable water body-specific targets. However, the majority of the working group declined this approach for containing too many assumptions and for not fully following the technical WFD implementation guidelines. Denmark would face similar problems with this alternative approach.

9.3 Effects of climate change on targets and MAI

Climate change shows its effects only gradually on a time horizon of decades, while the implementation of the WFD and measures to reach GES must take place within a decade. Further, depending on the emission scenario, climate change effects on countries and on regions within a country are uncertain, they show a large variability and are hard to predict. The Scientific Documentation Report addresses this topic and, in our opinion, provides sufficient evidence and reasons why climate change has not been taken into account in the definition of targets and in calculating MAI in Denmark.

However, several nutrient load reduction measures in river basins show the full effect only after decades. Major effects of climate change on Danish coastal waters, very likely, will result from changed nutrient loads as a result of altered spatial and seasonal precipitation and discharge patterns. Therefore, linked river basin – coastal water – sea models used for the assessment of the effectiveness of measures in the river basin should take into account climate change effects on river basin loads and shifts between nutrients. However, climate change can affect internal processes in coastal waters as well. Riemann et al (2016), for example, point out that more frequent stratification and higher water temperatures presumably hampered the improvement of bottom water oxygen conditions and counteracted the expected positive effects of reduced nutrient inputs in Denmark.

9.4 Relevance of typology on MAI

As indicated in Chapter 3, the Panel has the opinion that the Danish typology used in the Scientific Documentation Report does not sufficiently reflect the individual properties of the many Danish fjords and inner coastal waters. This is also true for the typology reported in Dahl et al (2005). Type-specific targets for the indicators, especially Chlorophyll a, that are applied to a wide range of significantly different water bodies do not sufficiently reflect their properties and behaviour to loads reductions. Consequences are less reliable water body-specific MAI. This may cause an underestimation of the required load reduction for some water bodies and an overestimation for others.

9.5 Relevance of indicator choice on MAI

The Panel agrees that Chlorophyll a is a core indicator, and coastal water body-specific Chlorophyll a concentrations are a sound basis for calculating water body-specific MAI. Further, the Panel agrees that water transparency has to be restored as one necessary condition to enable the recovery of eelgrass in coastal waters. Potentially, Kd can serve as an indicator for describing suitable growing conditions for eelgrass. Eelgrass can serve to indicate the status of macrophytes, a biological element in the WFD. Therefore, Kd has the potential to be an important parameter for calculating MAI.

However, as pointed out in Chapter 4, the relationship between Kd in coastal waters and external nutrient loading is sometimes very weak. Further, Kd and the insufficient relationship have different consequences for and are differently treated in the mechanistic and the statistical modelling exercises. In the statistical modelling approach, for example, the use of Kd in some cases causes impossible N load reduction requirements of above 100%. Further, Kd shows only a slow response to load reduction, the data are subject to high variability, and it shows a correlation to Chlorophyll a. Altogether, we consider Kd as a less suitable indicator in many Danish coastal water bodies. A strong weight of Kd in the calculation of MAI should be avoided and would add uncertainty to water body-specific MAI. Chapter 4 outlines possible solutions to overcome or at least to deal with some of these problems. In the Scientific Documentation Report, other indicators are sometimes mentioned and used in the statistical model. We do not see a major advantage of these indicators for the calculation of MAI, because they do not provide significant new information or show correlations to the exiting indicators.

9.6 Relevance of model quality and approach for MAI

In general, the mechanic model has a very good potential for calculating water body-specific MAI, but in the present state it does not cover all water bodies. The statistical modelling is based on real monitoring data, and in most coastal water bodies it can serve as a valuable tool to assess long-term trends as well as the mechanistic model performance. As indicated in Chapter 8, the model application and the process of calculating water body-specific MAI are complex and not entirely convincing. Most problematic is the averaging of Chlorophyll a reference values across both models and within coastal water types. This has negative consequences for the meta-modelled water bodies as well.

9.7 Conclusion and perspectives

Many of these aspects and shortcomings were mentioned and pointed out by several stakeholders as well. The Panel picked up the stakeholder comments and examined in some detail the MAI for specific areas with very high nutrient load reduction demands. Altogether, the Panel largely shares the stakeholder concerns.

The calculation of water body-specific MAI is a challenging task, but potentially has one major advantage: It allows the development of water body-specific management options and solutions. For this purpose, the coastal water and sea models should be combined with river basin models providing information about the quantitative potential and efficiency of single (or sets of) measures and providing load reduction scenarios for coastal models. If river basin models are able to provide nutrient load data on a monthly basis, this would allow the development of scenarios that take into account the seasonality of emissions. Assessing how seasonally differentiated emissions affect the status of coastal water bodies could lead to optimised, cost-effective management.

Taking into account all aspects and associated problems, the Panel has the impression that the water body-specific MAI are not sufficiently reliable to serve as a basis for decision-making and planning of load reduction measures. Further, the MAI are only addressing nitrogen load reductions and leaving out the possibility of potentially managing water bodies via phosphorus load reduction. However, models, competences and data are available in Denmark to meet the challenge to calculate water body-specific MAI. Even a modified processing of the existing model results might lead to much more reliable MAI.

10. Overall assessment and conclusions

The Water Framework Directive aims at restoring Good Ecological Status in surface waters in Europe. The Scientific Documentation Report proposes measures of nutrient load reduction to reach this Good Ecological Status in Danish transitional and coastal waters. The Panel fully endorses the importance attached to nutrient reductions as a necessary requirement to reach this Good Ecological Status and stresses the importance of nutrient conditions as a modulating factor interacting with any additional measures taken to improve the state of the system.

In comparison with many other European countries, Denmark has excellent databases, models and scientific expertise as a basis for the implementation of the Water Framework Directive. The Panel was delighted to see that these resources have been mobilised to achieve a leading position at the European scale. The Panel was impressed by the openness and transparency of the interaction between government, researchers and stakeholders as well as by the high intellectual level of the discussions. This open exchange of ideas and opinions is a perfect basis for a further improvement of the scientific basis for the WFD implementation.

The Panel has reviewed the choice of indicators and procedures, in the context of the WFD requirements and specifications, and found that the indicators, the methods to determine reference conditions and the methods to determine required actions were WFD compliant. The Danish implementation is based on either direct historical observation or model determination of reference conditions. Little or no uncontrollable “expert judgement” is involved. In that respect, the Danish models are attaining the highest possible standard of WFD implementation.

The Panel has analysed the consequences of using a relatively coarse typology of coastal waters for calculating reference conditions, targets and Maximum Allowable Inputs of nitrogen. The Panel concludes that the use of a coarse typology has led to reduction requirements that are not optimal for each of the individual water bodies. The Panel is convinced that the full use of available data and models would allow Denmark to forego the typology and develop advanced, specific reduction targets for each water body. The Panel recommends focusing on the water body scale of resolution throughout the scientific process. The regional grouping of reduction measures should be decided upon only at the stage of translating scientific advice into management action plans.

The Panel has analysed the indicators used and concluded that Chlorophyll a is a useful intercalibrated indicator of phytoplankton, while Kd is less optimal as an indicator of benthic angiosperms and macrophytes. The other indicators, used in the statistical modelling only, currently present methodological problems and are not yet mature enough for inclusion in the management plans. The Panel has identified promising developments in the modelling with respect to angiosperm and macrophyte indicators and made recommendations on how to extend and develop the indicator set in the future.

In view of the large efforts in the past to remove P load from point sources, the Panel endorses the emphasis placed in the Scientific Documentation Report on reducing N loads from diffuse sources. However, at least in principle, there could be an additional role for P load reduction and for seasonal regulation of the N load. The Panel is of the opinion that these options merit further scientific exploration, especially in watersheds where high efforts for N load reduction are required.

Although the maintenance of two parallel modelling lines (statistical and mechanistic) may seem redundant at first sight, the Panel strongly endorses maintaining these lines. Given the wealth of data available, it provides unique possibilities for evidence-based checking of mechanistic model results. The Panel assesses the mechanistic model as a state-of-the-art, very comprehensive tool, but emphasises that independent checking on data as well as uncertainty analysis remain necessary and can be performed by the statistical approach. This coherence can be optimised by improving the approach and methods of the statistical modelling.

The Panel endorses the general logic of the methodology to derive reference and target values from the models and to calculate the required N load reduction to reach the targets. The Panel has identified several points in the workflow where averaging is performed. This results in interdependence of model types, loss of indicator resolution and loss of spatial resolution. It also adds complexity to the procedure and makes it very difficult to understand. None of these losses are necessary since the model results and database do permit a fully transparent derivation of water body-specific required nutrient reduction.

Summing up these different aspects of the work, the Panel positively evaluates that nutrient load reductions are based on **solid** scientific evidence and generally high-level modelling approaches. The Panel is very positive about the near lack of expert judgment in the work and is of the opinion that in the few places where it does occur, it is not necessary and can be removed. The general (country-averaged) level of required nutrient load reduction compares favourably with independent efforts in similar areas and seems a **robust** measure of what is needed. At the same time, the Panel assesses the spatial resolution of the required efforts as **unnecessarily coarse**. The Panel is convinced that the rich database, combined with an **improved statistical approach** and the high-resolution mechanistic modelling tools, are able to derive improved, water body-specific MAI values. Current scientific insight endorses the view that the overall reductions proposed are **necessary**, but cannot guarantee that they will be **sufficient**. Especially for benthic angiosperms and macrophytes, additional measures may be needed.

11. Recommendations for going further

Monitoring: The Danish national monitoring programme used in the Scientific Documentation Report includes more than 90 stations along the coast and in the sea. It is very comprehensive and is generally well adjusted to the WFD requirements. It forms the basis for the further development of models, for most calculations and is required to evaluate the success of measures and whether the targets of the WFD are met. The Panel recommends maintaining this monitoring system at full strength and assessing if additional monitoring stations will be required for a water body-specific management.

Typology: The typology has weaknesses in reflecting the individual properties of fjordic water bodies. Instead of suggesting a refinement of the existing typology, we recommend calculating reference conditions and targets for each of the 119 water bodies in Denmark. Denmark is one of the few countries in Europe, where the necessary data, expertise and models are available for such a comprehensive approach. By taking specific conditions and individuality of every water body into account, the calculated targets and water body-specific Maximum Allowable Inputs will be optimised and lead to minimal waste of resources. For purposes of intercalibration, a robust typology can be based on the results of the water body-specific analyses.

Choice of indicators: Chlorophyll a is a generally accepted and intercalibrated indicator of phytoplankton. Kd, as a measure for macrophytes and angiosperms, has certain limitations. The Panel recommends building on recent efforts towards comprehensive modelling of eelgrass in order to derive a better indicator of macrophytes, but to keep Kd as a proxy meanwhile. The other indicators used in the statistical modelling address important ecological questions, but are not mature in the sense that they lack a clear quantitative relation with nutrient loading. The Panel recommends leaving them out of the present modelling and developing targeted modelling directed at their incorporation into the indicator system.

Statistical modelling: The Panel sees great merit in the strategy to maintain two independent lines of modelling, one based on statistical data analysis and the other based on mechanistic modelling. The Panel recommends reorienting the statistical modelling towards optimal estimation of the long-term slopes of the indicators on nutrient loading in a cross-systems analysis way and keeping in principle both N and P loading as explanatory variables. The Panel recommends elaborating the uncertainty analysis in the statistical modelling and suggests that this will be facilitated when a single cross-system advanced modelling approach is chosen.

Mechanistic models: The mechanistic models are state-of-the-art, both in terms of numerical technique and included processes. They are powerful tools for providing a sound scientific basis for the implementation of the WFD in Denmark. A shortcoming is that they do not cover all water bodies. As a consequence, different approaches were used for the definition of reference conditions, targets and MAI in different water bodies. We recommend extending a mechanistic modelling approach to as many water bodies as possible to ensure that, in future, a uniform methodology can be used for the definition of water body-specific MAI.

Methods to derive targets and MAI from the models: The Panel recommends simplifying the calculation procedure by removing the averaging steps between models, between indicators, between water bodies within types and between water bodies on a regional basis. In this way, the differences and correspondences between modelling approaches, indicators and water bodies will become clear and can be further analysed. Cross-checking of results of the statistical and mechanistic model approaches in systems, where both are available, will form a basis for extrapolation to all systems. The Panel recommends deriving one MAI per water body in this way and only deciding in a later phase on regional averaging or lumping, when scientific results are translated into management actions.

River basin interactions: River basin models allow calculating the load reduction potential of nitrogen and phosphorus for each river basin, the development of water body-specific nitrogen and phosphorus load reduction scenarios and cost estimates. Further, they allow addressing seasonal load and limitation patterns. The Panel recommends a combination of river basin and coastal water models to enable the development of water body-specific optimised management concepts that consider both nitrogen and phosphorus.

International approach: The technical WFD implementation guidelines force similar approaches in all member states. As a consequence, requirements, modelling and challenges are similar in different countries. Further, the WFD asks for an intercalibration and harmonisation of targets with neighbouring countries. Therefore, the Panel recommends a co-ordinated joint scientific approach, especially between Denmark, Germany and Sweden.

12. List of references

Andersen, J.H., Harvey, T., Kallenbach, E., Murray, C., Al-Hamdani, Z., Stock, A. (2017). Under the Surface. Report L.NR. 7128-2017 DK6 by NIVA Denmark.

Birk, S., Willby, N.J. , Kelly, M.G., Bonne, W., Borja, A., Poikane, S. , van de Bund, W. (2013). Intercalibrating Classifications of Ecological Status: Europe's Quest for Common Management Objectives for Aquatic Ecosystems. Sci Total Environ 454-455, 490-499.

Borja, A., Prins, T.C., Simboura, N., Andersen, J.H., Berg, T., Marques, J.C., Neto, J.M., Papadopoulou, N., Reker, J., Teixeira, H. and Uusitalo, L., 2014. Tales from a thousand and one ways to integrate marine ecosystem components when assessing the environmental status. Frontiers in Marine Science, 1, p.72.

Canal-Vergés P, Petersen JK, Rasmussen EK, Erichsen AC, Flindt MR (2016) Validating GIS tool to assess eelgrass potential recovery in the Limfjorden (Denmark). Ecological Modelling 338:135-148. http://dx.doi.org/10.1016/j.ecolmodel.2016.04.023.

Christensen et al. (subm.) *Referred to in the Scientific Documentation Report, but not found back*

Cloern, J.E. (2001). Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series, 210, 223-253.

Dahl K (red.), Andersen JH (red.), Riemann B (red.), Carstensen J, Christiansen T, Krause-Jensen D, Josefson AB, Larsen MM, Petersen JK, Rasmussen MB, Stand J (2005) Redskaber til vurdering af miljø- og naturkvalitet i de danske farvande. Typeinddeling, udvalgte indikatorer og eksempler på klassifikation. Danmarks Miljøundersøgelser. 158 s. - Faglig rapport fra DMU nr. 535.

Droop, M.R. (1968). Vitamin B12 and marine ecology IV: the kinetics of uptake, growth and inhibition in Monochrysis lutheri. J. Mar Biol. Ass UK, 48, 689-733.

Eilola, K., Gustafsson, B. G., Kuznetsov, I., Meier, H. M., Neumann, T., & Savchuk, O. P. (2011). Evaluation of biogeochemical cycles in an ensemble of three state-of-the-art numerical models of the Baltic Sea. Journal of Marine Systems, 88, 267-284.

Flindt, M. R., Rasmussen, E. K., Valdemarsen, T., Erichsen, A., Kaas, H., & Canal-Vergés, P. (2016). Using a GIS-tool to evaluate potential eelgrass reestablishment in estuaries. Ecological Modelling, 338, 122-134. http://doi.org/10.1016/j.ecolmodel.2016.07.005

HELCOM. (2013). Summary report on the development of revised Maximum Allowable Inputs (MAI) and updated Country Allocated Reduction Targets (CART) of the Baltic Sea Action Plan. Background Document for the 2013 HELCOM Ministerial Meeting, 1-23.

Kuusemäe, K., Rasmussen, E. K., Canal-Vergés, P., & Flindt, M. R. (2016). Modelling stressors on the eelgrass recovery process in two Danish estuaries. Ecological Modelling, 333, 11-42. http://doi.org/10.1016/j.ecolmodel.2016.04.008

Malve, O., Qian, S. 2006. Estimating nutrients and chlorophyll a relationships in Finnish Lakes. Environmental Science & Technology, 40(24),pp. 7848-7853. DOI: 10.1021/es061359b.

Markager S, Storm L, Stedmon C (2006) Limfjordens miljøtilstand 1985 til 2003. Sammenhæng mellem næringsstoftilførsler, klima og hydrografi belyst ved hjælp af emperiske modeller. Danmarks Miljøundersøgelser. 219 s. - Faglig rapport fra DMU, nr. 577.

Orth RJ, Moore KA, Marion SR, Wilcox DJ, Parrish DB (2012) Seed addition facilitates eelgrass recovery in a coastal bay system. Marine Ecology Progress Series 448:177-195.

Rasmussen, B., & Josefson, A. B. (2002). Consistent Estimates for the Residence Time of Micro-tidal Estuaries. *Estuarine, Coastal and Shelf Science*, *54*(1), 65-73. http://doi.org/10.1006/ecss.2001.0836

Riemann, B., Carstensen, J., Dahl, K., Fossing, H., Hansen, J. W., Jakobsen, H. H., et al (2016). Recovery of Danish Coastal Ecosystems After Reductions in Nutrient Loading: A Holistic Ecosystem Approach. *Estuaries and Coasts*, *39*(1), 82-97. http://doi.org/10.1007/s12237-015-9980-0

Scavia D, Rabalais NN, Turner RE, Justić D, Wiseman WJ (2003) Predicting the response of Gulf of Mexico hypoxia to variations in Mississippi River nitrogen load. Limnology and Oceanography 48:951-956. doi:10.4319/lo.2003.48.3.0951

Schernewski, G., Friedland, R., Carstens, M., Hirt, U., Leujak, W., Nausch, G., Neumann, T., Petenati, T., Sagert, S., Wasmund, N., von Weber, M. (2015). Implementation of European marine policy: New water quality targets for German Baltic waters. *Marine Policy*, 51, 305-321. https://doi.org/10.1016/j.marpol.2014.09.002.

Timmermann, K., Dinesen, G. E., Markager, S., Ravn-Jonsen, L., Bassompierre, M., Roth, E., & Støttrup, J. G. (2014). Development and Use of a Bioeconomic Model for Management of Mussel Fisheries under Different Nutrient Regimes in the Temperate Estuary of the Limfjord, Denmark. *Ecology and Society*, *19*(1), art 14-11. http://doi.org/10.5751/ES-06041-190114

van der Heide T, van Nes E., van Katwijk MM, Olff H, Smolders AJ (2011) Positive Feedbacks in Seagrass Ecosystems – Evidence from Large-Scale Empirical Data. PLoS ONE 6(1): e16504.

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Rapporten fra

"International evaluation of the Danish marine models"

Baggrunden

Den danske miljømålslov (2003) skulle især implementere EU's vandrammedirektiv (2000) og habitatdirektiv (1992), men også fuglebeskyttelsesdirektivet og skaldyrvandedirektivet (begge 1979). Loven foreskriver udarbejdelse af regionale vandplaner.

I forbindelse med Folketingets vedtagelse af "*Fødevare- og Landbrugspakken*" (2015-16) nedsattes et internationalt "panel" af fagligt kompetente forskere fra Sverige, Norge, Finland, Tyskland og Holland (Nederlandene).

Panelet skulle foretage en videnskabelig "*evaluering af kvælstofmodeller bag vandområdeplanerne*". Resultatet skal bruges ved udvikling af grundlaget for krav i de nye vandplaner for 2021-2027 ("*3. generation"*).

Panelets rapport (10. oktober 2017) baseres på den danske RBMP[1]-rapport, *"Development of models and methods to support the establishment of Danish River Basin Management Plans, Scientific documentation"* (Udvikling af modeller og metoder til understøttelse for fastlæggelse af danske vandplaner, Videnskabelig dokumentation), maj 2017. Forfatterne er fra Århus Universitets "DCE" (Nationalt Center for Energi og Miljø, tidligere Danmarks Miljøundersøgelser) og DHI (Institut for Vand og Miljø, et GTS-institut), som her sammenskriver flere tidligere rapporter.

I Panelets indledende spørgsmål til DCE/ DHI om deres rapport indgår bl.a.: *"Både panelet og interessenterne savner motivering for det fundamentale valg af at fokusere alene på reduktion af N-udledninger fra det åbne land som middel til at forbedre vandkvaliteten. Situationen er kompleks, da der er rigelige beviser for, at algevækst i mange systemer begrænses af et samspil mellem N* (kvælstof) *og P* (fosfor), *med visse sæson-variationer. Desuden kan N-fiksering i Østersøen forværre dette problem og ophæve virkningen af N-reduktions-foranstaltningerne, hvor der er overskud af P."*

(Both the panel and the stakeholders miss a justification of the fundamental choice to focus exclusively on reduction of (diffuse) N sources as the main means to improve water quality. The situation is complex, as there is ample evidence that in many systems there is co-limitation of phytoplankton growth by N and P, with some seasonal pattern in most systems. In addition, N fixation in the Baltic may aggravate the problem and undo N reduction measures where ample P is available.)

Inden færdiggørelsen af Panelets rapport 10. oktober 2017 blev den forelagt bl.a. landbrugets organisationer, DCE og DHI samt diverse "NGO'er" i høring 19. september - 2. oktober 2017, for bemærkninger og spørgsmål.

Den endelige rapport er stort set uændret i forhold til hørings-udgaven.

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[1]) River Basin Management Plan

1. **Indledningen**.

Panelet skriver, at danske miljøforskere bør justere deres modeller, og derpå publicere deres arbejde i internationalt anerkendte videnskabelige tidsskrifter, for at hævde den danske førerposition på området.

Det anføres, at forskningen i årtier har været gennemført i nært samarbejde med administrationen (Miljøministeriet og dets styrelser). For at undgå forvirring og misforståelser anbefales det, at ministeriet udarbejder klare kommissorier og valg af indikatorer.

Panelet udtrykker derpå håbet om, at interessenternes og forskernes fremsatte synspunkter vil kunne opbygge tillid mellem dem og bidrage til et godt resultat.

2. **Om overensstemmelse med vandrammedirektivet**

Det nævnes, at direktivets typer for overfladevand omfatter åer, søer, overgangsvande og kystvande, men at ”kunstige eller stærkt modificerede vandområder” også er en mulighed.

Det påpeges, at Danmark har valgt at slå overgangsvande sammen med kystvande til én type, så et fiske-”BQE” (biologisk kvalitets-element) ikke er nødvendigt.

Vandrammedirektivet foreskriver national fastsættelse af type-specifikke reference-tilstande for typer af overflade-vandområder *(surface water body types)*. Reference-tilstandene kan enten være geografisk *(spatially)* baserede, bygge på modellering eller en kombination heraf.

Danmark har valgt modellering og en basislinie ved år 1900, da der ikke findes uberørte områder. Panelet godkender dette, da det er bedre end alene et ekspert-skøn *(expert judgement)*.

I den danske RBMP-rapport valgtes indikatorerne klorofyl a (et mål for bladgrønt), ”Kd” (et mål for vandets gennemsigtighed) og et indeks for bund-organismer. Desuden nogle sekundære indikatorer til brug for den statistiske modellering (se afsnit 4).

Panelet kritiserer, at de danske modeller lægger alt for stor vægt på Kd, som kun er et indirekte mål for bundplanter og alger, og derfor ikke et direkte mål for udbredelsen af ålegræs.[2]

Ydermere er Kd ikke uafhængigt af klorofyl a, og er ikke som dette, og dybdegrænsen for ålegræs, ”interkalibreret” (sat i forhold til svenske og tyske farvande).

Panelet kritiserer, at de fleste modelberegninger for N-målsætning kun baseres på klorofyl a og Kd, hvor Kd er et utilstrækkeligt mål for ålegræs.

Panelet anfører, at eutrofierings[3]-problemer skal betragtes som et samspil mellem næringsstoffer og mange andre stress-faktorer: Kemisk forurening, invasive arter, habitat-ændringer, fiskeri m.v., og at disse ikke bør betragtes som additive faktorer, men netop i et samspil.

3. **Typologi for kystvande**

Panelet kritiserer, at de mange danske fjorde skæres over én kam *(are represented by only one target value)*, selv om der er store individuelle forskelle: ”*Konsekvensen er en stor og ikke*

[2]) Ålegræs er ikke en art af tang (makroalger) men en blomsterplante, der periodevis begrænses af naturlige sygdomme.

[3]) overgødsknings-

videnskabeligt underbygget variation i de krævede udledningsreduktioner for de enkelte vandområder."

Der udtales rosende ord om det omfattende danske moniteringsprogram (over 90 målestationer i de i alt 119 vandområder). Dog med det forbehold, at flere store østjyske fjorde kun har én målestation, selv om de er opdelt i flere vandområder.

Til sammenligning anføres, at Tyskland har udarbejdet 35 specifikke klorofyl a-referencer og målværdier alene for sine baltiske vandområder.

Panelet skriver, at moniteringsprogrammet muliggør individuelle klorofyl a-referencer og -målværdier for de enkelte vandområder. Det anbefales kraftigt, at dette gennemføres, da *"Danmark er et af de få europæiske lande, hvor der foreligger tilstrækkelige data, ekspertise og modeller for en så omfattende fremgangsmåde."*

4. Brugen af ålegræs og Kd som miljø-indikatorer

Det påtales, at ålegræs indtager for stor en rolle som indikator, selv om der i f.eks. Odense Fjord er store forekomster af andre undersøiske blomsterplanter, som ifølge vandrammedirektivet også skal tages i betragtning.

Panelet understreger endvidere, at der ikke er lineær sammenhæng mellem lysintensitet og Kd, hvorfor det lys, der når bunden, kan afvige signifikant (statistisk sikkert) fra, hvad der er beregnet via brug af Kd.

Derfor finder Panelet det usandsynligt, at brug af Kd som eneste indikator kan dække alle betingelserne for genopretning af ålegræsset.

Panelet finder kun ringe sammenhæng mellem Kd og N-belastning i årets løb, og ingen reel ændring *(no material changes)* set over flere år, trods ændret N-belastning. Dette understøttes af *DHI's mekanistiske model*[4], som ikke kan reproducere de (ca. år 1900) observerede Kd-reference-værdier ved at modellere reference-belastninger for år 1900.

Heller ikke *DCE's statistiske modeller*[5] eller analyser for perioden 1990 - 2013 synes i stand til at vise, at Kd skulle have en stærk afhængighed af næringsstofudledninger.

Selv om DCE's model synes at vise en sammenhæng mellem de gennemsnitlige værdier for klorofyl a og Kd, ser Panelet ingen fælles udvikling af disse to indikatorer i perioden 1990 - 2012.

Panelet konkluderer, at både klorofyl a og Kd er udtryk for eutrofierings-effekter, men at klorofyl a er en mere pålidelig indikator end Kd.

DHI's model vurderer, hvor meget af afstanden mellem mål og tilstand, der kan nås ved N-reduktion, og korrigerer for dette ved beregningen af reduktionskrav. Panelet finder dette passende, og at det ikke fører til uforsvarlig overvurdering af nødvendige krav.

Det fremhæves, at DCE's model for nogle vandområder medfører krav om langt over 100 % reduktion af N-udledningen for at bringe Kd ned under målværdien. Endvidere, at dette søges løst ved at "oversætte" dette til mere realistiske krav, f.eks. 75 % reduktion, hvor modellen kræver over

[4]) i det følgende omtalt som DHI's model eller DHI-modellen.
[5]) i det følgende omtalt som DCE's model eller DCE-modellen.

200 % reduktion. De danske forskere argumenterer med, at dette er et ekspert-skøn *(expert judgement)*.

Panelet skriver: *"Trods spørgsmål til forskerne har panelet ikke været i stand til at finde* (discover) *logikken bag denne oversættelse."* Panelet finder, at dette indfører et unødvendigt element af vilkårlighed, der står i kontrast til generelt erfaringsbaserede løsninger, og derfor udsætter hele proceduren for uproduktiv kritik.

Her henviser Panelet i øvrigt til afsnit 7, hvor det påvises, at målværdierne heller ikke med DHI-modellen i alle tilfælde kan opnås ved reduktion af landbaserede N-udledninger (landbruget). Derfor foreslår panelet, at de to modeller "harmoniseres" ved at inkorporere DHI-modellens metoder i DCE-modellen.

Panelet anbefaler, at man i det videre arbejde genovervejer det hele og starter med den basale observation, at det reelle kriterium ikke er Kd, men derimod overlevelse og genopretning af vand-blomsterplanterne (og ikke alene ålegræs).

Pga. Kd's utilstrækkelighed og dets høje korrelation med klorofyl a foreslår Panelet at "nedveje" Kd i de afsluttende beregninger af reduktionskrav: Der bør bruges et vejet gennemsnit af indikatorerne Kd og klorofyl a, fremfor de to hver for sig.

Panelet er overrasket over, at DCE-modellen omfatter indikatorer, der ikke indgår i DHI-modellen: 1) Iltsvind, 2) økologiske tegn på iltsvind pga. næringsstoffer, 3) klorofyl samt 4) antal dage, hvor N begrænser væksten af planteplankton. Dette mindsker sammenligneligheden og troværdigheden af de følgende gennemsnitsbetragtninger, f.eks. ved meta-modellering[6].

Desuden påpeges, at meta-modellering baseret på DCE-modellen sommetider omfatter disse ekstra indikatorer, og sommetider ikke.

Panelet citerer DCE's konklusioner om iltsvind i RBMP-rapporten: *"Der er direkte bevis på sammenhæng mellem næringsstofbelastninger og iltkoncentrationer i bundvandet (Markager et al. 2006) og størrelsen af iltsvindsområder (Scavia et al. 2003; Christensen et al., submitted). Imidlertid kompliceres disse sammenhænge af en betydelig tidsmæssig forsinkelse og høj følsomhed for klimavariationer såsom vandtemperatur og vindpåvirkning."*

("There is direct evidence for a relationship between nutrient loadings and oxygen concentrations in bottom water (Markager et al. 2006) and the size of hypoxic/anoxic areas (Scavia et al. 2003; Christensen et al, submitted). However, these relationships are complicated by a considerable time lag and a high sensitivity to climate variables like water temperature and wind stress.")

Panelet betvivler brugbarheden af DCE's ekstra indikatorer. Iltsvinds stærke afhængighed af skiftende vejrforhold giver betydelige variationer, der slører effekten af næringsstof-reduktioner. Udover disse problemer påpeges, at det har været nødvendigt at bruge en temmelig vilkårlig opslagstabel for at skønne hvilke næringsstof-reduktioner, der behøves for forbedring af iltsvinds-indikatorerne.[7]

("There is essentially only one (on/off) observation per year. In addition to these difficulties, a rather arbitrary look-up table approach has to be used in order to estimate the required nutrient reduction for improvement in the hypoxia indicators.")

[6]) Modellering ud fra en anden models resultater.
[7]) Tabel 8.7 i RBMP-rapporten.

Sammenfattende finder Panelet, at de ekstra indikatorer i DCE's model ikke bidrager væsentligt til formålet. Panelet anbefaler derfor, at man i stedet bruger DHI-modellen for bedre at kunne studere, hvordan iltsvind kan forbindes direkte med krav om reduceret udledning af næringsstoffer, før det bruges i praksis.

5. Kvælstof kontra fosfor

Dette afsnit behandler spørgsmålet om, hvorvidt forvaltningens krav har været unødigt fokuseret alene på reduktion af N-udledningerne.

Siden 1980'erne har byernes rensningsanlæg drastisk reduceret P-udledningerne, hvilket forbedrede vandkvaliteten betydeligt. Derpå koncentreredes opmærksomheden om landbrugets N-udledninger, da man mente, at N var den begrænsende faktor for væksten af planteplankton i de kystnære farvande.

Imidlertid er der i de senere år en stigende opmærksomhed på mere komplicerede samspil og dynamikker, såsom N-fiksering og P-frigørelse fra havbunden. Flere studier viser, at P tit er den begrænsende faktor om foråret, med variationer fra sted til sted. Derfor finder Panelet, at reduktion af P-udledningerne kunne være relevant i flere områder.

Basis for alle RBMP-rapportens beregninger er indikatorerne klorofyl a og Kd i sommerens løb. Dette har potentielle implikationer for, at man udelukkende har fokuseret på N-reduktioner.

Panelet mener, at indikatorer alene for sommerforholdene er for restriktivt et grundlag, bl.a. fordi der senere kan udløses P-frigivelse fra havbunden.

Endvidere påpeges, at den udvalgte periode for DCE-modellen er 1990-2013, dvs. efter 1980'ernes store udvikling af byernes rensningsanlæg, som gav store reduktioner af P-udledningen. Derfor kan DCE-modellen have maskeret P's rolle.

DCE-modellen giver en skævhed i retning af N: *"Overalt, hvor N vælges som den dominerende variabel, ses der bort fra en mulig P-afhængighed, fordi P ikke længere betragtes som en sekundær uafhængig variabel. Og hvis P vælges som dominerende variabel, bruges regressionsmodellen ikke. Således sker der ikke yderligere undersøgelse af den mulige indflydelse af reduceret P-belastning eller kombinationer af N- og P-reduktion."*

DHI-modellen inkluderer alle de relevante processer for både N og P samt kombinationer af begge. Men fokuseringen er hovedsagelig på N-scenarier, og de få scenarier, der også omfatter P-reduktion, er ikke detaljerede eller optimale for undersøgelse af P-reduktions virkning.

Det konkluderes, at resultaterne *(the evidence)* ikke er stærke nok til at udelukke P-reduktion eller kombinerede N/P-reduktioner som middel til at reducere års-gennemsnit for klorofyl og ilt-krav for sedimentet.

6. **Statistisk modellering** (DCE-modellen)

Modellens vigtigste resultater er kurverne (*slopes*, hældningskoefficienter) for sammenhængen mellem N-belastning og indikatorerne klorofyl a og Kd. Kurverne er kun beregnet, hvor N blev valgt som den vigtigste uafhængige variabel.

Panelet påpeger, at *"Der opgives ingen formel usikkerhedsanalyse af modellen som helhed, eller variansvurdering af de estimerede parametre (især ikke vedr. kurven for N-belastnings-indikatoren)."*

I modsætning til danske forskere sætter Panelet spørgsmålstegn ved *(In contrast to the researchers, however, the Panel questions)* den valgte metode, der blander to statistiske metoder. Dette antyder også *(leads to the suggestion)*, at N-belastning i nogle systemer overhovedet ikke var involveret som årsag til klorofyl a og Kd.

Det anføres, at i systemer, hvor N valgtes som vigtigste bestemmende faktor, kan mulige sekundære effekter af P ikke vises, og er slettet. Imidlertid er den vigtigste konsekvens, at dette kan føre til skævheder i *(biased)* beregning af kurver og "MAI" (*maximum allowable input*, nedenfor forstået som "max. tilladelige udledning").

Panelet finder ingen begrundelse for beregning af indikatorers korttids-afhængighed (år til år) af næringsstoffer, da der forekommer betydelige variationer i tilløb af ferskvand. Derfor skal der fokuseres mere på langtids-kurver og samspil mellem systemer.

Afslutningsvis godkender Panelet dog den fortsatte brug af både DCE's statistiske og DHI's mekanistiske modellering, fordi de exceptionelt mange indsamlede danske data muliggør evidens-baseret kontrol af sidstnævnte. Imidlertid anbefales, at de to modeller holdes mere adskilt og uafhængige.

Der anbefales en formel usikkerhedsanalyse af DCE-modellen, og Panelet gentager mistanken om skævheder i de beregnede kurver og referenceværdier pga. valget af variable.

7. Mekanistisk modellering (DHI-modellen)

Panelet finder, at modellen (modellerne) er ganske omfattende, inkluderer alle de relevante processer samt adskiller forskellene i klorofyl a, Kd og næringsstoffer mellem de enkelte vandtyper.

(Der følger en del komplicerede teoretisk/statistiske betragtninger).

Modellen synes at overvurdere uorganisk P (fosfat) om sommeren i nogle områder, men generelt er der god overensstemmelse mellem modelleret og observeret planktonvækst *(primary production)*.

Panelet konkluderer, at DHI-modellen er klart *"state-of-the-art"*, dvs. det nyeste og bedste på området, og resultaterne af en høj standard. Klorofyl a, Kd og næringsstoffer modelleres nøjagtigt på tværs af vandområde-typer, og beregninger af hydrodynamikken synes fremragende.

Svagest er næringsstof-beregningerne i "IDW"-modellen (Østersøen op til Skagerrak), hvor fosfat synes overvurderet sommer eller tidligt efterår, og N overvurderet om vinteren.

N-reduktions-scenarierne er relevante og passende opstillet, men P-reduktions-scenariet er ikke tilstrækkeligt beskrevet til, at det kan vurderes, om det kan danne basis for udelukkende at fokusere på N.

Afslutningsvis nævnes, at det vil være yderst værdifuldt at udstrække DHI-modellerne til at omfatte så mange vandområder som muligt.

8. Beregningsmåder for estimering af MAI fra model-resultater

Panelet skriver, at *"det er ikke let at følge og afveje de forskellige trin i beregningen af maximum allowable input, MAI* (max. tilladelige udledning) *for vandområderne."*

(Der følger en del komplicerede teoretisk/statistiske betragtninger).

Som et resultat af gennemsnitsberegninger kan der opstå N-reduktionskrav på over 100 %, og det er uklart, hvordan dette problem løses.

Der påpeges også problemer med forklaringen/forståelsen af andre trin i anvendelsen af gennemsnits-beregninger som basis for reduktionskrav.

Mest problematisk finder Panelet gennemsnitsberegninger af klorofyl a -referenceværdier på tværs af modeltyper og inden for vandområdetyper. Dette kan give for stor fokusering på ét system og for lille fokusering på et andet, og føre til både økonomisk og økologisk tab af effektivitet.

Panelet finder gennemsnits-procedurerne så usikre, at de bør udsættes til allersidste trin, så de to modellers beregninger er uafhængige, indtil den endelige sammenligning og valget af strategi.

Panelet foreslår at udelade de (i afsnit 4 omtalte) ekstra indikatorer af DCE-modellen, da de er stærkt korrelerede med klorofyl a. Derved kan de to modelberegninger lettere sammenlignes.

En anden potentiel uoverensstemmelse mellem de to modeller er, at kun DHI-modellen explicit viser, hvor meget af forskellen mellem tilstand og målsætning der kan nås alene ved at reducere danske N-udledninger fra land.

Panelet mener, at meta-modelleringen (se afsnit 4) for Nordsø-vandområder er mindre pålidelig end for de øvrige vandområder, da den baseres på dristige ekstrapolationer. Der anbefales flere studier af Nordsø-vandområderne for at forbedre de estimerede reduktionskrav.

9. Evaluering af MAI-resultaterne

MAI er definitionen på (af) den årlige næringsstofbelastning (her N), der er acceptabel for at kunne holde et kystvandområde i "god økologisk status" i henhold til vandrammedirektivet, eller tillade tilbagevenden dertil.

For at opnå god økologisk status, især for Fyn og Jylland, kræver den danske RBMP-rapport langt større reduktioner end de 3 %, der er stipuleret i den seneste HELCOM-aktionsplan for Østersøen. Den danske rapport kræver for hele landet en reduktion på 29-34 %, afhængigt af den valgte model.

RBMP-rapporten bruger år 1900 som reference, og Panelet finder dette velunderbygget. Imidlertid har arealernes anvendelse og befolkningens størrelse og bosætning undergået store forandringer siden da, og byerne udledte allerede dengang betydelige mængder urenset spildevand. Fastlæggelse af regionale MAI'er ud fra historiske forhold er derfor vanskelig, især for områder med store ændringer.

Som nævnt i afsnit 3 finder Panelet ikke, at den valgte danske typologi tilstrækkeligt afspejler de individuelle forhold i de mange danske fjorde og kystvande. Målsætningen for især indikatoren klorofyl a varierer derfor ikke tilstrækkeligt, og konsekvensen er mindre pålidelige vandområde-specifikke MAI'er. Dette kan give for store reduktionskrav i nogle vandområder, og for små i andre.

Som nævnt i afsnit 4 er sammenhængen mellem Kd i kystvande og ekstern næringsstofbelastning sommetider meget svag. Desuden har dette forskellige konsekvenser for - og behandles forskelligt i - DCE- og DHI-modellerne. Kd brugt i DCE-modellen giver derfor sommetider umulige N-reduktionskrav på over 100 %. Yderligere reagerer Kd kun langsomt på N-reduktioner, data varierer meget, og Kd viser korrelation med klorofyl a.

Alt i alt betragter Panelet Kd som en mindre egnet indikator i mange danske kystvandområder. Stor vægt på Kd i MAI-beregning vil give usikre vandområde-specifikke MAI'er, og bør undgås. I RBMP-rapporten bruges sommetider ekstra indikatorer i DCE-modellen, men Panelet ser ingen større fordel herved m.h.t. MAI; de ekstra indikatorer tilfører ikke væsentlig ny information, og de viser ikke korrelation med de eksisterende indikatorer.

Generelt finder Panelet, at DHI-modellen har et godt potentiale for beregning af vandområde-specifikke MAI'er, men den dækker endnu ikke alle vandområder.

DCE-modellen er baseret på reelle måledata, og kan for de fleste kystvandområder være værdifuld til vurdering af både langtidstendenser og DHI-modellens resultater.

(The statistical modelling is based on real monitoring data, and in most coastal water bodies it can serve as a valuable tool to assess long-term trends as well as the mechanistic model performance.)

Beregningen af vandområde-specifikke MAI'er er kompleks og ikke helt overbevisende. Mest problematisk er gennemsnitsberegning af klorofyl a -referenceværdier på tværs af begge modeller og indenfor kystvandstyper. Dette har også negative konsekvenser for meta-modellerede vandområder.

Panelet anfører, at det har læst og er enig i mange af interessenternes udtrykte bekymringer vedr. MAI'erne.

Beregning af vandområde-specifikke MAI'er har dog én stor fordel: Det tillader udvikling af vandområde-specifikke valgmuligheder for forvaltningen. Derfor bør kystvands- og hav-modeller kombineres med vandområde-modeller. Hvis sidstnævnte kan give månedlige data for næringsstofbelastningen, kan der udvikles scenarier, som tager "højde" for de sæsonmæssige udledninger, m.h.p. kost-effektiv forvaltning.

Sammenfattende finder Panelet ikke, at vandområde-specifikke MAI'er er tilstrækkeligt pålidelige som basis for beslutning og planlægning af foranstaltninger til reduktion af næringsstofbelastningen. Endvidere tager MAI kun sigte på N-reduktioner, og ser bort fra muligheden af at forvalte vandområder via reduktion af P-belastningen.

Imidlertid finder Panelet, at Danmark råder over modeller, kompetencer og data til at møde udfordringen om at beregne vandområde-specifikke MAI'er. Selv en modificeret behandling af eksisterende model-resultater kan måske føre til meget mere pålidelige MAI'er.

10. Samlende vurdering og konklusioner

I sammenligning med mange andre europæiske lande har Danmark udmærkede databaser, modeller og videnskabelig ekspertise.

Panelet har analyseret konsekvenserne af at bruge en relativt grov typeinddeling af de danske kystvande for beregning af referencetilstande, målsætninger og MAI.

Panelet konkluderer, at den grove typeinddeling har ført til reduktionskrav, der ikke er optimale for de enkelte vandområder. Det anbefales, at der i hele den videnskabelige proces fokuseres på de enkelte vandområder, og at regional gruppering først sker ved udarbejdelsen af de forvaltningsmæssige krav.

Klorofyl a er nyttig som indikator for planteplankton, mens Kd ikke er optimal som indikator for bundplanter og -tang. De andre indikatorer, der kun bruges i DCE-modellen, giver metodiske problemer, og er ikke modne til inddragelse i forvaltningsplanerne. Panelet ser dog en lovende udvikling i modelleringen for bundvegetationen, og har fremsat anbefalinger i denne henseende.

Panelet godkender RBMP-rapportens vægt på reduktion af N-udledninger fra det åbne land, men vurderer principielt også, at P-reduktion og sæsonmæssig N-reduktion kunne spille en rolle.

(In view of the large efforts in the past to remove P load from point sources, the Panel endorses the emphasis placed in the Scientific Documentation Report on reducing N loads from diffuse sources. However, at least in principle, there could be an additional role for P load reduction and for seasonal regulation of the N load.)

Panelet finder, at disse muligheder fortjener yderligere videnskabelig undersøgelse, især i områder, hvor der kræves store N-reduktioner.

Skønt opretholdelse af to parallelle modeller (DCE's og DHI's) kan synes overflødig, tilslutter Panelet sig dette, da mængden af data giver unikke muligheder for erfaringsbaseret check af DHI-modellens resultater.

Panelet vurderer DHI-modellen som *"state-of-the-art"* (det nyeste og bedste på området), men understreger, at uafhængige kontroldata samt usikkerhedsanalyser fortsat er nødvendige, og kan fås fra DCE-modellen. Forbedring af DCE-modellen kan optimere denne sammenhæng.

Panelet godkender den generelle logik bag udledning fra modellerne af reference- og målværdier samt beregning af N-reduktionskrav, men har fundet adskillige punkter, hvor der er brugt gennemsnitsberegninger. Dette fører bl.a. til indbyrdes afhængighed mellem modellerne, komplicerer proceduren og gør den vanskeligt forståelig. Intet af dette var nødvendigt, da databasen og model-resultaterne tillader en fuldt gennemskuelig udledning af de vandområde-specifikke reduktionskrav.

Sammenfattende har Panelet en positiv vurdering af, at reduktionskravene er baseret på **solid** videnskabelig erfaring og modellering på et generelt højt niveau.

Panelet ser meget positivt på, at der næsten ikke indgår ekspert-meninger *(near lack of expert judgment)*. Panelet mener, at i de få tilfælde, hvor sådant forekommer, er det unødvendigt, og kan fjernes.

Det generelle niveau (landsgennemsnit) af reduktionskravene svarer godt til uafhængige tiltag i lignende områder, og synes at være et **robust** mål for nødvendige tiltag.

På den anden side vurderer Panelet, at den arealmæssige (spatial) fordeling af de krævede foranstaltninger er **unødvendigt grov**. Panelet er overbevist om, at de rige databaser, kombineret med en **forbedret statistisk fremgangsmåde** (DCE) og den højopløsnings-mekanistiske modellering (DHI), kan føre til forbedrede vandområde-specifikke MAI'er.

Med den nuværende videnskabelige indsigt godkendes det synspunkt, at de foreslåede samlede reduktioner er **nødvendige**, men det kan ikke garanteres, at de vil være **tilstrækkelige**. Især for bundvegetationen kan yderligere tiltag blive nødvendige.

11. Anbefalinger for det videre forløb

Monitering:

Det danske moniteringsprogram med 90 målestationer ved kysterne og i havet er meget omfattende, og generelt godt tilpasset vandrammedirektivets krav. Det danner basis for yderligere udvikling af modellerne og de fleste beregninger, og er nødvendigt for vurdering af midlernes effekt, og om direktivets mål er nået.

Panelet anbefaler opretholdelse af programmet på fuld styrke samt vurdering af, om en vandområde-specifik forvaltning kræver flere målestationer.

Typologi:

Der er svagheder i afspejlingen af de enkelte fjordes egenskaber. Derfor anbefaler Panelet, at der beregnes referencetilstande for hvert enkelt af de 119 vandområder. Danmark er et af de få lande, hvor nødvendige data, ekspertise og modeller muliggør en så omfattende fremgangsmåde.

Derved kan målsætninger og vandområde-specifikke MAI’er optimeres og minimere spild af ressourcer. Tillige kan en ”robust typologi” danne basis for ”interkalibrering” (sammenligning med andre EU-lande).

Valg af indikatorer:

Klorofyl a er en generelt accepteret indikator, mens Kd som mål for bundvegetationen har visse begrænsninger. Panelet anbefaler at bygge på nyere ålegræs-modelleringer for udledning af bedre indikatorer, og kun i mellemtiden lade Kd være en slags stedfortræder *(proxy)*.

Panelet anbefaler at udelade DCE-modellens ekstra indikatorer (iltsvind, økologiske tegn på iltsvind pga. næringsstoffer, klorofyl samt antal dage, hvor N begrænser væksten af planteplankton). De angår vigtige økologiske spørgsmål, men er ikke ”modne”, da de mangler ”kvantitativ relation” til næringsstof-belastningen. I stedet anbefales udvikling af målrettet modellering, der kan inkorporeres i indikator-systemet.

Statistisk modellering:

Panelet ser stor værdi i opretholdelsen af to uafhængige modeller, men anbefaler, at DCE-modellen reorienteres hen imod optimal vurdering af indikatorernes langtidsafhængighed af næringsstoffer i en analyse på tværs af systemer. Det anbefales at betragte både N- og P-belastning som ”forklarings-variable.”

Panelet anbefaler, at der arbejdes videre med usikkerheds-analyser af DCE-modellen, og at dette lettes ved en avanceret kryds-modellering af de to systemer.

Mekanistiske modeller:

DHI-modellen er *”state-of-the-art”* (det nyeste og bedste på området), både m.h.t. talbehandling og de brugte modeller, som kraftfulde værktøjer for en sund videnskabelig basis for vandrammedirektivets implementering.

Men modellerne dækker ikke alle vandområder, hvorfor der er brugt forskellige metoder til definition af referencetilstande, mål og MAI i forskellige vandområder.

Panelet anbefaler udstrækning af DHI-modellen til så mange vandområder som muligt for at sikre, at ensartede metoder i fremtiden kan bruges for definition af vandområde-specifikke MAI'er.

Metoder for udledning af mål og MAI fra modellerne:

Panelet anbefaler forenkling af beregningsmetoderne ved at fjerne gennemsnitsberegninger mellem modeller, indikatorer, vandområder og -typer, samt mellem regionale vandområder. Derved kan forskelle og ligheder mellem disse klargøres og analyseres.

Kryds-tjek af resultaterne fra de to modeller kan danne basis for ekstrapolering til alle systemer.

Panelet anbefaler, at der udledes ét MAI pr. vandområde, og at der først i senere faser træffes beslutning om regionale gennemsnit eller grupperinger, når de videnskabelige resultater omsættes til forvaltnings-foranstaltninger.

Interaktioner mellem vandområder:

Vandområde-modeller muliggør beregning af mulige N-og P-reduktioner for hvert enkelt vandområde, udvikling af vandområde-specifikke N- og P- reduktions-scenarier samt vurdering af omkostningerne.

Ydermere tillader vandområde-modellerne fokusering på sæsonvariationer i næringsstofbelastningen og begrænsnings-mønstre *(seasonal load and limitation patterns.)*

Panelet anbefaler en kombination af vandområde- og kystvande-modeller for udvikling af en vandområde-specifik optimeret forvaltning, der omfatter både N og P.

International indfaldsvinkel:

Vandrammedirektivets tekniske implementerings-vejledning foreskriver ensartede fremgangsmåder i alle medlemslande, hvorfor krav, modellering og udfordringer også ligner hinanden.

Vandrammedirektivet kræver, at målene "interkalibreres" og "harmoniseres" med nabolande.

Derfor anbefaler Panelet en koordineret, fælles videnskabelig indsats især mellem Danmark, Tyskland og Sverige.

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Marine nitrogen: Phosphorus stoichiometry and the global N:P cycle

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Key words: freshwater, limitation, marine, nitrogen, phosphorus, ratio, stoichiometry, trace elements

Abstract. Nitrogen supply is often assumed to limit marine primary production. A global analysis of total nitrogen (N) to phosphorus (P) molar ratios shows that total N:P is low (<16:1) in some estuarine and coastal ecosystems, but up to 100:1 in open oceans. This implies that elements other than N may limit marine production, except in human impacted, estuarine or coastal ecosystems. This pattern may reconcile conflicting enrichment studies, because N addition frequently increases phytoplankton growth where total N:P is expected to be low, but P, Fe, or Si augment phytoplankton growth in waters where total N:P is high. Comparison of total N:P stoichiometry between marine and freshwaters yields a model of the form of the aquatic N:P cycle.

Introduction

Marine primary production yields >90 billion kg of food to the world economy each year (FAO 1993). Marine ecosystems store 50-times more inorganic carbon than the earth's atmosphere, so marine primary production may play an important role in establishing global climate (Mackenzie et al. 1993; Ritschard 1992). Factors regulating marine production are therefore of broad societal interest.

There is currently disagreement about the nutrient elements limiting marine primary production (Howarth 1988; Smith 1984). On one hand, some have concluded (e.g. Boynton et al. 1982) that because ratios of dissolved inorganic nitrogen (N) and phosphorus (P) are often lower than the average intracellular N:P ratio of marine organisms (Redfield 1934; Redfield et al. 1963), nitrogen is the element in shortest supply and therefore must limit marine primary production. On the other hand, geochemical budgets suggest that inorganic phosphorus should be in shortest supply (Meybeck 1982) in part because atmospheric N_2 can be fixed (Redfield 1958; Vitousek & Howarth 1991). Experimental additions of inorganic nutrients to seawater samples from various marine ecosystems suggest that several elements may play limiting roles, including N, P, Si and Fe (Boynton et al. 1982; Martin & Fitzwater 1988; Martin et al. 1994).

Although N and P are present in inorganic, organic and particulate forms, much of marine nutrient stoichiometry has been based only upon the relative amounts of dissolved inorganic N and P found in marine ecosystems. Frequently measured NO_2^-, NO_3^-, NH_4^+ and PO_4^{3-} may represent only a fraction of the N and P that can be used by the biotic community, however. Many aquatic organisms, from bacteria to fish, consume and cycle dissolved organic matter (e.g. Amon & Benner 1994; Bentzen et al. 1992; Glibert et al. 1991; Hoegh-Goldberg 1994), although dissolved organic N and P may be cycled at different rates (e.g. Smith et al. 1986). Particulate N and P can also be ingested and rapidly converted to dissolved inorganic and organic forms by zooplankton (e.g. Ikeda et al. 1982). Thus, many analyses of nutrient stoichiometry in marine ecosystems are based on analyses of fragments of the complete N and P pools.

Analyses of total N and P pools are used to distinguish N from P limitation in freshwaters. Phosphorus, usually measured as PO_4^{3-} was initially recognized as the principal limiting nutrient in freshwaters, but limitation by nitrogen has since been reported in many freshwater ecosystems (Elser et al. 1990). Dissolved N and P are cycled so rapidly by living organisms that they are absorbed as fast as they are produced and thus are often difficult to detect in the soluble inorganic state. Because both organic and inorganic N and P can be used by aquatic organisms, analyses of the total amounts of nutrient elements (i.e. the sum of all dissolved, particulate, organic and inorganic nutrients) yield better measurements of freshwater nutrient supply than analyses of soluble, inorganic N and P alone (Sakamoto 1966).

This approach has been successfully applied in the management of freshwater ecosystems. A recent study shows that comparison of the total N:P ratios of whole water samples with the average N and P needs of algal cells is useful in distinguishing freshwater ecosystems limited by N from those limited by P (Downing & McCauley 1992). Ecosystems with N:P molar ratios less than the average required cellular ratio of 16:1 (Hecky & Kilham 1988) are generally N-limited and those with ratios >16:1 are P-limited. Unfortunately, no analysis of marine data on total N and P stoichiometry has been performed due to the scarcity of total N and P data in marine environments. The purpose of this analysis is to summarize existing marine data to find how total inventories of N and P vary in the world's marine ecosystems.

Methods

Marine data on total N and total P are rare because marine studies seldom analyze all fractions of N and P. Some estimates of total N and P concentrations are scattered throughout the world literature, including data from a polluted European estuary, Chesapeake estuary, a North Carolina inlet, several harbors, the Pacific Ocean near Peru, California and New Zealand, the Gulf of Bothnia, the Baltic, the Gulf of Finland, the Strait of Gibraltar, the Indian Ocean, the Adriatic, and the Mediterranean. Several sources of total N and total P data are analyzed here: Ben-Taleb et al. (1987), Black et al. (1981), Coste et al. (1988), Dame et al. (1986), Degobbis & Gilmartin (1990), Jordan et al. (1991), Lahdes & Leppanen (1988) Le Rouzic & Bertru (1992), Perttila et al. (1980), Pietikainen et al. (1978), Pitkanen & Malin (1980), Ramadan et al. (1984), Rydberg & Sundberg (1986), Updegraff et al. (1977), Valderrama (1981), and Williams (1967). Data were restricted to averages for depth strata at sampling stations where analyses of total N and P or all components (inorganic, organic and particulate) were made using accepted standard methods. These methods usually consist of a digestion of samples for TP analysis followed by a colorimetric detection of PO_4^{3-}. Total nitrogen was analyzed by several methods, frequently determining concentrations of N fractions separately then summing, or by digestion to a soluble form, followed by inorganic N determination. Although some of these analyses may underestimate the abundance of dissolved, organic N (Suzuki et al. 1985), such underestimates, if present, would tend toward underestimation of TN:TP ratios (Karl et al. 1993). Standard TP and TN methods employed were those described in: Dal Pont et al. (1974), D'Elia et al. (1977), Grasshoff (1976), Grasshoff (1983), Menzel & Corwin (1965), Strickland & Parsons (1968), Strickland & Parsons (1972), Treguer & Le Corre (1975), and Valderrama (1981).

Results and discussion

We collected published data on total N and P inventories (dissolved inorganic and organic plus particulate forms) for 191 sample series taken at 88 marine sampling stations throughout the world. The total nitrogen concentration in marine ecosystems, ranging from estuaries to oligotrophic seas, only varied by about 30-fold (6–200 μM), whereas total phosphorus varied 650-fold (0.03–20 μM) (Figure 1). Therefore, total N:P ratios varied between about 5 and 310 (in moles), averaging N:P = 37, over all.

The majority of marine data analyzed here show N:P greater than the ratio required by planktonic marine algae (Figure 2). The N:P in living marine organisms varies greatly but averages about 16 (Hecky & Kilham 1988;

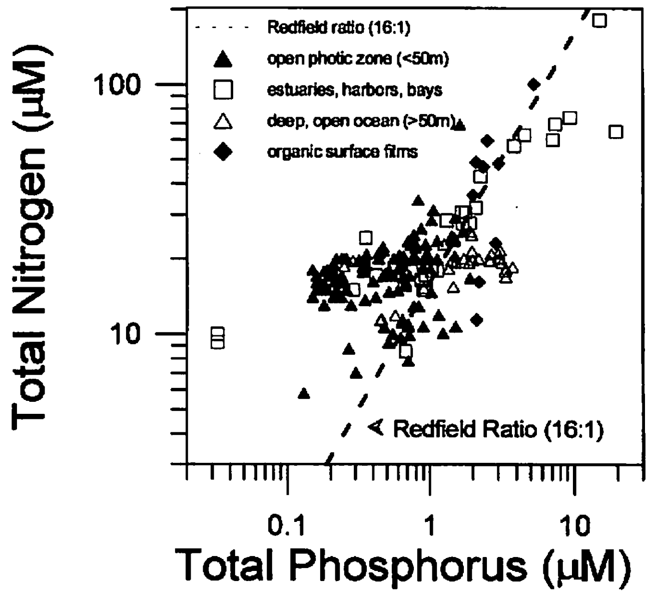

Figure 1. Relationship between total nitrogen and total phosphorus concentrations of marine ecosystems. Total N and P data were usually averages calculated from sets of samples taken at the same site on at least 3 different dates within the same year. Stations in the upper photic zone were defined conservatively as those <50 m in depth (Tett 1990) while nearshore sites were determined from information presented in published studies. The dashed line is the approximate average ratio of N and P in living marine plankton.

Meybeck 1982; *cf* Elser & Hassett 1994), therefore ambient N:P < 16 would indicate N limitation, while N:P > 16 would suggest P limitation of production. N:P is especially high in the upper photic zone where most of the ocean's planktonic production occurs and where nutrient limitation of production is most acute (Tett 1990). Only 12% of the open ocean photic zone sites had total N:P < 16:1. Only 3% had N:P indicating strong nitrogen deficiency (N:P < 10), 19% had N:P near to biological requirements (10 < N:P < 20), while 78% of open, photic zone sites had N:P indicating a deficiency of phosphorus or some element other than nitrogen (Figure 2). Both parametric and nonparametric statistical analyses show that the average total N:P ratio ($\overline{N{:}P}$ = 43) for open ocean photic zone sites (conservatively estimated as <50 m depth) was significantly ($p < 0.0001$) greater than 16:1. Oligotrophic sites have highest total N:P illustrated by the negative correlation between the logarithm of N:P and the logarithm of total P (Figure 3; $p < 0.00001$, $r^2 = 0.81$). The significant tendency for oligotrophic, open seas to have N:P greater than 16, indicates that P or some nutrient other than N limits production in open seas. Ratios of N:P are frequently >50:1 in the nutrient poor sites examined (Figure 3), often as great as 100:1. Total N:P > 16 occurred at 61

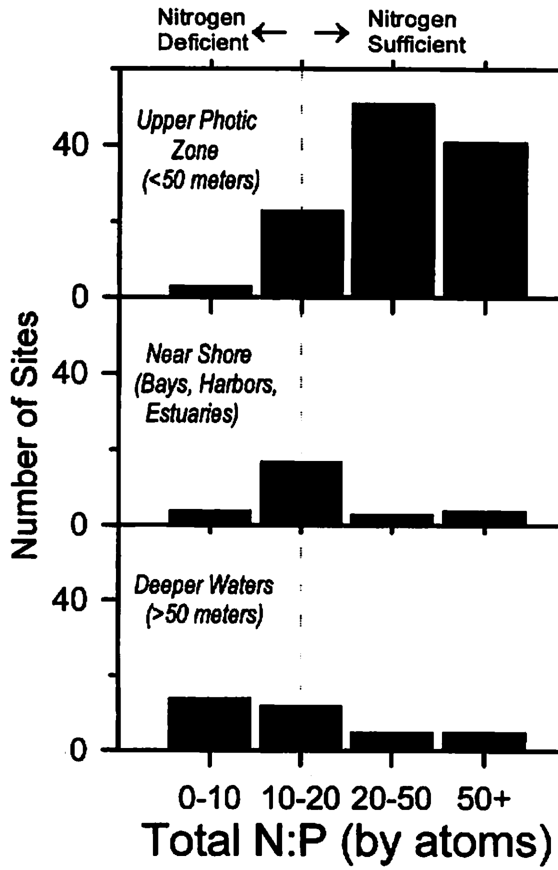

Figure 2. Nitrogen deficiency in marine waters relative to the usual N:P of living organisms. Shown are total N:P calculated from inventories of dissolved and particulate inorganic and organic N and P at several marine sampling stations. Environments include a broad variety of conditions and represent station-depth averages. Stations in the upper photic zone were defined conservatively as those <50 m in depth (Tett 1990) while nearshore sites were determined from information presented in published studies.

sites in 34 ecosystems, analyzed independently in 13 different studies, using several different standard methods. Provided that there is not a great divergence in the fraction of total N and P that are biologically labile, total N and P stoichiometry suggests that phosphorus or other factors limit production in the open oceans.

The data analyzed here suggest that N:P is very high in oligotrophic, open seas but often very low in estuaries and coastal ecosystems (Table 1). Estuarine, coastal, or enclosed parts of the sea had N:P ratios less than the required cellular ratio of 16:1 significantly more frequently (χ^2-test, p = 0.005) than samples taken from the surface waters of the open ocean (Figures

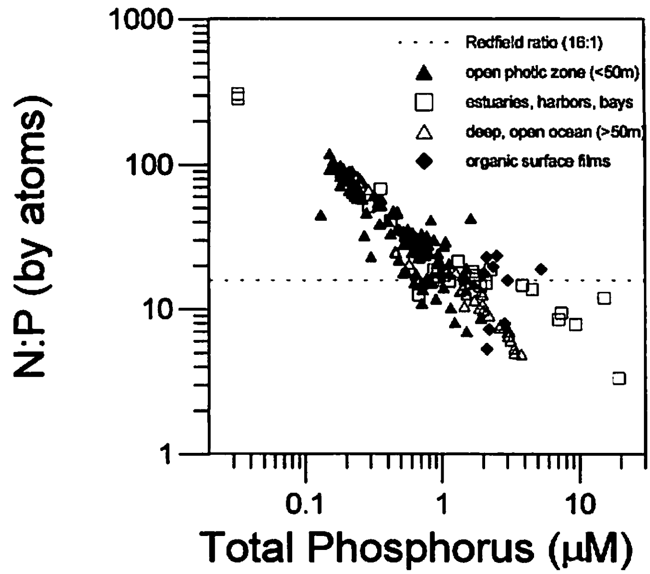

Figure 3. Total N:P ratios of data plotted in Figure 1 distinguishing types of marine environments. Deep and shallow sites were determined as in Figure 2. Data obtained from estuaries, harbors and bays were differentiated as such by the authors of studies. The dashed line is the approximate average ratio of N and P in living marine plankton.

Table 1. Frequency of total N:P molar ratios (Figure 3) that are greater than the approximate average ratio of N and P in living marine plankton (16:1). Ratios >16 imply that nitrogen is probably not the principal limiting nutrient.

Environment	Frequency of total N:P molar ratios: <16	≥16
Open oceans, <50m depth	14	104
Open oceans, surface films	3	6
Estuaries, harbors, bays	11	17
Deep, open oceans, >50m depth	23	13
Total	51	140

2–3; Table 1). Nitrogen is therefore probably the primary limiting nutrient in estuarine and coastal ecosystems, especially those heavily loaded by human influence (Howarth et al. 1995).

The unexpectedly great frequency of high N:P in open oceans is surprising because, except for a few cases (e.g. Kron et al. 1991), most analyses have concluded that N:P is usually <16 in marine ecosystems. This pattern has probably been missed because measurements of dissolved organic, particulate or whole N and P fractions have lagged behind measurements of NO_2^-, NO_3^-, NH_4^+ and PO_4^{3-}.

Some modern studies measuring several N and P pools at oligotrophic, open ocean sites also indicate total N:P frequently >16 in the photic zone. For example, the fractionation of N and P at an oligotrophic station in the open Pacific can be approximated from data collected under the Hawaii Ocean Time-series project of the Joint Global Ocean Flux Study (Winn et al. 1993). Water samples, collected in February, March, April, May, June, July, August, September, October, and December of 1991, were analyzed for several N and P fractions. N and P in different fractions and depths can be estimated from averaged N and P profiles in different sample series. Averaging over all cruises during the year, the approximate average total nitrogen concentration in the upper 200 m was 9.7 μM and total phosphorus was 0.4 μM. The annual average N:P of the photic zone was therefore about 24, but average N:P of the photic zone during individual cruises was as high as 120 and was >16 on 8 out of 10 dates. Karl et al. (1995) have also discussed the lack of evidence for N limitation shown in these samples. Although Karl et al. (1993) suggest that the average N:P (considering dissolved forms alone) does not depart from the Redfield ratio, more recent work shows that photic zone TDN:TDP ratios in the photic zone are stongly skewed toward high, non-Redfield ratios (David M. Karl, pers. comm.).

In spite of the high total N:P found at this site, traditional analyses of dissolved inorganic N:P would have suggested N limitation. The average dissolved inorganic N:P in the upper 200 m was usually <5. The particulate and dissolved organic fractions, however, had N:P as high as 200:1 in many samples. The fact that N:P was very high in particulate matter further counter-indicates N-limitation at this open-ocean site since much of the particulate matter may be living plankton.

Other studies also show that particulate and dissolved organic N:P can be very high at a variety of oligotrophic sites in the open sea (Copin-Montégut & Copin-Montégut 1983; Jackson & Williams 1985). The organic N and P pools are therefore essential to our understanding of N and P cycling in the seas but measurements of the size of these pools and their dynamics have been beyond the reach of standard marine chemical methodology. We therefore need much more extensive knowledge of the total stoichiometry of marine ecosystems.

Differences in the total N:P stoichiometry among marine ecosystems may reconcile the conflicting results obtained in nutrient bioassays. Many such analyses have experimentally enriched nutrients in natural seawater samples to see which nutrients lead to increased phytoplankton growth (Table 2). In polluted estuaries and bays, where total N:P ratios are frequently low (Figure 3), such bioassays usually find that N addition leads to greatest increases in algal activity (Granéli 1987; Le Rouzic & Bertru 1992; Ryther & Dunstan 1971), although some coastal systems may vary seasonally from N to P limitation (D'Elia et al. 1986; D'Elia et al. 1992; Fisher et al. 1992; Rinne & Tarkiainen 1975) depending upon the N:P of inflowing waters. Less polluted coastal waters also show N limitation (Edmondson 1956; Smayda 1974; Tarkiainen et al. 1974) but addition of both N and P together often results in greater phytoplankton growth than addition of N alone (Granéli 1984; Rudek et al. 1991; Vince & Valiela 1973). The N:P supply ratio may therefore be very close to 16:1 in some less polluted coastal systems. In the open ocean, however, where I found total N:P generally >16 (Figure 3), nitrogen is rarely the principal nutrient limiting phytoplankton abundance or activity (Table 2). Instead, nutrient bioassays usually find that production is limited by phosphorus (Berland et al. 1980; Berland et al. 1987; Bonin et al. 1989; Lapointe 1986), iron (DiTullio et al. 1993; Martin & Fitzwater 1988; Menzel & Ryther 1961; Menzel et al. 1963; Ryther & Guillard 1959; Tranter & Newell 1963), or silica (Smayda 1971). Nitrogen is probably rarely the principal element limiting phytoplankton in the open oceans because its supply appears to be ample relative to phosphorus.

Global N:P model

The combined patterns of variation of total N:P ratios in marine and freshwater environments provides a model of the global N:P cycle. In upstream freshwaters, where input is mostly derived from high N:P precipitation and high N:P run-off from undisturbed soils (Downing & McCauley 1992), P concentrations are low, N:P is high (Figure 4) and production is strongly P-limited. As water moves downstream, it is enriched by high P, low N:P run-off from terrestrial systems, P increases and N:P declines, resulting in frequent N-limitation of primary production (Elser et al. 1990) and blooms of N-fixing cyanobacteria (Smith 1983). This enrichment may proceed to a varying degree, depending upon the size, land use, and human inhabitation in the drainage system (Peierls et al. 1991).

When nutrient laden rivers flow into coastal marshes and estuaries, anoxic sediments and organic matter are usually abundant, leading to rapid denitrification (Seitzinger 1988), which may explain the drop in N:P as freshwater discharge passes through estuarine environments (Figure 4). As coastal waters move out into open seas, P may be sedimented more rapidly than N, as evidenced by low N:P in deep waters (Figures 2–3), marine sediments and sedimentary rocks (Downing & McCauley 1992). Phosphorus concentrations decline more rapidly than N concentrations in open seas, perhaps because atmospheric N_2 can be fixed in surface waters (Capone & Carpenter 1982;

Table 2. Examples of nutrient enrichment studies arranged in approximately increasing order of expected total N:P, based on the pattern observed in Figure 3. Elements that increased algal growth, production or biomass are listed in decreasing order of their importance (most important first), as perceived by the authors of the studies. "N+P" indicates that adding N and P together increased biomass over addition of N or P alone.

Place	Reference	Limiting elements
Polluted, enclosed estuary: Gulf of Morbihan, France	Le Rouzic & Bertru (1992)	N, N+P
Shallow, polluted Long Island Bays	Ryther & Dunstan (1971)	N
Brackish Laholm Bay	Granéli (1987)	N
Chesapeake Bay	Fisher et al. (1992)	N, P*
Chesapeake Bay	D'Elia et al. (1986; 1992)	N, P†
Neuse River Estuary, North Carolina.	Rudek et al. (1991)	N, N+P
New York Harbor, polluted Long Island Sound	Ryther & Dunstan (1971)	N
Lower Narragansett Bay	Smayda (1974)	N
Polluted Baltic near Helsinki	Rinne & Tarkiainen (1975)	N, P*
Baltic near Helsinki	Tarkiainen et al. (1974)	N, P‡
Woods Hole Harbor	Edmondson (1956)	N, P
Woods Hole Harbor	Vince & Valiela (1973)	N, N+P
Oresund narrows near Copenhagen	Granéli (1984)	N, N+P
Ocean between Bermuda and Puerto Rico	Smayda (1971)	N, P, Si
Open Ocean between Montauk Pt. and Bermuda	Ryther & Guillard (1959)	Fe, N+P
Indian Ocean	Tranter & Newell (1963)	Fe, N+P
Sargasso Sea	Menzel & Ryther (1961)	Fe, N+P
Sargasso Sea	Ryther & Guillard (1959)	Fe, N+P
Sargasso Sea	Menzel et al. (1963)	Fe, N+P
Sargasso Sea, Sargassum	Lapointe (1986)	P
Subarctic Pacific	Martin & Fitzwater (1988)	Fe
Mediterranean Sea	Bonin et al. (1989)	P
Mediterranean Sea	Berland et al. (1977)	P
Mediterranean Sea	Berland et al. (1987)	P, N+P

* changes seasonally. † changes according to N:P of inflowing waters. ‡ usually N limited.

Smith & Veeh 1989) and aeolian N deposition can be substantial (Fanning 1989). Surface N:P ratios can increase dramatically in offshore marine environments, leading to frequent limitation of production by P or nutrients other than N.

Taken together, the results of this study and those of Downing & McCauley (1992) suggest the form of general longitudinal N:P profiles of temperate zone rivers in developed countries (Figure 5A). Upland, headwater streams should have high N:P that largely reflect nutrient ratios in precipitation. Thus, in pristine areas, N:P ratios should be very high because N in rain is enriched by atmospheric fixation and there should be little P in aeolian deposition. N:P in precipitation is sometimes lower in urbanized or agricultural areas, and may alter headwater N:P (Loehr 1974). Downstream freshwaters will usually be

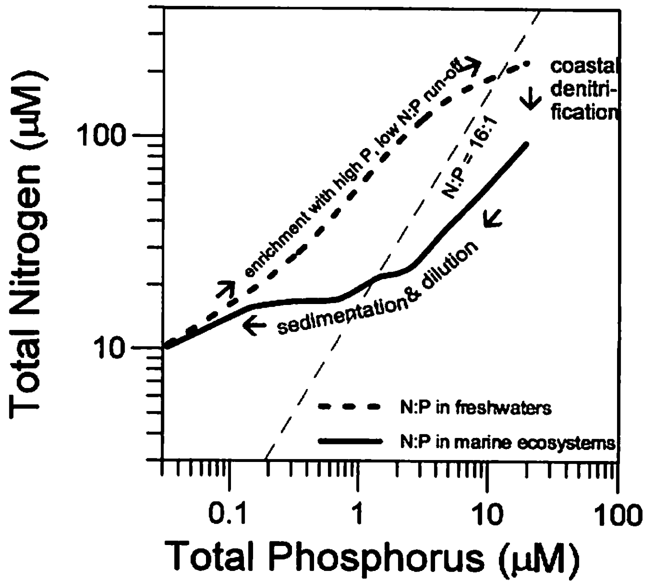

Figure 4. Observed global aquatic N:P cycle. Average trends shown are locally weighted sequentially smoothed fits to the actual observations. Trends were determined using LOWESS, a model-free, unbiased method for finding average trends in data (Cleveland & McGill 1985). LOWESS parameters were $\Delta = 0$, n-steps = 2, $f = 0.5$. The freshwater line is from Downing & McCauley (1992) and the marine line is fitted to the data in Figure 1.

richer in both N and P but have lowered N:P unless P is selectively removed or N is enriched.

A conservative mixing of two solutions that have different relative concentrations of two elements results in a linear relationship between the concentrations of the two elements in all possible admixtures of the two solutions. Therefore, if the variations in observed N:P are simply due to conservative enrichment of rainwater by polluted waters or the dilution of river water by dilute oceanic waters, relationships between N and P should be approximately linear across marine and freshwater systems. In freshwater systems, there is a clear change in mechanism of relative N and P enrichment as one moves from headwaters to downstream systems (Figure 5B). Since the slope of the relationship is directly related to the N:P of the enriching solution, this pattern is suggestive of a progressive decrease in the N:P of effluents. In marine systems however, the decline in N concentration with decreasing P in open oceans is nearly linear, suggesting that variations in marine N:P may be related to the conservative dilution of freshwater effluent by oceanic waters.

This study adds to the mounting evidence that productivity is limited by elements other than N in many of the world's open oceans (Martin & Fitzwater 1988; Martin et al. 1994). These data imply that N:P > 16 is found

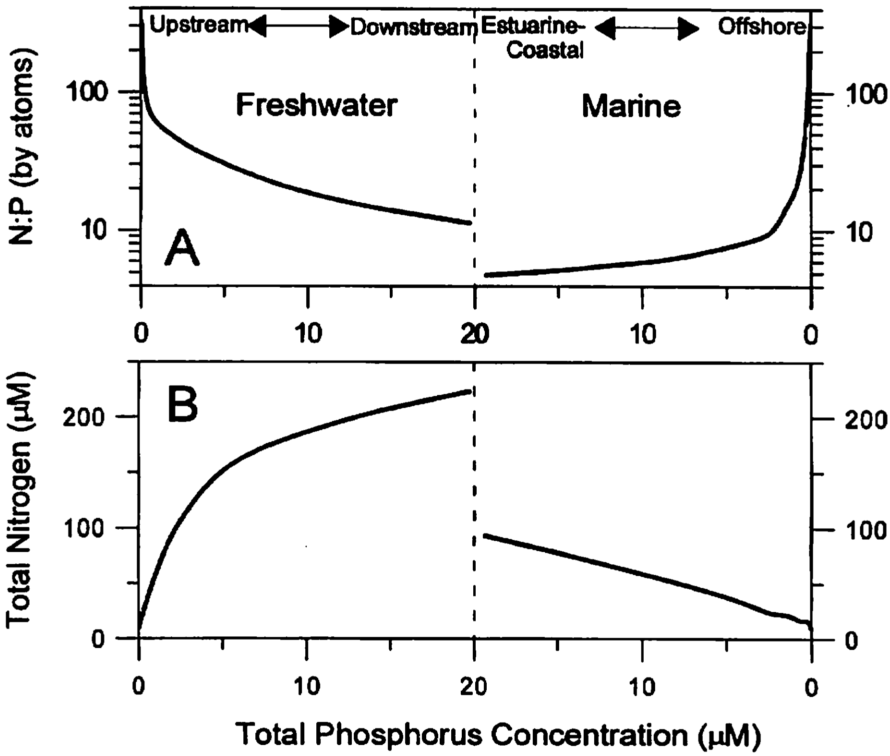

Figure 5. Predicted gradient in total N and P along the axis of rivers in temperate zone developed countries. Trends are actual LOWESS relationships (*see* Figure 4) in data from upstream and downstream lakes (Downing & McCauley 1992) and inshore vs. offshore marine ecosystems. (A) shows trends in N:P while (B) shows trends in N concentration. Data are arranged in order of increasing enrichment (0–20 μM P) in freshwaters and in order of decreasing enrichment (20–0 μM P) in marine systems.

most frequently below total P concentrations of 1 μM. Broad-scale surveys of total P in marine environments suggest that concentrations >1 μM may be rare away from shore or upwellings (Gibbs et al. 1986; Rochford 1958). The concept of nitrogen limitation of marine productivity and carbon cycling may therefore have its principal relevance to the shallow and coastal areas of the world's oceans where waters are polluted by low N:P freshwaters.

One might correctly suggest that <200 sets of observations of total N and total P can scarcely be expected to represent all of the world's oceans. However, it is significant that these data are all of the total N and P inventories that I could discover in an extensive literature review and letter writing campaign. This is the most extensive collection of data on total N and P stoichiometry in marine waters. It strongly suggests that open ocean nutrient ratios do not support the marine nitrogen limitation model.

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References

Amon RMW & Benner R (1994) Rapid cycling of high-molecular-weight dissolved organic matter in the ocean. Nature (London) 369: 549–552

Ben-Taleb KS, El-Khattal AO, Ramadan ZM & El-Belaazi AM (1987) Some physico-chemical characteristics of Tajura sea-shore water: East of Tripoli (S.P.L.A.J.) Bulletin of the Marine Biological Research Center of Tajura 8: 89–124

Bentzen E, Taylor WD & Millard ES (1992) The importance of dissolved organic phosphorus to phosphorus uptake by limnetic plankton. Limnol. and Oceanography 37: 217–231

Berland BR, Bonin DJ & Maestrini SY (1977) Role du phosphore dans la limitation de la production primaire dans les eaux Mediterranéenes. Actualités de Biochimie Marine 2: 243–246

Berland BR, Burlakova ZP, Georgieva LV, Izmestieva M, Khologov VI, Krupatkina DK, Maestrini SY & Zaika VE (1987) Phytoplancton estival de la mer du Levant, biomasse et facteurs limitants. IFREMER Acta Colloquia 5: 61–83

Black RE, Lukatelich RJ, McComb AJ & Rosher JE (1981) Exchange of water, salt, nutrients and phytoplankton between Peel Inlet, Western Australia, and the Indian Ocean. Australian J. Marine and Freshwater Res. 32: 709–720

Bonin DJ, Bonin MC & Berman T (1989) Mise en évidence expérimentale des facteurs nutritifs limitants de la production du micro-nanoplancton et de l'ultraplancton dans une eau cotière de la Méditerranée orientale (Haîfa, Israël). Aquatic Sciences 51(2): 129–152

Boynton WR, Kemp WM & Keefe CW (1982) A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In: Kennedy VS (Ed) Estuarine Comparisons (pp 69–70). Academic, NY

Capone D & Carpenter EJ (1982) Nitrogen fixation in the marine environment. Science 217: 1140–1142

Cleveland W & McGill SR (1985) Graphical perception and graphical methods for analyzing scientific data. Science 229: 828–833

Copin-Montégut C & Copin-Montégut G (1983) Stoichiometry of carbon, nitrogen, and phosphorus in marine particulate matter. Deep-Sea Research 30: 31–46

Coste B, Le Corre P & Minas HJ (1988) Re-evaluation of the nutrient exchanges in the Strait of Gibralter. Deep-Sea Research 35: 767–775

D'Elia CF, Steudler PA & Corwin N (1977) Determination of total nitrogen in aqueous samples using persulfate digestion. Limnology and Oceanography 22: 760–764

D'Elia CF, Sanders JG & Boynton WR (1986) Nutrient enrichment studies in a coastal plain estuary: Phytoplankton growth in large-scale, continuous cultures. Can. J. Fisheries and Aquatic Sci. 43: 397–406

D'Elia CF, Harding LW Jr., Leffler M & Mackiernan GB (1992) The role and control of nutrients in Chesapeake Bay. Water Science and Technology 26: 2635–2644

Dal Pont GK, Hogan M & Newell B (1974) Laboratory techniques in marine chemistry II – a manual. CSIRO Australia Division of Fisheries and Oceanography, Report 55
Dame R, Chrzanowski T, Bildstein K, Kjerfve B, McKellar H, Nelson D, Spurrier J, Stancyk S, Stevenson H, Vernberg J & Zingmark R (1986) The outwelling hypothesis and North Inlet, South Carolina. Marine Ecology Progress Series 33: 217–229
Degobbis D & Gilmartin M (1990) Nitrogen, phosphorus, and biogenic silicon budgets for the northern Adriatic Sea. Oceanologica Acta 13: 31–45
DiTullio GR, Hutchins DA & Bruland KW (1993) Interaction of iron and major nutrients controls phytoplankton growth and species composition in the tropical North Pacific Ocean. Limnol. and Oceanography 38: 495–508
Downing JA & McCauley E (1992) The nitrogen:phosphorus relationship in lakes. Limnol. and Oceanography 37: 936–945
Edmondson WT (1956) Factors affecting productivity in fertilized salt water. Deep-Sea Research Supplement 3: 451–464
Elser JJ & Hassett RP (1994) A stoichiometric analysis of the zooplankton-phytoplankton interaction in marine and freshwater ecosystems. Nature 370: 211–213
Elser JJ, Marzolf ER & Goldman CR (1990) Phosphorus and nitrogen limitation of phytoplankton in freshwaters of North America: A review and critique of experimental enrichments. Can. J. Fisheries and Aquatic Sci. 47: 1468–1477
Fanning KA (1989) Influence of atmospheric pollution on nutrient limitation in the ocean. Nature 339: 460–463
FAO (1993) Fishery Statistics Yearbook, vol. 72 Rome.
Fisher TR, Peele ER, Ammerman JW & Harding LW Jr (1992) Metabolic adaptations of deep-sea benthic foraminifera to seasonally varying food input. Marine Ecology Progress Series 82: 51–63
Gibbs CF, Tomczak M & Longmore AR (1986) The nutrient regime of Bass Strait. Australian J. Marine and Freshwater Res. 37: 451–466
Glibert PM, Garside C, Fuhrman JA & Roman MR (1991) Time-dependent coupling of inorganic and organic nitrogen uptake and regeneration in the plume of the Chesapeake Bay estuary and its regulation by large heterotrophs. Limnol. and Oceanography 36: 895–909
Granéli E (1984) Algal growth potential and limiting nutrients for phytoplankton production in Öresund water of Baltic and Kattegat origin. Limnologica 15: 563–569
Granéli E (1987) Nutrient limitation of phytoplankton biomass in a brackish water bay highly influenced by river discharge. Estuarine, Coastal and Shelf Science 25: 555–565
Grasshoff K (1976) Methods of Seawater Analysis. 1st edn. Verlag ChemieWeinheim
Grasshoff K (1983) Methods of Seawater Analysis. 2nd edn. Verlag Chemie Weinheim
Hecky RE & Kilham P (1988) Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment. Limnol. and Oceanography 33: 796–822
Hoegh-Goldberg O (1994) Uptake of dissolved organic matter by larval stage of the crown-of-thorns starfish *Acanthaster planci*. Marine Biology 120: 55–63
Howarth RW (1988) Nutrient limitation of net primary production in marine ecosystems. An. Rev. of Ecology 19: 89–110
Howarth RW, Jensen H, Marino R & Postma H (1995) Transport to and processing of phosphorus in near-shore and oceanic waters (pp 323–345). In: Tiessen H (Ed) Phosphorus in the Global Environment: Transfers, Cycles and Management. SCOPE 54, Wiley & Sons, Chichester
Ikeda T, Carleton JH, Mitchell AW & Dixon P (1982) Ammonia and phosphate excretion by zooplankton from the inshore waters of the Great Barrier Reef. II. Their *in situ* contributions to nutrient regeneration. Australian J. Marine and Freshwater Res. 33: 683–698
Jackson GA & Williams PM (1985) Importance of dissolved organic nitrogen and phosphorus to biological nutrient cycling. Deep-Sea Research 32: 223–235

Jordan TE, Correll DL, Miklas J & Weller DE (1991) Nutrients and chlorophyll at the interface of a watershed and an estuary. Limnol. and Oceanography 36: 251–267

Karl DM, Leteller R, Hebel D, Tupas L, Dore J, Christian J & Winn C (1995) Ecosystem changes in the North Pacific subtropical gyre attributed to the 1991–92 El Niño. Nature (London) 373: 230–234

Karl DM, Tien G, Dore J & Winn CD (1993) Total dissolved nitrogen and phosphorus concentrations at US-JGOFS station ALOHA: Redfield reconciliation. Marine Chemistry 41: 203–208

Krom MD, Kress N, Brenner S & Gordon LI (1991) Phosphorus limitation of primary productivity in the eastern Mediterranean Sea. Limnol. and Oceanography 36: 424–432

Lahdes E & Leppanen JM (1988) Cycling of organic matter during the vernal growth period in the open Northern Baltic proper. II. Nutrient development and chemical composition of particulate matter. Finnish Marine Research 255: 19–35

Lapointe BE (1986) Phosphorus-limited photosynthesis and growth of *Sargassum natans* and *Sargassum fluitans* (Phaeophyceae) in the western North Atlantic. Deep-Sea Research 33: 391–399

Le Rouzic & Bertru BG (1992) Détermination par bioessai de la biodisponibilité des ressources azote et phosphore, dans les eaux du Golfe du Morbihan. Revue des Sciences de l'Eau 5: 97–111

Mackenzie FT, Ver LM, Sabine C, Lane M & Lerman A (1993) C, N, P, S global biogeochemical cycles and modeling of global change. In: Wollast R, Mackenzie FT & Chou L (Eds) Interactions of C, N, P and S, Biogeochemical Cycles and Global Change (pp 1–61). Springer-Verlag, Berlin

Martin JH & Fitzwater SE (1988) Iron deficiency limits phytoplankton growth in the north-east pacific subarctic. Nature 331: 341–343

Martin JH, Coale KH, Johnson KS, Fitzwater SE, Gordon RM, Tanner SJ, Hunter CN, Elrod VA, Nowicki JL, Coley TL, Barber RT, Lindley S, Watson AJ, Van Scoy K, Law CS, Liddicoat MI, Ling R, Stanton T, Stockel J, Collins C, Anderson A, Bidigare R, Ondrusek M, Latasa M, Millero FJ, Lee K, Yao W, Zhang JZ, Friederich G, Sakamoto C, Chavez F, Buck K, Kolber Z, Greene R, Falkowski P, Chisholm SW, Hoge F, Swift R, Yungel J, Turner S, Nightingale P, Hatton A, Liss P & Tindale NW (1994) Testing the iron hypothesis in ecosystems of the equatorial Pacific Ocean. Nature 371: 123–129

Menzel DW & Corwin N (1965) The measurement of total phosphorus in seawater based on the liberation of organically bound fractions by persulfate oxidation. Limnol. and Oceanography 10: 280–282

Menzel DW & Ryther JH (1961) Nutrients limiting the production of phytoplankton in the Sargasso Sea, with special reference to iron. Deep-Sea Research 7: 276–281

Menzel DW, Hulburt EM & Ryther JH (1963) The effects of enriching Sargasso Sea water on the production and species composition of the phytoplankton. Deep-Sea Research 10: 209–219

Meybeck M (1982) Carbon, nitrogen, and phosphorus transport by world rivers. Am. J. Sci. 282: 401–450

Peierls B, Caracao N, Pace M & Cole J (1991) Human influence on river nitrogen. Nature 350: 386–387

Perttilä MP, Tulkki P & Pietikainen S (1980) Mean values and trends of hydrographical and chemical properties in the Gulf of Finland 1962–1978. Finnish Marine Research 247: 38–50

Pietikäinen S, Niemi A, Tulkki P & Aurimaa K (1978) Mean values and trends of physical and chemical properties in the Gulf of Bothnia 1962-1975. Finnish Marine Research 244: 64–75

Pitkänen H & Malin V (1980) The mean values and trends of some water quality variables in winter in the Gulf of Finland 1966–1978. Finnish Marine Research 247: 51–60

Ramadan ZM, Ben-Taleb KS & Trozosinska A (1984) Ecological conditions in the Mediterranean coastal zone. A case study: Fishery Harbour of Tripoli, Libya. Bulletin of the Marine Research Center of Tripoli 5: 69–106

Redfield AC (1934) On the proportions of organic derivatives in sea water and their relation to the composition of plankton. In: James Johnstone Memorial Volume (pp 176–192). Univ. Press, Liverpool

Redfield AC (1958) The biological control of chemical factors in the environment. Am. Sci. 46: 205–221

Redfield AC, Ketchum BH & Richards FA (1963) The influence of organisms on the composition of sea-water. In: Hill MN (Ed) The Sea, vol. 2 (pp 26–77) Wiley, NY

Rinne IE & Tarkiainen E (1975) Chemical factors affecting algal growth off Helsinki. Merentutkimuslait. Julk. Havsforskninginst. Skr. 239: 91–99

Ritschard RL (1992) Marine algae as a CO_2 sink. Water Air Soil Poll. 64: 289–303

Rochford D (1958) Total phosphorus as a means of identifying East Australian water masses. Deep-Sea Research 5: 89–110

Rudek J, Paerl HW, Mallin MA & Bates PW (1991) Seasonal and hydrological control of phytoplankton nutrient limitation in the lower Neuse River Estuary, North Carolina. Marine Ecology Progress Series 75: 133–142

Rydberg L & Sundberg J (1986) Seasonal nutrient supply to coastal waters. In: Marine Interfaces Ecohydrodynamics, 17th Annual Liege Colloquium on Ocean Hydrodynamics (pp 467–485). Elsevier Oceanography Series 42

Ryther JH & Dunstan WM (1971) Nitrogen, phosphorus, and eutrophication in the coastal marine environment. Science 171: 1008–1013

Ryther JH & Guillard RRL (1959) Enrichment experiments as a means of studying nutrients limiting to phytoplankton production. Deep-Sea Research 6: 65–69

Sakamoto M (1966) Primary production by phytoplankton community in some Japanese lakes and its dependence on lake depth. Archiv für Hydrobiologie 62: 1–28

Seitzinger SP (1988) Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. Limnol. and Oceanography 33: 702–724

Smayda TJ (1974) Bioassay of the growth potential of the surface water of lower Narragansett Bay over an annual cycle using the diatom *Thalassiosira pseudonana* (oceanic clone, 13-1). Limnol. and Oceanography 19: 889–901

Smith SV (1984) Phosphorus vs. nitrogen limitation in the marine environment. Limnol. and Oceanography 29: 1149–1160

Smith SV & Veeh HH (1989) Mass balance of biogeochemically active materials (C, N, P) in a hypersaline gulf. Estuarine, Coastal and Shelf Science 29: 195–215

Smith SV, Kimmerer WJ & Walsh TW (1986) Vertical flux and biogeochemical turnover regulate nutrient limitation of net organic production in the North Pacific Gyre. Limnol. and Oceanography 31: 161–167

Smith VH (1983) Low nitrogen to phosphorus ratios favor dominance by blue-green algae in lake phytoplankton. Science 221: 669–671

Strickland JH & Parsons TR (1968) A Practical Handbook of Seawater Analysis. Fisheries Res. Board Canada, Ottawa

Strickland JDH & Parsons TR (1972) A Practical Handbook of Seawater Analysis. Fisheries Res. Board Canada 167

Suzuki Y, Sugimura Y & Ito T (1985) A catalytic oxidation method for the determination of total nitrogen dissolved in seawater. Marine Chemistry 16: 83–97

Smayda TJ (1971) Further enrichment experiments using the marine centric diatom *Cyclotella nana* (clone 13-1) as an assay organism. In: Costlow JD (Ed) Fertility of the Sea, vol. 2 (pp 493–511). Gordon and Breach, NY

Tarkiainen E, Rinne I & Niemisto L (1974) On the chemical factors regulating the primary production of phytoplankton in the Baltic proper. Merentutkimuslait. Julk. Havsforskningsinst. Skr. 238: 39–52

Tett P (1990) The photic zone. In: Herring PJ, Campbell AK, Whitfield M & Maddock L (Eds) Light and Life in the Sea (pp 59–87). Cambridge

Tranter DJ & Newell BS (1963) Enrichment experiments in the Indian Ocean. Deep-Sea Research 10: 1–9

Treguer P & Le Corre P (1975) Manuel d'analyse des sels nutritifs dans l'eau de mer. Lab. Océanogr. Chimique, Univ. Brétagne Occident. Brest

Updegraff DM, Stanton DJ & Spencer MJ (1977) Surface waters of Waimea Inlet and Nelson Haven: A preliminary assessment of quality. New Zealand Journal of Marine and Freshwater Research 11: 559–575

Valderrama JC (1981) The simultaneous analysis of total nitrogen and total phosphorus in natural waters. Marine Chemistry 10: 109–122

Vince S & Valiela I (1973) The effects of ammonium and phosphate enrichments on chlorophyll a, pigment ratio, and species composition of phytoplankton of Vineyard Sound. Marine Biology 19: 69–73

Vitousek PM & Howarth RW (1991) Nitrogen limitation on land and in the sea: How can it occur? Biogeochem. 13: 87–115

Williams PM (1967) Sea surface chemistry: Organic carbon and organic and inorganic nitrogen and phosphorus in surface films and subsurface waters. Deep-Sea Research 14: 791–800

Winn C, Lukas R, Karl D & Firing E (1993) Hawaii Ocean Time-series Data Report 3, 1991. University of Hawaii, Honolulu

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META-ANALYSIS OF MARINE NUTRIENT-ENRICHMENT EXPERIMENTS: VARIATION IN THE MAGNITUDE OF NUTRIENT LIMITATION

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Abstract. Nutrient bioassay experiments have been performed in many marine and estuarine environments around the world. Although protocols have been relatively uniform, these experiments have yielded mixed results, implicating nitrogen, phosphorus, silica, iron, and several other elements as factors limiting phytoplankton growth. Meta-analysis has the potential to explain much of this variation by exploring the relationship between the magnitude of limitation and various environmental characteristics. We quantified limitation with a simple metric, Δr, that estimates the change in the per unit growth rate of phytoplankton directly attributable to addition of a specific nutrient, such as nitrogen, iron, or phosphorus. Preliminary analyses indicated that experiments lasting $\leq$1 d exhibited time lags in the numerical response of phytoplankton to nutrient addition, while experiments lasting >7 d confounded nutrient limitation with processes such as increased grazing or depletion of other nutrients. Thus, we restricted the meta-analysis to results from 2–7 d experiments. These analyses showed that phosphorus enrichment usually had little impact on phytoplankton growth, while enrichments of nitrogen and iron increased phytoplankton growth by 0.1–0.3 d^{-1}. Nutrient limitation due to N, P, and Fe varied significantly among sites. Nitrogen limitation was greatest in nearshore, nutrient-polluted, and temperate environments (where most experiments have been performed), while phosphorus and iron limitation were strongest in open ocean, unpolluted, and tropical ecosystems, or those receiving pollutants with high N:P ratios. Because phosphorus-enrichment studies have been most often performed in relatively polluted coastal waters, the possible role of phosphorus in limiting primary production in unpolluted oceanic systems may have been underestimated. Examining heterogeneity of responses of different systems to experiments is a valuable application of meta-analysis and can facilitate the development of new ecological insights.

Key words: bioassay; coastal and estuarine studies; iron; marine studies; meta-analysis; nitrogen; nutrient-enrichment experiments; nutrient limitation; oceanic systems, pollution; phosphorus; phytoplankton.

SPECIAL FEATURE

INTRODUCTION

Marine primary production is of major ecological and economic importance. Phytoplankton production fuels the production of higher trophic levels, which supply more than 90×10^9 kg of food to the world economy each year (FAO 1993). Marine ecosystems (including estuarine, coastal, and marine habitats) are also essential in the global carbon budget, storing 50 times more inorganic carbon than the earth's atmosphere, suggesting that marine primary production may play a global climatological role (Ritschard 1992, Mackenzie et al. 1993). Consequently, understanding the factors that limit or regulate phytoplankton production in the sea is of vital global interest. One of the most important factors potentially limiting primary production is the availability of nutrients.

The identity of the nutrient that limits primary production has been inferred from geochemical budgets, ratios of elements dissolved in seawater or contained in phytoplankton-sized particles, and experimental nutrient additions. These approaches have yielded a diversity of answers. For example, geochemical budgets suggest that inorganic phosphorus (P) should be in shorter supply than nitrogen (N), and therefore limiting (Meybeck 1982), because atmospheric N_2 can be fixed (Redfield 1958, Vitousek and Howarth 1991). In contrast, ratios of dissolved inorganic nitrogen (N) and phosphorus (P) are often lower than average intracellular N:P ratios of marine organisms (Redfield 1934, Redfield et al. 1963), suggesting that nitrogen should limit marine primary production (e.g., Boynton et al. 1982). Such assessments provide only indirect evidence about the nutrients that limit phytoplankton growth, and offer little insight into the magnitude or severity of nutrient limitation.

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Experimental manipulation of inorganic nutrients constitutes a more direct approach to assessing both the quality and strength of nutrient limitation in phytoplankton assemblages, and such experiments have been a mainstay of marine research for more than 50 yr. Although these experiments are sometimes criticized because of the small scales on which they often are conducted (Hecky and Kilham 1988), experimental additions performed at small scales have been validated by large-scale addition experiments in both lakes and oceans (Schindler 1978, Granéli and Sundback 1985, D'Elia et al. 1986, Elser et al. 1990, Oviatt et al. 1995, Taylor et al. 1995).

Although most nutrient-addition experiments in oceans have been directed toward the study of nitrogen (Capone and Carpenter 1982), it is clear that nitrogen is not always limiting in the ocean. Enrichments of phosphorus, iron, and silica have also been shown to induce significant increases in marine primary producers in at least some experiments (Boynton et al. 1982, Martin and Fitzwater 1988, Martin et al. 1994). The variation in results of experimental studies suggests two new approaches. First, it may be more profitable to estimate the magnitude of limitation than the presence or absence of it, for example, by quantifying how much the ambient growth rate of phytoplankton is depressed below the potential growth rate under nutrient-saturated conditions (e.g., Hecky and Kilham 1988, Osenberg and Mittelbach 1996). This is a question with a quantitative answer that can be extracted from a number of available studies that have been conducted in a broad variety of marine habitats and at different times. This meta-analytic approach obviates problems inherent in using the frequency of occurrence of statistically significant effects of nutrient enrichment as a de facto criterion of the importance of limitation by a given nutrient, as used implicitly in expert reviews or explicitly in vote-counting syntheses (e.g., see Gurevitch et al. 1992).

A second approach that might be fruitful would be to directly examine the variation in results and seek a framework to explain the variation of results rather than seeking a single global answer (e.g., ''is it iron, or is it nitrogen''). Indeed, the exploration of variation in effects is one of the most valuable applications of meta-analysis. For example, a meta-analysis of nutrient limitation in freshwaters (thought to be primarily P limited) uncovered a surprisingly important role of N (Elser et al. 1990). Furthermore, clarifying sources of variation in nutrient limitation has important implications for both applied and basic research. For example, the current paradigm in ocean sciences remains focused on nitrogen limitation (Jacques and Tréguer 1986, Barnes and Hughes 1988, Valiela 1995). If nitrogen is not the principal element limiting primary production in all (or even most) locales, then scientific resources, and considerable remedial action, might be misdirected. Meta-analysis could help to interpret the overall conclusions that can be drawn from a set of disparate experiments, by revealing patterns, offering hypotheses to explain variation in effects, and pointing to conceptual and empirical holes in the current level of understanding.

In this paper, we present a meta-analysis of results from nutrient-enrichment-experiments performed in marine habitats around the world. Our objective is to look for patterns in the magnitude of nutrient limitation in marine phytoplankton assemblages to highlight the application of meta-analysis. The analysis is not meant to be exhaustive, but a first step to illustrate the utility of meta-analysis and the importance of conceptual steps required in its application (a future analysis of an expanded data set examining co-limitation and other facets of this question will be presented elsewhere). To this end, we take a three-tiered approach: (1) we provide a conceptual, quantifiable definition of nutrient limitation; (2) we conduct a preliminary analysis to assess the confounding influence of experiment duration; and (3) based on these considerations, we restrict our primary meta-analysis to the most appropriate data and seek to explain the variation in limitation observed among different studies. We limit this paper's focus to variation among marine habitats that differ in the degree of anthropogenic pollution or geographic locale. In so doing, we hope to take a first step in the explanation of this variation and provide a means for prediction.

Methods and Analyses

The available data

We began our analysis by searching the literature for experiments that assessed nutrient limitation. All of the studies listed in Downing (1997) were included, to which we added observations from an extensive literature review of marine journals (details available from J. A. Downing). Below, we describe the standard experimental protocol of these studies. We then provide an explicit discussion of our conceptual and operational definition of limitation. Based on this definition, time-scale considerations, and other criteria, we restricted the studies included in the meta-analysis. Because this refinement of the question and evaluation of time-scale issues are both important parts of ecological meta-analysis (Osenberg et al. 1997, 1999), and because selection criteria can affect the outcome of a meta-analysis (Englund et al. 1999), we specifically highlight the details of this approach below (see *The quantification of nutrient limitation:* . . . and *Time-scale considerations*).

The typical experiment found in this literature was initiated by taking a large water sample (usually many liters) from a given marine sampling station, usually near the surface in the photic zone. This large sample was usually sieved to remove macrozooplankton grazers that might mask phytoplankton responses to nutrient additions (microzooplankton cannot be removed by sieving). The water sample was divided into subsamples, some of which received a dose of various dis-

Special Feature

solved, inorganic nutrients; others were left unamended to serve as controls. The most common experiment dosed some subsamples with nitrogen as NO_3, and others with phosphorus as PO_4. Other nutrients commonly added included iron and silica. The subsamples were then incubated in situ or under simulated conditions of ambient temperature and light. During or after the incubation period, samples were withdrawn from the incubation vessels and phytoplankton biomass estimated, usually as chlorophyll *a* concentration, but sometimes as light absorbance, phytoplankton biovolume, or cell numbers. The influence of the various nutrient additions was typically assessed from comparisons of estimated phytoplankton biomass in unenriched control samples to that found in the enriched samples. The published comparisons were usually in the form of a statistical test or a graphical comparison.

The quantification of nutrient limitation: choosing an effect-size metric

At first glance, many of the results of published nutrient-addition experiments appear to be lacking in comparability. Experimental results have been expressed as plots of biomass over time (often without further analysis), as ratios of treatment to control biomass, as differences in biomass between treatment and control, as cell growth rates in treatment and control, as differences in the slopes of temporal trends in biomass accumulation, as differences in photosynthetic rates, etc. Some metrics of effect size, such as *d,* borrowed from the statistical literature on meta-analysis (e.g., Gurevitch et al. 1992), would only be calculable in a minority of these experiments, would be difficult to interpret biologically, and could give misleading answers (Osenberg et al. 1997). Nevertheless, with careful attention to the conceptual definition of nutrient limitation, many measures reported in published experiments can be converted to a single, biologically meaningful measure of nutrient limitation that is comparable across studies.

A conceptual definition of limitation requires explicit consideration of how population growth responds to the augmentation of a growth-limiting factor (Hecky and Kilham 1988). Here, we define nutrient limitation as "the change in the per unit (per gram, per unit carbon, or per chlorophyll *a*) growth rate of an algal assemblage following the addition of surplus nutrients" (Osenberg and Mittelbach 1996). Hence, if $N_{x,E}$ and $N_{x,C}$ are respectively the amount of algae in the Experimental (nutrient addition) treatment and the Control (ambient) at the start ($x = 0$) and end ($x = t$) of the experiment, then the degree of limitation, Δr, can be estimated as:

$$\Delta r = r_E - r_C = \frac{dN_E}{N_E\,dt} - \frac{dN_C}{N_C\,dt} = \frac{\ln\left(\frac{N_{t,E}}{N_{0,E}}\right) - \ln\left(\frac{N_{t,C}}{N_{0,C}}\right)}{t}. \tag{1}$$

Because control and enriched treatments typically begin with the same amount of algae (i.e., $N_{0,E} = N_{0,C}$), Eq. 1 often reduces to

$$\Delta r = \frac{\ln\left(\frac{N_{t,E}}{N_{t,C}}\right)}{t}. \tag{2}$$

As conceptualized, Δr should efficiently measure the magnitude of nutrient limitation of the extant algal assemblage. This growth-based definition of limitation is not the only one possible (although it is also the one suggested by Hecky and Kilham 1988), but is most appropriate for the type of data reported in most nutrient-addition experiments, which tend to focus on phytoplankton biomass over relatively short-term experiments (e.g., several days).

Our definition of limitation assumes that nutrients were added in excess. The quantity of added nutrients varied among studies, but nutrients were usually added in great excess relative to the range of concentrations observed in the studied system, i.e., often exceeding the observed range seen across the range of marine ecosystems (Downing 1997). Consequently, we feel it is reasonable to assume that these nutrient-addition experiments measure how far the phytoplankton were from nutrient-saturated growth, as required by our model. If nutrients were not provided above saturating levels, then our estimates of limitation may be biased low.

Time-scale considerations

Because organismal responses occur on various time scales, critical consideration in the application of this metric to published data is the time scale of experimental observations (see a more theoretical development of this question in Osenberg et al. [1999]). To highlight the importance of time scale, we contrast two alternative metrics of effect size: Δr (Eq. 2) and the log response ratio (Hedges et al. 1999; the logarithm of the ratio of algal biomass in the two treatments, $\ln(N_{t,E}/N_{t,C})$). For example, at the start of a nutrient-addition experiment, the biomass of algae in the control and nutrient-addition treatment will diverge at a constant specific (i.e., per unit) rate. As a result, the log response ratio will increase linearly with time (Fig. 1A). The slope of this relationship will be equal to Δr. As feedbacks arise (e.g., due to numerical increases in micrograzers, or limitation by other nutrients), this slope will decline, and after some period of transient dynamics, biomass will eventually settle down to a constant value after the two treatments each reach new equilibria. This asymptote (as $t \rightarrow \infty$, see Fig. 1a) will be determined not only by nutrient limitation, but also by all other feedbacks and forms of density dependence that operate in the system, as well as the structure of the food web within the experimental vessels (Schaffer 1981, Bender et al. 1984). As a result, two studies with

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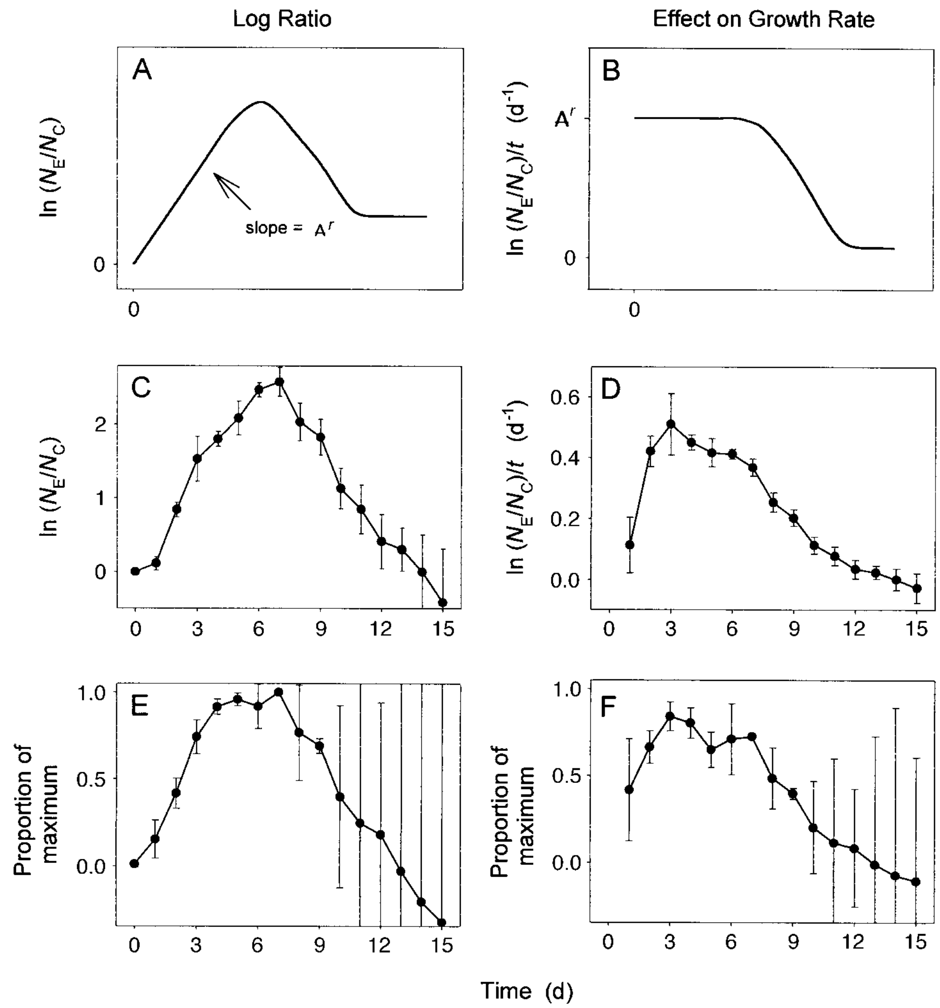

FIG. 1. The response of algal biomass to nutrient enrichment, expressed using the log ratio response ($\ln[N_{t,\mathrm{C}}/N_{t,\mathrm{E}}]$), and the effect on the per capita growth rate ($\Delta r = \ln[N_{t,\mathrm{C}}/N_{t,\mathrm{E}}]/t$). The top panels (A and B) give theoretical expectations, which at short time scales yield estimates of nutrient limitation of growth rate (the initial slope in A; the initial ceiling in B), but at longer time scales include system feedbacks that interfere with the estimation of limitation. The middle panels (C and D) give results obtained from data reported in Fig. 2 of D'Elia et al. (1986). The bottom panels (E and F) summarize results from seven studies, including Thomas et al. (1974: Fig. 3 [Sewage, All, All−Fe, All−Trace elements, All−Vitamins, All+Sewage] and Fig. 4 [N, Sewage, All, All−P, All−Si, All−Fe, All−Trace elements, All−Vitamins, All+Sewage]); D'Elia et al. (1986: Fig. 2 [ammonium addition] and Fig. 3 [NO_3^- addition]); Harrison et al. (1990: Fig. 2 [P, N+P]); and Hein and Riemann (1995: Fig. 1A [N+P addition]). In (E) and (F), all estimates of the log ratio and Δr were first standardized by dividing by the maximum that was observed in that time series. These standardized responses were then averaged across all time series. The bottom four panels (C–F) show means and 95% confidence intervals. Confidence intervals in (C) and (D) were estimated using the formula for variance of the log response ratio from Hedges et al. (1999). Sample sizes in (C) and (D) were $N_{t,\mathrm{C}} = 5$, $N_{t,\mathrm{E}} = 12$; in (E) and (F) the number of comparisons varied from 7 to 22 for days 1–6 and from 2 to 3 for days 7–15.

precisely the same dynamics would have radically different log ratios if they were of different durations but had not yet re-equilibrated (Osenberg et al. 1997, 1999).

In contrast to the log response ratio, Δr can provide more comparable results over short time scales, which are more appropriate to most nutrient-enrichment studies. For example, during the period that the two treatments diverge at a constant rate, Δr should have a constant value (ignoring initial time lags: Fig. 1B). This initial value of Δr provides a direct estimate of growth-rate limitation (Eq. 2; see also Osenberg and Mittelbach 1996, Osenberg et al. 1997, Laska and Wootton 1998). As time passes, however, Δr should eventually decline in magnitude toward zero, as the added nutrient is depleted below saturating concentrations, other nutrients become limiting, grazers increase in abundance, etc. Thus, studies used to estimate limitation of growth rates should be restricted to the early phase of divergence in which Δr is time invariant (i.e., during the flat portion of the curve in Fig. 1B).

Given that it is clearly critical to determine the time scale over which Δr can be applied (i. e., the experiment duration over which Δr is time invariant), we evaluated the behavior of Δr in a subset of appropriate studies, and then only extracted data for the meta-analysis from the appropriate time frame. Ideally, time-scale issues should be investigated for each study, but unfortunately, few time series were available from the literature (most investigators only sample algal biomass at the end of the experiments). Therefore, we had to estimate the appropriate time frame from a subset of studies and then apply this assessment to the entire collection. We examined how Δr and, for comparison, the log ratio ($\ln(N_{t,E}/N_{t,C})$) changed through time for the only study (D'Elia et al. 1986) that sampled algal biomass over at least 10 d, found nutrient limitation in some treatment, and reported variances in $N_{t,C}$ and $N_{t,E}$ (necessary for estimating error in the estimate of Δr; see Hedges et al. 1999). We also examined the general trend over time by using all available studies that reported time series and found nutrient limitation (seven papers, yielding 22 different comparisons). To put data from these studies on the same scale (studies varied in the magnitude of limitation), we first adjusted each day's response by the maximum response observed over the sampled time period, and then averaged across the studies for each day of the experiments.

Data from D'Elia et al. (1986), as well as the aggregated data set from all seven papers with time-series data, demonstrated three notable features (Fig. 1d and f): (1) Δr, as expected, appeared to converge toward 0 at relatively long time scales ($\geq$12 d); (2) At shorter time scales, on the order of 2–7 d, Δr was relatively independent of time, while the log ratio was strongly time dependent; and (3) There was an initial ($\leq$1 d) lag in algal response (i.e., Δr at day 1 is less than Δr on days 2–7), which may reflect a physiological lag between nutrient uptake and conversion to new biomass. Given that algal division rates under good conditions are on the order of once per day, an initial 1-d lag is not surprising. Despite this slight lag, the Experimental and Control treatments diverged from one another at an approximately exponential rate for the first 2–7 d (Fig. 1C and E, i.e., except for the inflection at day 1, the log ratio showed an approximately linear increase up to day 5 or so, which led to the relative constancy in Δr over the same time period—from roughly day 2 through day 7). After ~7 d the direct effects of nutrient limitation on algal growth was counteracted by other factors (e.g., numerical responses of micro-grazers), which caused Δr to decline. This analysis suggests that studies lasting $\leq$1 d are too short to quantify limitation (due to the time lag in the numerical response of algae), and studies lasting more than ~7 d are too long (due to feedbacks involving other components of the system).

The restricted data set

Based on the preliminary analyses of time scale, we restricted the final meta-analysis to estimates of Δr derived from reports of phytoplankton biomass in enriched and control treatments collected 2–7 d after nutrient addition. When data were reported for multiple sample dates within this 6-d period, an average Δr was calculated for the study. Unfortunately, this restriction on duration meant that almost half of the available experiments (48%) could not be used because they either lasted $\leq$1 d, or they were not sampled during the first week of a long-term enrichment experiment. This severe selection criterion is justified given our conceptual definition of limitation and the errors that could result from inclusion of studies that did not measure nutrient limitation as defined here (e.g., see Fig. 1).

We also excluded studies that reported data that could not easily be incorporated into our quantitative definition of limitation (Eqs. 1 and 2). For example, some studies provided estimates of total plankton production rates (e.g., ^{14}C assimilation, oxygen evolution, expressed at the level of the entire assemblage), after several days of incubation with nutrients, rather than biomasses or abundances of cells. These studies give biased (over-) estimates of limitation because they "double count" the effect of nutrient limitation. Total production reflects both the enhancement of per cell (or per unit biomass) production rates stimulated by nutrient addition, as well as the increased biomass of phytoplankton in the vessel, which had accumulated over the course of the experiment. Rates of change in production rates will therefore be greater than rates of change in biomass, so incorporating both sources into a quantitative estimate would essentially "double count" the effect of nutrient limitation.

Many studies also included treatments that consisted of the addition of multiple nutrients, most commonly N plus P. The most extreme form of this experiment (often termed "nutrient-deletion" experiments) consisted of the addition of a vast suite of nutrients and then the sequential removal of single nutrients from this enrichment solution. These deletion experiments assessed the response of the phytoplankton assemblage to the omission of a particular nutrient, when all other nutrients were present in excess (e.g., Paerl and Bowles 1987). We excluded all mixed nutrient results because they did not measure the magnitude of nutrient stress due to a single nutrient—they altered the background nutrient environment in addition to the availability of the target nutrient. Furthermore, although mixed algal assemblages can show co-limitation, whereas single-species systems should not (as argued in Hecky and Kilham 1988), we were most interested in the amount of limitation imposed by single nutrients, especially nitrogen, phosphorus, silicon or iron (for which we found the most data). Experiments examining co-limitation of algal assemblages will be analyzed in a future, more extensive manuscript.

Most frequently, the natural phytoplankton community was incubated in the collected sea water, but some researchers (18% of studies) filtered out the natural algal community, replacing it with a cultured algal species. In this case, the decision whether to include such studies is not obvious and meta-analysts may diverge on how they proceed (Englund et al. 1999). We chose to include these experiments (because they were always performed using indigenous, dominant algae) and examine their influence on the data set empirically. The results of these studies did not yield estimates of Δr that differed significantly overall from those obtained using only natural algal assemblages (resampling test; $P = 0.78$).

After restricting our data set to studies as defined above, we were left with data compiled from 16 different papers (Ryther and Dunstan 1971, Vince and Valiela 1973, Thómas et al. 1974, Granéli and Sundbäck 1985, Martin and Fitzwater 1985, D'Elia et al. 1986, Granéli 1987, Bonin et al. 1989, Le Rouzic and Bertru 1989, Granéli et al. 1990, Harrison et al. 1990, Pederson and Borum 1991, Rudek et al. 1991, Martin et al. 1994, Hein and Riemann 1995, Taylor et al. 1995), which provided 303 different comparisons of nutrient-addition and control treatments. For data availability see the Appendix. The large number of comparisons per paper exists because single papers usually provided data from multiple experiments (e.g., multiple sites and multiple times) or from multiple treatments (e.g., separate N, P, Fe, and Si additions in the same experiment). Therefore, our estimates are not wholly independent. The degree of non-independence is difficult to assess, however, and is likely to vary depending on the specific source of non-independence (e.g., two estimates from different sites, but reported in the same paper may well be less dependent on one another than two estimates from the same locale but different times and reported in different papers). At this point, the data set is too sparse to permit a rigorous evaluation of non-independence, so we raise this primarily as a cautionary note (e.g., see Gurevitch and Hedges 1999).

Analytical procedures

Experiments were performed in a wide variety of locations, spanning latitudes from 5° S to 60° N, in heavily polluted areas (e.g., New York City Harbor) to pristine open oceans (e.g., Atlantic and Pacific oceans), in estuaries, bays and harbors, coastal zones, sounds and nearly landlocked seas, shelf zones, and oceanic waters. We therefore coded each study based on the environment in which the study was conducted. We defined six levels of pollution based on descriptions of sites presented by the authors of the studies. ''Very polluted'' sites were those that were indicated as directly receiving polluted effluent, ''moderate pollution'' was indicated if study sites were adjacent to, but not directly within heavily polluted waters, ''light pollution'' was indicated if sites were between polluted and unpolluted waters, ''unpolluted'' waters were those where authors indicated that there was no obvious source of nutrient pollution, whereas ''pristine'' waters were distinguished as those where authors specifically indicated that analyses showed a lack of nutrient pollution. We categorized these sites as ''high N:P'' when the authors indicated the habitat received polluted waters with N:P consistently above the ratio most frequently required by phytoplankton (N:P = 16, as atoms). We then investigated variation in the response to nutrient enrichment among these different types of environments.

Tests for significant differences in responses among treatment types and 95% confidence intervals of effect sizes were made by using resampling methods and unweighted analyses (Adams et al. 1997) programmed in Meta Win 1.00 (Rosenberg et al. 1997). Although a mixed model using unequal weights would be more desirable (see Gurevitch and Hedges [1999] for further discussion of model types), a weighted analysis cannot be made without estimates of within-study sampling variability, which were not available from most of the studies. The 95% confidence intervals were generated from 5000 resampling iterations using equal weights for each study (i.e., assuming that all variances of Δr were approximately equal; J. Gurevitch, *personal communication*). This assumption is undoubtedly not strictly accurate, but does not bias estimated group means, although it probably inflates the sizes of estimated confidence intervals (Gurevitch and Hedges 1999).

Results and Discussion

Phytoplankton response to nutrient enrichment

Overall general effects of nutrient enrichment.—Additions of nitrogen or iron elicited similarly large average magnitudes of response in phytoplankton specific growth rates when data from all zones and levels of pollution were considered together (Fig. 2). The average responses to these two nutrients can also be expressed in terms of relative doubling times: on average, the nutrient-addition treatment achieved twice the algal biomass as the control in 3.3 d for nitrogen and 4.1 d for iron. Silicate also stimulated phytoplankton growth (Fig. 2), but doubling times in enriched treatments averaged nearly 10 d. In contrast, addition of phosphorus did not, on average, stimulate phytoplankton growth (Fig. 2). These results confirm the prevailing view among marine scientists that N and Fe are the two most important limiting nutrients in marine environments. Note, however, that these results give average responses, which do not address potential variation in nutrient limitation among different habitats.

Variation in limitation among different environments.—Comparisons were classified based on the amount and form of nutrient pollution present at the study site. The environmental conditions of available studies are strongly biased, with ~60–70% of all comparisons coming from studies conducted in waters we classified as ''very polluted.'' Approximately 30% of the comparisons were from moderately polluted sites,

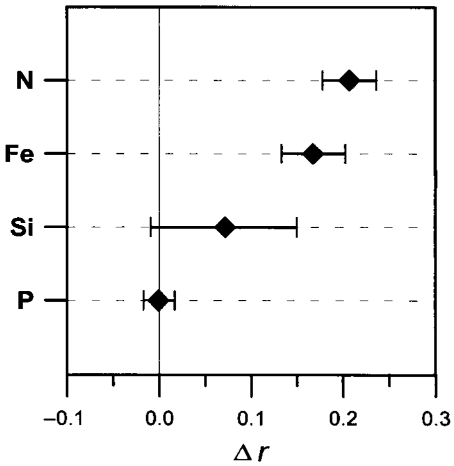

FIG. 2. Effects of nutrient addition on phytoplankton growth, as measured by Δr, the change in per unit (per gram; per unit carbon, or per chlorophyll *a*) growth rate of an algal assemblage following the addition of surplus nutrients. Results give responses for each nutrient added singly to phytoplankton assemblages and show, using an unweighted analysis with resampling procedures, that nutrients varied in their effect on phytoplankton growth ($P = 0.002$). Error bars represent 95% confidence intervals of Δr based on the resampling procedures with 5000 iterations. N denotes experiments enriching with nitrogen, P denotes phosphorus addition, Si denotes silicate addition, and Fe denotes iron addition. The means are based on 148 (N), 114 (P), 35 (Fe), and 6 (Si) experiments.

and only ~5% were from sites that were pristine or relatively unlikely to have been polluted. In contrast, most of the ocean waters on the globe receive little nutrient pollution and would be classed as "unpolluted" or "pristine" based on our classification scheme. Experiments to assess limitation by specific nutrients also have not been performed uniformly in all nutrient environments (Fig. 3). For example, >75% of the experimental nitrogen enrichments have been performed in very polluted or moderately polluted waters, while >90% of the experiments on iron limitation have been performed in "pristine" habitats. This biased distribution of studies has important effects on the results of the overall meta-analysis (i.e., Fig. 2).

The magnitude of response to nutrient enrichment varied significantly among marine environments (Fig. 4). Phosphorus limitation was high (nearly as high as the average effects of nitrogen and iron limitation) in pristine, unpolluted waters (probably high in N:P; Downing 1997) and those polluted with nutrients with a high N:P ratio (higher than the average ratio in plankton tissue). Further, experiments performed on waters taken from geographic areas unlikely to receive nutrient pollution (e.g., unenclosed coastal zones and those in open oceans) show that responses to P addition were significantly positive and differed significantly from the negligible responses to P enrichment in estuaries and enclosed bays ($P = 0.011$; Fig. 5). Since much of the world's oceans are relatively free of nutrient pollution and are far from coastal zones, phosphorus may play a greater role in nutrient limitation than is currently believed.

Nitrogen limitation tended to show the opposite pattern across the pollution gradient (Fig. 4), although limitation did not vary significantly among environments receiving differing degrees of nutrient pollution (Fig. 4; $P = 0.19$) or among habitats (e.g., Fig. 5; *P*

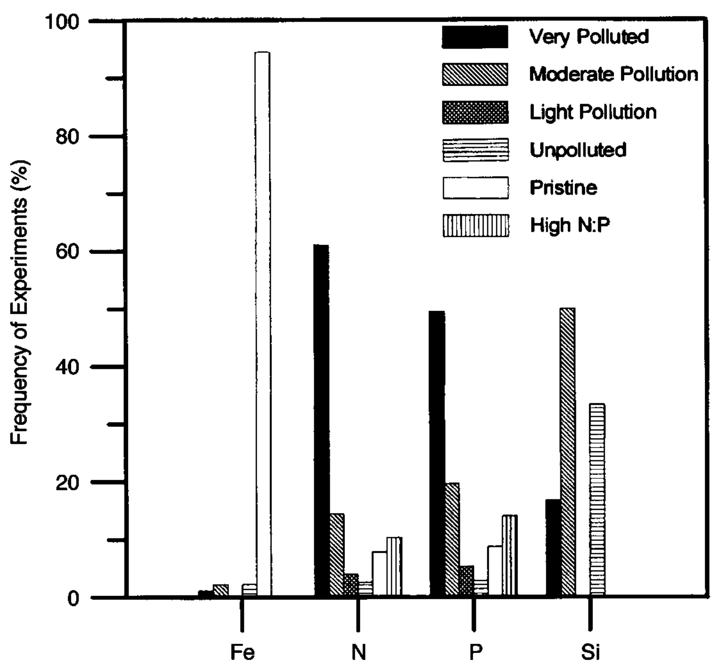

FIG. 3. Distribution of different kinds of nutrient-enrichment experiments performed in environments differing in probable degree of nutrient pollution. Total sample sizes are as in Fig. 2. Percentages are calculated over all experiments performed using each element; thus, e.g., "silica" experiments sum to 100% across all pollution categories.

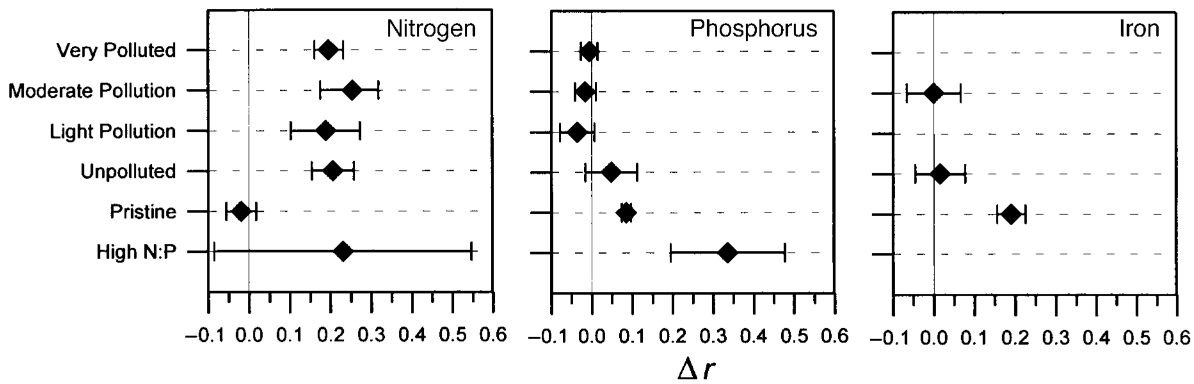

FIG. 4. Variation in nitrogen, phosphorus, and iron limitation among sites that differed in the degree of nutrient pollution. Pollution categories were assigned based on data presented in published reports (see *Methods and analyses: Analytical procedures*). Error bars give 95% confidence intervals of Δr from resampling procedures (5000 iterations). For N ($P = 0.19$), the means are based on 98 (very polluted), 37 (moderately polluted), 5 (lightly polluted), 4 (unpolluted), 2 (pristine), and 2 (high N:P) experiments; for P ($P = 0.0002$), the means are based on 64 (very polluted), 37 (moderately polluted), 5 (lightly polluted), 4 (unpolluted), 2 (pristine), and 2 (high N:P) experiments; and for Fe ($P = 0.0008$), the means are based on 2 (moderately polluted), 2 (unpolluted), and 30 (pristine) experiments. P values are probability estimates (from resampling tests) that experiments performed in different nutrient environments were sampled from identical distributions (e.g., with equal values of Δr).

= 0.16). Significant differences in nitrogen limitation among nutrient environments were somewhat masked by the highly variable responses found in waters that were extremely high in their N:P ratio (Fig. 4).

Although based on a much smaller sample size, the response to Fe enrichment also varied across habitats. Iron enrichment appeared to have little influence on phytoplankton growth in moderately polluted or unpolluted habitats, but yielded significantly ($P = 0.0008$) greater increases in growth in pristine habitats (Fig. 4). Likewise, whereas Fe had little impact on coastal phytoplankton, it increased algae growth significantly ($P = 0.0002$) in oceanic waters and at similar rates to those seen for N enrichment in inshore and polluted waters.

These responses to nutrient enrichment generally agree with biogeochemical hypotheses concerning differences in nutrient limitation among marine environments (Downing 1997). This analysis also reveals a very strong tendency for marine-nutrient bioassays to be performed primarily in polluted or land-impacted ecosystems that may not be representative of marine ecosytems as a whole. This meta-analysis thus suggests that inferences about the potential impact of nutrient enrichment on marine production or the global carbon cycle must be tempered with an understanding of the

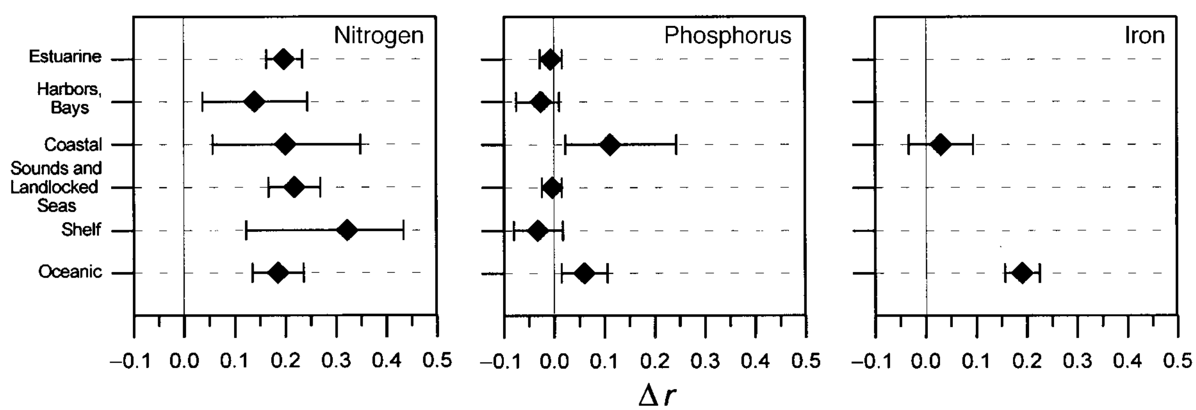

FIG. 5. Variation in nitrogen, phosphorus, and iron limitation among different marine habitats. Marine habitat classes were discerned from data and information presented in published reports. Error bars give 95% confidence intervals of Δr from resampling procedures (5000 iterations). For N ($P = 0.16$), the means are based on 92 estuarine, 10 harbors and bays, 8 coastal zones, 24 sounds and landlocked seas, 12 shelf zones, and 2 oceanic experiments; for P ($P = 0.011$), the means are based on 59 estuarine, 9 harbors and bays, 8 coastal zones, 24 sounds and landlocked seas, 12 shelf zones, and 2 oceanic experiments; and for Fe ($P = 0.0002$), the means are based on 3 coastal zones and 30 oceanic experiments. P values are probability estimates (from resampling tests) that experiments performed in different habitats were sampled from identical distributions (e.g., with equal values of Δr).

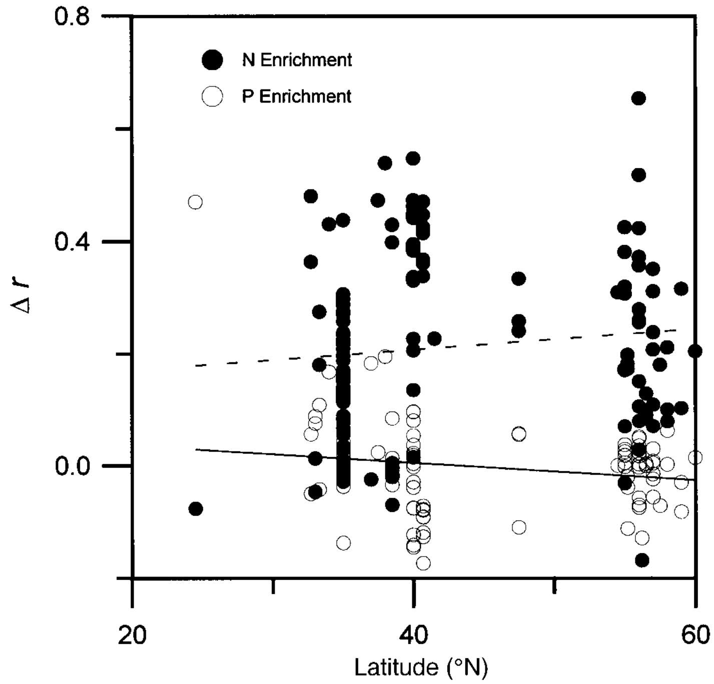

FIG. 6. Relationship between Δr in all nitrogen- and phosphorus-enrichment experiments and the latitude of the marine habitat from which water samples were taken. The dashed and solid lines represent (respectively) the least-squares regression relationship of the weak positive correlation between Δr and latitude in N-enrichment experiments ($r = 0.11$, $n = 147$, $P = 0.18$) and the relationship of the significant negative correlation between Δr and latitude in P-enrichment experiments ($r = -0.19$, $n = 114$, $P = 0.042$).

sensitivity of experimental results to the ambient nutrient environment.

Latitudinal patterns in experimental results.—It has been suggested by several authors that tropical marine ecosystems, especially those dominated by carbonate-rich sedimentary environments, are more frequently phosphorus limited than temperate ones (Smith 1984, Short et al. 1985, Fouqurean et al. 1993, Feller 1995). One might therefore hypothesize that Δr for P-enrichment experiments might be negatively correlated with latitude, that is, be highest in the low-latitude tropics, and lowest in the temperate zone. Conversely, Δr for N-enrichment experiments might be lower in experiments performed at low latitudes than in experiments performed at high latitudes. Although regression and correlation analyses of meta-analytical data are somewhat controversial when one cannot weight by appropriate variance estimates (Cooper and Hedges 1994, Hedges and Olkin 1985, Gurevitch and Hedges 1999), the correlations shown in Fig. 6 support this theory. There is a significant ($P = 0.042$) negative correlation between Δr and latitude in P-enrichment experiments, and a very weak positive relationship ($P = 0.18$) between Δr and latitude in N-enrichment experiments (Fig. 6). The correlations are probably weak due to the influences of a variety of other variables that we have not accounted for (such as seasonality and degree of pollution [Fig. 4]). It is encouraging, however, that any trend is seen in this preliminary analysis. Tests of hypotheses of this type would be extremely costly if they required newly collected primary data. Thus, meta-analysis, or comparative analyses, of existing data can greatly facilitate tests of important hypotheses that might not otherwise be feasible.

Conclusions

Meta-analytical methods offer powerful instruments for the interrogation of experiments performed under disparate conditions and can lead to the exploration of large-scale trends in responses reflecting differences in the functioning of ecosystems. Interpretation of the large number of analyses of marine nutrient limitation was facilitated by the choice of an appropriate metric reflecting the response of interest and the discovery of technical biases that can mask trends in experimental results.

Our exploratory and illustrative analysis revealed several general lessons regarding the application of meta-analysis in ecology. For example, most of the available studies did not report or allow calculation of variances among replicates, either because there were no replicates, or because variances were not included in publications. In previous ecological meta-analyses, such data sets would have been discarded (e.g., Gurevitch et al. 1992, Curtis 1996). We were, nonetheless, able to gain valuable insights about the strength of nutrient limitation and its variation across systems because (1) we used a metric (i.e., Δr) that did not require estimates of variance, and (2) we did not mandate that effect sizes be weighted by the inverse of their within-study variances (e.g., see Gurevitch and Hedges 1999). Although the absence of weights might have led to some bias in our calculation of confidence intervals (Gurevitch and Hedges 1999) and a reduction in statistical power, we believe it is far better to obtain approximate answers to important questions, than to remain ignorant of the answers because the data might be judged less than ideal.

Finally, our analyses demonstrated the value in conducting preliminary analyses to ascertain the relevance of particular data (e.g., collected over different time scales) and in performing exploratory analyses that investigate possible sources of variation in effect size. Indeed, the variation in nutrient limitation among different marine habitats has conceptual implications that affect how we apply literature reviews to the global oceanic system. In particular, very few nutrient-enrichment bioassays could be found that examined nutrient limitation in unpolluted open oceans. As a result, the average estimates of nutrient limitation were heavily biased toward polluted coastal systems, which are often the most easily studied. This has probably led to bias in our inferences about the relative importance of various nutrients in limiting production in marine systems, and its impact on major issues such as marine resources and global carbon budgets. Our analyses not only suggested the nature of this bias, but also pointed out holes in the existing data that should help direct future primary investigations. Like many other important questions that ecologists test using experimental data, the salient questions to pose concerning nutrient limitation in marine habitats are clearly more interesting than asking ''is the ocean limited by N, or is it P?''

ACKNOWLEDGMENTS

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LITERATURE CITED

Adams, D. C., J. Gurevitch, and M. S. Rosenberg. 1997. Resampling tests for meta-analysis of ecological data. Ecology **78**:1277–1283.

Barnes, R. S. K., and R. N. Hughes. 1988. An introduction to marine ecology. Blackwell, Oxford, UK.

Bender, E. A., T. J. Case, and M. E. Gilpin. 1984. Perturbation experiments in community ecology: theory and practice. Ecology **65**:1–13

Bonin, D. J., M. C. Bonin, and T. Berman 1989. Mise en évidence expérimentale des facteurs nutritifs limitants de la production du micro-nanoplancton et de l'ultraplancton dans une eau cotière de la Méditerranée orientale (Haifa, Israel). Aquatic Sciences **51**:129–152.

Boynton, W. R., W. M. Kemp, and C. W. Keefe. 1982. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. Pages 69–70 *in* V. S. Kennedy, editor. Estuarine comparisons. Academic Press, New York, New York, USA.

Capone, D., and E. J. Carpenter. 1982. Nitrogen fixation in the marine environment. Science **217**:1140–1142.

Cooper, H., and L. V. Hedges. 1994. The handbook of research synthesis. Russell Sage Foundation, New York, New York, USA.

Curtis, P. S. 1996. A meta-analysis of leaf gas exchange and nitrogen in trees grown under elevated carbon dioxide. Plant, Cell and Environment **19**:127–137.

D'Elia, C. F., J. G. Sanders, and W. R. Boynton. 1986. Nutrient enrichment studies in a coastal plain estuary: phytoplankton growth in large-scale, continuous cultures. Canadian Journal of Fisheries and Aquatic Sciences **43**:397–406.

Downing, J. A. 1997. Marine nitrogen:phosphorus stoichiometry and the global N:P cycle. Biogeochemistry **37**:237–252.

Elser, J. J., E. R. Marzolf, and C. R. Goldman. 1990. Phosphorus and nitrogen limitation of phytoplankton in freshwaters of North America: a review and critique of experimental enrichments. Canadian Journal of Fisheries and Aquatic Sciences **47**:1468–1477.

Englund, G., O. Sarnelle, and S. Cooper. 1999. The importance of data-selection criteria: meta-analyses of stream predation experiments. Ecology **80**:1132–1141.

FAO [Food and Agriculture Organization]. 1993. Fishery statistics yearbook. Volume 72. Rome, Italy.

Feller, I. C. 1995. Effects of nutrient enrichment on growth and herbivory of dwarf red mangrove (*Rhizophora mangle*). Ecological Monographs **65**:477–505.

Fourqurean, J. W., R. D. Jones, and J. C. Zieman. 1993. Processes influencing water column nutrient characteristics and phosphorus limitation of phytoplankton biomass in Florida Bay, FL, USA: inferences from spatial distributions. Estuarine Coastal and Shelf Science **36**:295–314.

Granéli, E. 1987. Nutrient limitation of phytoplankton biomass in a brackish water bay highly influenced by river discharge. Estuarine, Coastal and Shelf Science **25**:555–565.

Granéli, E., and K. Sundbäck. 1985. The response of planktonic and microbenthic algal assemblage to nutrient enrichment in shallow coastal waters, southwest Sweden. Journal of Experimental Marine Biology and Ecology **85**: 253–268.

Granéli, E., K. Wallstrom, U. Larsson, W. Granéli, and R. Elmgren. 1990. Nutrient limitation of primary production in the Baltic Sea area. Ambio **19**:142–151.

Gurevitch, J., and L. V. Hedges. 1999. Statistical issues in ecological meta-analyses. Ecology **80**:1142–1149.

Gurevitch, J., L. L. Morrow, A. Wallace, and J. S. Walsh. 1992. A meta-analysis of competition in field experiments. American Naturalist **140**:539–572.

Harrison, P. J., M. H. Hu, Y. P. Yang, and X. Lu. 1990. Phosphate limitation in estuarine and coastal waters of China. Journal of Experimental Marine Biology and Ecology **140**:79–87.

Hecky, R. E., and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: a review of recent evidences of the effects of enrichment. Limnology and Oceanography **33**:796–822.

Hedges, L. V., J. Gurevitch, and P. Curtis. 1999. The meta-analysis of response ratios in experimental ecology. Ecology **80**:1150–1156.

Hedges, L. V., and I. Olkin. 1985. Statistical methods for meta-analysis. Academic Press, New York, New York, USA.

Hein, M., and B. Riemann. 1995. Nutrient limitation of phytoplankton biomass or growth rate: an experimental approach using marine enclosures. Journal of Experimental Marine Biology and Ecology **188**:167–180.

Jacques, G., and P. Tréguer. 1986. Écosystèmes pélagiques marins. Masson, Paris, France.
Laska, M. S., and J. T. Wootton. 1998. Theoretical concepts and empirical approaches to measuring interaction strength. Ecology **79**:461–476.
Le Rouzic, B., and B. G. Bertru. 1989. Détermination par bioessai de la biodisponibilité des ressources azote et phosphore, dans les eaux du Golfe du Morbihan. Revue des Sciences de l'Eau **5**:97–111.
Mackenzie, F. T., L. M. Ver, C. Sabine, M. Lane, and A. Lerman. 1993. C, N, P, S global biogeochemical cycles and modeling of global change. Pages 1–61 *in* R. Wollast, F. T. Mackenzie, and L. Chou, editors. Interactions of C, N, P and S, biogeochemical cycles and global change. Springer-Verlag, Berlin, Germany.
Martin, J. H. et al. 1994. Testing the iron hypothesis in ecosystems of the equatorial Pacific Ocean. Nature **371**:123–129.
Martin, J. H., and S. E. Fitzwater. 1988. Iron deficiency limits phytoplankton growth in the northeast pacific subarctic. Nature **331**:341–343.
Meybeck, M. 1982. Carbon, nitrogen, and phosphorus transport by world rivers. American Journal of Science **282**:401–450.
Osenberg, C. W., and G. G. Mittelbach. 1996. The relative importance of resource limitation and predator limitation in food chains. Pages 134–148 *in* G. A. Polis and K. O. Winemiller, editors. Food webs: integration of patterns and dynamics, Chapman & Hall, New York, New York, USA.
Osenberg, C. W., O. Sarnelle, and S. D. Cooper. 1997. Effect size in ecological experiments: the applicaton of biological models to meta-analysis. American Naturalist **150**:798–803.
Osenberg, C. W., O. Sarnelle, S. D. Cooper, and R. D. Holt. 1999. Resolving ecological questions through meta-analysis: goals, metrics, and models. Ecology **80**:1105–1117.
Oviatt, C., P. Doering, B. Nowicki, L. Reed, J. Cole, and J. Frithsen. 1995. An ecosystem level experiment on nutrient limitation in temperate coastal marine environments. Marine Ecology Progress Series **116**:171–179.
Paerl, H. W., and D. Bowles. 1987. Dilution bioassays: their application to assessments of nutrient limitation in hypereutrophic waters. Hydrobiologia **146**:265–273.
Pedersen, M. F., and J. Borum. 1996. Nutrient control of algal growth in estuarine waters. Nutrient limitation and the importance of nitrogen requirements and nitrogen storage among phytoplankton and species of macroalgae. Marine Ecology Progress Series **142**:261–272.
Redfield, A. C. 1934. On the proportions of organic derivatives in sea water and their relation to the composition of plankton. Pages 176–192 *in* James Johnstone Memorial Volume. University Press, Liverpool, UK.
———. 1958. The biological control of chemical factors in the environment. American Scientist. **46**:205–221.
Redfield, A. C., B. H. Ketchum, and F. A. Richards. 1963. The influence of organisms on the composition of sea-water. Pages 26–77 *in* M. N. Hill, editor. The sea. Volume 2. John Wiley, New York, New York, USA.
Ritschard, R. L. 1992. Marine algae as a CO_2 sink. Water, Air, and Soil Pollution **64**:289–303.
Rosenberg, M. S., D. C. Adams, and J. Gurevitch. 1997. MetaWin. Statistical software for conducting meta-analysis: fixed effects models, mixed effects models, and resampling tests. Version 1.0. Sinauer Associates, Sunderland, Massachusetts, USA.
Rudek, J., H. W. Paerl, M. A. Mallin, and P. W. Bates. 1991. Seasonal and hydrological control of phytoplankton nutrient limitation in the lower Neuse River Estuary, North Carolina. Marine Ecology Progress Series **75**:133–142.
Ryther, J. H., and W. M. Dunstan. 1971. Nitrogen, phosphorus, and eutrophication in the coastal marine environment. Science **171**:1008–1013.
Schaffer, W. M. 1981. Ecological abstraction: the consequences of reduced dimensionality in ecological models. Ecological Monographs **51**:383–401.
Schindler, D. W. 1978. Factors regulating phytoplankton production and standing crop in the world's freshwaters. Limnology and Oceanography **23**:478–486.
Short, F. T., M. W. Davis, R. A. Gibson, and C. F. Zimmermann. 1985. Evidence for phosphorus limitation in carbonate sediments of the seagrass *Syringodium filiforme*. Estuarine, Coastal and Shelf Science **20**:419–430.
Smith, S. V. 1984. Phosphorus vs. nitrogen limitation in the marine environment. Limnology and Oceanography **29**:1149–1160.
Taylor, D., S. Nixon, S. Granger, and B. Buckley. 1995. Nutrient limitation and the eutrophication of coastal lagoons. Marine Ecology Progress Series **127**:235–244.
Thomas, H. W., D. L. R. Seibert, and A. N. Dodson. 1974. Phytoplankton enrichment experiments and bioassays in natural coastal sea water and in sewage outfall receiving waters off southern California. Estuarine and Coastal Marine Science**2**:191–206.
Valiela, I. 1995. Marine ecological processes. Springer-Verlag, New York, New York, USA.
Vince, S., and I. Valiela. 1973. The effects of ammonium and phosphate enrichments on chlorophyll a, pigment ratio, and species composition of phytoplankton of Vineyard Sound. Marine Biology **19**:69–73.
Vitousek, P. M., and R. W. Howarth. 1991. Nitrogen limitation on land and in the sea: how can it occur? Biogeochemistry **13**:87–115.

APPENDIX

Marine nutrient-enrichment data from 16 different papers (published dates: 1971–1995) and used in this paper are available in digital form from ESA's Electronic Data Archive: *Ecological Archives* E080-009. The data are also available from the data depository at the National Center for Ecological Analysis and Synthesis (NCEAS), Santa Barbara, California, USA.

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Phosphorus versus nitrogen limitation in the marine environment[1]

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Abstract

Limnological and marine geochemical opinion favors phosphorus limitation of organic production in aquatic environments, while marine biological opinion favors nitrogen limitation. Clues in the literature and nutrient budgets for selected marine ecosystems suggest that phosphorus vs. nitrogen limitation is a function of the relative rates of water exchange and internal biochemical processes acting to adjust the ratio of ecosystem N:P availability.

A limiting factor to biological activity is that material available in an amount most closely approaching the critical minimum required to sustain that activity (Odum 1971). This definition can be applied at any scale from cellular metabolism to global biogeochemical cycles. This paper deals with inorganic plant nutrients as limiting factors for the net production of new organic material in marine systems.

Marine geochemists and biologists hold antithetical views about nutrient limitation in the ocean. The view held by most marine geochemists (e.g. Lerman et al. 1975; Meybeck 1982; Broecker and Peng 1982) can apparently be traced to the seminal paper by Redfield (1958). Redfield concluded that phosphorus availability limits net organic production in the sea. He pointed out that any nitrogen deficits can be met by the biological fixation of atmospheric nitrogen, hence nitrogenous compounds can accumulate until the available phosphorus is utilized.

Among marine biologists, Redfield's view has largely been replaced with the view expressed by Ryther and Dunstan (1971) that nitrogen, not phosphorus, is the limiting factor to algal growth in coastal waters. Those workers (p. 1008) accepted the possibility that nitrogen fixation might "be important in regulating the level or balance of nutrients in the ocean as a whole and over geological time," but they concluded: "It (nitrogen fixation) is certainly not effective locally or in the short run."

Similarly, Thomas (1970*a,b*) and many subsequent workers have relied on experimental cultures of phytoplankton to evaluate nutrient limitation in the marine environment (*see also* Goldman et al. 1979). With a few exceptions (e.g. Myers and Iverson 1981) nitrogen appears to be the nutrient which is most often limiting to the specific growth rate of natural populations of phytoplankton grown in such cultures.

A further antithesis emerges. In contrast to the marine biologists, biological limnologists now generally subscribe to the view that phosphorus availability is ordinarily the primary limit to net organic carbon pro-

[1] Contribution 690 of the Hawaii Institute of Marine Biology.

duction in lakes. This view is exemplified by the phosphorus loading models put forth by Vollenweider (1968, 1969, 1976), by the long term monitoring studies conducted by Edmondson and his colleagues on Lake Washington (e.g. Edmondson 1970; Edmondson and Lehman 1981), and by the elegant whole-lake nutrient enrichment experiments summarized by Schindler (1971, 1974, 1977). In some recent analyses of lakes (e.g. Smith 1982) nitrogen is considered to have a significant, but secondary, effect on net production.

Schindler's work is particularly important, because he has explicitly addressed the need to assess questions about ecosystem nutrient limitation at the scale of ecosystems, not just at the scale of individual bottle incubation experiments or other studies of isolated components of the ecosystem. Factors limiting the growth rate of individual organisms should be distinguished from factors limiting the net production rate of entire ecosystems. Conclusions about ecosystems based on single-component studies persist, nevertheless.

We are left with the following questions: Can the geochemical vs. the biological views about nutrient limitation in the ocean be resolved into some kind of single model? What, if anything, is the inherent difference between nutrient limitation in lakes and in the ocean? I will suggest some general clues as to the resolution of these questions, and then I will present some of my own data which I believe further help to resolve these questions.

In a recent summary article on lacustrine nutrient limitation and its contrast with nutrient limitation in estuaries, Schindler (1981, p. 78) stated: "The 'evolution' of optimal ratios for phytoplankton growth . . . may not be able to occur in estuaries due to their short water residence times and domination by physical processes. Nitrogen fixed from the atmosphere or returned from sediments is swept into the open ocean so rapidly that it cannot accumulate to the degree which is common in freshwater lakes."

Schindler's comment is a direct outgrowth of whole-lake models (e.g. Vollenweider 1976; Schindler et al. 1978; Dillon 1975) which consider mass loading rates, water renewal times, depth, and various parameters of biomass response. The statement clearly implies that the "discrepancy" between phosphorus limitation in lakes and nitrogen limitation in the coastal marine environment lies with the difference between relative rates of biochemical reactions of nitrogen and water exchange in the environment. Water exchange may be fast relative to internal nitrogen fluxes in coastal marine ecosystems.

Broecker and Peng (1982) pointed out that competing, biologically mediated nitrogen fixation and fixed nitrogen loss reactions in the ocean interact with terrestrial nutrient inputs, oceanic circulation, and sedimentation, and tend to push the N:P ratio of dissolved inorganic and particulate organic nutrient cycling between surface and deep water toward a "geochemically balanced" ratio—the Redfield ratio (atomic N:P ratio = 16:1). Because phosphorus is not exchanging between the ocean and an atmospheric reservoir as nitrogen does, the delivery of phosphorus—not nitrogen—limits net production (and sedimentation) of organic material in the ocean as a whole.

Based on estimates of the rate of N_2O production from NO_3^-, Broecker and Peng argued that the upper limit on the time needed to balance the N:P ratio in the oceans as a whole is about 10^6 yr; they further pointed out that the real adjustment time is likely to be $\ll 10^6$ yr and is unknown. They did not attempt to assess global nitrogen fixation and fixed nitrogen loss rates in order to resolve the adjustment time any more precisely. Because a million years is relatively short geologically, an uncertainty of this magnitude is not entirely unsatisfactory to geochemists. However, most ecologists or environmental managers have trouble dealing seriously with 10^6 yr, and water exchange rates in shoal-water marine ecosystems are obviously very much shorter than that.

The caution with which these turnover times are given by Broecker and Peng is appropriate. The loss of fixed nitrogen back to gaseous nitrogen (N_2, N_2O) may arise from either denitrification or nitrification, each with its own particular biochemical pathways and constraints (Cohen and Gordon

1978; Hashimoto et al. 1983; Webb 1981; Hattori 1982). The constraints involve availability of nitrogen species, oxygen, and light, each of which can show microenvironmental variations not readily amenable to large-scale budgets. From the vantage of such budgets, the pathways do not really matter, but these alternate pathways can affect calculation of turnover times from N_2O data. Unless I am specifically referring to a particular biochemical pathway, I will use the noncommittal term "fixed nitrogen loss."

Ecologists turn by default to the statement by Ryther and Dunstan (1971) that nitrogen fixation is not really of local relevance in the marine environment on an ecologically meaningful time scale. That conclusion is certainly not disproven by the estimates of Broecker and Peng (1982), and Schindler's (1981) comments further help to put the subject into the perspective of one class of marine environments—estuaries.

If direct data are not available, then nutrient (usually nitrogen) limitation is often inferred from a deviation of N:P concentration or loading ratios from the composition of primary producers in the system (*see* e.g. Jaworski 1981). Is there any suggestion that the rate of internal N:P adjustment in the marine environment relative to N:P delivery and export ratios might ever be sufficiently rapid to be directly relevant to the understanding of individual ecosystems in the ocean?

Myers and Iverson (1981) implicated phosphorus, rather than nitrogen, as the nutrient most limiting to phytoplankton growth in estuaries along the northeastern margin of the Gulf of Mexico. McComb et al. (1981) have suggested that an estuarine system in Western Australia may shift seasonally between nitrogen and phosphorus limitation. Are these localized anomalies, or can we find some more general clues as to the rate of N:P adjustment in marine systems?

Martinez et al. (1983) argued that nitrogen fixation in the open ocean has been seriously underestimated because of experimental artifacts associated with incubation procedures. Moreover, there is evidence (reviewed by Doremus 1982) that the metabolism of nitrogen-fixing organisms can itself be limited by the availability of phosphorus. Dominance by nitrogen-fixing organisms in lakes with low N:P ratios has been well documented (Schindler 1977; Flett et al. 1980; V. H. Smith 1983). These observations suggest that nitrogen fixation—hence nitrogen availability—in the marine environment can be high but that it can be regulated by the availability of phosphorus.

Phosphorus and nitrogen budgets for three ecosystems which I have investigated provide further insight into this subject. These studies were not initially designed to test hypotheses about nitrogen vs. phosphorus limitation. Rather, the ideas have arisen from retrospective analyses of available data sets. Consequently, the analyses presented here do not constitute clean, well designed, repeated experiments. In a sense, these shortcomings are also a major value of the data sets. Despite the fact that these studies were not initially designed to test hypotheses about phosphorus vs. nitrogen limitation of net ecosystem production of organic material, the general conclusions that emerge seem inescapable.

Ecosystem nutrient budgets

The three sites discussed here can all be loosely described as "embayments" with one relatively restricted passage through which water and material exchange with the ocean occurs. These sites differ from "positive estuaries" in having no significant input of water or other materials from land, hence no advective throughput of materials. All three sets of nutrient budgets are tied to water, salt, and carbon budgets discussed in the references cited. Three kinds of data enter into the calculations of nutrient flux: the net oceanic delivery of materials, the immediate molar uptake ratio of N:P in primary producers, and the long term net molar uptake ratio of the sediments.

Oceanic delivery is established by assuming that salinity and water volume in each system remain constant through time: salt advection, driven by net water flow to or from the system (the direct result of rainfall minus evaporation), is balanced by salt diffusion down the concentration gradient. As developed by Atkinson and Smith (1983), the deviation of any material from a strict

Table 1. Generalized procedure for calculating nutrient fluxes. Details for each system are discussed in the text.

Flux	Symbol	Procedure
Phosphorus		
Oceanic delivery	O_p	$= f$(salt, water budgets)
Net uptake	P	$= O_p$
Nitrogen		
Oceanic inorganic N delivery	O_{din}	$= f$(salt, water budgets)
Oceanic organic N export	O_{don}	$= f$(salt, water budgets)
Immediate uptake	N_i	$= P \times$ biotic N:P
Long term uptake	N_l	$= P \times$ sediment N:P
N fixation		
Lower limit	F_l	$= N_l - (O_{din} - O_{don})$
Upper limit	F_u	$= N_i - (O_{din} - O_{don})$
Fixed N loss	D	$< F_u - F_l$

proportional relationship with salinity is a quantitative measure of net uptake or release of that material within the system. This uptake or release is scaled to a rate function with the water budget. Ecosystem net organic production must equal or slightly exceed 0 for the biomass of an isolated ecosystem to be maintained (Smith and Atkinson 1984). Hence, we are considering only net nutrient uptake for all of these systems.

Table 1 presents the generalized procedure used for calculating nutrient fluxes. The details differ for each system and are discussed in the text. An oceanic delivery budget entirely describes net ecosystem phosphorus uptake, because there are no significant additional sources of phosphorus. The sediments and organisms are important to the internal cycling of phosphorus, but neither of these reservoirs can constitute a long term phosphorus source in a steady state system. Nor, by the choice of systems examined, is there a significant terrestrial phosphorus source.

Such oceanic budgets provide only partial records of nitrogen uptake or release within the system because of the additional pathways of gas (N_2 or N_2O) transfer across the air–water interface coupled with N_2 fixation or fixed nitrogen losses from the water column or sediments back to gaseous form. As described below, insight into these additional, nonoceanic deliveries can be derived from the discrepancies between budgetary derivations of net nitrogen and phosphorus fluxes and the N:P uptake ratios of the organisms and sediments.

The minimum amount of nitrogen that must be available for net metabolism in these isolated ecosystems is established as the net phosphorus uptake (from the oceanic delivery budget) multiplied by the molar N:P ratio in the sediments (the long term net uptake ratio of the systems). The systems in question have no significant input of fixed nitrogen from terrestrial sources, so some fraction of this minimum net nitrogen requirement must be met by the oceanic delivery budget (adjusted in two of the systems for DON export), the remainder by nitrogen fixation. This calculation defines the minimum nitrogen fixation rate which must occur. Maximum nitrogen fixation would occur if there were no internal nitrogen recycling and would be represented by the difference between the oceanic delivery and the immediate uptake requirements (from the plant molar N:P ratios multiplied by the oceanic phosphorus uptake). Maximum fixed nitrogen loss back to N_2 or N_2O is the difference between the maximum and minimum rates of nitrogen fixation.

Shark Bay—The first location under consideration is Shark Bay, a large (13,000 km^2), hypersaline embayment in Western Australia. As discussed by Smith and Atkinson (1983, 1984), organic metabolism in that ecosystem is dominated by seagrass communities and by extensive soft-bottom infaunal communities. Rates of plankton production are not known but appear to be low. Net ecosystem production of organic carbon has been estimated to be 1.2 mmol $C \cdot m^{-2} \cdot d^{-1}$ (Smith and Atkinson 1983).

Smith and Atkinson (1983) developed a detailed phosphorus budget for the bay from a salt and water budget, and Smith and Atkinson (1984) inferred a nitrogen budget from the salt, water, and phosphorus budgets, plant composition, and sediment composition. Because of the very slow water turnover in the bay (average residence time

Table 2. Material budgets of the three study sites. Relevant references are discussed in the text.

	Location		
	Shark Bay	Christmas Island	Canton Atoll
Incoming water composition (all measured)			
Salinity (‰)	36	35	36
Reactive P (mmol·m^{-3})	0.2	0.3	0.6
Inorganic N (mmol·m^{-3})	0.5	2.7	3.6
Average embayment composition (all measured)			
Salinity (‰)	46	32	38
Reactive P (mmol·m^{-3})	0.05	0.1	0.2
Inorganic N (mmol·m^{-3})	0.5	0.9	0.7
Mean water residence time (all calculated)			
Days	400	50	50
Average biotic composition			
Plankton molar N:P	16:1*	—	—
Benthos molar N:P	30.1*	27:1†	30:1‡
Average sediment composition (measured, except as noted)			
Organic C (mmol·g^{-1})	2.1	0.5	0.7
Total N (mmol·g^{-1})	0.05	0.02	0.02‡
Total P (mmol·g^{-1})	0.007	0.006	0.006‡
Sediment molar N:P	7:1	3:1	3:1‡
Net oceanic nutrient export or import (all calculated, μmol·m^{-2}·d^{-1})			
P import	4	17	52
Dissolved inorganic N import	2	100	380
Dissolved organic N export	?	220	708
Internal nitrogen reactions (all calculated, μmol·m^{-2}·d^{-1})			
Immediate N uptake	64–120	467	1,560
Long term net N uptake	28	51	156
N fixation	62–118	171–587	484–1,888
Fixed N loss	<92	<416	<1,404

* Measured.
† Calculated.
‡ Assumed.

is about 400 days), we assumed that virtually all particulate material produced in the system is sedimented there rather than exported; the sediments therefore provide a reliable record of net nitrogen and phosphorus accumulation by the system. In the case of nitrogen, this net accumulation reflects uptake of nitrogen supplied via water exchange with the ocean, plus nitrogen fixation, minus fixed nitrogen loss. The bay is surrounded by desert, and there are no significant terrestrial inputs to this system.

Table 2 summarizes the nutrient budgets, both known and inferred. Several important points emerge: Net metabolism of Shark Bay must be almost entirely dependent on internally fixed nitrogen, rather than on the delivery of nitrogen from the ocean. More than half of the gaseous nitrogen fixed into organic materials within the bay may be subsequently lost back to the gaseous phase.

The ratio of dissolved inorganic nitrogen to phosphorus concentration in water entering Shark Bay is about 2.5:1, immediately suggesting nitrogen limitation. Phosphorus concentration decreases as salinity increases with increasing net evaporation and water residence time in the bay (Fig. 1), while nitrogen concentration remains at a low (but constant) level. The steep negative slope of phosphorus as a function of salinity indicates that there will be diffusive, as well as advective, input of phosphorus (Smith and Atkinson 1983). The zero slope for nitrogen indicates that there is advective, but not diffusive, nitrogen import. Hence, the oceanic import N:P ratio (0.5:1) is lower than the concentration ratio (Table 2).

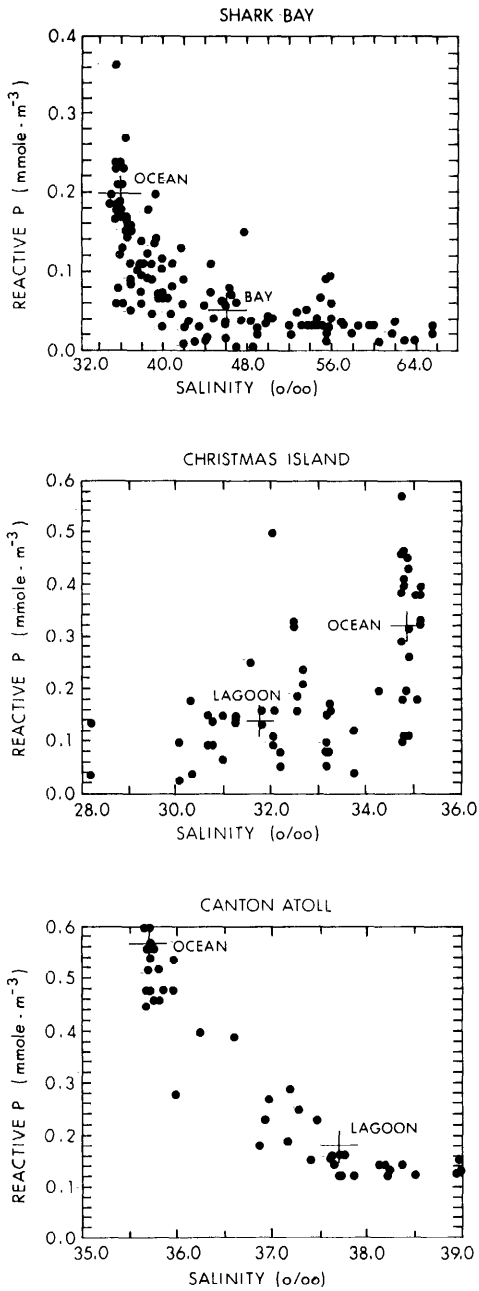

Fig. 1. Reactive phosphorus vs. salinity at Shark Bay, Christmas Island lagoon, and Canton Atoll lagoon. In all cases, ocean water enters these embayments with a salinity of 35–36‰. As water ages in Shark Bay and the Canton lagoon, it becomes more saline via net evaporation. As water ages in the Christmas lagoon, it becomes less saline through net rainfall.

Both of these ratios are very low relative to the net N:P uptake ratio indicated by plant composition. The immediate biotic uptake in the system is adjusted to a net N:P uptake ratio between about 16 and 30:1 (i.e. between the average N:P compositional ratio of plankton or benthic plants; Atkinson and Smith 1983, confirmed for Shark Bay). This adjustment occurs over an average water residence time of about 1 year, and the nitrogen made available by the adjustment is sufficient to account for the uptake of virtually all reactive phosphorus supplied to the system (Fig. 1).

The system-wide nitrogen fixation rate being invoked in this budget is reasonable in comparison with estimates of nitrogen fixation rates in seagrass communities and estuaries elsewhere. We estimated that nitrogen fixation provides $<120\ \mu mol \cdot m^{-2} \cdot d^{-1}$; Capone and Carpenter (1982) summarized data which indicate that estuaries and seagrass communities fix nitrogen at rates between 20 and 1,000 $\mu mol \cdot m^{-2} \cdot d^{-1}$.

Billen (1982) summarized available experimental data for shoal-water denitrification rates; the median value in his tabulation is about 1,000 $\mu mol \cdot m^{-2} \cdot d^{-1}$. Seitzinger et al. (1984) derived a denitrification rate of about 1,400 $\mu mol \cdot m^{-2} \cdot d^{-1}$ for Narragansett Bay, Rhode Island. That system has a relatively high organic loading (Nixon 1981), so it is not surprising that its rate of fixed nitrogen loss substantially exceeds that of Shark Bay ($<90\ \mu mol \cdot m^{-2} \cdot d^{-1}$).

Christmas Island—The second nutrient budget presented here is for the main lagoon of Christmas Island, a coral atoll in the central Pacific Ocean (Univ. Hawaii Coop. Rep. UNIHI-SEAGRANT-CR-84-02). Typical quiet-water coral reef communities and interreef soft-bottom infaunal communities dominate the organic metabolism of the system. The area of this lagoon is about 200 km^2, and water residence time averages about 50 days. Net ecosystem production of organic carbon has been estimated to be about 6 mmol $C \cdot m^{-2} \cdot d^{-1}$ (Univ. Hawaii Coop. Rep. UNIHI-SEAGRANT-CR-84-02). Results of the nutrient budgets are summarized in Table 2.

Estimates of dissolved inorganic nitrogen and phosphorus and organic phosphorus import and dissolved organic nitrogen export were derived from salt and water budgets. Again any significant terrestrial nutrient source can be dismissed. Dissolved organic nitrogen export exceeds inorganic nitrogen import, so there must be an internal nitrogen source.

As a minimum, internal nitrogen fixation must offset the export of dissolved organic nitrogen. Based on the following argument, we concluded that it was somewhat larger than this minimum. From Webb et al. (1975), we estimated that about ⅜ of the nitrogen fixed on coral reef flats is exported from the flats as dissolved organic nitrogen, and the remainder is exported as dissolved inorganic nitrogen. In the case of a relatively enclosed lagoon like Christmas Island, we assumed that any inorganic nitrogen fixed on the lagoonal reef flats and exported from them is taken up elsewhere in the lagoon, while the apparently less labile organic nitrogen is largely exported from the entire system.

Water enters the lagoon with an inorganic N:P ratio of about 9:1, perhaps suggesting nitrogen limitation. The oceanic delivery ratio adjusted for advection and diffusion is about 6:1 (Table 2). From the inferred flux pathways, the calculated N:P uptake ratio is 27:1, virtually identical with the 30:1 average compositional N:P ratio of benthic marine plants (Atkinson and Smith 1983). These budgetary calculations indicate that about 60–85% of the nitrogen being incorporated into organic material by this system is made available from internal fixation. Reactive phosphorus in the Christmas Island lagoon is depleted substantially below oceanic levels (Fig. 1), although not to the dramatic extent that occurs in Shark Bay.

Again, the nitrogen fixation rate being invoked for this system is modest in comparison with accepted rates of nitrogen fixation in similar environments elsewhere. The estimated nitrogen fixation rate is <600 $\mu mol \cdot m^{-2} \cdot d^{-1}$, in comparison with rates of up to 13,000 estimated for other coral reef flats (Wiebe et al. 1975).

Previously unpublished data on sediment nitrogen and phosphorus for the Christmas Island lagoon have been included in the calculations. These calculations suggest a fixed nitrogen loss of up to about 400 $\mu mol \cdot m^{-2} \cdot d^{-1}$. From both water turbidity and sediment composition it is obvious that a large fraction of the organic material produced in that lagoon is exported rather than being sedimented there. It is therefore entirely possible that the N:P ratio of the sedimentary materials yields a substantial overestimate of the denitrification rate of the entire system. Nevertheless I offer these preliminary estimates for eventual comparison with other system-wide fixed nitrogen loss rates. The maximum fixed nitrogen loss rate is about 70% of the estimated nitrogen fixation rate for the system.

Canton Atoll—The third site for these nutrient budgets is the lagoon of Canton Atoll, also in the central Pacific Ocean (Smith and Jokiel 1978). This reef-dominated lagoon has an area of about 40 km^2, and the average water residence time is about 50 days. After correcting the original carbon budget with a more precise estimate than that first used for net CO_2 gas flux across the air–water interface, I have estimated the net organic carbon production rate of the Canton lagoon to be 14 mmol $C \cdot m^{-2} \cdot d^{-1}$ (S. V. Smith 1983; Univ. Hawaii Coop. Rep. UNIHI-SEAGRANT-CR-84-02).

We lack data on sediment nitrogen or phosphorus or dissolved organic nitrogen for Canton. Nevertheless, the somewhat more complete budget for Christmas Island suggests that a great deal can be inferred from the expected N:P uptake ratio of benthic marine plants. These results are also summarized in Table 2.

Water enters the Canton lagoon with an inorganic N:P concentration ratio of 6:1, again perhaps suggesting nitrogen limitation. The oceanic delivery ratio is 7:1. In order that an immediate N:P uptake ratio approximating the average composition of benthic marine plants be achieved (30:1; Atkinson and Smith 1983), 1,180 $\mu mol \cdot m^{-2} \cdot d^{-1}$ of gaseous nitrogen must be fixed and incorporated with the phosphorus. As for Christmas Island, this fixation is estimated to represent ⅝ of the total fixation, the remainder being exported from the lagoon as dissolved organic nitrogen. The estimated fixation rate is still very low in comparison with rates found on coral reefs elsewhere. Reactive phosphorus in the ecosystem is largely consumed (Fig. 1), even though the input rate is much higher than it is for the other two systems.

I do not have sediment nutrient data for Canton, but I would predict that the sediment N:P ratio would again be low, reflecting fixed nitrogen loss largely balancing this higher rate of nitrogen fixation. Such an interpretation is entirely consistent with the assessment by Nixon (1981) for Narragansett Bay. If the sediment N:P ratio at Canton is assumed to equal that at Christmas, then the calculated maximum fixed nitrogen loss rate is about 70% of the calculated maximum nitrogen fixation rate.

Discussion

What has been learned from these budgets that cannot be readily derived from other ecosystem nutrient budgets? Many nutrient budgets for estuaries are dominated by advective input and output (e.g. Jaworski 1981; Monbet et al. 1981). Jaworski summarized estuarine nutrient loading rates of up to about 200 mmol $N \cdot m^{-2} \cdot d^{-1}$ and 20 mmol $P \cdot m^{-2} \cdot d^{-1}$; most of that loading is exported. In systems with high advective throughput, nutrient concentrations are likely to be nearly conservative with respect to nonreactive tracers such as salinity (Boyle et al. 1974; Biggs and Cronin 1981; Schindler 1981), and details of internal fluxes are not easily deciphered. The rate of N:P adjustment under such circumstances is—as previously discussed—slow compared to the rate of advective nutrient throughputs via water exchange.

Near-conservative behavior of nitrogen and phosphorus with respect to salinity indicates that nutrient composition is controlled primarily by the physical processes of advection and mixing, not by net biological uptake; uptake and release may introduce scatter into the data set but roughly balance one another (Imberger et al. 1983). If the net flux of both nitrogen and phosphorus is approximately conservative with respect to salinity, then neither nutrient limits net ecosystem production in any operational sense. The specific growth rate of individual organisms in the system may be limited by the local concentration of nitrogen or phosphorus, but net ecosystem production must be limited by some other factor (e.g. another nutrient, light, temperature, grazing pressure, or biomass washout).

It is also misleading to interpret low N:P concentration or loading ratios as indicative of nitrogen limitation. There may, for example, be substantial gaseous nitrogen transformation reactions (nitrogen fixation and fixed nitrogen loss) which are coupled with rapid internal turnover of nitrogen (e.g. *see* Imberger et al. 1983), effectively obscuring any net ecosystem nitrogen gains by the scatter which is ordinarily interpreted as "analytical noise." Net gains in nitrogen availability may be sufficient to deplete available phosphorus without being statistically detectable in data sets derived from such advectively dominated systems.

Isolated measurements of low (or no, or high) nitrogen fixation or denitrification in incubation chambers also do not address the question of nitrogen fixation at the scale of the ecosystem. In order to evaluate the environmental significance of nitrogen fixation and fixed nitrogen loss, we need to reconstruct what must be occurring at the scale of the ecosystem. As demonstrated here, these reconstructions become more easily achieved as advective throughputs diminish.

The confined ecosystems examined here have very slow advective throughput; water residence time in these systems ranges from almost 2 months to >1 year. Clearly the rate for adjustment of the N:P availability ratio is fast compared to these exchange rates, and these systems have become phosphorus-limited. The minimum time scale for phosphorus limitation in the marine environment remains unclear, but it is short enough to be significant to the understanding and management of marine ecosystems.

Sediment data for two of the systems that I have examined in detail and water composition data for all of them indicate that relatively little nitrogen above the oceanically imported nitrogen actually accumulates in these systems. Rather, the locally fixed nitrogen is held up in the systems long enough for fixed nitrogen loss largely to offset nitrogen fixation. Sediment data consistently suggest that most nitrogen available for community metabolic needs is subsequently lost from the sediments. Differential rates of nitrogen and phosphorus regeneration might partially account for this loss, but N:P sediment regeneration ratios that are near or below plant uptake ratios, even though particulate fallout ratios approximate that of plant uptake (Nixon 1981; Smith et al. 1981), suggest that sediment denitrification, nitrification, or both are significant loss pathways.

We thus return to the bases on which nitrogen, not phosphorus, has come to be considered the nutrient most limiting to metabolism in the marine environment. Ryther and Dunstan (1971) concluded from ambient nitrogen and phosphorus concentration data and from bottle enrichment experiments that nitrogen fixation in the ocean is not effective locally or in the short run. On the basis of the composition of phytoplankton grown experimentally under various conditions of nutrient limitation, Goldman et al. (1979, p. 215) concluded that "severe phosphorus limitation probably does not occur in the world's oceans." These arguments are reminiscent of bottle-based arguments for lacustrine nutrient limitation (e.g. the work by Lange 1970 on carbon limitation; *see* Schindler 1971 for a detailed critique).

At a global scale, these arguments are unlikely to be correct. Nutrient availability, not concentration, is most relevant to limitation of net ecosystem production. From Hattori (1982) and Capone and Carpenter (1982), I estimate that at least 1×10^{12} mol of nitrogen are fixed annually in surface ocean waters. This rate seems likely to be low (Martinez et al. 1983). Dissolved inorganic nitrogen import from streams apparently totals about 8×10^{11} mol·yr^{-1} (Meybeck 1982), so total fixed inorganic nitrogen availability apparently is at least 2×10^{12} mol·yr^{-1}. From Lerman et al. (1975), Meybeck (1982), and Froelich et al. (1982), I estimate that dissolved phosphorus input from land totals about 4×10^{10} mol·yr^{-1}. We see that the N:P availability ratio (>50:1) substantially exceeds the Redfield ratio (16:1). The net oceanic nitrogen budget is clearly dominated by internal biochemical reactions, not by inputs from land.

Arguments which have been presented to substantiate nitrogen limitation are also unconvincing at the scale of ecosystems. The bottle experiments which "demonstrate" nitrogen limitation in the marine environment can obviously exclude or deactivate major nitrogen-fixing organisms (Martinez et al. 1983), so the interpretation from these experiments of nitrogen limitation should be restricted to the limitation of the specific growth rate of the plankton most actively growing in those bottles. This aspect of nitrogen limitation does not appear particularly relevant to the understanding and management of net production in the marine environment, although it is clearly of great physiological interest.

Although apparently unproven as a phenomenon at the scale of marine ecosystems, nitrogen limitation may nevertheless occur in the ocean. If the nitrogen and phosphorus budgets for a particular ecosystem were to demonstrate system-wide nitrogen limitation, then one might look to rapid throughput of locally fixed nitrogen or perhaps to large losses of relatively refractory dissolved organic nitrogen as the most obvious explanations.

There is not an inherent difference in the characteristics of nutrient limitation between lakes and the ocean. The apparent difference lies in part with the relative rates for material exchange via the physical processes of advection and eddy diffusion and the biochemical processes of nitrogen fixation and fixed nitrogen loss. If the exchange rates are faster, then net ecosystem production of organic material may be nitrogen-limited; if the biochemical rates are faster, then the net production will tend toward phosphorus limitation.

Different analytical approaches applied to lakes and the ocean also appear to have contributed to this perceived environmental difference between lakes and the ocean. As clearly demonstrated by the various limnological studies cited (Vollenweider 1968, 1969, 1976; Schindler 1971, 1974, 1977; Schindler et al. 1978; Flett et al. 1980; Dillon 1975; Smith 1982; Edmondson 1970; Edmondson and Lehman 1981), firm con-

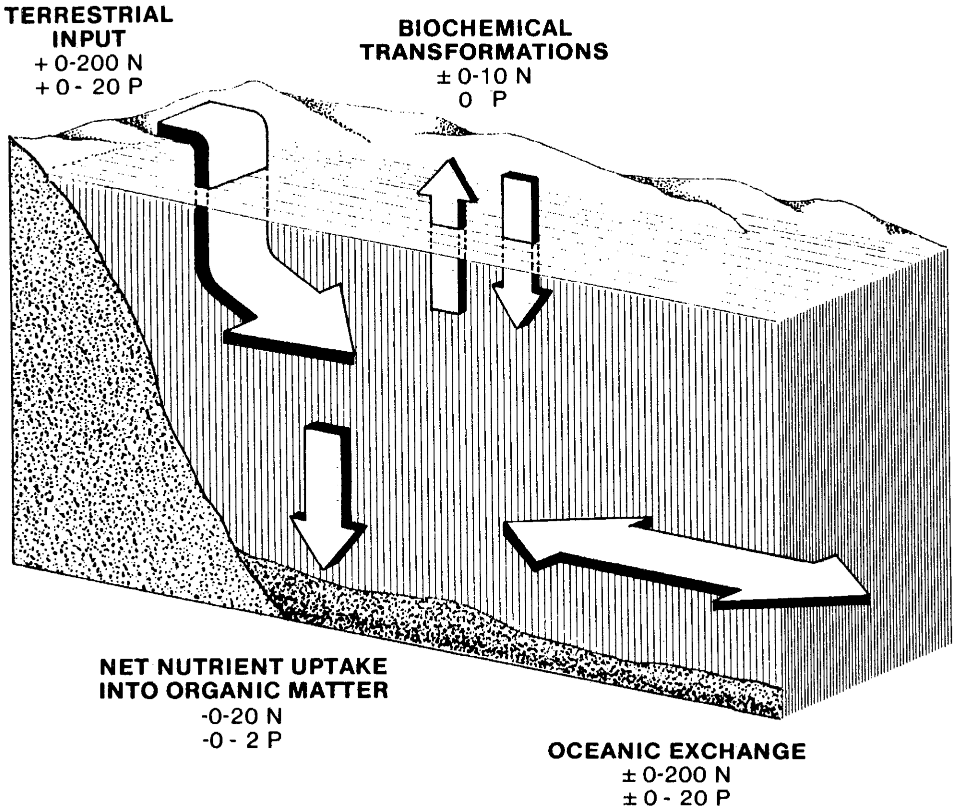

Fig. 2. Schematic diagram to illustrate the approximate ranges of nutrient fluxes (in $mmol \cdot m^2 \cdot d^{-1}$) contributing to the net production of organic matter in aquatic environments.

clusions about nutrient limitation of total-system organic carbon production should be derived at the scale of whole ecosystems rather than bottles. Each ecosystem of interest should be examined on the basis of its characteristics of nutrient supply, internal metabolic cycles, and export, in order that phosphorus vs. nitrogen limitation be assessed at this scale. The insights gained from budgeting nitrogen and phosphorus together greatly exceed those gained from single-nutrient budgets. The budgets should address the characteristics of internal biological and sedimentary nutrient pools, as well as advective and diffusive inputs and outputs. The kind of limitation being discussed—specific growth rate of taxa, primary production of communities, or net production of systems—should be carefully specified.

"Availability" is clearly a key aspect of the definition of limiting factors. Availability extends beyond the concentration and loading of materials. In the case of nutrient limitation of net ecosystem production, availability must include the ability of the ecosystem to supply materials by internal biochemical cycles.

Conclusions

I summarize my conclusions from these budgets with a general figure (Fig. 2) and equation:

Net ecosystem production of organic matter =
$R \times$ net nutrient uptake =
$R \times$ [(hydrographic nutrient additions $-$ losses) + (biochemical nutrient additions $-$ losses)].

R is the scaling ratio between the uptake of any nutrient and the desired measure of net production. Hydrographic gains and losses, via terrestrial inputs and oceanic exchange, can span a wide dynamic range; for discussion I suggest a range of 0–20 mmol $P \cdot m^{-2} \cdot d^{-1}$ and 0–200 mmol $N \cdot m^{-2} \cdot d^{-1}$. By contrast, net biochemical fluxes for phosphorus apparently do not occur and, for nitrogen, reach only about 10 mmol$\cdot m^{-2} \cdot d^{-1}$. Net ecosystem uptake of phosphorus and nitrogen is a response to nutrient availability and is constrained to equal the sum of these fluxes; but nutrient uptake is also constrained by other factors limiting net organic production. From data on net organic production rate, I estimate that the upper limit of nutrient uptake in aquatic systems is of the order of 2 mmol $P \cdot m^{-2} \cdot d^{-1}$ and 20 mmol $N \cdot m^{-2} \cdot d^{-1}$.

If hydrographic fluxes are small, then biochemical fluxes of nitrogen are likely to obviate nitrogen limitation while phosphorus is exhausted. If hydrographic fluxes are large, then neither biochemical fluxes nor hydrographic supply of either nitrogen or phosphorus is likely to limit net ecosystem production of organic matter. Other limiting factors must be sought.

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References

ATKINSON, M. J., AND S. V. SMITH. 1983. C:N:P ratios of benthic marine plants. Limnol. Oceanogr. **28:** 568–574.

BIGGS, R. B., AND L. E. CRONIN. 1981. Special characteristics of estuaries, p. 3–23. *In* B. J. Neilson and L. E. Cronin [eds.], Estuaries and nutrients. Humana.

BILLEN, G. 1982. An idealized model of nitrogen recycling in marine sediments. Am. J. Sci. **282:** 512–541.

BOYLE, E., AND OTHERS. 1974. On the chemical mass-balance in estuaries. Geochim. Cosmochim. Acta **38:** 1719–1728.

BROECKER, W. S., AND T.-H. PENG. 1982. Tracers in the sea. Eldigio.

CAPONE, D. G., AND E. J. CARPENTER. 1982. Nitrogen fixation in the marine environment. Science **217:** 1140–1142.

COHEN, Y., AND L. I. GORDON. 1978. Nitrous oxide in the oxygen minimum of the eastern tropical North Pacific: Evidence for its consumption during denitrification and possible mechanisms for its production. Deep-Sea Res. **25:** 509–524.

DILLON, P. J. 1975. The phosphorus budget of Cameron Lake, Ontario: The importance of flushing rate to the degree of eutrophy of lakes. Limnol. Oceanogr. **20:** 28–39.

DOREMUS, C. 1982. Geochemical control of dinitrogen fixation in the open ocean. Biol. Oceanogr. **1:** 429–436.

EDMONDSON, W. T. 1970. Phosphorus, nitrogen, and algae in Lake Washington after diversion of sewage. Science **169:** 690–691.

———, AND J. T. LEHMAN. 1981. The effect of changes in the nutrient income on the condition of Lake Washington. Limnol. Oceanogr. **26:** 1–29.

FLETT, R. J., D. W. SCHINDLER, R. D. HAMILTON, AND N. E. CAMPBELL. 1980. Nitrogen fixation in Canadian Precambrian Shield lakes. Can. J. Fish. Aquat. Sci. **37:** 494–505.

FROELICH, P. N., M. L. BENDER, N. A. LUEDTKE, G. R. HEATH, AND T. DEVRIES. 1982. The marine phosphorus cycle. Am. J. Sci. **282:** 474–511.

GOLDMAN, J. C., J. J. MCCARTHY, AND D. G. PEAVY. 1979. Growth rate influence on the chemical composition of phytoplankton in oceanic waters. Nature **279:** 210–215.

HASHIMOTO, L. K., W. A. KAPLAN, S. C. WOFSY, AND M. B. MCELROY. 1983. Transformations of fixed nitrogen and N_2O in the Cariaco Trench. Deep-Sea Res. **30:** 575–590.

HATTORI, A. 1982. The nitrogen cycle in the sea with special reference to biogeochemical processes. J. Oceanogr. Soc. Jap. **38:** 245–265.

IMBERGER, J., AND OTHERS. 1983. The influence of water motion on the distribution and transport of materials in a salt marsh estuary. Limnol. Oceanogr. **28:** 201–214.

JAWORSKI, N. A. 1981. Sources of nutrients and the scale of eutrophication problems in estuaries, p. 83–110. *In* B. J. Neilson and L. E. Cronin [eds.], Estuaries and nutrients. Humana.

LANGE, W. 1970. Cyanophyta-bacteria systems: Effects of added carbon compounds or phosphate on algal growth at low nutrient concentrations. J. Phycol. **6:** 230–234.

LERMAN, A., F. T. MACKENZIE, AND R. M. GARRELS. 1975. Modeling geochemical cycles: Phosphorus as an example. Geol. Soc. Am. Mem. **142:** 205–218.

MCCOMB, A. J., AND OTHERS. 1981. Eutrophication in the Peel-Harvey estuarine system, Western Australia, p. 323–342. *In* B. J. Neilson and L. E. Cronin [eds.], Estuaries and nutrients. Humana.

MARTINEZ, L. A., M. W. SILVER, J. M. KING, AND A. L. ALLDREDGE. 1983. Nitrogen fixation by floating diatom mats: A source of new nitrogen to oligotrophic ocean waters. Science **221:** 152–154.

MEYBECK, M. 1982. Carbon, nitrogen, and phosphorus transport by world rivers. Am. J. Sci. **282:** 401–450.

MONBET, Y., AND OTHERS. 1981. The use of nutrients, salinity and water circulation data as a tool for coastal zone planning, p. 343–372. *In* B. J. Neilson and L. E. Cronin [eds.], Estuaries and nutrients. Humana.
MYERS, V. B., AND R. I. IVERSON. 1981. Phosphorus and nitrogen limited phytoplankton productivity in northeastern Gulf of Mexico coastal estuaries, p. 569–582. *In* B. J. Neilson and L. E. Cronin [eds.], Estuaries and nutrients. Humana.
NIXON, S. W. 1981. Remineralization and nutrient cycling in coastal marine ecosystems, p. 111–138. *In* B. J. Neilson and L. E. Cronin [eds.], Estuaries and nutrients. Humana.
ODUM, E. P. 1971. Fundamentals of ecology. Saunders.
REDFIELD, A. C. 1958. The biological control of chemical factors in the environment. Am. Sci. **46**: 205–222.
RYTHER, J. H., AND W. M. DUNSTAN. 1971. Nitrogen, phosphorus, and eutrophication in the coastal marine environment. Science **171**: 1008–1013.
SCHINDLER, D. W. 1971. Carbon, nitrogen, and phosphorus and the eutrophication of freshwater lakes. J. Phycol. **7**: 321–329.
———. 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. Science **184**: 897–899.
———. 1977. The evolution of phosphorus limitation in lakes. Science **195**: 260–262.
———. 1981. Studies of eutrophication in lakes and their relevance to the estuarine environment, p. 71–82. *In* B. J. Neilson and L. E. Cronin [eds.], Estuaries and nutrients. Humana.
———, E. J. FEE, AND T. RUSZCZYNSKI. 1978. Phosphorus input and its consequences for phytoplankton standing crop and production in the Experimental Lakes Area and in similar lakes. J. Fish. Res. Bd. Can. **35**: 190–196.
SEITZINGER, S. P., S. W. NIXON, AND M. E. PILSON. 1984. Denitrification and nitrous oxide production in a coastal marine ecosystem. Limnol. Oceanogr. **29**: 73–83.
SMITH, S. V. 1983. Net production of coral reef ecosystems, p. 127–131. *In* The ecology of deep and shallow coral reefs. NOAA Symp. Ser. for Undersea Res. V. 1.
———, AND M. J. ATKINSON. 1983. Mass balance of carbon and phosphorus in Shark Bay, Western Australia. Limnol. Oceanogr. **28**: 625–639.
———, AND ———. 1984. Phosphorus limitation of net production in a confined aquatic ecosystem. Nature **307**: 626–627.
———, AND P. L. JOKIEL. 1978. Water composition and biogeochemical gradients in the Canton Atoll lagoon. Atoll Res. Bull. 221, p. 15–53.
———, W. J. KIMMERER, E. A. LAWS, R. E. BROCK, AND T. W. WALSH. 1981. Kaneohe Bay sewage diversion experiment: Perspectives on ecosystem responses to nutritional perturbation. Pac. Sci. **35**: 279–396.
SMITH, V. H. 1982. The nitrogen and phosphorus dependence of algal biomass in lakes: An empirical and theoretical analysis. Limnol. Oceanogr. **27**: 1101–1112.
———. 1983. Low nitrogen to phosphorus ratios favor dominance by blue-green algae in lake phytoplankton. Science **221**: 669–671.
THOMAS, W. H. 1970*a*. On nitrogen deficiency in tropical Pacific oceanic phytoplankton: Photosynthetic parameters in poor and rich water. Limnol. Oceanogr. **15**: 380–385.
———. 1970*b*. Effect of ammonium and nitrate concentration on chlorophyll increases in natural tropical Pacific phytoplankton populations. Limnol. Oceanogr. **15**: 386–394.
VOLLENWEIDER, R. A. 1968. Water management research. OECD, Paris. DAS/CSI/68.27. 183 p.
———. 1969. Moglichkeiten und Grenzen elementarer Modelle des Stoffbilanz von Seen. Arch. Hydrobiol. **66**: 1–36.
———. 1976. Advances in defining critical loading levels of phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol. **33**: 53–83.
WEBB, K. L. 1981. Conceptual models and processes of nutrient cycling in estuaries, p. 25–46. *In* B. J. Neilson and L. E. Cronin [eds.], Estuaries and nutrients. Humana.
———, W. D. DUPAUL, W. WIEBE, W. SOTTILLE, AND R. E. JOHANNES. 1975. Enewetak (Eniwetok) Atoll: Aspects of the nitrogen cycle on a coral reef. Limnol. Oceanogr. **20**: 198–210.
WIEBE, W. J., R. E. JOHANNES, AND K. L. WEBB. 1975. Nitrogen fixation in a coral reef community. Science **188**: 257–259.

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Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment

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Lake 227, a small lake in the Precambrian Shield at the Experimental Lakes Area (ELA), has been fertilized for 37 years with constant annual inputs of phosphorus and decreasing inputs of nitrogen to test the theory that controlling nitrogen inputs can control eutrophication. For the final 16 years (1990–2005), the lake was fertilized with phosphorus alone. Reducing nitrogen inputs increasingly favored nitrogen-fixing cyanobacteria as a response by the phytoplankton community to extreme seasonal nitrogen limitation. Nitrogen fixation was sufficient to allow biomass to continue to be produced in proportion to phosphorus, and the lake remained highly eutrophic, despite showing indications of extreme nitrogen limitation seasonally. To reduce eutrophication, the focus of management must be on decreasing inputs of phosphorus.

cyanobacteria blooms | Experimental Lakes | nutrient limitation | phosphorus

Eutrophication is the general term used by aquatic scientists to describe the suite of symptoms that a lake exhibits in response to fertilization with nutrients (1). Common symptoms include dense algal blooms causing high turbidity and increasing anoxia in the deeper parts of lakes from the decay of sedimenting plant material. The anoxia can in turn cause fish kills in midsummer. One of the most objectionable symptoms of eutrophication has been the appearance of floating algal "blooms" (Fig. 1). In freshwaters, these surface blooms are often of nitrogen (N)-fixing cyanobacteria (known popularly as blue–green algae) (2). Similar forms are also common in many eutrophied estuaries (3) although other types of nuisance algal blooms are also common (4).

The emphasis on controlling eutrophication in freshwater lakes has been focused heavily on decreasing inputs of phosphorus (P) (2, 5–7). Schindler (2, 7) noted that many lakes rendered eutrophic by the addition of P contained phytoplankton communities that showed signs of extreme N limitation in short-term bioassays such as N debt (8, 9) or nutrient addition bioassays (10), He concluded that N limitation was a symptom of overfertilization with P and proposed that short-term N limitation was not necessarily a reliable indication that N must be controlled to reverse eutrophication. Hecky and Kilham (11) also warned that short-term measures of N deficiency were not reliable indicators of ecosystem responses to N enrichment or removal. Despite these early warnings, many studies in lakes and estuaries still conclude that N must be controlled as well as, or instead of, P to reduce eutrophication (for review, see ref. 12). The subject is hotly debated with respect to reducing eutrophication in the Baltic Sea (3). Recently, there has been renewed advocacy of N control to mitigate eutrophication of both lakes and estuaries. N and P control is being proposed to halt the rapid increase in eutrophication in Lake Winnipeg (13) and the Baltic Sea (3). This is troubling because proponents of controlling N in lakes and estuaries are relying on the same bioassays or correlations with nutrient concentrations that we (2, 7, 11) found to lead to the erroneous conclusion that N inputs must be controlled to reduce eutrophication. These bioassays and the related assumptions have led to very expensive mitigation programs in several countries.

Aquatic scientists have often relied on the Redfield ratio to gauge whether nutrient supplies are sufficient. Redfield (14) observed that the ratio of carbon:nitrogen:phosphorus in marine phytoplankton was quite constant, with mean ratios by weight of ≈40:7:1. The Redfield ratio has subsequently been accepted as a general indicator for balanced growth with potential for near optimum growth rates (8). In the Experimental Lakes Area (ELA), lakes rendered eutrophic by experimental additions of N and P at N:P ratios less than Redfield ratio (7:1 weight ratio) have had N concentrations increase to above Redfield ratios as the result of N fixation by diazotrophic heterocystous cyanobacteria (2, 9, 15, 16). Algal biomass and chlorophyll *a* have remained proportional to P inputs regardless of the ratio of N:P added as fertilizer. Here, we describe a deliberate and extreme long-term experiment to test the effectiveness of controlling N on eutrophication.

The Lake and Its Experimental Treatments. Lake 227 in the ELA of northwestern Ontario, Canada, has a surface area of 5.0 ha, a mean depth of 4.4 m, and a maximum depth of 10.0 m (17). In June 1969, fertilization of the lake began with P and N to test the hypothesis then popular in North America that C could limit eutrophication of lakes (18). For the first five years (1969–1974), the ratio of N to P in fertilizer was added at 12: 1 by weight, well above the Redfield ratio, to ensure that phytoplankton had adequate N and P supplies during the period when we were testing the C limitation hypothesis. Lake 227 became highly eutrophic, producing phytoplankton blooms in proportion to P supplies, despite phytoplankton showing symptoms of extreme C limitation for most of the summer months. C deficiency reduced daily rates of photosynthesis, but phytoplankton biomass increased until limited by P (19). In a second experiment in nearby Lake 226, we deliberately tested the effects of N limitation, by adding N and C to two isolated basins, but phosphorus only to one basin (North). N:P ratios in North Basin fertilizer were 4.6 to 5.5:1 by weight, well below the Redfield ratio. Large algal blooms were again in proportion to P additions, but the respond-

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Fig. 1. Photograph of Grand Beach on the southern basin of Lake Winnipeg, August 2006. Photo by Lori Volkart.

ing species were primarily N-fixing cyanobacteria (2, 7). To test further the hypothesis that low N:P favored N-fixing species, the ratio of N to P in fertilizer added to Lake 227 was decreased to 4:1 beginning in 1975. The hypothesis was supported, and N fixation was high in subsequent years (2, 15, 16). Lake 227 continued to be fertilized at this N:P ratio through 1989. By that time there were signs that the lake was becoming both C- and N-sufficient because of slowly increasing concentrations of these elements as the result of several years of atmospheric invasion and net fixation and retention of N_2 and CO_2 (15). As nutrient balance was approached, the domination of phytoplankton by N-fixing cyanobacteria was decreasing (16), and short-term N limitation was less pronounced (9). From 1990 onward, no N fertilizer has been added to the lake. P continues to be added, and P inputs have remained relatively constant throughout the 37 years of fertilization (Table 1).

Superimposed on the nutrient fertilization was a short-term (4-year) food web manipulation (20). In 1993–1994, pike *Esox lucius* were added to the lake, which had contained only large numbers of forage fish, including fathead minnows (*Pimephales promelas*) and several species of dace (*Semotilus margarita*, *Phoxinus eos*, and *Phoxinus neogaeus*). By 1996, predation by pike had extirpated all forage fish. They have remained absent, and the lake fishless after all pike were removed in 1996 (ref. 20 and K. Mills, unpublished observation).

Nutrient Concentrations and Ratios. Concentrations of total phosphorus (TP) in the epilimnion during ice-free season in all years of fertilization averaged 42 μg/liter (Fig. 2*A*). Concentrations of total dissolved phosphorus (TDP) averaged 11 μg/liter (Fig. 2*B*).

Table 1. Summary of annual fertilizer additions to Lake 227, 1969–2005

Year	N, kg per year	P, kg per year	N:P by weight
1969	249	20.7	12.1
1970–1974	308	24.8	12.4
1975–1982	110	23.6	4.66
1983	110	19.8	5.54
1984–1989	110	23.6	4.66
1990–1997	0	23.6	0
1998	0	31.9	0
1999–2005	0	24.5	0

There was no significant long-term trend in either form. Soluble reactive P was not routinely measured, but it was generally well below the limit of detection except by radioactive P bioassays, i.e., in the nanogram per liter range (21).

Total nitrogen (TN) in Lake 227 averaged 825 μg of N per liter in 1969–1974, the period when high N fertilizer was added (Fig.

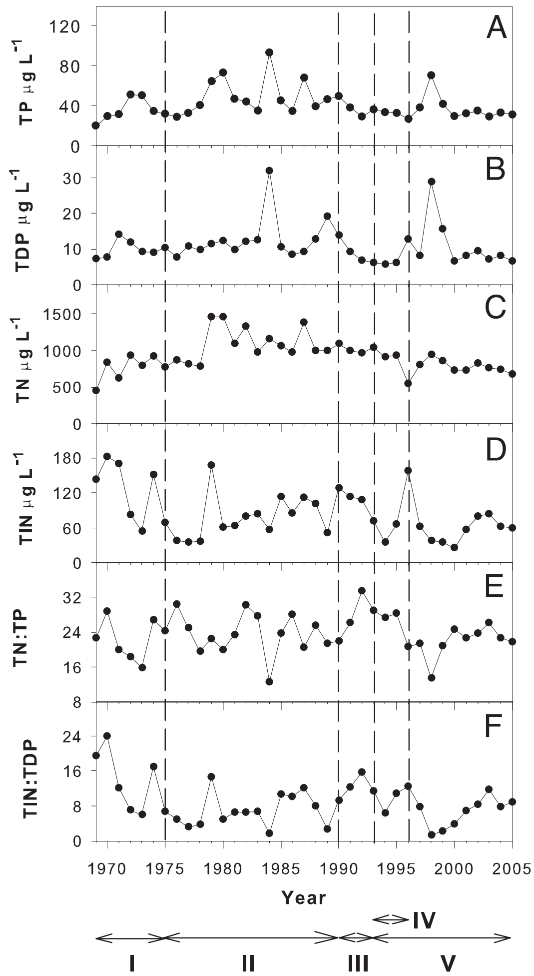

Fig. 2. Mean annual epilimnetic nutrient concentrations and ratios in Lake 227, 1969–2005. Periods separated by vertical dashed lines represent: I, the period of fertilization at high N:P (12:1 by weight) 1969–1974; II, the period of fertilization with low N:P (4:1), 1975–1989; III–V, the period when no N fertilizer was added to the lake. IV, the years (1993–1996) that pike were present in the lake. The lake was fishless after 1996. (*A*) Total P. (*B*) Total dissolved P. (*C*) Total N. (*D*) Total inorganic nitrogen (= $NH_4 + NO_2 + NO_3$). (*E*) Ratio by weight of total N to total P in the lake. (*F*) Ratio by weight of TIN to TDP.

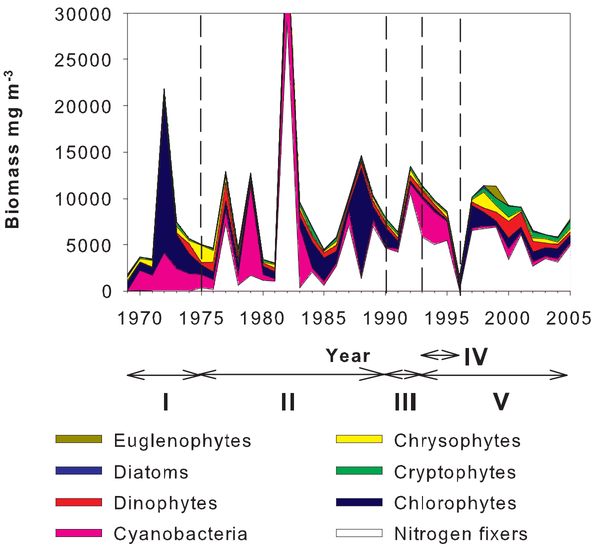

Fig. 3. Phytoplankton biomass in the epilimnion by algal group, 1969–2005. Vertical dashed lines were as in Fig. 2. In the Legend, ''cyanobacteria'' refers to cyanobacteria species that are not known to fix nitrogen. ''Nitrogen fixers'' refers to N-fixing species of cyanobacteria.

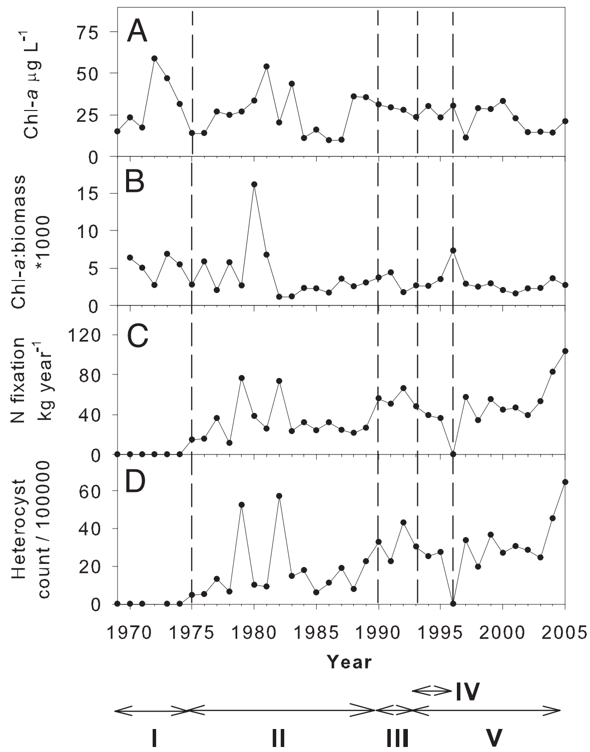

Fig. 4. Other measures of phytoplankton and nitrogen fixation, 1969–2005. (*A*) Chlorophyll *a*. (*B*) Ratio of chlorophyll *a*:phytoplankton biomass (μg/mm^3). (*C*) Nitrogen fixation calculated from heterocyst counts. (*D*) Heterocyst counts. Vertical dashed lines are as in Fig. 2.

2*C*). It stayed constant for several years after N fertilizer was decreased, then increased in 1978, staying at ≈1,200 μg/liter through 1989. After termination of N fertilization in 1990, it decreased very slowly to ≈800 μg of N per liter after 1999. Dissolved inorganic N, the sum of nitrate, nitrite, and ammonium (TIN) was highest in the 1969–1974 period, when high N fertilizer was applied (Fig. 2*D*). It averaged 128 μg of N per liter. After 1975, there was no significant long-term trend. Mean values for 1975–1989, when N fertilizer was added, and 1990–2004, when no N was added were almost identical, 77 and 76 μg/liter, respectively. Concentrations of TIN were high in winter and spring but decreased to low concentrations in early summer when the phytoplankton exhibited strong N deficiency that was subsequently relieved as N-fixing cyanobacteria increased to high midsummer biomasses (9). An analysis of the data through 1984 revealed that the lake had attained steady-state with inputs of both N and C by that time (15).

Ratios of TN:TP in the lake were usually much higher than Redfield ratios (7:1 by weight; Fig. 2*E*). Average values for 1969–1974, 1975–1989, and after 1990 were 22–28 by weight, above the value of 20:1 where N limitation is usually inferred in freshwater or marine systems (22, 23). It is noteworthy that average TIN:TDP ratios were occasionally below Redfield ratios after 1975 (Fig. 2*F*). However, average TIN:TDP ratios equaled or exceeded Redfield ratios in all three fertilization periods, at 14, 7, and 10 by weight. The N:P in fertilizer was zero after 1989, so these mean ratios were maintained entirely by N fixation and nitrogen recycled within the lake.

Response of the Phytoplankton and N Fixation. From 1969–1974, all groups of phytoplankton increased as the result of fertilization with high N:P (Fig. 3). The algal assemblage was dominated by small unicellular desmids with large populations of *Limnothrix redekei* occurring from late June until early September. N-fixing cyanobacteria were not detected (24) as N was being added in excess of algal demand and TIN:TDP ratios were high (Fig. 2*F*). C necessary for algal biomass was supplied by invasion of CO_2 to the lake from the atmosphere (25). Short-term bioassays indicated that C-limited photosynthesis and algal growth and did not predict the continued growth of algal biomass in proportion to P.

N-fixing cyanobacteria first appeared in significant numbers in 1975, within weeks of reducing the N:P ratio in loading (ref. 2 and Fig. 3). *Aphanizomenon schindlerii* appeared first, and by the early 1980s it dominated the summer assemblage. The average annual importance of N fixers varied considerably from 1975 through 1989. They were always the dominant part of the phytoplankton biomass in July and August. Annual *Aphanizomenon* biomass diminished slowly and became more variable during the late 1980s. After N fertilization ceased in 1990, N fixers (primarily *A. schindlerii*) always formed >50% of the total phytoplankton biomass, except in 1996, the terminal year of the food web manipulation (20). From 1996 when the lake became fishless to the present, the algal community became more diverse with increased representation by chrysophytes, diatoms, cryptophytes, and dinoflagellates. Nonfixing cyanobacteria, most notably *L. redekei*, remained prominent throughout the experiment but tended to increase after midsummer blooms of N-fixing cyanobacteria, suggesting a competitive advantage over the N fixers when recycled fixed N became available.

Phytoplankton biomass averaged 9,306 mg m^{-3}. There were no significant differences in annual average biomass among the periods when different N:P ratios were applied. Variability in phytoplankton biomass was high, ranging from 1,502 mg m^{-3} in 1996 to 39,000 mg m^{-3} in 1982.

Chlorophyll *a* is frequently used as a measure of phytoplankton biomass especially in short-term bioassays. Annual averages during the term of the experiment ranged from 17 to 59 μg liter^{-1}, with higher values occurring when the N:P ratio in fertilizer was high (Fig. 4*A*). Interannual variation in chlorophyll was higher during the periods when N fertilizer was added than after 1990. The chlorophyll *a*:biomass ratio was also less variable when no N fertilizer was added (Fig. 4*B*). It fluctuated in a narrow range, except for high values in 1980 and 1996. In those

2 years, N-fixing cyanobacteria populations were low, and other taxa dominated the phytoplankton. Average chlorophyll *a*:biomass in the first several years of the experiment was somewhat higher than in the last 20 years. These results demonstrate that chlorophyll *a* per unit of biomass is higher when N is in excess of algal requirements. Conversely, if the algal community is made more N-deficient, chlorophyll per unit biomass declines. Consequently, nutrient enrichment experiments that rely only on chlorophyll *a* as a measure of phytoplankton abundance may give misleading results about changes in biomass.

N fixation was negligible before 1975, when fertilizer N:P ratios were high (Fig. 4*C*). From 1975 onward, fixation by the lake varied from 0.2 to 2.1 g m^{-2} $year^{-1}$ with the exception of 1996, when no fixation occurred after food chain manipulation. By the late 1980s when indicators of N limitation were weakening (9), N fixation had stabilized at 0.5–0.6 g of N m^{-2} $year^{-1}$, approximately half of the highest values in earlier years. After fertilization with N was terminated in 1990, fixation increased again for several years. It decreased to zero in 1996, after which fixation returned to ≈1 g of N m^{-2} $year^{-1}$ then increased again in 2004–2005 to >2 g of N m^{-2} $year^{-1}$, the highest values recorded in the 37-year dataset.

Changes Caused by the "Trophic Cascade": 1996. The response of phytoplankton in 1996, after fish were extirpated, stands out. In that year, the zooplankton consisted primarily of large *Daphnia pulicaria*, as the result of decreased zooplanktivory by fish. Addition of pike in 1993–1994 and prior minnow trapping had extirpated populations of forage fish, as described earlier. Pike were removed in late 1995 and early 1996, rendering Lake 227 fishless. During that year, TIN and chlorophyll *a*:biomass were high and comparable with values observed before 1975, when ratios of added N:P were high. However, chlorophyll *a* and phytoplankton biomass were very low in 1996. N-fixing cyanobacteria and N fixation were unmeasurable. Increased sedimentation of P and excretion of TIN as the result of high grazing by *Daphnia* appeared to be responsible for the high TIN (20). N fixers returned to dominance in 1997 and in subsequent years as *Daphnia* populations returned to low abundances. Invertebrate predators such as *Chaoborus* have increased after fish removal and may now be the dominant predator on *Daphnia* and other zooplankton. Before 1996, predation by fish likely limited the abundance of invertebrate predators (26).

Interpretation. Although there was no increase in the abundance of N-fixing cyanobacteria after N fertilization ceased in 1990, heterocyst numbers and subsequent N fixation increased greatly (Fig. 3 *C* and *D*). Despite the reduction of N fertilizer to zero, P additions kept the lake eutrophic, with no substantial changes in biomass of phytoplankton. We suggest that in most freshwater lakes, attempts to manage N inputs will be equally futile. Indeed, addition of N to a hypereutrophic lake caused N-fixing cyanobacteria to be suppressed, allowing other species to thrive (27). In another case study, N addition was futile because it was rapidly denitrified. Water quality improved, but this was attributed to a fish kill and subsequent expansion of macrophyte cover (28). Unless N concentrations are of concern for human health, funds for eutrophication control are better spent on more complete removal of P sources. This work also demonstrates the necessity of using studies at whole-ecosystem scales for managing ecosystems. Bottle bioassays or mesocosm studies cannot properly account for important long-term processes such as atmospheric exchange, colonization by N-fixing cyanobacteria, changes in the grazer community, and increases in N as the result of N fixation and return from sediments that determine the long-term fate of eutrophied ecosystems.

Are Our Findings Applicable to Estuaries? It is generally believed that coastal marine waters are N-limited and that N must be controlled to reduce algal abundance (for review, see 12). However, these conclusions are largely based on bioassays similar to those that we have found to give spurious results for freshwaters. In low-salinity estuaries such as the Baltic Sea, N fixers are abundant, and rates of N fixation are high (3, 29). In stratified eutrophic lakes and estuaries, both P and N are generally plentiful in spring, before stratification develops. Once warming allows thermoclines to become established, initial concentrations of N are depleted, and N-fixing species of phytoplankton are favored. N fixation allows phytoplankton standing crops to develop that are proportional to P, as indicated by the constant ratios for suspended N:P and total N:P in Lake 227 (Fig. 1). On a long-term basis, if annual N fixation does not exceed sedimentation plus denitrification, chronic N deficiencies favor N-fixing cyanobacteria every summer.

In at least one well documented case of an estuarine recovery from eutrophication (29), chlorophyll *a* in the Stockholm Archipelago estuary decreased from 30 μg/liter to <16 μg/liter in the early 1970s, after the control of P alone from Swedish sewage treatment plants. TP decreased from 75–90 to <30 μg/liter in 3 years. As in Lake 227, at the time, the waters of the Archipelago were dominated by N-fixing cyanobacteria species in summer and showed signs of extreme N limitation (3, 30). N fixation [measured in 1972 (30)] was 2.25 g/m^2 per year, higher than any year in Lake 227. P and phytoplankton in the Archipelago declined, initially rapidly with the imposition of P removal in waste treatments then more slowly since the mid-1970s (3)., The recent removal of N at waste treatment plants beginning in the mid-1990s has also led to somewhat lower chlorophyll concentrations. However, the improved waste treatment also removed more P and reduced biological oxygen demand (BOD from ammonia loading) (3). The reduction in BOD is important because it has decreased the intensity and duration of anoxia at the sediment surface. Preliminary data suggest that P release from sediments is beginning to decrease in response to increasing oxygen (3). Overall, it is difficult to tell whether the slight improvements in recent years are the result of reduced inputs of N, P, BOD, or all three. However, we suggest that as in Lake 227, N fixation and short-term indicators of N deficiency such as low TIN:TDP ratios or sensitivity to N enrichment bioassays are a sign of overfertilization with P and not necessarily evidence that N control will cause decreased eutrophication.

As we found in the 227 trophic cascade experiment described above, in large mesocosm experiments using near full-strength seawater from Narragansett Bay, RI, nitrogen fixers prospered, and nitrogen fixation occurred as long as the abundance of grazing zooplankton was low (31, 32). Normal concentrations of zooplankton were able to suppress cyanobacteria populations by grazing, keeping colonies too small to form heterocysts. These observations appear to be comparable with Lake 227 in 1996, when high-grazer populations after decreased zooplanktivory prevented N-fixing cyanobacteria from dominating despite low N:P inputs in fertilizer. These results suggest that under the right conditions, grazers may be able to prevent cyanobacteria blooms, in both freshwaters and estuaries.

There is, however, still need for caution when extrapolating our results to estuaries. It is conceivable that molybdenum, iron or other trace nutrients might limit the rate of colony growth by N fixers (31, 32), although the rates of fixation observed (29, 33, 34) are as high as we have observed in freshwaters. In some estuaries, such as the Baltic Sea, haloclines isolate productive epilimnions from deepwater nutrient sources even more strongly than during the temperature-driven summer stratification of most lakes. Persistent haloclines can restrict oxygen circulation to deeper waters and return of nutrients to the euphotic zone. Haloclines also promote hypoxia and release of P from sedi-

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ments. The strength and duration of haloclines often depend on extreme weather events that are highly stochastic (3). High rates of denitrification in estuarine anoxic zones allow much of the accumulated fixed N to be returned to the atmosphere (34, 35). However, this would reinforce the chronic N deficiency that we have discussed above, which can only be relieved by reducing P loading. Also, unlike the Baltic, many estuaries have rather rapid flushing that would result in the continuous dilution of any N fixed by cyanobacteria (36), potentially keeping an estuary in a chronic N-deficient state despite high rates of N fixation. Finally, estuaries are often highly turbid, and light may limit N fixation.

In summary, the long-term experiment in Lake 227 and the early response of the Stockholm Archipelago to P control challenge the widely held belief that short-term N limitation in phytoplankton communities is evidence that external sources of N should be controlled to decrease eutrophication. N-fixing cyanobacteria cannot be limited by a shortage of dissolved N and instead are competitively favored. The increasing appreciation of the importance of N fixation to balancing the global ocean N budget (34, 37) demonstrates that salinity and marine geochemistry alone do not limit N-fixing species and N fixation has the potential to overcome N deficiencies in a wide range of aquatic environments. Our results suggest that controlling N inputs could actually aggravate the dominance of N-fixing cyanobacteria.

The adjustment of N deficiencies in Lake 227 required several years (15), indicating that conclusions meaningful for nutrient management are unlikely to be obtained from short-term experiments. The responses of Lake 227 over almost 4 decades of fertilization indicate that experiments to guide nutrient management confidently must be full-ecosystem scale and carried out for at least several years (38).

Materials and Methods

Fertilization of the lake began on June 26, 1969 and was done weekly during the ice-free seasons of all years since that time. The lake was sampled from weekly to monthly during the ice-free seasons and from two to four times under ice in most years. On each date, nitrate plus nitrate, ammonium, total dissolved N, particulate N, total dissolved P, total P, chlorophyll *a*, base cations and strong acid anions, and pH were measured. Samples of phytoplankton were taken for identification and counting and for measurements of primary production. Zooplankton samples were also taken on each sampling date. A temperature profile was also measured.

Full details of methods are given in the supporting information (SI) *Materials and Methods*.

ACKNOWLEDGMENTS. Reviews of an early draft by Suzanne Bayley, Frieda Taub, and Diane Orihel and of the semifinal draft by Val Smith, Steve Carpenter, and Peter Vitousek improved the manuscript and are greatly appreciated. Work during parts of the 37-year period was supported by the Fisheries Research Board of Canada, the Canadian Department of Fisheries and Oceans, and Natural Sciences and Engineering Research Council Discovery Grants (to D.W.S. and R.E.H.).

1. Hutchinson, GE (1973) Eutrophication. *Am Sci* 61:269–279.
2. Schindler DW (1977) Evolution of phosphorus limitation in lakes: Natural mechanisms compensate for deficiencies of nitrogen and carbon in eutrophied lakes. *Science* 195:260–262.
3. Boesch D, Hecky R, O'Melia C, Schindler DW, Seitzinger S (2006) *Eutrophication of Swedish Seas* (Final Report, Swedish Environmental Protection Agency, Stockholm Sweden).
4. Anderson DM, *et al.* (2002) Harmful algal blooms and eutrophication: Nutrient sources, composition, and consequences. *Estuaries* 25:704–726.
5. Vollenweider RA (1968) *Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters, with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication* (Tech Rep DAS/CS/68.27, OECD, Paris).
6. Vollenweider RA (1976) Advances in defining critical loading levels for phosphorus in lake eutrophication. *Mem 1st Ital Idrobiol* 33:53–83.
7. Schindler DW (1974) Eutrophication and recovery in experimental lakes: Implications for lake management. *Science* 184:897–899.
8. Healey FP, Hendzel LL (1980) Physiological indicators of nutrient deficiency in lake phytoplankton. *Can J Fish Aquat Sci* 37:442–453.
9. Hendzel LL, Hecky RE, Findlay DL (1994) Recent changes of nitrogen fixation in Canadian Precambrian Shield lakes. *Can J Fish Aquat Sci* 51:2247–2253.
10. Schindler DW (1971) Carbon, nitrogen and phosphorus and the eutrophication of freshwater lakes. *J Phycol* 7:321–329.
11. Hecky RE, Kilham P (1988) Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment. *Limnol Oceanogr* 33:796–822.
12. Howarth RW, Marino R (2006) Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: Evolving views over three decades. *Limnol Oceanogr* 51:364–376.
13. Lake Winnipeg Stewardship Board (2006) *Reducing Nutrient Loading to Lake Winnipeg and Its Watershed*: *Our Respective Responsibility and Commitment to Action* (Report to the Minister of Water Stewardship December 2006).
14. Redfield AC (1958) The biological control of chemical factors in the environment. *Am Sci* 46:205–221.
15. Schindler DW, Hesslein RH, Turner MA (1987) Exchange of nutrients between sediments and water after 15 years of experimental eutrophication. *Can J Fish Aquat Sci* 44(Suppl 1):26–33.
16. Findlay DL, Hecky RE, Hendzel LL, Stainton MP, Regehr GW (1994) Relationship between N_2-fixation and heterocyst abundance and its relevance to the nitrogen budget of Lake 227. *Can J Fish Aquat Sci* 51:2254–2266.
17. Brunskill GJ, Schindler DW (1971) Geography and bathymetry of selected lake basins in the Experimental Lakes Area (ELA), northwestern Ontario. *J Fish Res Board Can* 28:139–155.
18. Legge RF, Dingeldein D (March/April 1970) We hung the phosphates without a fair trial. *Can Res Develop* 3:19–42.
19. Schindler DW (1975) Whole-lake fertilization experiments with phosphorus, nitrogen, and carbon. *Int Ver Theor Angew Limnol Verh* 19:3221–3231.
20. Elser JJ, *et al.* (2000) Pelagic C:N:P stoichiometry in a eutrophied lake: Responses to a whole-lake food-web manipulation. *Ecosystems* 3:293–307.
21. Levine SN, Schindler DW (1980) Radiochemical analysis of orthophosphate concentrations and seasonal changes in the flux of orthophosphate to seston in two Canadian Shield lakes. *Can J Fish Aquat Sci* 37:479–487.
22. Smith VH (2006) Responses of estuarine and coastal marine phytoplankton to nitrogen and phosphorus enrichment. *Limnol Oceanogr* 51:377–384.
23. Guildford S, Hecky RE (2000) Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: Is there a common relationship? *Limnol Oceanogr* 45:1213–1223.
24. Schindler DW, *et al.* (1973) Eutrophication of Lake 227 by addition of phosphate and nitrate: The second, third, and fourth years of enrichment 1970, 1971, and 1972. *J Fish Res Board Can* 30:1415–1440.
25. Schindler DW, Brunskill GJ, Emerson S, Broecker WS, Peng T-H (1972) Atmospheric carbon dioxide: Its role in maintaining phytoplankton standing crops. *Science* 177:1192–1194.
26. Ramcharan CW, *et al*. (1995) A comparative approach to determining the role of fish predation in structuring limnetic ecosystems. *Arch Hydrobiol* 133:389–416.
27. Barica J, Kling H, Gibson J (1980) Experimental manipulation of algal bloom composition by nitrogen addition. *Can J Fish Aquat Sci* 37:1175–1183.
28. Lathrop RC (1988) Evaluation of whole-lake nitrogen-fertilization for controlling blue–green algal blooms in a hypereutrophic lake. *Can J Fish Aquat Sci* 45:2061–2075.
29. Brattberg G (1986) Decreased phosphorus loading changes phytoplankton composition and biomass in the Stockholm archipelago. *Vatten* 42:141–152.
30. Brattberg G (1977) Nitrogen fixation in a polluted brackish water archipelago. *Ambio (Spec Rep)* 5:27–42.
31. Marino R, Chan F, Howarth RW, Pace ML, Likens GE (2006) Ecological constraints on planktonic nitrogen fixation in saline estuaries. I. Nutrient and trophic controls. *Mar Ecol Prog Ser* 309:25–39.
32. Chan F, Marino RL, Howarth RW, Pace ML (2006) Ecological constraints on planktonic nitrogen fixation in saline estuaries. II. Grazing controls on cyanobacterial population dynamics. *Mar Ecol Prog Ser* 309:41–53.
33. Voss M, Emeis K-C, Hille S, Neumann T, Dippner JW (2005) Nitrogen cycle of the Baltic Sea from an isotopic perspective. *Global Biogeochem Cycles* 19: GB 3001, doi:10.1029/2004GB002338.
34. Gardner WS, *et al.* (2006) Nitrogen fixation and dissimilatory nitrate reduction to ammonium (DNRA) support nitrogen dynamics in Texas estuaries. *Limnol Oceanogr* 51:558–568.
35. Seitzinger SP (1988) Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. *Limnol Oceanogr* 33:702–724.
36. Mahaffy C, Michaels AF, Capone DG (2005) The conundrum of marine N_2 fixation. *Am J Sci* 305:546–595.
37. Karl D, *et al.* (2002) Dinitrogen fixation in the world's oceans. *Biogeochemistry* 57–58:47–98.

II

(Ikke-lovgivningsmæssige retsakter)

AFGØRELSER

KOMMISSIONENS AFGØRELSE

af 20. september 2013

om fastsættelse i overensstemmelse med Europa-Parlamentets og Rådets direktiv 2000/60/EF af værdierne for klassifikationerne i medlemsstaternes overvågningssystemer som resultat af interkalibreringen og om ophævelse af beslutning 2008/915/EF

(meddelt under nummer C(2013) 5915)

(EØS-relevant tekst)

(2013/480/EU)

EUROPA-KOMMISSIONEN HAR —

under henvisning til traktaten om Den Europæiske Unions funktionsmåde,

under henvisning til Europa-Parlamentets og Rådets direktiv 2000/60/EF af 23. oktober 2000 om fastlæggelse af en ramme for Fællesskabets vandpolitiske foranstaltninger [1], navnlig punkt 1.4.1, nr. ix), i dette direktivs bilag V, og

ud fra følgende betragtninger:

(1) Ifølge artikel 4, stk. 1, litra a), nr. ii), i direktiv 2000/60/EF har medlemsstaterne pligt til at beskytte, forbedre og genoprette alle overfladevandområder med henblik på at opnå en god tilstand for overfladevand senest 15 år efter datoen for direktivets ikrafttræden — med forbehold af visse undtagelser — i overensstemmelse med bestemmelserne i bilag V til dette direktiv. Ifølge artikel 4, stk. 1, litra a), nr. iii), i direktiv 2000/60/EF har medlemsstaterne pligt til at beskytte og forbedre alle kunstige og stærkt modificerede vandområder med henblik på at opnå et godt økologisk potentiale og god kemisk tilstand for overfladevand senest 15 år efter datoen for direktivets ikrafttræden — med forbehold af visse undtagelser — i overensstemmelse med bestemmelserne i bilag V til dette direktiv. I henhold til punkt 1.4.1, nr. i), i bilag V til direktiv 2000/60/EF skal henvisninger til »økologisk tilstand« opfattes som »økologisk potentiale«, hvad angår kunstige eller stærkt modificerede vandområder.

(2) Ved interkalibreringen anvendes en harmoniseret tilgang til definitionen af et af de vigtigste miljømål i direktiv 2000/60/EF, nemlig god økologisk tilstand.

(3) I punkt 1.4.1 i bilag V til direktiv 2000/60/EF er fastlagt en proces, hvormed der sikres sammenlignelighed mellem medlemsstaternes biologiske overvågningsresultater, som er en central del af klassifikationen af den økologiske tilstand. I den forbindelse skal medlemsstaternes biologiske overvågningsresultater og klassifikationerne i deres overvågningssystemer sammenlignes via et interkalibreringsnetværk bestående af overvågningslokaliteter i hver medlemsstat og hver økoregion i Unionen. Ifølge direktiv 2006/60/EF har medlemsstaterne pligt til i passende omfang at indsamle de nødvendige oplysninger om de lokaliteter, som indgår i interkalibreringsnetværket, så det er muligt at vurdere, om klassifikationerne i det nationale overvågningssystem stemmer overens med de normgivende definitioner i punkt 1.2 i bilag V til direktiv 2000/60/EF, og om resultaterne af klassifikationerne i de nationale overvågningssystemer er indbyrdes sammenlignelige.

(4) Med henblik på at udføre interkalibreringen er medlemsstaterne organiseret i geografiske interkalibreringsgrupper bestående af medlemsstater, der har de samme bestemte overfladevandområdetyper som defineret i punkt 2 i bilaget til Kommissionens beslutning 2005/646/EF af 17. august 2005 om oprettelse af et register over lokaliteter, der skal udgøre et interkalibreringsnetværk, i overensstemmelse med Europa-Parlamentets og Rådets direktiv 2000/60/EF [2].

[1] EFT L 327 af 22.12.2000, s. 1.

[2] EUT L 243 af 19.9.2005, s. 1.

(5) Det fastslås i punkt 1.4.1 i bilag V til direktiv 2000/60/EF, at interkalibreringen udføres med hensyn til biologiske elementer, idet der foretages en sammenligning mellem klassifikationsresultaterne fra de nationale overvågningssystemer for hvert biologisk element og for hver fælles overfladevandområdetype blandt medlemsstaterne i samme geografiske interkalibreringsgruppe, og det vurderes, om resultaterne er overensstemmende med de normgivende definitioner fastlagt i punkt 1.2 i bilag V til direktiv 2000/60/EF.

(6) Kommissionen har lettet to faser i arbejdet med interkalibreringen med hjælp fra Institut for Miljø og Bæredygtighed under Det Fælles Forskningscenter.

(7) Inden for rammerne af den fælles gennemførelsesstrategi for vandrammedirektivet er der blevet udarbejdet tre vejledninger (nr. 6 [1] og 14 (to versioner) [2]) for at lette arbejdet med interkalibreringen. De giver et overblik over hovedprincipperne og retningslinjerne for interkalibreringen, herunder frister og indberetningskrav.

(8) I 2007 havde Kommissionen modtaget interkalibreringsresultater for en række biologiske kvalitetselementer. De indgik i Kommissionens beslutning 2008/915/EF af 30. oktober 2008 om fastsættelse i overensstemmelse med Europa-Parlamentets og Rådets direktiv 2000/60/EF af værdierne for klassifikationerne i medlemsstaternes overvågningssystemer som resultat af interkalibreringen [3], hvori de værdier for grænselinjerne mellem tilstandsklasser, som medlemsstaterne skal anvende for klassifikationen i deres overvågningssystemer, fastlægges. Resultaterne af den første fase af interkalibreringen var ufuldstændige, idet alle biologiske kvalitetselementer ikke var omfattet. Det var imidlertid nødvendigt at godkende de tilgængelige resultater af interkalibreringen til tiden som grundlag for udformningen af de første vandområdeplaner og indsatsprogrammer i overensstemmelse med artikel 11 og 13 i direktiv 2000/60/EF.

(9) Resultaterne af denne første fase af interkalibreringen blev vedtaget i beslutning 2008/915/EF. Disse resultater blev indarbejdet midlertidigt, idet yderligere interkalibreringsresultater skulle indgå i en ny afgørelse, når medlemsstaterne havde tilvejebragt de relevante oplysninger i overensstemmelse med punkt 1.4.1 i bilag V til direktiv 2000/60/EF.

(10) For at lukke hullerne og forbedre sammenligneligheden af interkalibreringsresultaterne forud for vandområdeplanernes anden cyklus i 2015 iværksatte Kommissionen den anden fase af interkalibreringen.

(11) I bilag I til denne afgørelse redegøres for resultaterne af interkalibreringen, hvor interkalibreringen har været vellykket med de tekniske muligheder, der er i dag.

(12) I bilag II til denne afgørelse redegøres for resultaterne af interkalibreringen, hvor interkalibreringen har været delvis vellykket. Alle de nødvendige faser i interkalibreringen skal afsluttes, således at resultaterne kan indarbejdes i en ny afgørelse. Disse resultater er således midlertidige.

(13) Medlemsstaterne bør afslutte interkalibreringen senest den 22. december 2016, således at Kommissionen kan indarbejde de resultater, der findes i bilag I og II til nærværende afgørelse, i et enkelt bilag til en ny afgørelse. Disse resultater vil herefter kunne danne grundlag for vandområdeplanernes tredje cyklus.

(14) Alle nødvendige faser i interkalibreringen bør også afsluttes senest den 22. december 2016 for de geografiske interkalibreringsgrupper og biologiske kvalitetselementer, hvor der endnu ikke er nogen interkalibreringsresultater, som kan indarbejdes i denne afgørelse. Disse resultater vil herefter ligeledes kunne indarbejdes i en ny afgørelse og anvendes i vandområdeplanernes tredje cyklus.

(15) Selv om interkalibreringen i henhold til direktiv 2000/60/EF skal udføres med hensyn til biologiske elementer, anses enkelte parametre (f.eks. en koncentration af chlorophyll-a eller dybdegrænser for makroalger og dækfrøede planter) i visse tilfælde som værende repræsentative for et helt biologisk kvalitetselement. I disse tilfælde anføres resultaterne af interkalibreringen i bilag I.

(16) I nogle tilfælde har medlemsstaterne udviklet særskilte metoder, der kun omfatter en del af et biologisk kvalitetselement (f.eks. særskilt metode for makrofyter og bundvegetation for kvalitetselementet »makrofyter og bundvegetation«). I tilfælde, hvor interkalibreringen for sådanne subbiologiske kvalitetselementer er blevet afsluttet med et vellykket resultat, indarbejdes resultaterne af interkalibreringen i bilagene og identificeres som et subbiologisk kvalitetselement.

[1] Common implementation strategy for the Water Framework Directive (2000/60/EC), Guidance Document No 6, Towards a Guidance on Establishment of the Intercalibration Network and the Process on the Intercalibration Exercise, European Communities, 2003, ISBN 92-894-5126-2.

[2] Common implementation strategy for the Water Framework Directive (2000/60/EC), Guidance document No. 14, Guidance document on the Intercalibration Process 2004-2006, ISBN 92-894-9471-9. Common implementation strategy for the Water Framework Directive (2000/60/EC), Guidance document No. 14, Guidance document on the Intercalibration Process 2008-2011, ISBN 978-92-79-18997-5.

[3] EUT L 332 af 10.12.2008, s. 20.

(17) Resultaterne af interkalibreringen skal henvise til vandområdernes økologiske tilstand. Hvis vandområder svarende til de interkalibrerede typer udpeges som stærkt modificerede vandområder i overensstemmelse med artikel 4, stk. 3, i direktiv 2000/60/EF, kan resultaterne i bilag I og II til nærværende afgørelse i overensstemmelse med de normgivende definitioner i bilag V, punkt 1.2.5, i direktiv 2000/60/EF anvendes til at udlede deres gode økologiske potentiale, idet der er taget hensyn til de fysiske forandringer og den tilhørende vandanvendelse.

(18) Medlemsstaterne skal overføre resultaterne af interkalibreringen til deres nationale klassifikationssystem med henblik på at fastsætte grænselinjerne mellem høj og god tilstand og mellem god og moderat tilstand for alle nationale typer.

(19) De oplysninger, der bliver tilgængelige i kraft af etableringen af overvågningsprogrammerne i henhold til artikel 8 i direktiv 2000/60/EF og den gennemgang og ajourføring af vandområdedistrikternes karakteristika, der er fastlagt i artikel 5 i direktiv 2000/60/EF, kan afdække ny viden, som igen kan føre til, at medlemsstaternes overvågnings- og klassifikationssystemer tilpasses det videnskabelige og tekniske fremskridt, og i sidste ende til en gennemgang af interkalibreringsresultaterne med henblik på at forbedre deres kvalitet.

(20) Der skal derfor vedtages bestemmelser, der ophæver og afløser beslutning 2008/915/EF.

(21) Foranstaltningerne i denne afgørelse er i overensstemmelse med udtalelse fra det udvalg, der er nedsat i henhold til artikel 21, stk. 1, i direktiv 2000/60/EF —

VEDTAGET DENNE AFGØRELSE:

Artikel 1

1. I forbindelse med punkt 1.4.1, nr. iii), i bilag V til direktiv 2000/60/EF skal medlemsstaterne for klassifikationen i deres overvågningssystemer anvende de værdier for grænselinjerne mellem tilstandsklasser, som er fastlagt i bilag I og II til denne afgørelse.

2. Medlemsstaterne skal afslutte alle nødvendige faser i interkalibreringen for resultaterne i bilag II til denne afgørelse senest den 22. december 2016.

Artikel 2

Beslutning 2008/915/EF ophæves.

Artikel 3

Denne afgørelse er rettet til medlemsstaterne.

Udfærdiget i Bruxelles, den 20. september 2013.

På Kommissionens vegne
Janez POTOČNIK
Medlem af Kommissionen

BILAG I

VANDKATEGORI: Vandløb

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Alpine

Beskrivelse af almindelige interkalibreringstyper

Type	Vandløbs-karakteristika	Oplands-areal (km^2)	Højde og geomorfologi	Alkalinitet	Strømnings-forhold
R-A1	Foralpin, lille til mellemstort, højland, kalkholdig	10-1 000	800-2 500 m (opland), rulle-/strandsten	Høj (men ikke ekstremt høj) alkalinitet	
R-A2	Lille til mellemstort, højland, silikatholdig	10-1 000	500-1 000 m (maks. oplands-højde 3 000 m, intermediær 1 500 m), rulle-sten	Ikke-kalkholdig (granit, metamor-fisk), mellem til lav alkalinitet	Sne/is-betingede strømnings-forhold

Lande med samme interkalibrerede typer

Type R-A1: Tyskland, Østrig, Frankrig, Italien og Slovenien

Type R-A2: Østrig, Frankrig, Italien og Spanien

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE ALPINE – VANDLØB

Biologisk kvalitetselement: Bentisk invertebratfauna

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type R-A1			
Østrig	Vurdering af de biologiske kvalitetselementer – bentiske invertebrater [Erhebung der biologischen Qualitätselemente – Teil Makrozoobenthos (Detaillierte MZB-Methode)]	0,80	0,60
Frankrig	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-350 og Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,93	0,79
Tyskland	PERLODES – Bewertungsverfahren von Fließgewässern auf Basis des Makrozoobenthos	0,80	0,60
Italien	MacrOper, baseret på STAR Intercalibration Common Metric Index (STAR_ICMi)	0,97	0,73
Slovenien	Metodologija vrednotenja ekološkega stanja rek z bentoškimi nevretenčarji v Sloveniji (system til vurdering af vandløbs økologiske tilstand ved hjælp af bentiske invertebrater i Slovenien)	0,80	0,60
Type R-A2			
Østrig	Vurdering af de biologiske kvalitetselementer – bentiske invertebrater [Erhebung der biologischen Qualitätselemente – Teil Makrozoobenthos (Detaillierte MZB-Methode)]	0,80	0,60

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Frankrig (Alperne)	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-350 og Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,93	0,71
Frankrig (Pyrenæerne)	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-350 og Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,94	0,81
Italien	MacrOper, baseret på STAR Intercalibration Common Metric Index (STAR_ICMi)	0,95	0,71
Spanien	Iberian BMWP (IBMWP)	0,83	0,53

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE ALPINE — VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Bundvegetation (fytobentos)

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type R-A1			
Østrig	Vurdering af de biologiske kvalitetselementer – bundvegetation [Leitfaden zur Erhebung der biologischen Qualitätselemente, Teil A3 – Fließgewässer/Phytobenthos]	0,88	0,56
Frankrig	IBD 2007 (Coste et al, Ecol. Ind. 2009). AFNOR NF-T-90-354, december 2007. Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,94	0,78
Tyskland	Verfahrensanleitung für die ökologische Bewertung von Fließgewässern zur Umsetzung der EG-Wasserrahmenrichtlinie: Makrophyten und Phytobenthos (Phylib), Modul Diatomeen	0,735	0,54
Italien	ICMi (Intercalibration Common Metric) Index (Mancini & Sollazzo, 2009, Phytobenthos Intercalibration Common Metric (pICM: Kelly et al., 2009)	0,87	0,70
Slovenien	Metodologija vrednotenja ekološkega stanja rek s fitobentosom in makrofiti v Sloveniji; fitobentos (system til vurdering af vandløbs økologiske tilstand ved hjælp af bundvegetation og makrofyter i Slovenien; bundvegetation)	0,80	0,60

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat

Type R-A2

Østrig	Vurdering af de biologiske kvalitetselementer – bundvegetation [Leitfaden zur Erhebung der biologischen Qualitätselemente, Teil A3 – Fließgewässer/Phytobenthos]	0,88	0,56
Frankrig	IBD 2007 (Coste et al, Ecol. Ind. 2009). AFNOR NF-T-90-354, december 2007. Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,94	0,78
Spanien	IPS (Coste in Cemagref, 1982)	0,94	0,74
Italien	ICMi (Intercalibration Common Metric) Index (Mancini & Sollazzo, 2009, Phytobenthos Intercalibration Common Metric (pICM: Kelly et al., 2009)	0,85	0,64

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE ALPINE – VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

FINDER IKKE ANVENDELSE

VANDKATEGORI: Vandløb

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Central/Baltic

Beskrivelse af almindelige interkalibreringstyper

Type	Vandløbs-karakteristika	Oplands-areal (km^2)	Højde og geomorfologi	Alkalinitet (meq/l)
R-C1	Lille, lavland, silikatholdig, sand	10-100	Lavland, domineret af sandbund (lille partikelstørrelse), bredde 3-8 m (bredfyldt)	> 0,4
R-C2	Lille, lavland, silikatholdig – sten	10-100	Lavland, stenmateriale, bredde 3-8 m (bredfyldt)	< 0,4
R-C3	Lille, intermediær, silikatholdig	10-100	Intermediær, stenbund (granit) til grusbund, bredde 2-10 m (bredfyldt)	< 0,4
R-C4	Mellemstort, lavland, blandet	100-1 000	Lavland, sandbund til grusbund, bredde 8-25 m (bredfyldt)	> 0,4
R-C5	Stort, lavland, blandet	1 000-10 000	Lavland, barbezone, variation i strømningshastighed, maks. oplandshøjde: 800 m, bredde > 25 m (bredfyldt)	> 0,4
R-C6	Lille, lavland, kalkholdig	10-300	Lavland, grusbund (kalksten), bredde 3-10 m (bredfyldt)	> 2

Lande med samme interkalibrerede typer

Type R-C1: Belgien (Flandern), Belgien (Vallonien), Tyskland, Danmark, Frankrig, Italien, Litauen, Nederlandene, Polen, Sverige og Det Forenede Kongerige

Type R-C2: Spanien, Frankrig, Irland, Portugal, Sverige og Det Forenede Kongerige

Type R-C3: Østrig, Belgien (Vallonien), Tjekkiet, Tyskland, Polen, Portugal, Spanien, Sverige, Frankrig, Letland, Luxembourg og Det Forenede Kongerige

Type R-C4: Belgien (Flandern), Belgien (Vallonien), Tjekkiet, Tyskland, Danmark, Estland, Spanien, Frankrig, Irland, Italien, Litauen, Luxembourg, Nederlandene, Polen, Sverige og Det Forenede Kongerige

Type R-C5: Belgien (Vallonien), Tjekkiet, Estland, Frankrig, Tyskland, Spanien og Irland. Italien, Letland, Litauen, Luxembourg, Nederlandene, Polen, Sverige og Det Forenede Kongerige

Type R-C6: Belgien (Vallonien), Danmark, Estland, Spanien, Frankrig, Irland, Italien, Polen, Litauen, Luxembourg, Sverige og Det Forenede Kongerige

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE CENTRAL-BALTIC – VANDLØB

Biologisk kvalitetselement: Bentisk invertebratfauna

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Følgende resultater gælder for alle typer som beskrevet ovenfor.

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	Vurdering af de biologiske kvalitetselementer – bentiske invertebrater	0,80	0,60
Belgien (Flandern)	Flamsk multimetrisk makroinvertebratindeks (MMIF)	0,90	0,70
Belgien (Vallonien)	Indice Biologique Global Normalisé (IBGN) (Norme AFNOR NF T 90 350, 1992) og Arrêté du Gouvernement wallon du 13 septembre 2012 relatif à l'identification, à la caractérisation et à la fixation des seuils d'état écologique applicables aux masses d'eau de surface et modifiant le Livre II du Code de l'Environnement, contenant le Code de l'Eau. Moniteur belge 12.10.2012	0,97 (type R-C3, R-C5, R-C6) 0,94 (type R-C1)	0,74 (type R-C3, R-C5, R-C6) 0,75 (type R-C1)
Tjekkiet	Tjekkisk system til vurdering af vandløbs økologiske tilstand ved hjælp af bentiske makroinvertebrater	0,80	0,60
Danmark	Dansk Vandløbsfauna-indeks (DVFI)	1,00	0,71
Estland	Estisk vurdering af overfladevands økologiske tilstand – makroinvertebrater i vandløb	0,90	0,70
Tyskland	PERLODES – Bewertungsverfahren von Fließgewässern auf Basis des Makrozoobenthos	0,80	0,60
Frankrig	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-350 og Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,94	0,80
Irland	Quality Rating System (Q-value)	0,85	0,75
Italien	MacrOper, baseret på STAR_ICM-indeksberegning	0,96	0,72
Luxembourg	Classification luxembourgeoise DCE Indice Biologique Global Normalisé (IBGN) 1992, AFNOR NF-T-90-350 og Circulaire DCE 2007/22 MEDD/DE/MAGE/BEMA 07/n° 4 du 11 avril 2007	0,96	0,72

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Nederlandene	KRW-maatlat	0,80	0,60
Polen	$RIVECO_{macro}$ til vurdering af vandløbs økologiske tilstand ved hjælp af bentiske makroinvertebrater (Multimetrisk makroinvertebratindeks, baseret på STAR_ICM)	0,91(type RC1)	0,72 (type RC1)
Spanien	METI	0,93	0,70
Sverige	DJ-index (Dahl & Johnson 2004)	0,80	0,60
Det Forenede Kongerige	River Invertebrate Classification Tool (RICT) – WHPT	0,97	0,86

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE CENTRAL-BALTIC – VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Land	Interkalibrerede nationale klassifikationssystemer	Type	Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	AIM for Rivers (Austrian Index Macrophytes for rivers – østrigsk makrofytindeks for vandløb)	RC-3	0,875	0,625
Belgien (Flandern)	MAFWAT – Flamsk system til vurdering af makrofyter	R-C1	0,80	0,60
Belgien (Vallonien)	IBMR-WL – Biologisk makrofytindeks for vandløb (Arrêté du Gouvernement wallon du 13 septembre 2012 relatif à l'identification, à la caractérisation et à la fixation des seuils d'état écologique applicables aux masses d'eau de surface et modifiant le Livre II du Code de l'Environnement, contenant le Code de l'Eau. Moniteur belge 12.10.2012)	R-C3	0,925	0,607
Danmark	DVPI – Dansk Vandløbsplante-indeks	R-C1	0,70	0,50
		R-C4	0,70	0,50
Tyskland	Verfahrensanleitung für die ökologische Bewertung von Fließgewässern zur Umsetzung der EG-Wasserrahmenrichtlinie: Makrophyten und Phytobenthos (Phylib), Modul Makrophyten	R-C1	0,745	0,495
		R-C3	0,80	0,55
		R-C4	0,575	0,395
Frankrig	Fransk standard NF T90-395 (2003-10-01). Qualité de l'eau - Détermination de l'indice biologique macrophytique en rivière (IBMR)	R-C3	0,93	0,79
		R-C4	0,905	0,79
Irland	MTR – IE – Mean Trophic Ranking	R-C4	0,74	0,62

Land	Interkalibrerede nationale klassifikationssystemer	Type	Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Italien	IBMR – IT – Biologisk makrofytindeks for vandløb	R-C1	0,90	0,80
		R-C4	0,90	0,80
Luxembourg	IBMR – LU – Biologisk makrofytindeks for vandløb	R-C3	0,89	0,79
		R-C4	0,89	0,79
Polen	MIR – Makrofytindeks for vandløb	R-C1	0,90	0,65
		R-C3	0,91	0,684
		R-C4	0,90	0,65
Det Forenede Kongerige	LEAFPACS – Økologisk klassifikation af vandløb ved hjælp af makrofyter	R-C1	0,80	0,60
		R-C3	0,80	0,60
		R-C4	0,80	0,60

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE CENTRAL-BALTIC VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Bundvegetation (fytobentos)

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Land	Interkalibrerede nationale klassifikationssystemer	Type	Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	Vurdering af de biologiske kvalitetselementer – bundvegetation [Leitfaden zur Erhebung der biologischen Qualitätselemente, Teil A3 – Fließgewässer/Phytobenthos]	Alle typer, højde < 500 m	0,70	0,42
		Alle typer, højde > 500 m	0,71	0,43
Belgien (Flandern)	Proportions of Impact-Sensitive and Impact-Associated Diatoms (PISIAD)	Alle typer	0,80	0,60
Belgien (Vallonien)	IPS (Coste, in CEMAGREF, 1982; Lenoir & Coste, 1996 og Arrêté du Gouvernement wallon du 13 septembre 2012 relatif à l'identification, à la caractérisation et à la fixation des seuils d'état écologique applicables aux masses d'eau de surface et modifiant le Livre II du Code de l'Environnement, contenant le Code de l'Eau. Moniteur belge 12.10.2012)	Alle typer	0,98	0,73
Estland	Indice de Polluosensibilité Spécifique (IPS)	Alle typer	0,85	0,70
Frankrig	IBD 2007 (Coste et al, Ecol. Ind. 2009). AFNOR NF-T-90-354, december 2007. Arrêté ministeriel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	Alle typer	0,94	0,78

Land	Interkalibrerede nationale klassifikationssystemer	Type	Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Tyskland	Verfahrensanleitung für die ökologische Bewertung von Fließgewässern zur Umsetzung der EG-Wasserrahmenrichtlinie: Makrophyten und Phytobenthos (Phylib), Modul Diatomeen	R-C1	0,67	0,43
		R-C3	0,67	0,43
		R-C4	0,61	0,43
		R-C5	0,73	0,55
Irland	Revideret udgave af det trofiske diatome indeks (TDI)	Alle typer	0,93	0,78
Italien	ICMi (Intercalibration Common Metric) Index (Mancini & Sollazzo, 2009, Phytobenthos Intercalibration Common Metric (pICM: Kelly et al., 2009)	Alle typer	0,84	0,65
Luxembourg	Indice de Polluosensibilité Spécifique (IPS)	Alle typer	0,90	0,70
Nederlandene	KRW Maatlat	Alle typer	0,80	0,60
Polen	Indeks Okrzemkowy IO dla rzek (det diatome indeks for vandløb)	Alle typer	0,80	0,58
Spanien	Diatome multimetriske indeks (MDIAT)	R-C2, R-C3, R-C4	0,93	0,70
Sverige	Svenske vurderingsmetoder, svenske EPA-bekendtgørelser (NFS 2008:1) baseret på Indice de Polluosensibilité Spécifique (IPS)	Alle typer	0,89	0,74
Det Forenede Kongerige	Diatom Assessment for River Ecological Status (DARLEQ2)	Alle typer	1,00	0,75

VANDKATEGORI: Vandløb

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Eastern Continental

Beskrivelse af almindelige interkalibreringstyper

Type	Vandløbskarakteristika	Øko-region	Oplands-areal (km^2)	Højde (m)	Geologi	Substrat
R-E1a	Karpaterne: Lille til mellemstort, intermediær	10	10 – 1 000	500 – 800	Blandet	
R-E1b	Karpaterne: Lille til mellemstort, intermediær	10	10 – 1 000	200 - 500	Blandet	
R-E2	Sletterne: Mellemstort, lavland	11 og 12	100 – 1 000	< 200	Blandet	Sand og dynd
R-E3	Sletterne: Stort, lavland	11 og 12	> 1 000	< 200	Blandet	Sand, dynd og grus
R-E4	Sletterne: Mellemstort, intermediær	11 og 12	100 – 1 000	200 – 500	Blandet	Sand og grus
R-EX4	Stort, intermediær	10, 11 og 12	> 1 000	200 - 500	Blandet	Grus og rullesten
R-EX5	Sletterne: Lille, lavland	11 og 12	10 - 100	< 200	Blandet	Sand og dynd

Type	Vandløbskarakteristika	Øko-region	Oplands-areal (km^2)	Højde (m)	Geologi	Substrat
R-EX6	Sletterne: Lille, intermediær	11 og 12	10 - 100	200 - 500	Blandet	Grus
R-EX7	Balkan: Lille, kalkholdig, intermediær	5	10-100	200-500	Kalkholdig	Grus
R-EX8	Balkan: Lille til mellemstort, kalkholdig karstkilde	5	10-1 000		Kalkholdig	Grus, sand og dynd

Lande med samme interkalibrerede typer

Type R-E1a: Bulgarien, Tjekkiet, Rumænien og Slovakiet

Type R-E1b: Bulgarien, Tjekkiet, Ungarn, Rumænien og Slovakiet

Type R-E2: Bulgarien, Tjekkiet, Ungarn, Rumænien og Slovakiet

Type R-E3: Bulgarien, Tjekkiet, Ungarn, Rumænien og Slovakiet

Type R-E4: Østrig, Bulgarien, Ungarn, Rumænien, Slovakiet og Slovenien

Type R-EX4: Tjekkiet, Rumænien og Slovakiet

Type R-EX5: Bulgarien, Ungarn, Rumænien, Slovenien og Slovakiet

Type R-EX6: Bulgarien, Ungarn, Rumænien og Slovenien

Type R-EX7: Slovenien

Type R-EX8: Bulgarien og Slovenien

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE EASTERN CONTINENTAL – VANDLØB

Biologisk kvalitetselement: Bentisk invertebratfauna

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Land	Interkalibrerede nationale klassifikationssystemer	Type	Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	Vurdering af de biologiske kvalitetselementer – bentiske invertebrater	R-E4	0,80	0,60
Bulgarien	Irsk biotisk indeks	R-E1a, R-E1b	0,86	0,67
Tjekkiet	Tjekkisk system til vurdering af vandløbs økologiske tilstand ved hjælp af bentiske makroinvertebrater	R-E1a, R-E1b, R-E2, R-E3	0,80	0,60
Ungarn	Ungarsk multimetrisk makroinvertebratindeks	R-E1b, R-E3, R-E4, R-EX5, R-EX6	0,80	0,60
Rumænien	Metode til vurdering af vandområders økologiske tilstand baseret på makroinvertebrater	R-E1a, R-E1b, R-E3, R-EX4	0,74	0,58
Slovenien	Metodologija vrednotenja ekološkega stanja rek z bentoškimi nevretenčarji v Sloveniji	R-E4, R-EX5, R-EX6	0,80	0,60
Slovakiet	Slovakisk vurdering af bentiske invertebrater i vandløb	R-E1a, R-E1b, R-E2, R-E3, R-E4, R-EX4	0,80	0,60

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE EASTERN CONTINENTAL – VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Land	Interkalibrerede nationale klassifikationssystemer	Type	Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	AIM for Rivers (Austrian Index Macrophytes for rivers – østrigsk makrofytindeks for vandløb)	R-E4	0,875	0,625
Bulgarien	Referenceindeks	R-E2, R-E3	0,570	0,370
Bulgarien	Referenceindeks	R-E4	0,510	0,270
Ungarn	Referenceindeks	R-E2, R-E3	0,700	0,370
Slovenien	Makrofytindeks for vandløb	R-E2, R-E3, R-E4	0,800	0,600
Slovakiet	Biologisk makrofytindeks for vandløb	R-E2, R-E3, R-E4	0,800	0,600

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE EASTERN CONTINENTAL – VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Bundvegetation (fytobentos)

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Land	Interkalibrerede nationale klassifikationssystemer	Type	Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	Vurdering af de biologiske kvalitetselementer – bundvegetation	R-E4	0,70	0,42
Bulgarien	Vurdering af vandløbs økologiske tilstand i Bulgarien baseret på det diatome indeks IPS	R-E1a, R-E1b, R-E3	0,87 (national type R2, R4) 0,85 (national type R7, R8)	0,66 (national type R2, R4) 0,64 (national type R7, R8)
Tjekkiet	System til vurdering af vandløb ved hjælp af bundvegetation	R-E1a, R-E1b, R-E2, R-E3, R-EX4	0,80	0,60
Ungarn	Vurdering af vandløbs økologiske tilstand baseret på diatoméer	R-E2, R-E3, R-EX5	0,80	0,60
Slovenien	Metodologija vrednotenja ekološkega stanja rek s fitobentosom in makrofiti v Sloveniji; fitobentos (system til vurdering af vandløbs økologiske tilstand ved hjælp af bundvegetation og makrofyter i Slovenien; bundvegetation)	R-E4, R-EX5, R-EX6, R-EX7, R-EX8	0,80	0,60
Slovakiet	System til vurdering af vandløbs økologiske tilstand ved hjælp af bundvegetation	R-E1a, R-E1b, R-E2, R-E3, R-E4, R-EX4	0,90	0,70

VANDKATEGORI: Vandløb

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Mediterranean

Beskrivelse af almindelige interkalibreringstyper

Type	Vandløbskarakteristika	Oplandsareal (km^2)	Geologi	Strømningsforhold
R-M1	Lille vandløb i det mediterrane område	< 100	Blandet (undtagen silikatholdig)	Meget sæsonbetingede
R-M2	Mellemstort vandløb i det mediterrane område	100-1 000	Blandet (undtagen silikatholdig)	Meget sæsonbetingede
R-M4	Bjergvandløb i det mediterrane område		Ikkesilikatholdig	Meget sæsonbetingede
R-M5	Udtørrende vandløb			Udtørrende

Lande med samme interkalibrerede typer

Type R-M1: Frankrig, Grækenland, Italien, Portugal, Slovenien og Spanien

Type R-M2: Frankrig, Grækenland, Italien, Portugal, Slovenien og Spanien

Type R-M4: Cypern, Frankrig, Grækenland, Italien og Spanien

Type R-M5: Cypern, Italien, Portugal, Slovenien og Spanien.

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN – VANDLØB

Biologisk kvalitetselement: Bentisk invertebratfauna

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
R-M1			
Frankrig	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-350 og Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,940	0,700
Italien	MacrOper (baseret på STAR Intercalibration Common Metric Index ICMi)	0,970	0,720
Portugal	Rivers Biological Quality Assessment Method-Benthic Invertebrates (IPtIN, IPtIS)	0,870 (type 1) 0,850 (type 3)	0,678 (type 1) 0,686 (type 3)
Slovenien	Metodologija vrednotenja ekološkega stanja rek z bentoškimi nevretenčarji v Sloveniji (system til vurdering af vandløbs økologiske tilstand ved hjælp af bentiske invertebrater i Slovenien)	0,800	0,600
Spanien	Iberian Biological Monitoring Working Party (IBMWP)	0,845	0,698
Spanien	Iberian Mediterranean Multimetric Index – baseret på kvantitative data (IMMi-T)	0,811	0,707

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
R-M2			
Frankrig	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-350 og Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,940	0,700
Italien	MacrOper (baseret på STAR Intercalibration Common Metric Index ICMi)	0,940	0,700
Portugal	Rivers Biological Quality Assessment Method-Benthic Invertebrates (IPtIN, IPtIS)	0,830 (type 2) 0,880 (type 4)	0,693 (type 2) 0,676 (type 4)
Slovenien	Metodologija vrednotenja ekološkega stanja rek z bentoškimi nevretenčarji v Sloveniji (system til vurdering af vandløbs økologiske tilstand ved hjælp af bentiske invertebrater i Slovenien)	0,800	0,600
Spanien	Iberian Biological Monitoring Working Party (IBMWP)	0,845	0,698
Spanien	Iberian Mediterranean Multimetric Index – baseret på kvantitative data (IMMi-T)	0,811	0,707
R-M4			
Frankrig	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-350 og Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,940	0,700
Cypern	STAR Intercalibration Common Metric Index (STAR_ICMi)	0,972	0,729
Italien	MacrOper (baseret på STAR Intercalibration Common Metric Index ICMi)	0,940	0,700
Spanien	Iberian Biological Monitoring Working Party (IBMWP)	0,840	0,700
Spanien	Iberian Mediterranean Multimetric Index – baseret på kvantitative data (IMMi-T)	0,850	0,694
R-M5			
Cypern	STAR Intercalibration Common Metric Index (STAR_ICMi)	0,982	0,737
Italien	MacrOper (baseret på STAR Intercalibration Common Metric Index ICMi)	0,970	0,730
Portugal	Rivers Biological Quality Assessment Method-Benthic Invertebrates (IPtIN, IPtIS)	0,973 (type 5) 0,961 (type 6)	0,705 (type 5) 0,708 (type 6)
Slovenien	Metodologija vrednotenja ekološkega stanja rek z bentoškimi nevretenčarji v Sloveniji (system til vurdering af vandløbs økologiske tilstand ved hjælp af bentiske invertebrater i Slovenien)	0,800	0,600

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Spanien	Iberian Biological Monitoring Working Party (IBMWP)	0,830	0,630
Spanien	Iberian Mediterranean Multimetric Index – baseret på kvantitative data (IMMi-T)	0,830	0,620

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN – VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

Resultater: økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
R-M1, 2, 4			
Cypern	IBMR – Biologisk makrofytindeks for vandløb	0,795	0,596
Frankrig	Fransk standard NF T90-395 (2003-10-01) Qualité de l'eau - Détermination de l'indice biologique macrophytique en rivière (IBMR)	0,930	0,745
Grækenland	IBMR – Biologisk makrofytindeks for vandløb	0,750	0,560
Italien	IBMR – Biologisk makrofytindeks for vandløb	0,900	0,800
Portugal	IBMR – Biologisk makrofytindeks for vandløb	0,920	0,690
Slovenien	RMI – Makrofytindeks for vandløb	0,800	0,600
Spanien	IBMR – Biologisk makrofytindeks for vandløb	0,950	0,740

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN – VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Bundvegetation (fytobentos)

Resultater: økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
R-M1			
Frankrig	IBD 2007 (Coste et al, Ecol. Ind. 2009). AFNOR NF-T-90-354, december 2007. Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,940	0,780
Italien	ICMi (Intercalibration Common Metric) Index (Mancini & Sollazzo, 2009)	0,800	0,610

Type og land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Portugal	IPS (Coste in Cemagref, 1982)	0,970 (type 1) 0,910 (type 3)	0,730 (type 1) 0,680 (type 3)
Slovenien	Metodologija vrednotenja ekološkega stanja rek s fitobentosom in makrofiti v Sloveniji; fitobentos (system til vurdering af vandløbs økologiske tilstand ved hjælp af bundvegetation og makrofyter i Slovenien; bundvegetation)	0,800	0,600
Spanien	IPS (Coste in Cemagref, 1982)	0,937	0,727
R-M2			
Frankrig	IBD 2007 (Coste et al, Ecol. Ind. 2009). AFNOR NF-T-90-354, december 2007. Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,940	0,780
Italien	ICMi (Intercalibration Common Metric) Index (Mancini & Sollazzo, 2009)	0,800	0,610
Portugal	IPS (Coste in Cemagref, 1982)	0,910 (type 2) 0,970 (type 4)	0,680 (type 2) 0,730 (type 4)
Slovenien	Metodologija vrednotenja ekološkega stanja rek s fitobentosom in makrofiti v Sloveniji; fitobentos (system til vurdering af vandløbs økologiske tilstand ved hjælp af bundvegetation og makrofyter i Slovenien; bundvegetation)	0,800	0,600
Spanien	IPS (Coste in Cemagref, 1982)	0,938	0,727
R-M4			
Cypern	IPS (Coste in Cemagref, 1982)	0,910	0,683
Frankrig	IBD 2007 (Coste et al, Ecol. Ind. 2009). AFNOR NF-T-90-354, december 2007. Arrêté ministériel du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	0,940	0,780
Italien	ICMi (Intercalibration Common Metric) Index (Mancini & Sollazzo, 2009)	0,800	0,610
Spanien	IPS (Coste in Cemagref, 1982)	0,935	0,727
R-M5			
Cypern	IPS (Coste in Cemagref, 1982)	0,958	0,718
Italien	ICMi (Intercalibration Common Metric) Index (Mancini & Sollazzo, 2009)	0,880	0,650
Portugal	IPS (Coste in Cemagref, 1982)	0,940	0,700
Slovenien	Metodologija vrednotenja ekološkega stanja rek s fitobentosom in makrofiti v Sloveniji; fitobentos (system til vurdering af vandløbs økologiske tilstand ved hjælp af bundvegetation og makrofyter i Slovenien; bundvegetation)	0,800	0,600
Spanien	IPS (Coste in Cemagref, 1982)	0,935	0,700

VANDKATEGORI: Vandløb

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Northern

Beskrivelse af almindelige interkalibreringstyper

<table>
<tr><th>Type</th><th>Vandløbskarakteristika</th><th>Oplandsareal (km2)</th><th>Højde og geomorfologi</th><th>Alkalinitet (meq/l)</th><th>Organisk materiale (mg Pt/l)</th></tr>
<tr><td>R-N1</td><td>Lille, lavland, silikatholdig, moderat alkalinitet</td><td>10-100 km^2</td><td rowspan="3">< 200 m eller under den højeste kystlinje</td><td>0,2 - 1</td><td>< 30
(< 150 i Irland)</td></tr>
<tr><td>R-N3</td><td>Lille til mellemstort, lavland, organisk, lav alkalinitet</td><td>10-1 000 km^2</td><td>< 0,2</td><td>> 30</td></tr>
<tr><td>R-N4</td><td>Mellemstort, lavland, silikatholdig, moderat alkalinitet</td><td>100-1 000 km^2</td><td>0,2 - 1</td><td>< 30</td></tr>
<tr><td>R-N5</td><td>Lille, intermediær, silikatholdig, lav alkalinitet</td><td>10-100 km^2</td><td>Mellem lavland og højland</td><td>< 0,2</td><td>< 30</td></tr>
</table>

Lande med samme interkalibrerede typer

Type R-N1: Finland, Irland, Norge, Sverige og Det Forenede Kongerige

Type R-N3: Finland, Irland, Norge, Sverige og Det Forenede Kongerige

Type R-N4: Finland, Norge, Sverige og Det Forenede Kongerige

Type R-N5: Finland, Norge, Sverige og Det Forenede Kongerige

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTHERN – VANDLØB

Biologisk kvalitetselement: Bentisk invertebratfauna (metoder, der er følsomme over for organisk berigelse og generel nedbrydning)

Resultater: økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Følgende resultater gælder for alle typer som beskrevet ovenfor.

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Finland	Multimetrisk system, første version fastlagt	0,80	0,60
Irland	Quality Rating System (Q-value)	0,85	0,75
Norge	ASPT	0,99	0,87
Sverige	DJ-index (Dahl & Johnson 2004)	0,80	0,60
Det Forenede Kongerige	River Invertebrate Classification Tool (RICT) – WHPT	0,97	0,86

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTHERN – VANDLØB

Biologisk kvalitetselement: Bentisk invertebratfauna (metoder, der er følsomme over for forsuring)

Resultater: økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Følgende resultater gælder for alle typer klare vandløb med lav alkalinitet

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Norge	AcidIndex2 (Modified Raddum index2) (forsuring af vandløb)	0,675	0,515
Det Forenede Kongerige – Skotland	WFD-AWICsp: WFD Acid Water Indicator Community species	0,910	0,830
Det Forenede Kongerige – England og Wales	WFD-AWICsp: WFD Acid Water Indicator Community species	0,980	0,890

Resultater: økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Følgende resultater gælder for alle typer humøse vandløb med lav alkalinitet

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Sverige	MISA: Multimetric Invertebrate Stream Acidification index – multimetrisk invertebratindeks for forsuring i vandløb	0,550	0,400
Det Forenede Kongerige	WFD-AWICsp: WFD Acid Water Indicator Community species	0,930	0,830

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTHERN – VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Bundvegetation (fytobentos)

Resultater: økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Følgende resultater gælder for alle typer som beskrevet ovenfor.

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Finland	Indice de Polluosensibilité Spécifique (IPS)	0,91	0,80
Sverige	Indice de Polluosensibilité Spécifique (IPS)	0,89	0,74
Irland	Revideret udgave af det trofiske diatome indeks (TDI)	0,93	0,78
Det Forenede Kongerige	DARLEQ 2	1,00	0,75
Norge	Periphytonindeks for trofisk tilstand (PIT)	0,99 (Ca ≤ 1 mg/L) 0,95 (Ca > 1 mg/L)	0,83

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTHERN – VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

INTERKALIBRERING IKKE AFSLUTTET

VANDKATEGORI: Vandløb

GEOGRAFISKE INTERKALIBRERINGSGRUPPER: Alle

BIOLOGISK KVALITETSELEMENT: Fiskefauna

Oversigt over regionale grupper, som er blevet fastlagt for interkalibreringen af fisk i vandløb

Gruppen lavland – inde i landet –Belgien (Flandern), Belgien (Vallonien), Frankrig, Tyskland, Nederlandene, Litauen, Luxembourg, Det Forenede Kongerige (England og Wales), Polen, Letland, Estland, Danmark og Ungarn

Den nordiske gruppe – Finland, Irland, Sverige, Det Forenede Kongerige (Skotland og Nordirland) og Norge

Gruppen alpebjerge – Østrig, Frankrig, Tyskland, Slovenien

Den mediterrane og sydatlantiske gruppe – Portugal, Spanien, Italien og Grækenland

Donaugruppen – Tjekkiet, Rumænien, Slovakiet og Bulgarien

Resultater: økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Gruppen Lowland-Midland

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Belgien (Flandern)	IBI, opstrøms og lavland	0,850	0,650
Belgien (Vallonien)	IBIP (Arrêté du Gouvernement wallon du 13 septembre 2012 relatif à l'identification, à la caractérisation et à la fixation des seuils d'état écologique applicables aux masses d'eau de surface et modifiant le Livre II du Code de l'Environnement, contenant le Code de l'Eau. Moniteur belge 12.10.2012)	0,958	0,792
Frankrig	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-344. Arrêté du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	1,131	0,835
Tyskland	FIBS – fischbasiertes Bewertungssystem für Fließgewässer zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland	1,086	0,592
Luxembourg	Classification française DCE Indice Biologique Global Normalisé (IBGN). AFNOR NF-T-90-344. Arrêté du 25 janvier 2010 modifié relatif aux méthodes et critères d'évaluation de l'état écologique {…} des eaux de surface	1,131	0,835
Nederlandene	NLFISR	0,800	0,600
Litauen	LZI	0,940	0,720

Den nordiske gruppe

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Finland	Finsk fiskeindeks (FiFi) – type L2	0,665	0,499
Finland	Finsk fiskeindeks (FiFi) – type L3	0,658	0,493
Finland	Finsk fiskeindeks (FiFi) – type M1	0,709	0,532
Finland	Finsk fiskeindeks (FiFi) – type M2	0,734	0,550
Finland	Finsk fiskeindeks (FiFi) – type M3	0,723	0,542
Irland	FCS2 IRELAND	0,845	0,540
Sverige	Den svenske metode VIX	0,739	0,467
Det Forenede Kongerige – Nordirland	IR_FCS2	0,845	0,540
Det Forenede Kongerige – Skotland	FCS2 Scotland	0,850	0,600

Den mediterrane gruppe

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Portugal	F_IBIP	0,850	0,675
Spanien	IBIMED – type T2	0,816	0,705
Spanien	IBIMED – type T3	0,929	0,733
Spanien	IBIMED – type T4	0,864	0,758
Spanien	IBIMED – type T5	0,866	0,650
Spanien	IBIMED – type T6	0,916	0,764

Donaugruppen

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Tjekkiet	Den tjekkiske multimetriske metode CZI	0,780	0,585
Rumænien	EFI+ det europæiske fiskeindeks (cyprinid_wading type)	0,939	0,700
Rumænien	EFI+ det europæiske fiskeindeks (salmonid type)	0,911	0,755
Slovakiet	Det slovakiske fiskeindeks FIS	0,710	0,570

Alpegruppen

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	FIA	0,875	0,625
Frankrig	FBI	1,131	0,876
Tyskland	FIBS – fischbasiertes Bewertungssystem für Fließgewässer zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland	1,086	0,592
Slovenien	SIFAIR	0,800	0,600

VANDKATEGORI: Vandløb

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Alle – Meget store vandløb

Beskrivelse af almindelige interkalibreringstyper

Type	Vandløbskarakteristika	Oplandsareal (km^2)	Alkalinitet (meq/l)
R-L1	Meget store vandløb med lav alkalinitet	> 10 000 km^2	< 0,5
R-L2	Meget store vandløb med mellem til høj alkalinitet	> 10 000 km^2	> 0,5

Lande med samme interkalibrerede typer

Type R-L1: Finland, Norge og Sverige

Type R-L2: Østrig, Belgien (Flandern), Bulgarien, Kroatien, Tjekkiet, Estland, Frankrig, Tyskland, Grækenland, Ungarn, Italien, Letland, Nederlandene, Norge, Polen, Portugal, Rumænien, Slovakiet, Slovenien, Spanien og Sverige

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEGET STORE VANDLØB

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Bundvegetation (fytobentos)

Resultater: økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Følgende resultater gælder for meget store vandløb med lav alkalinitet (type R-L1)

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Finland	Indice de Polluosensibilité Spécifique (Specific Pollution Sensitivity Index SPI)	0,80	0,60
Sverige	Bentiske alger i vandløb – diatomanalyse	0,89	0,74

Følgende resultater gælder for meget store vandløb med mellem til høj alkalinitet (type R-L2)

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	Vurdering af de biologiske kvalitetselementer – bundvegetation	0,85	0,57
Tjekkiet	System til vurdering af vandløb ved hjælp af bundvegetation	0,80	0,60

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Estland	Estisk vurdering af overfladevands økologiske tilstand – bundvegetation i vandløb	0,83	0,64
Tyskland	Verfahrensanleitung für die ökologische Bewertung von Fließgewässern zur Umsetzung der EG-Wasser-rahmenrichtlinie: Makrophyten und Phytobenthos (Phylib), Modul Diatomeen	0,725	0,545
Ungarn	Vurdering af vandløbs økologiske tilstand baseret på diatoméer	0,762	0,60
Nederlandene	VRD-metrikker for naturlige vandområder	0,80	0,60
Slovakiet	System til vurdering af vandløbs økologiske tilstand ved hjælp af bundvegetation	0,90	0,70
Slovenien	Metodologija vrednotenja ekološkega stanja rek s fitobentosom in makrofiti v Sloveniji; fitobentos (system til vurdering af vandløbs økologiske tilstand ved hjælp af bundvegetation og makrofyter i Slovenien; bundvegetation)	0,80	0,60

VANDKATEGORI: Vandløb

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Alle – Meget store vandløb

BIOLOGISKE KVALITETSELEMENTER: Makrofyter, fytoplankton, fisk, bentiske invertebrater

INTERKALIBRERING IKKE AFSLUTTET

VANDKATEGORI: Søer

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Alpine

Beskrivelse af almindelige interkalibreringstyper

Type	Søkarakteristika	Højde (moh)	Middeldybde (m)	Alkalinitet (meq/l)	Søens størrelse (km^2)
L-AL3	Lavland eller intermediær, dyb, moderat til høj alkalinitet (alpin indflydelse), stor	50 - 800	> 15	> 1	> 0,5
L-AL4	Intermediær, lavvandet, moderat til høj alkalinitet (alpin indflydelse), stor	200 - 800	3 - 15	> 1	> 0,5

Lande med samme interkalibrerede typer

Type L-AL3: Østrig, Frankrig, Tyskland, Italien og Slovenien

Type L-AL4: Østrig, Frankrig, Tyskland og Italien

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE ALPINE – SØER

Biologisk kvalitetselement: Fytoplankton

Medlemsstat	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	Vurdering af de biologiske kvalitetselementer – del B2 – fytoplankton	0,80	0,60

Medlemsstat	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Tyskland	PSI (Phyto-Seen-Index) – Bewertungsverfahren für Seen mittels Phytoplankton zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland	0,80	0,60
Italien	Italian Phytoplankton Assessment Method (IPAM) – italiensk fytoplanktonbaseret vurderingsmetode	0,80	0,60
Slovenien	Metodologija vrednotenja ekološkega stanja jezer s fitoplanktonom v Sloveniji (system til vurdering af søers økologiske tilstand ved hjælp af fytoplankton i Slovenien)	0,80	0,60

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE ALPINE – SØER

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

Medlemsstat	Interkalibrerede nationale klassifikationssystemer		Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	AIM for Lakes (Austrian Index Macrophytes for lakes – østrigsk makrofytindeks for søer)	L-AL3 + L-AL4	0,80	0,60
Frankrig	IBML (fransk makrofytindeks for søer)	L-AL3+ L-AL4	0,92	0,72
Tyskland	PHYLIB for Lakes (det tyske system til vurdering af makrofyter/bundvegetation for søer som led i gennemførelsen af vandrammedirektivet): Modul Makrofyter	L-AL3+ L-AL4	0,76	0,51
Tyskland	PHYLIB for Lakes (det tyske system til vurdering af makrofyter/bundvegetation for søer som led i gennemførelsen af vandrammedirektivet): Moduler Makrofyter og bundvegetation	LAL4	0,74	0,47
Italien	MacroIMMI (makrofytindeks til vurdering af de italienske søers økologiske tilstand)	L-AL3+ L-AL4	0,80	0,60
Slovenien	SMILE (det slovenske makrofytbaserede indeks for økosystemer i søer)	L-AL3	0,80	0,60

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE ALPINE – SØER

Biologisk kvalitetselement: Bentiske invertebrater

Medlemsstat	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Slovenien	Metodologija vrednotenja ekološkega stanja jezer z bentoškimi nevretenčarji v Sloveniji (system til vurdering af søers økologiske tilstand ved hjælp af bentiske invertebrater i Slovenien)	0,80	0,60
Tyskland	AESHNA – Bewertungsverfahren für das eulitorale Makrozoobenthos in Seen zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland	0,80	0,60

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE ALPINE — SØER

Biologisk kvalitetselement: Fiskefauna

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Østrig	ALFI (Austrian lake fish index): Et multimetrisk indeks til vurdering af alpine søers økologiske tilstand baseret på fiskefauna.	0,80	0,60
Tyskland	DELAFI_SITE — Deutsches probennahmestandort-spezifisches Bewertungsverfahren für Fische in Seen zur Umsetzung der EG-Wasserrahmenrichtlinie	0,85	0,69
Italien	Lake Fish Index (LFI)	0,82	0,64

VANDKATEGORI: Søer

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Central/Baltic

Beskrivelse af almindelige interkalibreringstyper

Type	Søkarakteristika	Højde (moh)	Middeldybde (m)	Alkalinitet (meq/l)	Opholdstid (år)
L-CB1	Lavland, lavvandet, kalkholdig	< 200	3 - 15	> 1	1 - 10
L-CB2	Lavland, meget lavvandet, kalkholdig	< 200	< 3	> 1	0,1 - 1

Lande med samme interkalibrerede typer

Type L-CB1: Belgien, Tyskland, Danmark, Estland, Irland, Litauen, Letland, Nederlandene, Polen og Det Forenede Kongerige

Type L-CB2: Belgien, Tyskland, Danmark, Estland, Irland, Litauen, Letland, Nederlandene, Polen og Det Forenede Kongerige

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE CENTRAL/BALTIC — SØER

Biologisk kvalitetselement: Fytoplankton

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Belgien (Flandern)	Flamsk fytoplanktonbaseret vurderingsmetode for søer	0,80	0,60
Tyskland	PSI (Phyto-See-Index) — Bewertungsverfahren für Seen mittels Phytoplankton zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland — tysk fytoplanktonindeks for søer (Phyto-See-Index)	0,80	0,60
Danmark	Dansk fytoplanktonindeks	0,80	0,60
Estland	Estisk vurdering af overfladevands økologiske tilstand — fytoplankton i søer	0,80	0,60

	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Irland	IE Lake Phytoplankton Index – irsk fytoplanktonindeks for søer	0,80	0,60
Nederlandene	VRD-metrikker for naturlige vandområder	0,80	0,60
Polen	Phytoplankton method for Polish Lakes (PMPL) – fytoplanktonbaseret metode for polske søer	0,80	0,60
Det Forenede Kongerige	Phytoplankton Lakes Assessment Tool (PLUTO) – fytoplanktonbaseret redskab til vurdering af søer	0,80	0,60

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE CENTRAL/BALTIC – SØER

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationssystemer	IC type	Økologiske kvalitetsratioer	
			Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Belgien (Flandern)	Flamsk system til vurdering af makrofyter	Alle typer	0,80	0,60
Danmark	Dansk makrofytindeks for søer	Alle typer	0,80	0,60
Estland	Estisk vurdering af overfladevands økologiske tilstand – makrofyter i søer	LCB1	0,78	0,52
		LCB2	0,76	0,50
Tyskland	Verfahrensanleitung für die ökologische Bewertung von Seen zur Umsetzung der EG-Wasserrahmenrichtlinie: Makrophyten und Phytobenthos (Phylib), Modul Makrophyten)	Alle typer	0,80	0,60
Litauen	Litauisk metode til vurdering af makrofyter	Alle typer	0,75	0,50
Letland	Lettisk metode til vurdering af makrofyter	Alle typer	0,80	0,60
Nederlandene	VRD-metrikker for naturlige vandområder	Alle typer	0,80	0,60
Polen	Makrofytbaseret vurderingsmetode for søer – Ecological Status Macrophyte Index ESMI (multimetrisk)	Alle typer	0,68	0,41
Det Forenede Kongerige	LEAFPACS makrofytbaseret klassifikationsredskab for søer (*)	Alle typer	0,80	0,66

(*) Vil blive anvendt i England, Wales og Skotland

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE CENTRAL/BALTIC – SØER

Biologisk kvalitetselement: Bentiske invertebrater

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Belgien (Flandern)	Flamsk multimetrisk makroinvertebratindeks (MMIF)	0,90	0,70

Medlemsstat	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Tyskland	AESHNA – Bewertungsverfahren für das eulitorale Makrozoobenthos in Seen zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland	0,80	0,60
Estland	Estisk vurdering af overfladevands økologiske tilstand – makroinvertebrater i søer	0,86	0,70
Litauen	Litauisk makroinvertebratindeks for søer	0,74	0,50
Nederlandene	VRD-metrik for naturlige vandområder	0,80	0,60
Det Forenede Kongerige	Chironomid Pupal Exuvial Technique (CPET)	0,77	0,64

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE CENTRAL/BALTIC – SØER

Biologisk kvalitetselement: Fiskefauna

INTERKALIBRERING IKKE AFSLUTTET

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE EASTERN CONTINENTAL – SØER

Biologisk kvalitetselement: Fytoplankton

INTERKALIBRERING IKKE AFSLUTTET

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE EASTERN CONTINENTAL – SØER

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

INTERKALIBRERING IKKE AFSLUTTET

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE EASTERN CONTINENTAL – SØER

Biologisk kvalitetselement: Bentiske invertebrater

INTERKALIBRERING IKKE AFSLUTTET

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE EASTERN CONTINENTAL – SØER

Biologisk kvalitetselement: Fiskefauna

INTERKALIBRERING IKKE AFSLUTTET

VANDKATEGORI: Søer

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Mediterranean

Beskrivelse af almindelige interkalibreringstyper

Type	Søkarakteristika	Højde (m)	Årlig middelnedbør (mm) og temperatur (°C)	Middeldybde (m)	Areal (km^2)	Oplandsareal (km^2)	Alkalinitet (meq/l)
L-M5/7	Reservoir, dyb, stor, **silikatholdig, "våd**områder"	< 1 000	> 800 og/eller < 15	> 15	0,5-50	< 20 000	< 1
L-M8	Reservoir, dyb, stor, **silikatholdig**	< 1 000	—	> 15	0,5-50	< 20 000	> 1

Lande med samme interkalibrerede typer

Type L-M5/7: Grækenland, Frankrig, Italien, Portugal, Rumænien og Spanien

Type L-M8: Cypern, Frankrig, Italien, Rumænien og Spanien

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN – SØER

Biologisk kvalitetselement: Fytoplankton

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
LM 5/7			
Spanien	Mediterranean Assessment System for Reservoirs Phytoplankton (MASRP) – mediterant system til vurdering af fytoplankton i reservoir	Ingen data (*)	0,58
Portugal	Metode til vurdering af den biologiske kvalitet i reservoir – fytoplankton (New Mediterranean Assessment System for Reservoirs Phytoplankton: NMASRP).	Ingen data	0,60
Italien	Ny italiensk metode (NITMET)	Ingen data	0,60
L-M8			
Spanien	Mediterranean Assessment System for Reservoirs Phytoplankton (MASRP) – mediterant system til vurdering af fytoplankton i reservoir	Ingen data	0,60
Cypern	New Mediterranean Assessment System for Reservoirs Phytoplankton (NMASRP) – nyt mediterant system til vurdering af fytoplankton i reservoir	Ingen data	0,60
Italien	Ny italiensk metode (NITMET)	Ingen data	0,60

(*) Grænselinje mellem høj og god er ikke defineret for reservoir (både LM5/7- og LM8-typer er reservoir)

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN – SØER

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

INTERKALIBRERING IKKE AFSLUTTET

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN – SØER

Biologisk kvalitetselement: Bentiske invertebrater

INTERKALIBRERING IKKE AFSLUTTET

DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN – SØER

Biologisk kvalitetselement: Fiskefauna

INTERKALIBRERING IKKE AFSLUTTET

VANDKATEGORI: Søer

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Northern

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTHERN – SØER

Biologisk kvalitetselement: Fytoplankton

Beskrivelse af almindelige interkalibreringstyper

Type	Søkarakteristika	Højde (moh)	Middeldybde (m)	Alkalinitet (meq/l)	Farve (mg Pt/l)
L-N1	Lavland, lavvandet, moderat alkalinitet, klart	< 200	3 - 15	0,2 - 1	< 30
L-N2a	Lavland, lavvandet, lav alkalinitet, klart	< 200	3 - 15	< 0,2	< 30
L-N2b	Lavland, dyb, lav alkalinitet, klart	< 200	> 15	< 0,2	< 30
L-N3a	Lavland, lavvandet, lav alkalinitet, mesohumøst	< 200	3 - 15	< 0,2	30 - 90
L-N5	Intermediær, lavvandet, lav alkalinitet, klart	200-800	3 - 15	< 0,2	< 30
L-N6a	Intermediær, lavvandet, lav alkalinitet, mesohumøst	200-800	3 - 15	< 0,2	30 - 90
L-N8a	Lavland, lavvandet, moderat alkalinitet, mesohumøst	< 200	3 - 15	0,2 - 1	30 - 90

Lande med samme interkalibrerede typer

Type L-N1, L-N2a, L-N3a, LN-8a: Irland, Finland, Norge, Sverige og Det Forenede Kongerige

Type LN-2b: Norge, Sverige og Det Forenede Kongerige

Type LN-5, LN-6a: Norge og Sverige

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Finland	Finsk fytoplanktonbaseret vurderingsmetode for søer	0,80	0,60
Irland	IE Lake Phytoplankton Index – irsk fytoplanktonindeks for søer	0,80	0,60
Norge	Fytoplanktonbaseret metode til klassifikation af søers økologiske tilstand	0,80	0,60
Sverige	Metoder til vurdering af søers økologiske tilstand – kvalitetsfaktor fytoplankton	0,80	0,60
Det Forenede Kongerige	Phytoplankton Lakes Assessment Tool (PLUTO) – fytoplanktonbaseret redskab til vurdering af søer	0,80	0,60

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTHERN – SØER

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Makrofyter

Beskrivelse af almindelige interkalibreringstyper

Type	Søkarakteristika	Alkalinitet (meq/l)	Farve (mg Pt/l)
L-N-M 101	Lav alkalinitet, klart	0,05 - 0,2	< 30

Type	Søkarakteristika	Alkalinitet (meq/l)	Farve (mg Pt/l)
L-N-M 102	Lav alkalinitet, humøst	0,05 - 0,2	> 30
L-N-M 201	Moderat alkalinitet, klart	0,2 - 1,0	< 30
L-N-M 202	Moderat alkalinitet, humøst	0,2 - 1,0	> 30
L-N-M 301a	Høj alkalinitet, klart, atlantisk undertype	> 1,0	< 30
L-N-M 302a	Høj alkalinitet, humøst, atlantisk undertype	> 1,0	> 30

Lande med samme interkalibrerede typer

Type 101, 102, 201 og 202: Irland, Finland, Norge, Sverige og Det Forenede Kongerige

Type 301a: Irland og Det Forenede Kongerige

Type 302a: Irland og Det Forenede Kongerige

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Finland	Finsk makromytbaseret klassifikationssystem (Finnmac)	0,8 (alle typer)	0,6 (alle typer)
Irland	Free Macrophyte Index	0,9 (alle typer)	0,68 (alle typer)
Norge	Nationalt makrofytindeks (Trophic Index – TIc)	Type 101: 0,98 Type 102: 0,96 Type 201: 0,95 Type 202: 0,99	Type 101: 0,87 Type 102: 0,87 Type 201: 0,75 Type 202: 0,77
Sverige	Trofisk makrofytindeks (TMI)	Type 101: 0,93 Type 102: 0,93 Type 201: 0,89 Type 202: 0,91	Type 101: 0,80 Type 102: 0,83 Type 201: 0,78 Type 202: 0,78
Det Forenede Kongerige	LEAFPACS makrofytbaseret klassifikationsredskab for søer (*)	0,8 (alle typer)	0,66 (alle typer)
Det Forenede Kongerige	Free Macrophyte Index (**)	0,9 (alle typer)	0,68 (alle typer)

(*) Vil blive anvendt i England, Wales og Skotland
(**) Vil også blive anvendt i Det Forenede Kongerige (Nordirland)

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTHERN – SØER

Biologisk kvalitetselement: Bentiske invertebrater

Beskrivelse af almindelige interkalibreringstyper

Type	Søkarakteristika	Økoregion	Højde (m absl)	Alkalinitet (meq/l)	Farve (mg Pt/l)
Forsuring af søkyster					
L-N-BF1	Lavland/intermediær, lav alkalinitet, klart	Ingen data	< 800	0,05 - 0,2	< 30
Dyb eutrofiering i søer					
L-N-BF2	Økoregion 22, lav alkalinitet, klart og humøst	22	Areal > 1 km^2, maks. dybde > 6 m	< 0,2	Ingen data

Lande med samme interkalibrerede typer

Type L-N-BF1: Norge, Sverige, Det Forenede Kongerige, Irland og Finland

Type L-N-BF2: Finland og Sverige

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Forsuring af søkyster			
Sverige	MILA: Multimetric Invertebrate Stream Acidification index – multimetrisk invertebratindeks for forsuring i søer	0,85	0,60
Det Forenede Kongerige	LAMM (Lake Acidification Macroinvertebrate Metric – makroinvertebratindeks for forsuring i søer)	0,86	0,70
Norge	MultiClear: Multimetric Invertebrate Index for Clear Lakes – multimetrisk invertebratindeks for klare søer	0,95	0,74
Dyb eutrofiering i søer			
Sverige	BQI (Bentisk kvalitetsindeks)	0,84	0,67
Finland	BQI (Bentisk kvalitetsindeks)	0,75	0,63

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTHERN – SØER

Biologisk kvalitetselement: Fiskefauna

Beskrivelse af almindelige interkalibreringstyper

Type	Søkarakteristika	Søens størrelse km^2	Alkalinitet (meq/l)	Farve (mg Pt/l)
L-N-F1	Dimiktiske søer med klart vand	< 40	< 0,2	< 30
L-N-F2	Dimiktiske humøse søer	< 5	< 0,2	30-90

Lande med samme interkalibrerede typer

Type L-N-F1: Irland, Finland, Norge, Sverige og Det Forenede Kongerige

Type L-N-F2: Irland, Finland, Norge, Sverige og Det Forenede Kongerige

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Finland	EQR4	0,80	0,60
Irland	FIL2	0,76	0,53
Det Forenede Kongerige (Nordirland)	FIL2	0,76	0,53

VANDKATEGORI: Søer

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Kryds-GIG bundvegetation

Beskrivelse af almindelige interkalibreringstyper

Type	Søkarakteristika	Alkalinitet (meq/l)	Økoregioner
HA	Søer med høj alkalinitet	> 1	Central-Baltic, Mediterranean
MA	Søer med moderat alkalinitet	0,2-1	Central-Baltic, Northern
LA	Søer med lav alkalinitet	< 0,2	Northern

Lande med samme interkalibrerede typer

Type HA: Belgien, Tyskland, Ungarn, Irland, Italien, Polen, Sverige, Slovenien og Det Forenede Kongerige

Type MA: Belgien, Frankrig, Finland, Irland, Sverige, Det Forenede Kongerige

Type LA: Finland, Irland, Sverige og Det Forenede Kongerige

KRYDS-GIG INTERKALIBRERINGSRESULTATER FOR SØER

Biologisk kvalitetselement: Makrofyter og bundvegetation

Subbiologisk kvalitetselement: Bundvegetation (fytobentos)

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
HA-type			
Belgien (Flandern)	Proportions of Impact-Sensitive and Impact-Associated Diatoms (PISIAD)	0,80	0,60
Tyskland	Verfahrensanleitung für die ökologische Bewertung von Seen zur Umsetzung der EG-Wasserrahmenrichtlinie: Makrophyten und Phytobenthos (Phylib), Modul Phytobenthos	0,80	0,55
Ungarn	MIL – Multimetrisk indeks for søer	0,80	0,69
Irland	Trofisk diatomindeks for søer (IE)	0,90	0,63
Polen	PL IOJ (Multimetryczny Indeks Okrzemkowy dla Jezior = Multimetrisk diatomindeks for søer)	0,91	0,76
Sverige	IPS	0,89	0,74
Slovenien	Trofisk indeks (TI)	0,80	0,60
Det Forenede Kongerige	DARLEQ 2	0,92	0,70
MA-type			
Belgien (Flandern)	Proportions of Impact-Sensitive and Impact-Associated Diatoms (PISIAD)	0,80	0,60
Finland	IPS	0,80	0,64

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Irland	Trofisk diatomindeks for søer (IE)	0,90	0,63
Sverige	IPS	0,89	0,74
Det Forenede Kongerige	DARLEQ 2	0,93	0,66
LA-type			
Irland	Trofisk diatomindeks for søer (IE)	0,90	0,66
Det Forenede Kongerige	DARLEQ 2	0,92	0,70

VANDKATEGORI: Kystvande/overgangsvande

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Baltic

Beskrivelse af almindelige interkalibreringstyper

Type	Saltindhold på overfladen psu	Saltindhold på bunden	Eksponering	Isdage	Andre karakteristika
BT 1	0-8 Oligohalint	0 - 8	Meget beskyttet	—	Vistulabugten i Polen og Den Kuriske Bugt i Litauen
BC1	0,5 - 6 Oligohalint	1 -6	Eksponeret	90 - 150	Lokaliteter mellem Quark og Det Botniske Hav og Skærgårdshavet (for fytoplankton er sidstnævnte ikke omfattet, men integreret i type BC9) Humøse stoffers indvirkning
BC3	3 - 6 Oligohalint	3 - 6	Beskyttet	90 - 150	Finske og estiske kyster ved Den Finske Bugt
BC4	5 - 8 Nedre mesohalint	5 - 8	Beskyttet	< 90	Estiske og lettiske lokaliteter i Rigabugten
BC5	6 - 8 Nedre mesohalint	6 - 12	Eksponeret	< 90	Lokaliteter i den sydøstlige del af Østersøen langs Letlands, Litauens og Polens kyst
BC6	8 - 12 Mellem mesohalint	8 -12	Beskyttet	< 90	Lokaliteter langs den vestlige Østersø ved den sydlige svenske kyst og den sydøstlige danske kyst
BC7	6 - 8 Mellem mesohalint	8 - 11	Eksponeret	< 90	Den polske vestkyst og den tyske østkyst
BC8	13 -18 Øvre mesohalint	18 -23	Beskyttet	< 90	Danske og tyske kyster i den vestlige Østersø
BC9	3 - 6 Nedre mesohalint	3 - 6	Moderat eksponeret til eksponeret	90 - 150	Lokaliteter i den vestlige del af Den Finske Bugt, Skærgårdshavet og Askö (kun for fytoplankton)

Lande med samme interkalibrerede typer

Kystvande

Type BC1: Finland og Sverige

Type BC3: Finland og Estland

Type BC4: Estland og Letland

Type BC5: Litauen, Letland og Polen

Type BC6: Sverige og Danmark

Type BC7: Tyskland og Polen

Type BC8: Tyskland og Danmark

Type BC9: Finland, Sverige og Estland (type, der kun er relevant for fytoplankton)

Overgangsvande

Type BT1: Litauen og Polen

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE BALTIC

Biologisk kvalitetselement: Bentisk invertebratfauna

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Kystvande

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
BC1			
Finland	BBI – Finsk bentisk indeks for brakvand	0,96	0,56
Sverige	BQI – Svensk multimetrisk biologisk kvalitetsindeks (infauna i blødt sediment)	0,77	0,31
BC3			
Estland	ZKI – Estisk indeks for makrozoobenthos i kystvande	0,39	0,24
Finland	BBI – Finsk bentisk indeks for brakvand	0,94	0,56
BC6			
Danmark	DKI ver2 – Dansk kvalitetsindeks version 2	0,84	0,68
Sverige	BQI – Svensk multimetrisk biologisk kvalitetsindeks (infauna i blødt sediment)	0,76	0,27
BC8			
Danmark	DKI ver2 – Dansk kvalitetsindeks version 2	0,86	0,72
Tyskland	MarBIT – Marint biotisk indeksredskab	0,8	0,6

Overgangsvande:

INTERKALIBRERING IKKE AFSLUTTET

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE BALTIC

Biologisk kvalitetselement: Fytoplankton

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Kystvande

Medlemsstat	Nationale interkalibrerede klassifikationsmetoder	Økologiske kvalitetsratioer i de nationale klassifikationssystemer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
BC7			
Tyskland	Tysk fytoplanktonbaseret metode for kystvande	0,8	0,6
Polen	Polsk fytoplanktonbaseret metode for kystvande	0,8	0,6
BC8			
Danmark	Dansk fytoplanktonbaseret metode for kystvande	0,8	0,6
Tyskland	Tysk fytoplanktonbaseret metode for kystvande	0,8	0,6

Resultater for parameter, der indikerer biomasse (klorofyl a): SE BILAG II

Overgangsvande:

INTERKALIBRERING IKKE AFSLUTTET

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE BALTIC

Biologisk kvalitetselement: Makroalger og dækfrøede planter

Kystvande

Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
BC3			
Estland	EPI – Estisk indeks for bundvegetation i kystvande (makroalger og dækfrøede planter)	0,98	0,86
Finland	Dybdegrænse for Fucus (makroalger)	0,92	0,79

Økologiske kvalitetsratioer og parameterværdier for parameter, der indikerer abundans (dybdegrænse for ålegræs *Zostera marina*): Økologiske kvalitetsratioer og parameterværdier

Type og land	Økologiske kvalitetsratioer i de nationale klassifikationssystemer		Parameterværdier/-områder Dybdegrænse (m) Ålegræs *Zostera marina*	
	Grænselinje mellem høj og god	Grænselinje mellem god og moderat	Grænselinje mellem høj og god	Grænselinje mellem god og moderat
BC8				
Danmark og Tyskland Åben kyst	0,90	0,74	8,5	7

Overgangsvande:

INTERKALIBRERING IKKE AFSLUTTET

VANDKATEGORI: Kystvande/overgangsvande

GEOGRAFISK INTERKALIBRERINGSGRUPPE: North-East Atlantic

Beskrivelse af almindelige interkalibreringstyper

Type	Karakterisering	Saltindhold (psu) Tidevandsområde (m) Dybde (m)	Bølgehastighed (knob) Eksponering	Blanding Opholdstid
Type opportunistisk blomsterbærende makroalger, havgræs, harrilsiv og bentisk invertebratfauna				
NEA1/26	Åbent oceansk eller lukket hav, eksponeret eller beskyttet, euhalint, lavvandet	> 30 Mesotidevand 1-5 < 30	Medium 1-3 Eksponeret eller beskyttet	Fuldt blandet Dage (til uger i Vade-havet)
Undertyper af makroalger i tidevandszonen				
NEA1/26 A2	Åbent oceansk, eksponeret eller beskyttet, euhalint, lavvandet Tempererede vande (primært, > 13 °C) og høj stråling (primært, PAR > 29 Mol/m^2 dag)	> 30 Mesotidevand 1–5 < 30	Medium 1-3 Eksponeret eller beskyttet	Fuldt blandet Dage
NEA1/26 B21	Åbent oceansk eller lukket hav, eksponeret eller beskyttet, euhalint, lavvandet Afkølede vande (primært, > 13 °C) og medium stråling (primært, PAR > 29 Mol/m^2 dag)	> 30 Primært mesotide-vand 1–5 < 30	Medium 1-3 Eksponeret eller beskyttet	Fuldt blandet Dage
Subtyper af fytoplankton				
NEA1/26a	Åbent oceansk, eksponeret eller beskyttet, euhalint, lavvandet	> 30 Mesotidevand 1-5 < 30	Medium 1-3 Eksponeret eller beskyttet	Fuldt blandet Dage
NEA1/26b	Lukket hav, eksponeret eller beskyttet, euhalint, lavvandet	> 30 Mesotidevand 1-5 < 30	Medium 1-3 Eksponeret eller beskyttet	Fuldt blandet Dage
NEA1/26c	Lukket hav, lukket eller beskyttet, delvis stratificeret	> 30 Mikro-/mesotidevand < 1-5 < 30	Medium 1-3 Eksponeret eller beskyttet	Delvis stratificeret Dage til uger
NEA1/26d	Skandinavisk kyst, eksponeret eller beskyttet, lavvandet	> 30 Mikrotidevand < 1 < 30	Lav < 1 Eksponeret eller moderat eksponeret	Delvis stratificeret Dage til uger
NEA1/26e	Områder med upwelling, eksponeret eller beskyttet, euhalint, lavvandet	> 30 Mesotidevand 1-5 < 30	Medium 1-3 Eksponeret eller beskyttet	Fuldt blandet Dage
Typer fytoplankton, makroalger, havgræs, harrilsiv, bentisk invertebratfauna og fisk (overgangsvande)				
NEA3/4	Polyhalint, eksponeret eller moderat eksponeret (Vade-havstype)	Polyhalint 18-30 Mesotidevand 1-5 < 30	Medium 1-3 Eksponeret eller moderat eksponeret	Fuldt blandet Dage
NEA7	Fjordsystem	> 30 Mesotidevand 1-5 > 30	Lav < 1 Beskyttet	Fuldt blandet Dage

Type	Karakterisering	Saltindhold (psu) Tidevandsområde (m) Dybde (m)	Bølgehastighed (knob) Eksponering	Blanding Opholdstid
NEA8a	Indre Skagerrak bueformet, polyhalint, mikrotidevand, moderat eksponeret, lavvandet	Polyhalint 25-30 Mikrotidevand < 1 > 30	Lav < 1 Moderat eksponeret	Fuldt blandet Dage til uger
NEA8b	Indre Skagerrak bueformet, polyhalint, mikrotidevand, moderat beskyttet, lavvandet	Polyhalint 10-30 Mikrotidevand < 1 < 30	Lav < 1 Beskyttet til moderat eksponeret	Delvis stratificeret Dage til uger
NEA9	Fjord med en lavvandet forhøjning ved mundingen og meget stor maksimal dybde i det centrale bækken med ringe vandskifte i dybden	Polyhalint 25-30 Mikrotidevand < 1 > 30	Lav < 1 Beskyttet	Delvis stratificeret Uger
NEA10	Ydre Skagerrak bueformet, polyhalint, mikrotidevand, eksponeret, dybt	Polyhalint 25-30 Mikrotidevand < 1 > 30	Lav < 1 Eksponeret	Delvis stratificeret Dage
NEA11	Overgangsvande	Oligohalint 0-35 Mikro- til makrotidevand < 30	Variabel Beskyttet eller moderat eksponeret	Delvis permanent stratificeret Dage til uger

Lande med samme interkalibrerede typer

Kystvande

Type NEA1/26 opportunistisk blomsterbærende makroalger, søgræs, harrilsiv:	Belgien, Frankrig, Tyskland, Irland, Nederlandene, Portugal, Spanien og Det Forenede Kongerige
Type NEA1/26 A2 makroalger i tidevandszonen:	Frankrig, Spanien og Portugal
Type NEA1/26 B21 makroalger i tidevandszonen:	Frankrig, Irland, Norge og Det Forenede Kongerige
Type NEA1/26a fytoplankton:	Spanien, Frankrig, Irland, Norge og Det Forenede Kongerige
Type NEA1/26b fytoplankton:	Belgien, Frankrig, Nederlandene og Det Forenede Kongerige
Type NEA1/26c fytoplankton:	Tyskland og Danmark
Type NEA1/26d fytoplankton:	Danmark
Type NEA1/26e fytoplankton:	Portugal og Spanien
Type NEA3/4:	Tyskland og Nederlandene
Type NEA7:	Norge og Det Forenede Kongerige
Type NEA8a:	Norge og Sverige
Type NEA8b:	Danmark og Sverige
Type NEA9:	Norge og Sverige
Type NEA10:	Norge og Sverige

Overgangsvande

Type NEA11: Belgien, Tyskland, Spanien, Frankrig, Irland, Nederlandene, Portugal og Det Forenede Kongerige

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTH-EAST ATLANTIC

Biologisk kvalitetselement: Bentisk invertebratfauna

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Resultaterne gælder kun for habitater i blødt sediment (mudder-/sandhabitater under tidevandszonen).

Kystvande

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type NEA8b			
Danmark	DKI	0,84	0,68
Sverige	BQI	0,71	0,54
Types NEA8a/9/10			
Norge	NQI	0,82	0,63
Sverige	BQI	0,71	0,54

Resultater for kystvande, TYPE NEA 1/26 OG NEA7: SE BILAG II

Overgangsvande:

INTERKALIBRERING IKKE AFSLUTTET

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTH-EAST ATLANTIC

Biologisk kvalitetselement: Fytoplankton

Kystvande

Fytoplankton: Parameter, der indikerer biomasse (klorofyl a)

Resultater: Økologiske kvalitetsratioer og parameterværdier

Parameterværdierne udtrykkes i µg/l som 90-percentilværdien beregnet over den definerede vækstsæson i en seksårig periode. Resultaterne relaterer sig til geografiske områder inden for typerne som beskrevet i den tekniske rapport.

Medlemsstat	Økologiske kvalitetsratioer		Værdier (µg/l, 90-percentil)	
	Grænselinje mellem høj og god	Grænselinje mellem god og moderat	Grænselinje mellem høj og god	Grænselinje mellem god og moderat
NEA1/26c				
Danmark	0,67	0,44	5	7,5
Tyskland	0,67	0,44	5	7,5

Resultater for kystvande, TYPE NEA 1/26a, NEA 1/26b, NEA1/26e, NEA 3/4, NEA9, NEA10: SE BILAG II

Overgangsvande:

INTERKALIBRERING IKKE AFSLUTTET

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTH-EAST ATLANTIC

Biologisk kvalitetselement: Makroalger og dækfrøede planter

Kystvande

Resultater: Makroalger – parameter for makroalger på sublittoral klippegrund eller klippegrund i tidevandszonen

Kystvande

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type NEA1/26 A2 makroalger i tidevandszonen			
Frankrig	CCO – Cover, Characteristic species, Opportunistic species on intertidal rocky bottoms – dækning, karakteristiske arter, opportunistiske arter på klippegrund i tidevandszonen	0,80	0,60
Portugal	PMarMAT – Redskab til vurdering af marine makroalger	0,80	0,61
Spanien	CFR – Klippegrundens kvalitet	0,81	0,60
Spanien	RICQI – Rocky Intertidal Community Quality Index – kvalitetsindeks for klippesamfund i tidevandszonen	0,82	0,60
Spanien	RSL – Reduced Species List – reduceret artsfortegnelse	0,75	0,48
Type NEA1/26 B21 makroalger i tidevandszonen			
Irland	RSL – Reduced Species List – klippekyst, reduceret artsfortegnelse	0,80	0,60
Norge	RSLA – Rocky Shore Reduced Species List – klippekyst, reduceret artsfortegnelse	0,80	0,60
Det Forenede Kongerige	RSL – Reduced Species List – klippekyst, reduceret artsfortegnelse	0,80	0,60
Type NEA7 makroalger i tidevandszonen			
Norge	RSLA – Rocky Shore Reduced Species List – klippekyst, reduceret artsfortegnelse med abundans	0,80	0,60
Det Forenede Kongerige	RSL – Reduced Species List – klippekyst, reduceret artsfortegnelse	0,80	0,60
Type NEA8a/9/10 makroalger i tidevandszonen			
Norge	MSMDI – Multi Species Maximum Depth Index – indeks for flere arter, maksimal dybde	0,80	0,60
Sverige	MSMDI – Multi Species Maximum Depth Index – indeks for flere arter, maksimal dybde	0,80	0,60

Resultater for makroalger – parameter for blomsterbærende makroalger i tidevandszonen – Type NEA1/26: SE BILAG II

Overgangsvande:

Resultater for makroalger – parameter for blomsterbærende makroalger i tidevandszonen – NEA11: SE BILAG II

Resultater: Blomsterplanter – subBQE, der indikerer havgræs

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Kystvande

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type NEA3/4			
Tyskland	SG – Bewertungssystem für Makroalgen und Seegräser der Küsten- und Übergangsgewässer zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland	0,80	0,60
Nederlandene	Overvågede havgræsenge pr. vandområde ved hjælp af luftfotos, feltundersøgelse og angivelse af overflade og tæthed pr. art	0,80	0,60

Resulter for blomsterplanter (subBQE, der indikerer havgræs) – type 1/26: SE BILAG II

Overgangsvande:

Resulter for blomsterplanter (subBQE, der indikerer havgræs) – NEA11: SE BILAG II

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTH-EAST ATLANTIC

Biologisk kvalitetselement: Fisk (Overgangsvande)

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Belgien	EBI – Zeeschelde estuarint biotisk indeks	0,85	0,615
Frankrig	ELFI – Estuarine and Lagoon Fish Index – delta- og lagunefiskeindeks	0,91	0,675
Tyskland	FAT – TW – Fischbasiertes Bewertungswerkzeug für Übergangsgewässer der norddeutschen Ästuare	0,84	0,62
Irland	TFCI – Transitional Fish Classification Index – indeks til klassifikation af fisk i overgangsvande	0,81	0,58
Nederlandene	FAT – TW – WFD Fish index for transitional waters, type O2 – VRD-fiskeindeks for overgangsvande	0,80	0,60
Portugal	EFAI – Estuarine Fish Assessment Index – indeks til vurdering af estuarine fisk	0,865	0,70
Spanien	AFI – AZTI's fiskeindeks	0,78	0,55
Spanien	TFCI – Transitional Fish Classification Index – indeks til klassifikation af fisk i overgangsvande	0,90	0,65
Det Forenede Kongerige (Nordirland)	TFCI – Transitional Fish Classification Index – indeks til klassifikation af fisk i overgangsvande	0,81	0,58

VANDKATEGORI: Kystvande/overgangsvande

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Mediterranean

Typologi med regionale almindelige interkalibreringstyper er kun blevet defineret for fytoplankton (se nedenfor).

For bentisk invertebratfauna, makroalger og havgræs gælder interkalibreringsresultaterne for hele Middelhavsområdet i medlemsstaterne.

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN

Biologisk kvalitetselement: Bentisk invertebratfauna

Resultater: Økologiske kvalitetsratioer i de nationale klassifikationssystemer

Kystvande

Følgende resultater gælder kun for bløde sedimenter.

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Metoder, der omfatter diversitetsparameter			
Italien	M-AMBI	0,81	0,61
Slovenien	M-AMBI	0,83	0,62
Metoder, der ikke omfatter diversitetsparameter			
Cypern	Bentix	0,75	0,58
Frankrig	AMBI	0,83	0,58
Grækenland	Bentix	0,75	0,58
Spanien	BOPA	0,95	0,54
Spanien	MEDOCC	0,73	0,47

Overgangsvande:

INTERKALIBRERING IKKE AFSLUTTET

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN

Biologisk kvalitetselement: Fytoplankton

Beskrivelse af typer for kystvande, der er blevet interkalibreret (gælder kun fytoplankton)

Type	Beskrivelse	Tæthed (kg/m^3)	Årlig middelværdi for saltindhold (psu)
Type I	Meget påvirket af ferskvandstilførsel	< 25	< 34,5
Type IIA: *IIA Adriatic*	Moderat påvirket af ferskvandstilførsel (kontinentalindflydelse)	25-27	34,5-37,5
Type IIIW	Kontinentalkyst, ikke påvirket af ferskvandstilførsel (vestlige bækken)	> 27	> 37,5
Type IIIE:	Ikke påvirket af ferskvandstilførsel (østlige bækken)	> 27	> 37,5
Type Island-W	Økyst (vestlige bækken)	Alle områder	Alle områder

Lande med samme interkalibrerede typer

Type I: Frankrig og Italien

Type IIA: Frankrig, Spanien og Italien

Type IIA Adriatic: Italien og Slovenien

Type Island-W: Frankrig, Spanien og Italien

Type IIIW: Frankrig, Spanien og Italien

Type IIIE: Grækenland og Cypern

Kystvande

Resultater for parameter, der indikerer biomasse (klorofyl a): SE BILAG II

Overgangsvande:

INTERKALIBRERING IKKE AFSLUTTET

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE MEDITERRANEAN

Biologisk kvalitetselement: Makroalger og dækfrøede planter

Kystvande

Makroalger: sub-BQE, der indikerer makroalger og dækfrøede planter

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Følgende resultater gælder øvre infralittorale zone (3,5-0,2 m dybde) ved klippekyster:

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Cypern	EEI-c – Ecological Evaluation Index – økologisk evalueringsindeks	0,76	0,48
Frankrig	CARLIT – Kartografi over littorale og øvre sublit-terale klippekystsamfund	0,75	0,60
Grækenland	EEI-c – Ecological Evaluation Index – økologisk evalueringsindeks	0,76	0,48
Italien	CARLIT – Kartografi over littorale og øvre sublit-terale klippekystsamfund	0,75	0,60
Slovenien	EEI-c – Ecological Evaluation Index – økologisk evalueringsindeks	0,76	0,48
Spanien	CARLIT – Kartografi over littorale og øvre sublit-terale klippekystsamfund	0,75	0,60

Havgræs: sub-BQE, der indikerer makroalger og dækfrøede planter

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Cypern	PREI – Posidonia oceanica Rapid Easy Index	0,775	0,55
Frankrig	PREI – Posidonia oceanica Rapid Easy Index	0,775	0,55
Italien	PREI – Posidonia oceanica Rapid Easy Index	0,775	0,55
Spanien	POMI – Posidonia oceanica Multivariate Index	0,775	0,55
Spanien	Valencian-CS	0,775	0,55

Makroalger og dækfrøede planter

Overgangsvande:

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Medlemsstat	Interkalibrerede nationale klassifikationsmetoder	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Frankrig	Exclame	0,80	0,60
Grækenland	EEI-c – Ecological Evaluation Index – økologisk evalueringsindeks	0,70	0,40
Italien	MaQI – Makrofytkvalitetsindeks	0,80	0,60

VANDKATEGORI: Kystvande/overgangsvande

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Black Sea

Beskrivelse af almindelige interkalibreringstyper

Type	Beskrivelse
CW-BL1	Kystvande Mesohalint, mikrotidevand (< 1 m), lavvandet (< 30 m), moderat eksponeret, blandet bund

Lande med samme interkalibrerede typer

Bulgarien og Rumænien

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE BLACK SEA

Biologisk kvalitetselement: Fytoplankton

Kystvande

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Bulgarien	IBI	0,80	0,63
Rumænien	IBI	0,80	0,63

BILAG II

VANDKATEGORI: Kystvande/overgangsvande

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Baltic

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE BALTIC

Biologisk kvalitetselement: Fytoplankton

Resultater for parameter, der indikerer biomasse (klorofyl a): Økologiske kvalitetsratioer og parameterværdier

Følgende resultater henviser til sommergennemsnitsværdier maj/juni-september

Kystvande

Medlemsstat	Økologiske kvalitetsratioer i de nationale klassifikationssystemer		Parameterværdier/-områder Klorofyl a (μg/l)	
	Grænselinje mellem høj og god	Grænselinje mellem god og moderat	Grænselinje mellem høj og god	Grænselinje mellem god og moderat
BC1				
Finland	0,76	0,59	1,7	2,2
Sverige	0,87	0,65	1,5	2,0
BC9				
Estland	0,82	0,67	2,2	2,7
Finland	0,79	0,65	1,9	2,3
Sverige	0,80	0,67	1,5	1,8

VANDKATEGORI: Kystvande/overgangsvande

GEOGRAFISK INTERKALIBRERINGSGRUPPE: North-East Atlantic

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTH-EAST ATLANTIC

Biologisk kvalitetselement: Bentisk invertebratfauna

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Resultaterne gælder kun for habitater i blødt sediment (mudder-/sandhabitater under tidevandszonen).

Kystvande

Type NEA 1/26 og NEA7

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type NEA1/26 og NEA 7 (Indekser, der primært afspejler organisk berigelse og giftig forurening, som belaster habitater i blødt sediment)			
Danmark	DKI	0,67	0,53
Frankrig	M-AMBI	0,77	0,53
Tyskland	M-AMBI	0,85	0,70
Irland	IQI	0,75	0,64

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Nederlandene	BEQI2	0,78	0,58
Norge	NQI	0,92	0,81
Portugal	P-BAT	0,79	0,58
Spanien	M-AMBI	0,77	0,53
Spanien	BO2A	0,78	0,44
Det Forenede Kongerige	IQI	0,75	0,64
Type NEA1/26 (Indekser, der afspejler multiple belastninger i multiple habitater)			
Belgien	BEQI	0,80	0,60

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTH-EAST ATLANTIC

Biologisk kvalitetselement: Fytoplankton

Kystvande

Fytoplankton: Parameter, der indikerer biomasse (klorofyl a)

Resultater: Økologiske kvalitetsratioer og parameterværdier

Parameterværdierne udtrykkes i µg/l som 90-percentilværdien beregnet over den definerede vækstsæson i en seksårig periode. Resultaterne relaterer sig til geografiske områder inden for typerne som beskrevet i den tekniske rapport.

Type	Økologiske kvalitetsratioer		Værdier (µg/l, 90-percentil)	
	Grænselinje mellem høj og god	Grænselinje mellem god og moderat	Grænselinje mellem høj og god	Grænselinje mellem god og moderat
NEA 1/26a				
Frankrig	0,67	0,33	5	10
Irland	0,67	0,33	5	10
Norge	0,67	0,33	2,5	5
Sydspanien	0,67	0,33	5	10
Nordspanien Den østlige del af Det Cantabriske Hav	0,67	0,33	1,5	3
Spanien, den norlige og centrale del af Det Cantabriske Hav	0,67	0,33	3	6
Det Forenede Kongerige	0,67	0,33	5	10
NEA1/26b				
Belgien	0,67	0,44	10	15
Frankrig	0,67	0,44	10	15
Nederlandene	0,67	0,44	10	15
Det Forenede Kongerige	0,67	0,44	10	15
NEA3/4				
Tyskland	0,66	0,44	7-10	11-15

Type	Økologiske kvalitetsratioer		Værdier (µg/l, 90-percentil)	
	Grænselinje mellem høj og god	Grænselinje mellem god og moderat	Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Nederlandene	0,66	0,44	10-14	15-21
NEA1/26e				
Portugal	0,67	0,44	6 – 8	9 – 12
Spanien	0,67	0,44	6 – 8	9 – 12
NEA9				
Norge	0,67	0,33	2,5	5
Sverige	0,67	0,33	2,5	5
NEA10				
Norge	0,67	0,33	3	6
Sverige	0,67	0,33	3	6

RESULTATER FOR DEN GEOGRAFISKE INTERKALIBRERINGSGRUPPE NORTH-EAST ATLANTIC

Biologisk kvalitetselement: Makroalger og dækfrøede planter

Makroalger: parameter for blomsterbærende makroalger i tidevandszonen i blød bund, der indikerer abundans

Resultater: Økologiske kvalitetsratioer for interkalibrerede nationale parametre

Kystvande

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type NEA1/26			
Tyskland	Bewertungssystem für opportunistische Makroalgen auf eulitoralen Weichböden der Küstengewässer	0,80	0,60
Irland	OGA-redskab – Abundans af opportunistiske grønne makroalger	0,80	0,60
Det Forenede Kongerige	OMBT-redskab for opportunistiske blomsterbærende makroalger	0,80	0,60

Overgangsvande

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type NEA11			
Irland	OGA-redskab – Abundans af opportunistiske grønne makroalger	0,80	0,60
Portugal	BMI – Indeks for blomsterbærende makroalger (vurdering af blomsterbærende makroalger)	0,80	0,60
Det Forenede Kongerige	OMBT-redskab for opportunistiske blomsterbærende makroalger	0,80	0,60

Resultater: Blomsterplanter – sub-BQE, der indikerer makroalger og dækfrøede planter

Kystvande:

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Land	Interkalibrerede nationale klassifikationssystemer	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type NEA1/26			
Frankrig	SBQ – Kvaliteten af havgræsenge i kyst- og overgangsvandområder	0,80	0,60
Tyskland	SG – Bewertungssystem für Makroalgen und Seegräser der Küsten- und Übergangsgewässer zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland	0,80	0,60
Irland	Havgræsabundans i tidevandszonen og artssammensætning	0,80	0,63

Overgangsvande:

Resultater: Økologiske kvalitetsratioer i de interkalibrerede nationale klassifikationssystemer

Type og land	Interkalibrerede nationale parametre	Økologiske kvalitetsratioer	
		Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type NEA11			
Frankrig	SBQ – Kvaliteten af havgræsenge i kyst- og overgangsvandområder	0,80	0,60
Tyskland	SG – Bewertungssystem für Makroalgen und Seegräser der Küsten- und Übergangsgewässer zur Umsetzung der EG-Wasserrahmenrichtlinie in Deutschland	0,80	0,60
Irland	Havgræsabundans i tidevandszonen og artssammensætning	0,83	0,70
Nederlandene	Overvågede havgræsenge pr. vandområde ved hjælp af luftfotos, feltundersøgelse og angivelse af overflade og tæthed pr. art	0,80	0,60
Portugal	SQI – Seagrass quality index for intertidal TW – indeks for kvaliteten af havgræs i overgangsvande i tidevandszonen	0,80	0,60

VANDKATEGORI: Kystvande/overgangsvande

GEOGRAFISK INTERKALIBRERINGSGRUPPE: Mediterranean

Biologisk kvalitetselement: Fytoplankton

Fytoplankton: Parameter, der indikerer biomasse (klorofyl a)

Kystvande

Resultater: Økologiske kvalitetsratioer og parameterværdier

Parameterværdierne er udtrykt i µg/l klorofyl a for 90-percentilen beregnet for et år i mindst en femårig periode. Resultaterne relaterer sig til geografiske områder inden for typerne som beskrevet i den tekniske rapport.

Type	Økologiske kvalitetsratioer		Værdier (μg/l, 90-percentil)	
	Grænselinje mellem høj og god	Grænselinje mellem god og moderat	Grænselinje mellem høj og god	Grænselinje mellem god og moderat
Type II-A				
Frankrig	0,80	0,53	2,38	3,58
Spanien	0,80	0,53	2,38	3,58
Italien (Tyrrhenian)	0,76	0,59	1,06	2,19
Type II-A Adriatic				
Italien	0,75	0,58	1,58	3,81
Slovenien	0,75	0,58	1,58	3,81
Type Island-W				
Frankrig	0,80	0,50	0,75	1,20
Spanien	0,80	0,50	0,75	1,20
Type III-W				
Frankrig	0,80	0,50	1,13	1,80
Spanien	0,80	0,50	1,13	1,80
Type III-E				
Cypern	0,80	0,20	0,10	0,40
Grækenland	0,80	0,20	0,10	0,40

Danske fjorde og kystnære havområder

Fastlæggelse af klorofyl *a* grænseværdier i fjorde og kystområder ved brug af modelværktøjer

Rapport fra DHI og DCE

Dato: 7. maj 2015

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AARHUS UNIVERSITET
DCE – NATIONALT CENTER FOR MILJØ OG ENERGI

DHI

Indhold

1 Introduktion

Naturstyrelsen har med projektet "Implementering af marine modeller til brug for vandforvaltningen" udviklet modelværktøjer til vandforvaltningen, herunder værktøjer til vurderinger af miljøtilstand og indsatsbehov for tilførsel af næringsstoffer.

En forudsætning for at anvende værktøjerne er, at der er fastlagt miljømål for den interkalibrerede klorofylindikator for alle danske vandområder, jævnfør vandrammedirektivet (VRD). I forbindelse med EU's interkalibrering blev der kun fastlagt klorofylgrænseværdier for enkelte vandområder, og Naturstyrelsen gennemførte derfor nærværende projekt med henblik på at definere miljømål for klorofylindikatoren for alle fjorde samt kystnære og åbne vandområder i de indre danske farvande. Analysen er gennemført med de samme modelværktøjer, som efterfølgende er anvendt til at bestemme hvilke indsatsbehov, der er nødvendige for at opnå en god miljøtilstand.

Denne rapport beskriver klorofylmålsundersøgelsens metoder og resultater og giver en oversigt over de beregnede klorofyl-miljømålsværdier.

1.1 Formål

Formålet med projektet er at fastlægge god-moderat grænseværdier for den interkalibrerede klorofylindikator for de danske vandområder, som er inkluderet af vandrammedirektivet. Der er særligt fokus på grænseværdien mellem god og moderat tilstand (miljømål), da denne grænseværdi er afgørende for, om et vandområde har acceptabel miljøtilstand eller ej, og dermed om der skal iværksættes en indsatsplan for vandområdet.

1.2 EU rammen

Ifølge VRD er målet, at der er mindst god økologisk tilstand i alle europæiske floder, søer og kystvande (medmindre eksisterende miljøforhold og samfundsforhold forhindrer dette). I marine områder skal vurderingen af den økologiske tilstand hovedsagligt baseres på indikatorer knyttet til de biologiske kvalitetselementer: fytoplankton, bundvegetation og bundfauna, og for alle kvalitetselementerne er grænsen mellem god økologisk tilstand og moderat tilstand beskrevet i forhold til en uberørt tilstand (referencetilstanden) som beskrevet i *tabel 1.1.*

Tabel 1.1. Normgivende definitioner af klassifikationer af økologisk tilstand i henhold til vandrammedirektivet. Tilstanden "under uberørte forhold" betegnes i forvaltningen af vandrammedirektivet som referencetilstanden.

God tilstand	Moderat tilstand
Værdierne for de biologiske kvalitetselementer for den pågældende type overfladevandområde udviser niveauer, der er svagt ændret som følge af menneskelig aktivitet, men afviger kun lidt fra, hvad der normalt gælder for denne type overfladevand under uberørte forhold.	Værdierne for de biologiske kvalitetselementer for den pågældende type overfladevand afviger i mindre grad fra, hvad der normalt gælder for denne type overfladevand under uberørte forhold. Værdierne viser mindre tegn på ændring som følge af menneskelig aktivitet og er signifikant mere forstyrrede end under forhold med god tilstand.

Den uberørte tilstand eller referencetilstanden er således central i forhold til at karakterisere god-moderat grænsen. Ifølge vandrammedirektivet (Annex II, 1.3) skal referencetilstanden helst bestemmes ud fra enten data (fra eksisterende uberørte områder eller historiske data) og/eller ved brug af modeller. Hvis dette ikke er muligt, kan der alternativt anvendes ekspertvurderinger. Dette gælder for alle de kvalitetselementer, som indgår i vandrammedirektivet.

Fytoplankton-kvalitetselementet

For kvalitetselementet fytoplankton er det ikke muligt at finde data for uberørte områder eller historiske data, som svarer til en referencetilstand. Referencetilstanden skal derfor bestemmes ved modellering, eventuelt kombineret med ekspertvurdering.

For alle kvalitetselementer gennemføres der på EU plan interkalibreringer for at sikre, at der er overensstemmelse i landenes bedømmelse af den økologiske tilstand i vandområder i samme økoregion. I økoregion Østersøen udgøres den interkalibrerede klorofylindikator af den gennemsnitlige klorofylkoncentration i sommermånederne maj-september (Anon. 2013).

Ved interkalibreringerne er der fastsat såkaldte EQR (ecological quality ratio) værdier, som mål for hvor meget indikatorværdien må afvige fra værdien under uberørte forhold (referencetilstand) og for klorofylindikatoren er EQR defineret som:

$$EQR_{klorofyl} = \frac{Referenceværdi\ for\ indikator}{Tilstandsværdi\ for\ indikator}$$

I vandrammedirektivet opereres med fem økologiske tilstandsklasser (hhv. dårlig, ringe, moderat, god og høj), der alle er defineret i forhold til referencetilstanden, og der er tilsvarende fire EQR-værdier, der afgrænser de fem økologiske tilstandsklasser. De fire interkalibrerede EQR-værdier, der anvendes til at adskille de fem tilstandsklasser for klorofylindikatoren, er angivet i *tabel 1.2*.

Tabel 1.2. Interkalibrerede EQR-værdier for klorofylindikatoren for danske vandområder i Østersøen (inden for Skagerrak) (Anon. 2013). EQR-værdierne definerer grænserne mellem de fem økologiske tilstandsklasser i vandrammedirektivet. Værdierne udtrykker det mål, der er sat for forholdet mellem referenceværdi for klorofylindikatoren og tilstandsværdien for indikatoren.

Tilstandsklasser	Høj-god grænseværdi	God-moderat grænseværdi	Moderat-ringe grænseværdi	Ringe-dårlig grænseværdi
EQR-værdi	0,8	0,6	0,4	0,2

Den afgørende grænseværdi er den, som adskiller god og moderat tilstand, da den fastsætter grænsen mellem acceptabel (god eller bedre) og ikke-acceptabel (moderat eller dårligere) tilstand (se definition ifølge direktivet i *tabel 1.1*). For klorofylindikatoren er EQR for god-moderat grænsen fastsat til 0,6. Ved kendskab til referenceværdien af klorofylindikatoren kan god-moderat (GM) grænseværdien for klorofylindikatoren derfor beregnes efter:

$$GM\ grænseværdi_{klorofyl} = \frac{Referenceværdi}{0{,}6}$$

Tilsvarende er det muligt at beregne grænseværdierne for de andre klasser ved brug af de fastlagte EQR-værdier, som fremgår af *tabel 1.2*.

1.3 Vandplanmodellerne

Projektet "Implementering af marine modeller til brug for vandforvaltningen" har udviklet to modeltyper: statistiske modeller og mekanistiske modeller, som begge er anvendt i nærværende projekt. Modellerne er udviklet af DHI og Aarhus Universitet.

De statistiske modeller er opstillet på basis af multivariate analyser af sammenhængen mellem data for miljøindikatoren (i dette tilfælde sommerklorofylkoncentration) og de miljøfaktorer, der påvirker miljøindikatoren, herunder belastningen med kvælstof og fosfor. Modellerne bygger på målte data og modeltilgangen er nærmere beskrevet i Timmermann et al. (2015). Der indgår 22 statistiske modeller, som dækker 19 vandområder i klorofyl-analysen.

De mekanistiske modeller giver dynamiske beskrivelser af økosystemets processer og virkningen på økosystemets kemiske og biologiske komponenter (fx næringssalte, bundvegetation og fytoplankton). Modellerne simulerer økosystemets reaktion på variationer i de ydre påvirkningsfaktorer (som meteorologi, hydrodynamik og næringsstofbelastning) og i de kemiske og biologiske komponenter. Modeltilgangen er nærmere beskrevet i Erichsen & Kaas (2015). Der indgår 4 mekanistiske modeller i klorofylanalysen; 3 fjordmodeller (Limfjorden, Roskilde Fjord, Odense Fjord) og en farvandsmodel for de indre danske farvande (benævnt IDF). Hver model dækker flere vandområder.

2 Metodebeskrivelse

2.1 Overordnede principper

Proceduren til bestemmelse af reference- og grænseværdier for klorofylindikatoren er opdelt i to dele som adresserer hhv. a) de fjord- og kystnære områder og b) de åbne vandområder. For både fjordtype- og åbentvandsområderne består den overordnede metode i først at beregne klorofylkoncentrationen i en referencesituation (uberørt tilstand) ved brug af modelværktøjerne og derfra beregne grænseværdien mellem god og moderat tilstand ved brug af den interkalibrerede EQR-værdi.

Den marine referencesituationen defineres i landene rundt om Østersøen (herunder Danmark og Tyskland) generelt som en situation svarende til perioden omkring år 1900 (± ca. 15 år), hvor de menneskelige aktiviteter, der kan påvirke de marine områder, har været begrænsede (før industrialiseringen og intensivering af landbruget). Denne ”referenceperiode” er bl.a. anvendt i tidligere studier af den historiske tilstand i relation til såvel vandrammedirektivet, Baltic Sea Action Plan og det marine havstrategidirektiv (Carstensen et al. 2013; Carstensen & Henriksen 2009; Gustafsson et al. 2012; Schernewski et al. 2015). I dette projekt benyttes derfor ligeledes perioden omkring år 1900 til at karakterisere en referencetilstand.

Efter aftale med Naturstyrelsen er klorofylindikatoren beregnet svarende til den interkalibrerede klorofylindikator for økoregion Østersøen; dvs. som gennemsnitlig klorofylkoncentrationen i sommermånederne maj-september (Anon. 2013) – uanset de indre farvande nord for Bælthavet tilhører økoregionen Nordatlanten, hvor der dog endnu ikke findes en interkalibreret indikator. Dette er gjort for at opnå et ensartet miljømål for alle indre danske farvande.

Til at fastlægge miljømål er der anvendt en typologibaseret tilgang, hvor grænseværdier beregnes for grupper af sammenlignelige vandområder og overføres til alle vandområder af samme type ifølge den danske typologi (Anon. 2014; Dahl et al. 2005). Dermed er det muligt at fastlægge klorofylgrænseværdier for alle danske vandområder beliggende i indre danske farvande og ikke kun for de områder, som er dækket af modellerne.

Den overordnede procedure til fastlæggelse af klorofylreference- og grænseværdier for de fjord- og kystnære områder er følgende:

- Etablering af inputdata som repræsenterer forholdene omring år 1900. Inputdata skal bruges til modelsimuleringer af en referencesituation (referencescenarie).
- For alle fjorde og kystnære vandområder, som er dækket af enten en statistisk og/eller mekanistisk model opstilles et referencescenarie med inputdata, som repræsenterer forholdene omkring år 1900, og klorofylindikatorværdierne i en referencesituation estimeres.
- Referenceværdierne transformeres til god-moderat grænseværdier ved brug af den interkalibrerede EQR-værdi.
- For at reducere usikkerhederne på de estimerede GM-grænseværdier benyttes ensemble modellering til fastlæggelse af klorofylgrænseværdierne.

Dvs. for de vandområder, hvor der er opstillet både statistiske og mekanistiske modeller, beregnes grænseværdien som gennemsnit af grænseværdier beregnet med hhv. den statistiske og mekanistiske model.

- De ensemble modellerede vandområder inddeles i kategorier baseret på graden af ferskvandspåvirkning, og der beregnes en kategori-specifik klorofylgrænseværdi for hver kategori.
- De kategori-specifikke grænseværdier overføres til samtlige vandområder via typologien.

For de åbne vandområder (OW-områderne ifølge den danske typologi) er proceduren den samme med den forskel, at der ikke er tale om ensemble modellering, da der alene er foretaget mekanistisk modellering for disse områder. Dvs. at der er fastlagt kategori-specifikke grænseværdier ud fra gennemsnittet af referenceklorofylværdier for de vandområder, der tilhører kategorien. Klorofylgrænseværdierne er beregnet på basis af resultaterne fra den mekanistiske modellering og EQR-værdien.

2.2 N og P tilførsler under referenceforhold

For at anvende de udviklede modeller til beregning af klorofylkoncentrationer under referenceforhold skal modellerne have inputdata om tilførslen af kvælstof og fosfor i en referencesituation. Naturstyrelsen har derfor rekvireret data fra forskellige kilder, se *tabel 2.1*. For de statistiske modeller er der udelukkende behov for data om næringsstoftilførsler fra dansk opland, hvorimod de mekanistiske modeller også kræver data om referencetilførsler fra Østersøen og fra atmosfæren.

Tabel 2.1. Leverandører af data til opgørelse af næringsstoftilførsler under referenceforhold.

Tilførsler	Kilde; anvendelse
Dansk referencetilførsel	NST, data rekvireret fra Aarhus Universitet (Bøgestrand et al. 2014a; Bøgestrand et al. 2014b); anvendt i den statistiske og mekanistiske modellering
Østersø referencetilførsel	Aarhus Universitet, oprindelige kilde: Baltic Nest Institute[1] (Gustafsson et al. 2012); anvendt til den mekanistiske modellering
Atmosfære referencedeposition	Aarhus Universitet (Geels et al. 2012); anvendt til den mekanistiske modellering

Opgørelsen af kvælstof- og fosfortilførsler til de marine områder er ikke en del af nærværende projekt, men som baggrund for de beskrevne analyser gives nedenfor en kort gennemgang af de data, som projektet er baseret på.

2.2.1 Dansk referencetilførsel

Data for referencetilførsler fra dansk opland til marine recipienter er leveret af Naturstyrelsen via DCE/Aarhus Universitet, Institut for Bioscience. Referencetilførslen er estimeret ud fra: a) baggrundskoncentrationen af total-kvælstof, nitrat, total-fosfor og opløst fosfor i tilførsler til de marine områder, og b) den nutidige vandføring i m^3 pr. måned. Metoden til beregning af baggrundskoncentrationer er beskrevet af Bøgestrand et al. (2014b). Estimaterne bygger på nutidsdata fra oplande med lav antropogen påvirkning, og der er foretaget en arealvægtet beregning af gennemsnitskoncentrationen i tilførsler til 4. ordens farvande. Specielt for fosfor er baggrundskoncentrationerne som ud-

[1] BNI er et internationalt forskningssamarbejde med deltagelse af blandt andet Aarhus Universitet

gangspunkt beregnet for større geografiske områder (georegioner), og usikkerheden er større end for N. Der findes kun spredte målinger af næringsstofindholdet i danske vandløb fra slutningen af 1800-tallet og der eksisterer ikke modelberegninger af historiske næringsstofbelastninger. Forfatterne anbefaler derfor, at data for baggrundsbelastningen som et muligt alternativ anvendes som estimat for tilførslerne omkring år 1900 (Bøgestrand et al. 2014a).

Til den mekanistiske modellering, som bygger på en mere detaljeret beskrivelse af næringsstoftilførslen, er data efterbehandlet for at a) koble data fra 4. ordens oplande til de tilsvarende VRD-vandområder, b) opsplitte total-kvælstof yderligere (bl.a. for at skelne mellem organisk og uorganisk tilførsel) og c) opnå en større tidslig opløsning. Efterbehandlingsmetoden er beskrevet i Erichsen & Kaas (2015).

2.2.2 Østersø-referencetilførsel

Data for referencetilførsler fra lande omkring den øvrige Østersø er stillet til rådighed af Baltic Nest Institute (BNI) via Aarhus Universitet. BNI har rekonstrueret historiske data om vand- og stoftilførsel til Østersøen ved modellering. De simulerede data er kvalitetssikret af BNI mod tilgængelige målinger. De anvendte metoder og data er beskrevet af Gustafsson et al. (2012).

Gustafsson et al. (2012) opgør næringsstoftilførslen for ammonium, nitrat, total N, uorganisk fosfor og total P pr. måned pr. år for 13 bassiner i Østersøen (inkl. eksempelvis Bælthavet og Kattegat) (Gustafsson et al. 2012). Data kan ikke benyttes direkte i den mekanistiske model, da den kræver en finere opløsning både i tid, i sted og i opdeling af næringsstoffraktioner. Data er derfor efterbehandlet. For at sikre det mest konsistente datasæt til modellen er efterbehandlingen baseret på ændringerne i tilførsler mellem perioden 1891-1909 og perioden fra 2000-2006[2]. I praksis udregnes den gennemsnitlige tilførsel pr. måned for de enkelte næringsstoffraktioner og Østersøbassiner opgjort af Gustafsson et al. (2012) og derefter beregnes den relative forskel mellem de månedlige tilførsler fra perioden 1891-1909 og perioden 2000-2006. Den relative månedlige forskel benyttes herefter til at omregne tilførslen i den mekanistiske model til en historisk tilførsel svarende til de enkelte bassintilførsler.

2.2.3 Atmosfærisk kvælstofdeposition

Data for den atmosfæriske kvælstofdeposition i referencetilstanden er leveret af Aarhus Universitet, Institut for Miljøvidenskab Den atmosfæriske kvælstofdeposition er beregnet med en atmosfæremodel, som beregner transport, omsætning og deposition af diverse kemiske forbindelser, herunder kvælstofforbindelser som NOx og ammoniak (Geels et al. 2012). Til beregninger af kvælstofdepositionen i en referencesituation er modellen blevet forceret med historiske emissioner fra år 1900 leveret af IIASA, "Representative Concentration Pathways" (RCPs; hentet fra
http://tntcat.iiasa.ac.at:8787/RcpDb/dsd?Action=htmlpage&page=welcome.
Kvælstofdepositionsdata er leveret som månedsgennemsnit over en 10 årig periode og med en rumlig opløsning på 5 × 5 km^2. De leverede modeldata er direkte anvendt som inputdata til de mekanistiske modeller.

[2] 2006 udgør det seneste år i datasættet fra Gustafsson et al. (2012).

2.3 Statistisk modellering af reference-sommerklorofyl

Den statistiske modellering af referenceforholdene for klorofyl er gennemført med de klorofylmodeller, som er beskrevet i Timmermann et al. (2015). Modellerne er udviklet til at kunne beskrive sammenhænge mellem klorofylindikatoren (klorofylkoncentrationen i perioden maj til september) og en række presfaktorer herunder kvælstoftilførslen. Udvælgelsen af de forklarende presfaktorer i et givent vandområde er baseret på multipel lineær regression, krydsvalideret ved brug af Jackknifing, hvor sub-samples af variablene udtages og forklaringskraften i de resulterende modeller analyseres. For de modeller, hvor der er fundet en signifikant sammenhæng mellem næringsstoftilførsler og klorofylkoncentrationen, kan modellerne benyttes som værktøj til analyse af klorofylkoncentration i en referencesituation.

Klorofylkoncentrationerne i en referencesituation estimeres ved at påtrykke modellerne med den danske referencetilførsel (Bøgestrand et al. 2014b) fra samme opland, som blev benyttet til opstilling af modellerne.

2.4 Mekanistisk modellering af reference-sommerklorofyl

Den mekanistiske modellering af referenceforholdene for klorofyl er gennemført med modificerede udgaver af de modeller, der er udviklet til beskrivelse af den nuværende tilstand: Limfjordsmodellen, Odense Fjord-modellen, Roskilde Fjord-modellen og farvandsmodellen for indre danske farvande (IDF-modellen). Modifikationen omfatter reduktioner af næringsstoftilførsel fra oplande, atmosfære, tilstødende vandområder og sedimentet. Ud over disse modifikationer er modelopsætningen identisk med den oprindelige opsætning. Det betyder, at modellerne er afviklet for en 10 års periode med meteorologiske forhold og ferskvandstilførsel svarende til perioden 2002-2011. Kun resultater for de sidste 5 år er anvendt til bestemmelse af klorofylkoncentrationen i en referencesituation.

I det følgende gives en kort beskrivelse af modelopsætningen. For en mere detaljeret beskrivelse henvises til Erichsen & Kaas (2015).

2.4.1 Næringsstoftilførsel

I den mekanistiske modellering indgår flere typer af data, der beskriver tilførslen af kvælstof og fosfor til recipienterne:

- Referencetilførsel fra dansk opland (alle modeller)
- Referencetilførsel fra øvrige Østersø-oplande (til model for indre danske farvande)
- Næringsstoftilførsel over modelrande (alle modeller)
- Intern belastning – næringsstofudvekling mellem havbund og –vand

Data vedrørende referencetilførsel fra danske og udenlandske oplande er beskrevet ovenfor (*afsnit 2.2*).

2.4.1.1 Næringsstoftilførsel over modelrande

Modellen for de indre danske farvande (IDF) har én modelrand, der ligger i midten af Skagerrak. Da der ikke findes historiske målinger, er de benyttede referencekoncentrationer ved randen bestemt på basis af litteraturen. Siden 1950'erne har koncentrationerne af næringsstoffer i den central del af Nordsøen og i Skagerraks overfladevand været konstant (Radach & Patsch 1997) uanset næringsstofkoncentrationerne i de floder, der udleder til Tyske Bugt, er reduceret med en faktor 4 (Topcu et al. 2011). Det indikerer, at referencekoncentrationerne i Skagerrak ikke har været meget forskellige fra i dag. På basis af Savchuk et al. (2008), der har modelleret historiske koncentrationer, er det beregnet, at referencekoncentrationerne på Skagerrak-randen har været ca. 85 % af de nuværende koncentrationer (2007-2012). For både kulstof, kvælstof og fosfor er randværdierne i den oprindelige modelopsætning derfor reduceret med 15 % for modellering af referencesituationen.

Randværdierne til fjordmodellerne er baseret på observerede data. Da der ikke findes historiske målinger, er de sandsynlige referencekoncentrationer bestemt ud fra IDF-modellens resultater for randområderne. Fjordmodellernes nutidsværdier er således reduceret med en procent svarende til de forskelle, der er fundet i randområderne ved IDF-modelleringen af nutids- og referencesituationen.

2.4.1.2 Intern belastning – næringsstofudvekling mellem havbund og -vand

En anden vigtig parameter for den mekanistiske modellering er inputdata om puljerne af kulstof, kvælstof og fosfor i sedimenterne. I en referencesituation forventes puljerne at være reducerede i forhold til nutidssituation, men de historiske puljestørrelser er ikke kendte, og den viden, der kan bruges til at fastlægge de historiske puljer, er meget begrænset. Fastlæggelsen af puljernes størrelse i en referencesituation er sket på forskellig måde for henholdsvis fjordmodellerne og IDF-modellen.

Sammenlignet med de åbne områder, er sedimenternes responstid i fjordene hurtigere pga. en væsentlig kortere opholdstid. Det er derfor valgt, at estimere referencesedimentpuljerne ved at ændre næringsstoftilførslen i fjordmodellerne fra nutidsforhold til referenceforhold og derefter simulere den deraf følgende ændring i sedimentpuljerne. Med andre ord er sedimentpuljerne ikke reduceret forud for modelleringen af den 10-årige periode, som anvendes i de mekanistiske modeller (2002-2011). Modelsimuleringerne viser, at de væsentligste reduktioner i sedimentpuljerne (øverste 10 cm) sker inden for de første 5 år af den 10-årige periode. Puljerne er efter 5 år endnu ikke i ny ligevægt, men det er vurderet, at den yderligere reduktion, der sker efter det 5. år, ikke har væsentlig indflydelse på de estimerede referenceværdier for klorofyl. Ved beregning af referencekoncentrationer for klorofyl er der, som nævnt tidligere, alene anvendt data fra de sidste 5 år af de 10 simulerede år.

For IDF-modellen (som dækker de åbne farvande) vil den store opholdstid i Østersøen, og i dele af de danske farvande, medføre en betydeligt langsommere indstilling af ligevægten i nogle områder (især i den centrale Østersø). Derfor er sedimentpuljerne i Østersøen og indre danske farvande reduceret inden start af modelleringen af 10-års perioden 2002-2011. Reduktioner (i %) er sket i forhold til de puljer, der er anvendt i nutidskørslerne (se Erichsen et al. 2015) og med anvendelse af følgende nøgle:

- Organisk kvælstof: Reduceret med 55 %
- Organisk kulstof: Reduceret med 34 %
- Organisk fosfor: Reduceret med 34 %
- Jernbunden fosfor: Reduceret med 34 %

Reduktionsprocenterne er baseret undersøgelser af historiske sedimentkoncentrationer (Almroth & Skogen 2010; Andersen et al. 2011; Carman & Cederwall 2001; Savchuk et al. 2008).

2.5 Anvendt typologi

For at give basis for at "udbrede" de fundne GM-grænseværdier til ikke modellerede vandområder er der i nærværende projekt defineret 5 kategorier, som de modellerede vandområder er opdelt i (*tabel 2.2*).

Tabel 2.2. Oversigt over de kategorier, som er brugt i nærværende projekt i forbindelse med fastlæggelse af klorofylgrænseværdier i modellerede danske vandområder.

Kategori	Beskrivelse
1	Åbent vand
2	Åbne fjorde/bugter, som er mindre ferskvandspåvirkede
3	Vandområder, som er noget ferskvandspåvirkede
4	Vandområder, som er meget ferskvandspåvirkede
5	Slusefjorde

For fjord/kyst-vandområderne er der for hver kategori fastlagt kategori-specifikke klorofylværdier for GM-grænsen, som efterfølgende er udbredt til alle vandområder ved at kombinere kategorierne med den danske typologi (Anon. 2014; Dahl et al. 2005), der klassificerer alle danske vandområder. Fjord/kyst-kategorierne er primært baseret på graden af ferskvandspåvirkning, idet det antages, at ferskvandspåvirkningen også i en referencesituation vil være afgørende for klorofylkoncentrationen i vandområdet. I den danske typologi indgår ligeledes et udtryk for ferskvandspåvirkningen (angivet ved et afstrømningsindex, F, der beskriver relationen mellem ferskvandsafstrømning og ferskvandets gennemsnitlige opholdstid (Anon. 2014; Dahl et al. 2005)), og generelt er der god overensstemmelse mellem nærværende projekts kategorisering og den danske typologi. *Tabel 2.3* giver den sammenhæng mellem de to klassifikationer, der er anvendt til at udbrede modelresultaterne til ikke-modellerede fjord- og kystområder.

Tabel 2.3. Sammenhæng mellem kategorier anvendt i denne undersøgelse og typerne i den danske typologi for fjord- og kystområder.

Kategorier	2	3	4	5
Den danske typologi	M1, M2, P1, P2	M3, M4, P3, P4	O3, O4	slusefjord

For åbent vand svarer kategori 1 til den danske typologis OW1-3 områder. Kategori 1 er opdelt i 4 undergrupper, som ud over ferskvandspåvirkning er baseret på de hydrografiske forhold (salinitet, upwelling, strøm, etc.). Da alle OW1-3-områderne er repræsenteret i IDF-modellen er der ikke behov for at udbrede resultaterne til ikke modellerede områder. *Tabel 2.4* giver sammenhængen mellem de åbenvandskategorier, der er anvendt i denne undersøgelse, og den danske typologi.

Den klassifikation, der er benyttet ift. de modellerede områder og bestemmelse af reference- og grænseværdier for klorofylindikatoren, består således af 4 fjordkategorier (2, 3, 4 og 5) og 4 åbentvandskategorier (1.1, 1.2, 1.3 og 1.4).

Tabel 2.4. Anvendte kategori 1 undergrupper med angivelse af de tilsvarende typer i henhold til den danske typologi.

Kategori	**Beskrivelse**	**Den danske typologi**
1.1	Åbent hav/kyster/bugter meget påvirket af Øster-søen, salinitet 5-18 psu	OW3a-c
1.2	Åbent hav/kyster/bugter, beskyttede for hovedstrømme, generelt mindre påvirkede af Østersøvand, salinitet: 5-18 psu	OW3a
1.3	Åbent hav/kyster/åbne bugter, Nordsøpåvirkede, ± direkte ferskvandspåvirkning, salinitet 18-30 psu	OW2, + et OW1
1.4	Åbent hav/kyster, østvendte åbne kyster med upwelling, Nordsøpåvirkede – salinitet 18-30 psu	OW1, OW2

3 Resultater

3.1 Fastsættelse af kategori-specifikke grænseværdier for fjord- og kystområder

For de fjord- og kystvandområder, hvor der både findes en mekanistisk og en statistisk model, er det muligt at bestemme klorofylreferenceværdier og god-moderat grænseværdier ved ensemble modellering, dvs. at basere bestemmelsen på resultaterne af forskellige modeltilgange, hvilket dels bevirker, at resultatet bliver mere sikkert og dels muliggør en beregning af en usikkerhed på modelprædiktionen. De fjord- og kystvandområder, hvor der både findes en mekanistisk og en statistisk model, tilhører alle kategori 2 eller 3.

Tabel 3.1 angiver de kategori 2 og 3 områder, der indgår i analysen, og deres modelbaserede god-moderat sommerklorofylgrænseværdier (hhv. for hver modeltilgang og ensemble-resultatet) samt gennemsnittet for hver kategori. Grænseværdierne er beregnet ud fra resultatet af referencemodelleringen og EQR-værdien. Ensemble-værdien er beregnet som gennemsnittet af grænseværdierne beregnet med de to modeltilgange. For den mekanistiske modellering er der i tabellen angivet GM-grænseværdi for en position svarende til overvågningsstationen og for hele vandområdet. Det er værdien for hele området, der er anvendt til ensemble-modelleringen, da den anses for at være mest repræsentativ for området.

Vandområdet Aarhus Bugt, Kalø og Begtrup Vig (ID 147) er ifølge den danske typologi et P3 område, men da området har en kort hydraulisk opholdstid, vurderes det at være mindre ferskvandspåvirket end indikeret af den danske typologi, og området er kategorisereret som et kategori 2 område med en ensemble-baseret god-moderat grænseværdi på 1,9 µg/l.

For vandområder Roskilde Fjord, indre, er det modsatte tilfældet. Inderfjorden har en lang hydraulisk opholdstid og vurderes derfor at være mere ferskvandpåvirket end indikeret af den danske typologi (M3), og området er kategoriseret som et kategori 3 vandområde.

En stor del af Limfjorden er samlet i ét vandområde: Nissum, Thisted, Kås, Løgstør, Nibe, Langerak (ID 156). Området er samlet set kategoriseret som et kategori 3 område, og ensembleværdien for god-moderat grænsen er beregnet til 3.5 µg/l. Med de opstillede modeller er det muligt at analysere værdier for delområder, og som det fremgår af *tabel 3.1* varierer den beregnede ensemble god-moderat grænseværdi mellem delområderne.

For vandområdet Bjørnholms Bugt, Riisgårde Bredning, Skive Fjord og Lovns Bredning (ID 157) er det vurderet, at det ikke kan indplaceres i kategorierne. Dette skyldes hovedsageligt, at området sandsynligvis også vil være påvirket af iltsvind i en referencesituation, hvilket betyder, at andre forhold end ferskvandstilførslen (iltsvind) har betydning for klorofyl-referencekoncentrationen. Hvis der er forekommet iltsvind i en referencesituation (naturligt iltsvind), er det sandsynligt, at klorofyl-referencekoncentrationen er højere end i en tilsvarende fjord uden "naturligt" iltsvind. I overensstemmelse hermed er der beregnet en højere klorofylgrænseværdi for vandområdet (*tabel 3.2*) end for de øvrige områder, der svarer til kategori 2 og 3 (ID157 er et P3 område ifølge den danske typologi). For Mariager Fjord, indre, er situationen sandsynligvis tilsvarende, dvs. at området også i en referencesituation er påvirket af iltsvind, men der er ikke datagrundlag for at udvikle en statistisk model for området, og der er ikke udviklet en mekanistisk fjordmodel for Mariager Fjord.

Kategorien "slusefjorde" omfatter Hjarbæk Fjord og Ringkøbing Fjord. Fjordene er så forskellige, at det ikke anses for pålideligt at opfatte dem som én type, og de er derfor analyseret hver for sig (*tabel 3.2*). For Hjarbæk Fjord er grænseværdien beregnet med den mekanistiske model og god-moderat grænsen er bestemt til 9 µg/l. For Ringkøbing Fjord er grænseværdien beregnet ud fra den statistiske model, der er opstillet for overvågningsstationen i fjorden, og god-moderat grænsen er bestemt til 8 µg/l. Nissum Fjord er ikke inkluderet i den mekanistiske modellering, og det har ikke været muligt at udvikle en statistisk model på basis af overvågningsdata.

Tabel 3.1. God-moderat grænseværdier for fytoplankton-indikatoren sommerklorofyl for kategori 2 og 3 fjord/kystområder analyseret ved ensemble modellering. Type i henhold til den danske typologi = vandområdets klassificering ifølge den danske typologi. Kategori = klassifikation anvendt i denne undersøgelse. MEK stn. GM = god-moderat grænsen beregnet for overvågningsstationen ved mekanistisk modellering. MEK Vomr. GM = god-moderat grænsen beregnet for hele vandområdet ved mekanistisk modellering. STAT stn. GM = god-moderat grænsen beregnet for overvågningsstationen ved statistisk modellering. Ensemble GM = gennemsnittet af MEK Vomr. og STAT stn. GM pr. kategori = gennemsnit af ensemble GM-værdier. Data med gråt er beregnede værdier for delområder af Limfjordsvandområdet Nissum Bredning m.fl., ID156.

Navn	Vandområde ID	Type i henhold til den danske typologi	Kategori	MEK stn. GM	MEK Vomr. GM	STAT stn. GM	Ensemble GM	GM pr. kategori
				God-moderat grænseværdier (µg/l)				
Isefjord ydre	165	M2	2			3,4		
Roskilde ydre	1	M2	2	1,9	1,7		2,6*	
Åbenrå Fjord	102	P1	2	1,6	1,6	1,9	1,8	
Aarhus Bugt, Kalø og Begtrup Vig	147	P3	2	1,7	1,6	2,2	1,9	2,1
Roskilde Fjord, indre	2	M2	3	4,2	4,6	3,7	4,2	
Odense Fjord, ydre	92	P3	3	2,1	1,3	6,9	4,1	
Vejle	123	P3	3	2,4	2,2	2,9	2,6	
Nissum, Thisted, Kås, Løgstør, Nibe, Langerak	156	P4	3	-	3,0	4,0	3,5	3,6
Delarealer af område 156								
Kås	156	P4	3	1,2		6,7	4,0	
Nibe	156	P4	3	0,7		3,4	2,1	
Løgstør	156	P4	3	5,4		4,8	5,1	
Nissum	156	P4	3	1,6		1,2	1,4	

* Gennemsnit af Isefjord, ydre og Roskilde Fjord, ydre.

Tabel 3.2. God-moderat grænseværdier for fytoplankton-indikatoren sommerklorofyl for øvrige fjord- og kystområder analyseret ved ensemble modellering. Type i henhold til den danske typologi = vandområdets klassificering ifølge den danske typologi. Kategori = klassifikation anvendt i denne undersøgelse. MEK Vomr. GM= god-moderat grænsen beregnet for hele vandområdet ved mekanistisk modellering. MEK stn. GM = god-moderat grænsen beregnet for overvågningsstationen ved mekanistisk modellering. STAT stn. GM = god-moderat grænsen beregnet for overvågningsstationen ved statistisk modellering. Ensemble GM = gennemsnittet af MEK Vomr. og STAT stn. GM pr. kategori = gennemsnit af ensemble GM-værdier.

Navn	Vand-område ID	Type i henhold til den danske typologi	Kategori	MEK stn. GM	MEK Vomr. GM	STAT stn. GM	Ensemble GM	GM pr. kategori
				God-moderat grænseværdier, sommerklorofyl (µg/l)				
Bjørnholms Bugt, Riisgårde Bredning, Skive Fjord og Lovns Bredning	157	P3	--	7	10,8	4,8	5,9	6
Ringkøbing	132	Sluse	5			7,8		8
Hjarbæk Fjord	158	Sluse	5	9,7	8,5/15,9		8,8	9

Grænseværdierne for hver kategori er opsummeret i *tabel 3.3*. For kategori 2 og 3 er kategoriens grænseværdi det afrundede gennemsnit for de vandområder, der tilhører kategorien. For kategori 5 (slusefjorde) er det på grund af den meget forskellige natur af de danske slusefjorde ikke muligt at fastlægge ét miljømål for typen (mht. mål for enkelte fjorde, se *tabel 3.2*). Det var ikke muligt at etablere en grænseværdi for kategori 4 vandområder, da der ikke er udført mekanistisk modellering, og der ikke har været datagrundlag for at opstille statistiske modeller for områder i denne kategori (områder med salinitet under 5 promille (O3-, O4 områder)).

Tabel 3.3. Sommerklorofylgrænseværdier for kategorierne for fjord- og kystområder.

Kategori	2	3	4	5
Sommerklorofylgrænseværdi	2,1	3,6	Ukendt	Individuel

3.2 Fastsættelse af kategori-specifikke grænseværdier for åbenvandsområder

Fastsættelse af grænseværdier for de åbne vandområder (OW1-3 i den danske typologi) er udelukkende baseret på resultater fra den mekanistiske model. *Tabel 3.4* angiver de modellerede vandområder, de mekanistisk model-estimerede god-moderat grænseværdier pr. vandområde og god-moderat grænsen for de 4 undergrupper af kategori 1 (kategori 1.1-1.4).

Tabel 3.4. God-moderat grænseværdier for fytoplankton-indikatoren sommerklorofyl for kategori 1 områder. Grænseværdier er baseret på mekanistisk modellering. Type i henhold til den danske typologi = vandområdets klassificering ifølge den danske typologi. Kategori = klassifikation anvendt i denne undersøgelse. MEK Vomr. GM = god-moderat grænsen beregnet for hele vandområdet ved mekanistisk modellering. GM pr. kategori = gennemsnit af ensemble GM-værdier.

Vandområde	Vand område ID	Type i henhold til den danske typologi	Kategori	MEK Vomr. GM	GM pr. kategori
	God-moderat grænseværdier (µg klorofyl/l)				
Nordlige Øresund	6	OW2	1.1	1,8	
Hjelm Bugt	44	OW3b	1.1	1,6	
Køge Bugt	201	OW3b	1.1	1,8	
Fakse Bugt	46	OW3b	1.1	1,6	
Østersøen, Bornholm	56	OW3c	1.1	1,7	
Storebælt, SV	95	OW3a	1.1	1,8	1,7
Langelandsbælt, øst	41	OW3a	1.2	1,5	
Femerbælt	208	OW3a	1.2	1,5	
Grønsund	45	OW3a	1.2	1,2	
Langelandssund	90	OW3a	1.2	1,6	
Smålandsfarvandet, åbne del	206	OW3a	1.2	1,4	
Det sydfynske Øhav, åbne del	214	OW3a	1.2	1,3	
Lillebælt, syd	216	OW3a	1.2	1,5	
Lillebælt, Bredningen	217	OW3a	1.2	1,5	1,5
Kattegat, Nordsjælland > 20 m	205	OW1	1.3	1,5	
Anholt	139	OW2	1.3	1,4	
Kattegat, Læsø	154	OW2	1.3	1,4	
Nordlige Lillebælt	224	OW2	1.3	1,6	
Kattegat, Nordsjælland	200	OW2	1.3	1,6	
Sejerø Bugt	28	OW2	1.3	1,5	
Hevring Bugt	138	OW2	1.3	1,7	
Aarhus Bugt syd, Samsø og Nordlige Bælthav	219	OW2	1.3	1,7	
Kattegat, Aalborg Bugt	222	OW2	1.3	1,7	1,6
Nordlige Kattegat - Ålbæk Bugt	225	OW1	1.4	1,9	
Djursland øst	40	OW2	1.4	1,8	
Storebælt, NV	96	OW2	1.4	1,9	1,9

3.3 Fastsættelse af grænseværdier for danske vandområder

Tabel 3.5 og *tabel 3.6* lister god-moderat miljømålene for de vandområder, der er inkluderet i nærværende projekt – dvs. eksklusiv vandområder i Nordsøen og Vadehavet. Af *tabel 3.5* fremgår miljømålene for kategori 1 områderne svarende til typerne OW1-3 i den danske typologi, mens *tabel 3.6* omfatter fjord- og kystområderne svarende til typerne M1-M4 og P1-P4 i den danske typologi.

Tabel 3.5. God-moderat miljømål for fytoplankton-indikatoren sommerklorofyl baseret på kategori 1 (åbne farvande) grænseværdier estimeret på basis af mekanistisk modellering af referencetilstand.

	Vandområde ID	**Type i henhold til den danske typologi**	**Kategori**	**GM miljømål Sommer-klorofyl (µg/l)**
Nordlige Øresund	6	OW2	1.1	1.7
Hjelm Bugt	44	OW3b	1.1	1,7
Køge Bugt	201	OW3b	1.1	1,7
Fakse Bugt	46	OW3b	1.1	1,7
Østersøen, Bornholm	56	OW3c	1.1	1,7
Storebælt, SV	95	OW3a	1.1	1,7
Langelandsbælt, øst	41	OW3a	1.2	1,5
Femerbælt	208	OW3a	1.2	1,5
Grønsund	45	OW3a	1.2	1,5
Langelandssund	90	OW3a	1.2	1,5
Smålandsfarvandet, åbne del	206	OW3a	1.2	1,5
Det sydfynske Øhav, åbne del	214	OW3a	1.2	1,5
Lillebælt, syd	216	OW3a	1.2	1,5
Lillebælt, Bredningen	217	OW3a	1.2	1,5
Kattegat, Nordsjælland > 20 m	205	OW1	1.3	1,6
Anholt	139	OW2	1.3	1,6
Kattegat, Læsø	154	OW2	1.3	1,6
Nordlige Lillebælt	224	OW2	1.3	1,6
Kattegat, Nordsjælland	200	OW2	1.3	1,6
Sejerø Bugt	28	OW2	1.3	1,6
Hevring Bugt	138	OW2	1.3	1,6
Aarhus Bugt syd, Samsø og nordlige Bælthav	219	OW2	1.3	1,6
Kattegat, Aalborg Bugt	222	OW2	1.3	1,6
Nordlige Kattegat - Ålbæk Bugt	225	OW1	1.4	1,9
Djursland Øst	140	OW2	1.4	1,9
Storebælt, NV	96	OW2	1.4	1,9

Tabel 3.6. God-moderat grænseværdier for fytoplankton-indikatoren sommerklorofyl baseret på de beregnede kategori-grænseværdier. For modellerede vandområder er de beregnede GM-kategoriværdier anvendt. For ikke-modellerede vandområder er sammenhængen angivet i *tabel 2.3* mellem den danske typologi og kategorierne anvendt til at "udbrede" GM-værdierne.

	Vandområde ID	**Type i henhold til den danske typologi**	**Kategori**	**GM miljømål**
Smålandsfarvandet, syd	34	M1	2	2,1
Helnæs Bugt	87	M1	2	2,1
Als Sund	104	M1	2	2,1
Nakskov Fjord	207	M1	2	2,1
Roskilde Fjord, ydre	1	M2	2	2,1
Musholm Bugt, indre	26	M2	2	2,1
Avnø Fjord	37	M2	2	2,1
Guldborgsund	38	M2	2	2,1
Stege Bugt	48	M2	2	2,1
Kløven	72	M2	2	2,1
Lunkebugten	89	M2	2	2,1
Faaborg Fjord	212	M2	2	2,1
Åbenrå Fjord	102	P1	2	2,1
Als Fjord	103	P1	2	2,1
Flensborg Fjord, ydre	114	P1	2	2,1
Ebeltoft Vig	141	P1	2	2,1
Kalø Vig, indre	145	P1	2	2,1
Isefjord	24	P2	2	2,1
Kalundborg Fjord	29	P3	2[1)]	2,1
Aarhus Bugt, Kalø og Begtrup Vig	147	P3	2[1)]	2,1
Augustenborg Fjord	105	M2	2	2,1
Flensborg Fjord, indre	113	P1	2	2,1
Roskilde Fjord, indre	2	M2	3[2)]	3,6
Karrebæk Fjord	35	M3	3	3,6
Nakkebølle Fjord	63	M3	3	3,6
Odense Fjord, Seden Strand	93	M4	3	3,6
Mariager ydre	160	P1	3	3,6
Nyborg Fjord	86	P3	3	3,6
Odense Fjord, ydre	92	P3	3	3,6
Vejle Fjord, ydre	122	P3	3	3,6
Vejle Fjord, indre	123	P3	3	3,6
Kolding Fjord, indre	124	P3	3	3,6
Kolding Fjord, ydre	125	P3	3	3,6
Horsens Fjord, ydre	127	P3	3	3,6
Horsens Fjord, indre	128	P3	3	3,6
Randers, ydre	137	P3	3	3,6
Dalby Bugt	61	P4	3	3,6
Nissum, Thisted, Kås, Løgstør, Nibe, Langerak	156	P4	3	3,6
Ringkøbing	132	Slusefjord	5	8
Hjarbæk Fjord	158	Slusefjord	5	9
Bjørnholms Bugt, Riisgårde Bredning, Skive Fjord og Lovns Bredning	157	P3	UK[3)]	6

1) Vandområder med kort hydraulisk opholdstid og som derfor vurderes at være mindre ferskvandspåvirkede end angivet i den danske typologi for kystvandstyper.

2) Vandområder med lang hydraulisk opholdstid og som derfor vurderes at være mere ferskvandpåvirkede end angivet i den danske typologi for kystvandstyper.

3) Vandområde, hvor det vurderes, at der også i en referencesituation kan forekomme iltsvind, som kan påvirke referencekoncentrationen af klorofyl. Vandområderne er derfor ikke indplaceret i dette projekts kategorier, men har fået tildelt en klorofylgrænseværdi baseret på områdespecifikke modelberegninger for vandområde 157.

3.4 Usikkerheder forbundet med bestemmelse af miljømål for klorofyl

Den anvendte metode til fastlæggelse af miljømål for klorofylindikatoren tager i dette projekt udgangspunkt i modelberegninger af klorofylkoncentrationen i en referencesituation (jf. definitionen i vandrammedirektivet). Referencesituation er defineret til perioden omkring år 1900. Det er imidlertid ikke nogen triviel opgave, at beregne troværdige niveauer for klorofylkoncentrationen i en "upåvirket tilstand". Derfor er der naturligvis usikkerheder forbundet med beregningerne af reference-klorofylkoncentrationerne. Et væsentlig bidrag til usikkerhederne stammer fra usikkerheder på de inputparametre (fx næringsstoftilførsler, sedimentforhold og klima), der skal bruges for at simulere en referencesituation. Det er ikke muligt at fastlægge de nødvendige inputparametre på basis af målinger, da der i bedste fald kun findes spredte og ikke standardiserede historiske målinger fra referenceperioden. Derfor må inputparametrene fastlægges ud fra modeller og/eller ved antagelser om, hvordan forholdene var.

Et andet væsentligt bidrag til usikkerhed kommer af, at de marine modeller er udviklet og kalibreret til den nuværende situation, herunder den nuværende næringsstoftilførsel. Ved simulering af en referencesituation skal modellerne derfor ekstrapoleres ud over det område, de er kalibreret og valideret for. Da der ikke findes historiske klorofyl-observationer, er det selvsagt ikke muligt at kvantificere, hvor pålidelige modellerne er til at beskrive forholdene i en uberørt situation, men generelt set vil modelusikkerheden øges med afstanden fra kalibreringsområdet. For at reducere usikkerhederne er der for fjord/kysttype vandområderne benyttet en ensemble modeltilgang med to uafhængige modeller, hvilket øger sikkerheden på referenceestimatet. Endvidere er der for både kysttype- og åbentvandsområderne benyttet en typologitilgang, hvilket yderligere bidrager til, at estimatet for referenceklorofyl bliver mere robust. Typologitilgangen betyder dog, at det enkelte vandområde reelt kan afvige fra gennemsnittet af de modellerede vandområder tilhørende samme type, da der naturligt forekommer variationer imellem de enkelte vandområder, fx pga. gradienter, inden for typerne, forskelle i vanddybder m.m., men det vurderes, at anvendelsen af typespecifikke mål generelt giver en større sikkerhed på bestemmelsen af miljømålet.

4 Referencer

Almroth, E. & Skogen, M.D. 2010: A North Sea and Baltic Sea Model Ensemble Eutrophication Assessment. – Ambio 9: 59-69.

Andersen, J.H., Axe, P., Backer, H., Carstensen, J., Claussen, U., Fleming-Lehtinen, V., Jarvinen, M., Kaartokallio, H., Knuuttila, S., Korpinen, S., Kubiliute, A., Laamanen, M., Lysiak-Pastuszak, E., Martin, G., Murray, C., Mohlenberg, F., Nausch, G., Norkko, A. & Villnas, A. 2011: Getting the measure of eutrophication in the Baltic Sea: towards improved assessment principles and methods. - Biogeochemistry 106: 137-156.

Anon. 2013: Commision decision of 20 September 2013 (2013/480/EU) establishing, pursuant to Directive 2000/60/EC of the European Parliament of the Council, the values of the Member State monitoring system classification as a result of the intercalibration exercise and repealing Decision 2008/915/EC. The European Commission. Official Journal of the European Union.

Anon. 2014: BEK 1400 af 15/12/2014. Bekændtgørelse om basisanalyser. Miljøministeriet.

Bøgestrand, J., Windolf, J. & Kronvang, B. 2014a: Næringsstofbelastningen til vandområder omkring år 1900. 9 s. Aarhus Universitet. Notat fra DCE - Nationalt Center for Miljø og Energi.

Bøgestrand, J., Windolf, J., Kronvang, B. & Kjeldgaard, A. 2014b: Baggrundsbelastning med total N og nitrat-N. 11 s.. Aarhus Universitet. Notat fra DCE-Dansk Center for Miljø og Energi.

Carman, R. & Cederwall, H. 2001: Sediments and Macrofauna in the Baltic Sea - Characteristics, Nutrient Content and Distribution. - In: A Systems analysis of the Baltic Sea. Eds. F. Wulff, R. A. Rahm, and R. A. Larsson. pp. 289-328.

Carstensen, J., Andersen, J.H., Dromph, K., Fleming-Lehtinen, V., Simis, S., Gustafsson, B. G., Norkko, A., Radtke, H., Petersen, D. L. J. & Uhrenholdt, T. 2013: Approaches and methods for eutrophication target setting in the Baltic Sea region. Helsinki Commission - Balt. Sea Environ. Proc. No. 133.

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

Dahl, K. (Red.), Andersen, J.H. (Red.), Riemann, B. (Red.), Carstensen, J., Christiansen, T., Krause-Jensen, D., Josefson, A.B., Larsen, M.M., Petersen, J.K., Rasmussen, M.B. & Stand, J. 2005: Redskaber til vurdering af miljø- og naturkvalitet i de danske farvande. Typeinddeling, udvalgte indikatorer og eksempler på klassifikation. Danmarks Miljøundersøgelser. 158 s. - Faglig rapport fra DMU nr. 535.

Erichsen, A.E. & Kaas, H. 2015: NST projektet ”Implementeringen af modeller til brug for vandforvaltningen”. Modeller for Danske Fjorde og Kystnære Havområder – Del 2. Mekanistiske modeller og metode til bestemmelse af indsatsbehov.

Geels, C., Hansen, K., Christensen, J., Skjoth, C., Ellermann, T., Hedegaard, G., Hertel, O., Frohn, L., Gross, A. & Brandt, J. 2012: Projected change in atmospheric nitrogen deposition to the Baltic Sea towards 2020. - Atmospheric Chemistry and Physics 12: 2615-2629.

Gustafsson, B.G., Schenk, F., Blenckner, T., Eilola, K., Meier, H.E.M., Muller-Karulis, B., Neumann, T., Ruoho-Airola, T., Savchuk, O. P. & Zorita, E. 2012: Reconstructing the Development of Baltic Sea Eutrophication 1850-2006. - Ambio 41: 534-548.

Radach, G. & Patsch, J. 1997: Climatological annual cycles of nutrients and chlorophyll in the North Sea. - Journal of Sea Research 38: 231-248.

Savchuk, O.P., Wulff, F., Hille, S., Humborg, C. & Pollehne, F. 2008: The Baltic Sea a century ago - a reconstruction from model simulations, verified by observations. - Journal of Marine Systems 74: 485-494.

Schernewski, G., Friedland, R., Carstens, M., Hirt, U., Leujak, W., Nausch, G., Neumann, T., Petenati, T., Sagert, S., Wasmund, N. & von Weber, M. 2015: Implementation of European marine policy: New water quality targets for German Baltic waters. - Marine Policy 51: 305-321.

Timmermann, K., Christensen, J., Murray, C. & Markager, S. 2015: NST projektet ”Implementeringen af modeller til brug for vandforvaltningen”. Modeller for Danske Fjorde og Kystnære Havområder – Del 3. Statistiske modeller og metode til bestemmelse af indsatsbehov.

Topcu, D., Behrendt, H., Brockmann, U. & Claussen, U. 2011: Natural background concentrations of nutrients in the German Bight area (North Sea). - Environmental Monitoring and Assessment 174, 361-388.

Development of models and methods to support the establishment of Danish River Basin Management Plans

Scientific documentation

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Development of models and methods to support the establishment of Danish River Basin Management Plans

Scientific documentation

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Data sheet

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Contents

1 Prologue

The Food and Agriculture Agreement (in Danish: "Fødevare- og landbrugspakken") adopted by the Danish parliament in February 2016 allocated resources for an international evaluation of the marine models and methods that constitute the scientific foundation for the Danish River Basin Management Plan 2015-2021 (RBMP 2015-2021). The evaluation is one of more initiatives to ensure a best practice basis for the third Danish RBMP to be implemented in 2021. With the mandate to evaluate the marine model tools and methodologies applied in the RBMP 2015-2021, a panel of European experts with well-established knowledge in marine ecosystems and marine management has been engaged by the government to conduct such an evaluation in 2017.

The expert panel has been instructed to (citation from their letter of instruction):

"for the expert panel to perform a thorough evaluation of the marine modelling tools that form the basis for the mitigation demands for land-based nitrogen (N) runoff in the Danish River Basin Management Plans, with regards to the importance of N, as well as other relevant pressures such as phosphorous, fisheries etc. In particular, the evaluation panel has to respond to points such as:

- *Evaluate the use of models for determination of type specific reference values (according to WFD annex 2) for the water quality element phytoplankton (chlorophyll).*
- *Evaluate the use of models to determine environmental targets (Max Allowable nitrogen Input MAI) and mitigation needs to achieve good environmental status – and evaluate differences and similarities between the use of different methods and model types for coastal waters with different typology.*
- *Evaluate the estimated nitrogen target loads and mitigation needs in the Danish River Basin Management Plans and evaluate the method for determining the Danish proportion of total mitigation needs. How is the current environmental status in Danish coastal waters determined by N runoff from Danish land areas in relation to other pressures such as N released from sediments and N loads from catchments in neighbouring countries and airborne N deposition (the Danish share of the total mitigation needs related to N)?"*

The marine models applied in the RBMP 2015-2021 were developed by DHI, an international research and consultancy institute, and Danish Centre for Environment and Energy at Aarhus University (hereafter AU). Both organisations are affiliated to the Ministry of Higher Education and Science. DHI and AU have been requested to prepare scientific documentation of the models and their use in English (existing material is in Danish) to be used as basis for the evaluation, with the tasks of the expert panel in mind. Main focus of this report is the models and methods developed to support the establishment of Danish RBMP 2015-2021, and other pressures such as phosphorous, fisheries etc. has been assessed according to their potential impact on the Danish RBMP as part of the final discussion (chapter 9).

Hence, this report presents the scientific documentation of the developed marine models and their application in the preparation of the Danish River

Basin Management Plan 2015-2021, including a brief discussion of other alternative/potential pressures than nitrogen.

Objectives and content of the marine development project

The overarching objective of the model development project was to support the implementation of the EU Water Framework Directive (WFD) in Danish coastal waters by providing modelling tools and methods for calculating the maximum allowable input (MAI) of nutrients, which should not be exceeded as this would prevent maintenance/achievement of good ecological status (GES) required by the WFD. The objective was supported by the following sub-objectives:

- To provide and implement a toolbox for defining and improving the Danish River Basin Management Plan 2015-2021, including development of indicators for biological quality elements and supporting elements.

- To ensure optimal coverage of Danish water bodies, including areas with no or few observations.

- To base the development on state-of-the-art knowledge.

The current report presents the scientific documentation of the models and methods developed, illustrated by examples for selected water bodies.

Allocation of roles

In the project on development and application of marine models for the Danish RBMP 2015-2021, which commenced in summer 2013, DHI developed models and methodologies for application of dynamic mechanistic models, while AU was responsible for the development and application of statistical models. Merging of methodologies and models into one aggregated output was a joint task. Briefly described, the tasks were separated as follows:

- DHI contributed with the development of mechanistic hydrodynamic and biogeochemical models, the subsequent calibration and evaluation of those models as well as processing of mechanistic model results and the following input to the shared management tools.

- AU contributed with the development of statistical models and their subsequent evaluation as well as processing of statistical model results and the following input to the shared management tools.

- Both institutions contributed to the development of reference values for chlorophyll-*a*, preparation of methods for combining model results into specific nutrient reduction targets and corresponding MAI to ensure GES in all Danish water bodies.

This separation of tasks implies that the individual chapters (chapter 6, 7, 8.3 and 8.4) have different authors and that quality assurance has been undertaken according to the rules of the responsible organisation.

2 Introduction

2.1 Eutrophication of Danish waters

During the last half of the 20th century, coastal ecosystems worldwide have been under extensive anthropogenic and climatic pressure. Nutrient enrichment, exploitation of coastal resources, overfishing, destruction of habitats, chemical pollution, physical changes (Boesch 2002) all impact the ecological systems as well the socio-economic systems (Newton et al. 2014) and cause changes in ecosystem services and human life. Coastal eutrophication, resulting from increased nutrient input, has been identified as the main driver of the deterioration of coastal ecosystems in Europe, North America, Asia and Oceania (Boesch et al. 2001, Boesch 2002, Conley et al. 2009a, Conley et al. 2009b). The eutrophication has resulted in higher phytoplankton production, blooms of opportunistic algae, decreased light penetration, loss of underwater macrophytes and increased occurrence of hypoxia and anoxia.

The first clear signs of eutrophication in Danish coastal waters appeared in the 1970s (Clarke et al. 2003, 2006; Ellegaard et al. 2006). From the beginning of the last century, the surplus of nitrogen (N) in Danish agriculture rose from approximately 100,000 tons N year^{-1} to about 500,000 tons N year^{-1} (*Figure 2.1*) and peaked during the late 1970s and early 1990s. Combined with, among other factors, changes in land use, such as inclusion of wetlands and drainage, this increased the diffuse loadings to rivers, lakes and the marine environment. Data on nutrient loadings are not available before the early 1980s. At that time the estimated N loadings from Danish land to the marine environment ranged between 100,000 and 120,000 tons year^{-1} (Kaas et al. 1996). This nitrogen originated from both diffuse loadings and point sources such as industry and wastewater discharges. Phosphorus (P) loadings were in the order of 15,000 tons year^{-1} (Kaas et al. 1996), deriving mainly from sewage treatment plants without tertiary treatment comprising phosphorus removal. Contrary to N loading, the estimated drop in P loadings to the marine environment occurred as early as in the beginning of the 1990s (data not shown). In the Baltic Sea (from the Arkona Basin and eastwards), similar peaks were estimated for the diffuse loads between the 1970s and the mid-1990s for both N and P (Gustafsson et al. 2012; Savchuk et al. 2012a).

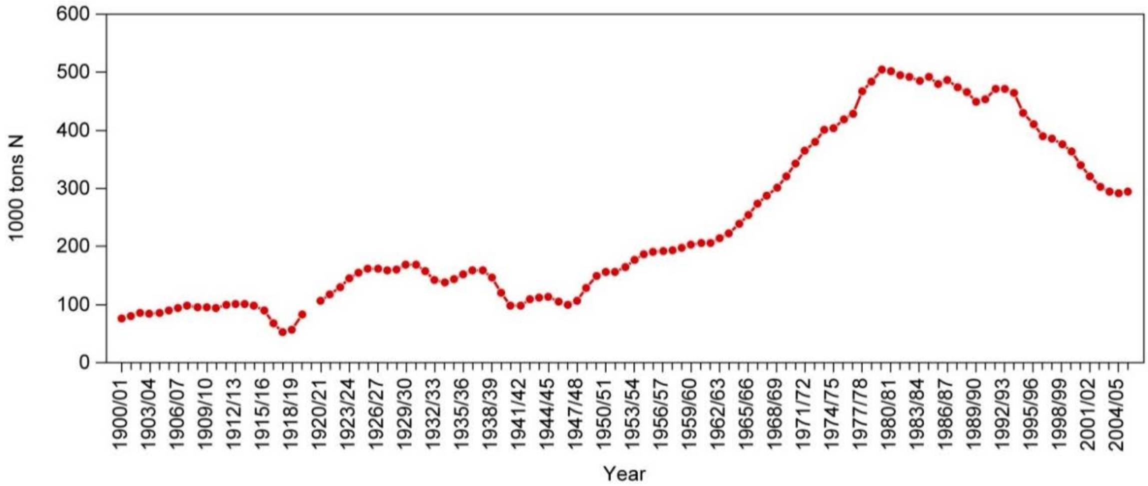


Figure 2.1. Surplus of nitrogen in Danish agriculture (three-year average) (from Kyllingsbæk 2008).

To combat coastal eutrophication, numerous local and regional treaties have been developed and implemented, especially in the US and EU. Although there was an emerging scientific understanding of the cause and effects of coastal eutrophication in the 1970s-1990s, quantitative relations between nutrient loading and environmental quality were not established, and hence, not an integral instrument in water management. Hence, the first action plans in Denmark as well as around the world focused on reducing nutrient inputs and not on obtaining a certain environmental state. Thus, in 1985, Denmark adopted its first action plan, the NPO-plan (Nitrogen-Phosphorous-Organic matter) with the aim to reduce N, P and O input from multiple sources, including nitrogen from agricultural production. Continued events of oxygen depletion accelerated new interventions, and in 1987 the first Danish water action plan (VMP I) was adopted. The quantitative goal of VMP I was to reduce nutrient loadings by 50% for nitrogen and 80% for phosphorous relative to the loadings in the late 1980s. The series of water action plans implemented in Denmark in order to counteract eutrophication now encompasses the NPO-plan (1985), the Action Plan on the Aquatic Environment (Vandmiljøplan (VMP I) in Danish) from 1987, VMP II (1997), VMP III (2004-2005), the RBMP 2009-2015 (2014) and now the RBMP 2015-2021 (2016).

Since the water action plans of the 1980s, focus has shifted from national stand-alone plans to region-wide plans such as the EU Water Framework Directive (WFD) that was adopted in 2000, and the HELCOM Baltic Sea Action Plan (BSAP), implemented by all the coastal Baltic countries and the EU in 2007. Hence, national statutes and regulations have changed to support implementation of multijurisdictional compacts administered by regional bodies such as the Helsinki Commission (HELCOM) for the Baltic Sea and the Oslo-Paris Commission (OSPAR) for the North Sea.

The early action plans were based on relatively arbitrary goals stipulating reductions in nutrient inputs by a certain percentage without quantitative understanding of how and when this would affect the coastal ecosystem (Boesch 2002). With BSAP I from 2007, the WFD from 2000 as well as the later Marine Strategy Framework Directive (MSFD-D5[1]) from 2010, a conceptual change has been introduced to create scientific coherence between the goals of achieving a certain – and political defined – environmental quality, and the required reduction of nutrient inputs.

The historic reduction goal originating from VMP I of an 80% reduction of the phosphorus input was reached in the mid 1990s, whereas the 50% goal for nitrogen was nearly met in 2012. Today, the action plans have not only significantly reduced nutrient loadings to the aquatic environment (*Figure 2.2*) but have also led to detectable environmental improvements (Riemann et al. 2016). However, according to the WFD only a few Danish areas have obtained good ecological status (GES).

[1] MSFD includes a number of different descriptors for monitoring the overall environmental status and encompasses many more factors than eutrophication. However, descriptor 5 (D5) is an eutrophication descriptor, and according to Naturstyrelsen 2012, D5 should be aligned with WFD targets in areas outside the 1 nm border defining the WFD areas

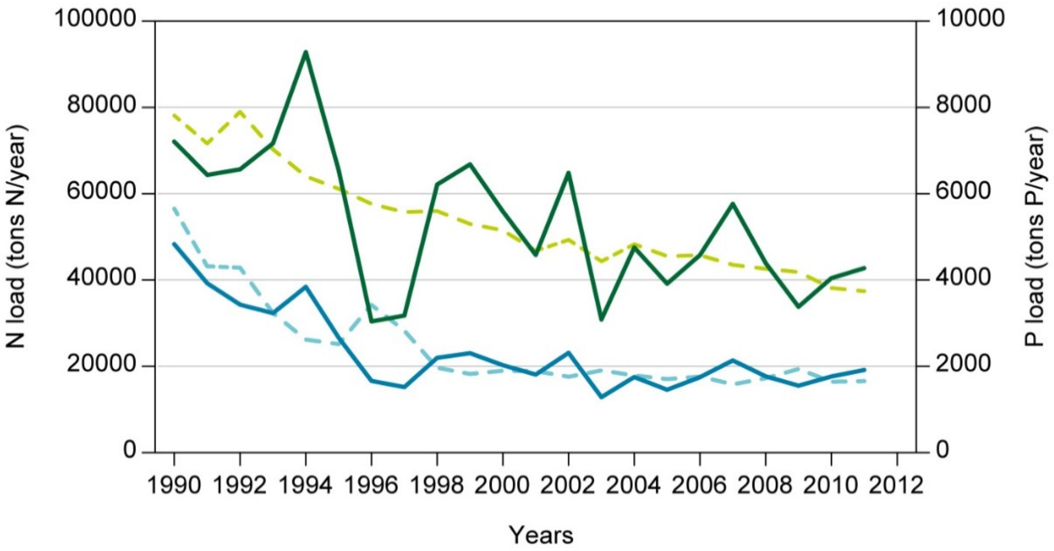


Figure 2.2. Time series of Danish land-based loading of nitrogen (green) and phosphorous (blue) from 1990 to 2011. Full line is actual loadings, dotted lines are flow-normalised loadings (Windolf et al. 2013).

2.2 Implementation of the Water Framework Directive in Denmark

After the Water Framework Directive (WFD) entered into force in EU in 2000 (Directive 2000/60/EC), it was adopted by the Danish parliament in 2003. The essence of the directive is that all surface waters (e.g. lakes, rivers, coastal waters) should achieve at least good ecological status (GES) and good chemical status (GCS).

2.2.1 Ecological status of Danish water bodies

"Ecological status" expresses the quality of the structure and functioning of an aquatic ecosystem. It is categorised into five classes (Bad, Poor, Moderate, Good and High) defined relative to an undisturbed (reference) condition. For coastal ecosystems, the ecological status is defined in terms of "biological quality elements" encompassing the composition, abundance and biomass of phytoplankton, the composition and abundance of other flora (macroalgae and angiosperms) and the composition and abundance of benthic invertebrate fauna. Achievement of GES requires that all biological quality elements fulfil the targets set for each quality element (the so called "one out all out principle").

For the Danish plan period 2015-2021, ecological status was classified according to three indicators:

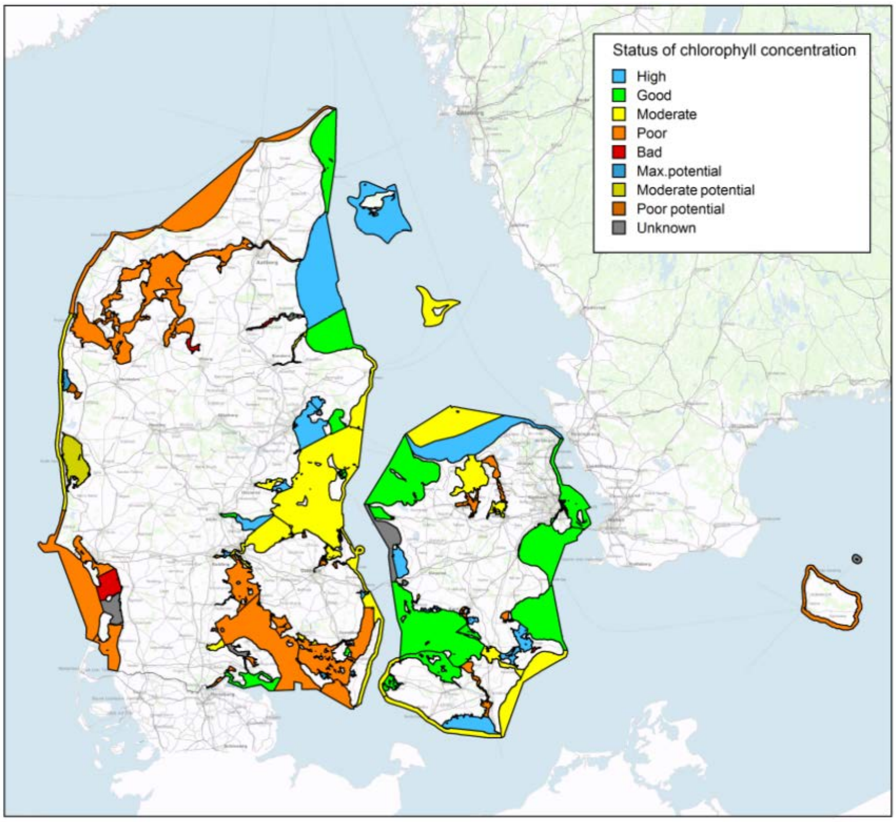
- Chlorophyll-*a* concentration is an indicator for phytoplankton biomass and is assessed as the average (May-September) chlorophyll-*a* concentration within the inner Danish waters and 90-percentile of the March to September chlorophyll-*a* concentrations for water bodies located in the North Sea and the Skagerrak.

- Eelgrass depth limit is an indicator for the quality element angiosperms and defined as the maximum depth with at least 10% cover.

- Danish Quality Index (in Danish: Dansk Kvalitets Index - DKI) is an indicator for the composition and abundance of benthic fauna and a multimetric index including both biodiversity and sensitivity/tolerance towards disturbance.

Based on targets for the three indicators and observations of status values, the Danish Environmental Protection Agency (Danish EPA) has classified the status of the different Danish water bodies (see *Figures 2.3* to *2.5*).

Figure 2.3. Assessment of chlorophyll-*a*[2] status of Danish marine water bodies



[2] The Good-Moderate targets adopted for this assessment result from the model development described in this report. See section 8.1 for more details.

Figure 2.4. Assessment of eelgrass depth limit status of Danish marine water bodies.

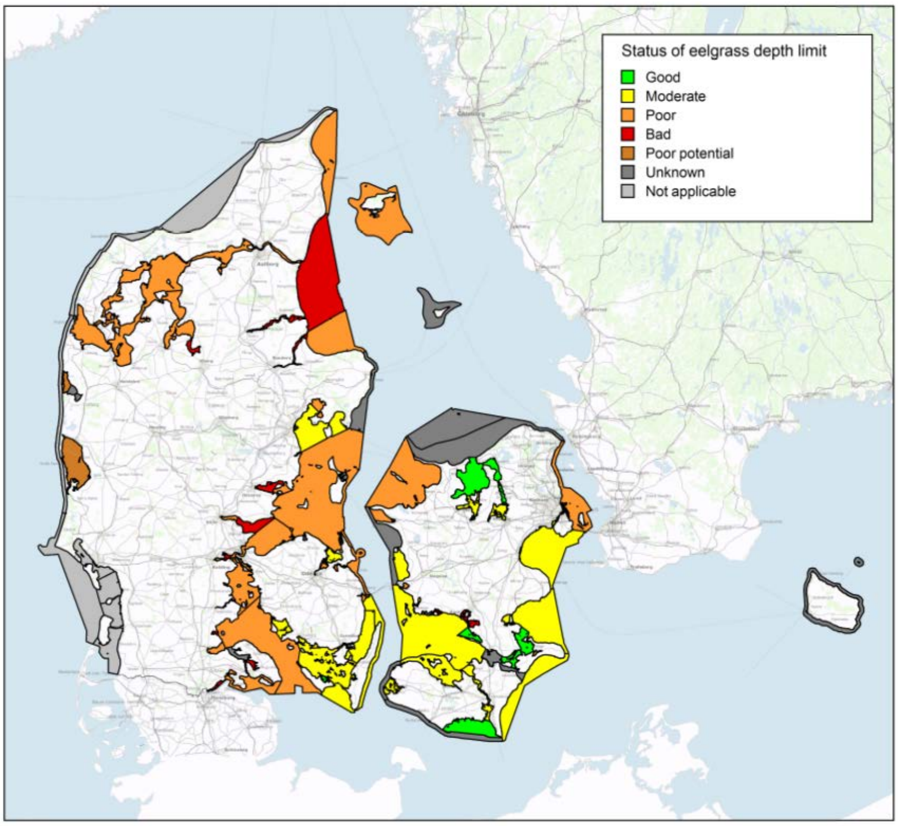
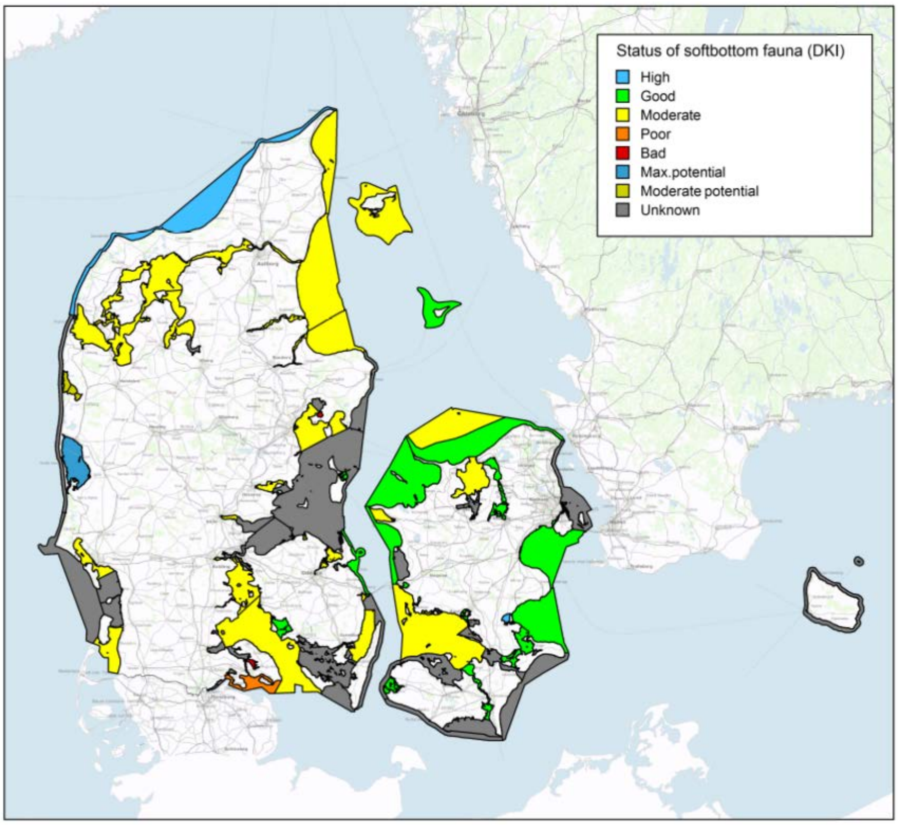


Figure 2.5. Assessment of DKI status of Danish marine water bodies.



2.2.2 The first RBMP and the request for improved modelling tools

The first Danish River Basin Management Plan (RBMP 2009-2015) only included the quality indicator "depth limit of eelgrass" and was largely based on monitoring data from the Danish National Aquatic Monitoring and Assessment Program (DNAMAP, in Danish: NOVANA) and a simple statistical model linking eelgrass depth limit to total nitrogen (TN) concentrations ("the Eelgrass Tool"). However, though the best available tool at that time, it was acknowledged and strongly backed up by a stakeholder consultation that development of new tools was required to support differentiated assessments taking into account the diversity of the water bodies and including additional quality elements.

With special emphasis on the use of eelgrass as quality indicator, during 2011 and 2012 several government appointed working groups reviewed the Eelgrass Tool and provided recommendations for improving the modelling and management tools to be used for the RBMP 2015-2021. The final recommendations were formulated by the "Eelgrass Working Group II" and published in December 2012 (Naturstyrelsens arbejdsgruppe 2012).

As a response to the raised criticism and the above-mentioned recommendations, by the end of 2012 the Danish EPA[3] developed an initiative with the objective to improve the scientific foundation of the RBMP 2015-2021 and reduce the uncertainties. The initiative consisted of three projects: i) a project aiming at developing a catchment model complex; ii) a project aiming at developing improved lake water quality models and iii) a project aiming at developing models for describing marine water quality and effort needed to obtain GES in the marine water bodies. The development of the latter is the subject of the present documentation report.

The development of the marine model tools was largely founded on the recommendations of the "Eelgrass Working Group II" regarding the need for mechanistic modelling tools covering the Danish coastal waters and selected estuaries, as well as more simple statistical tools for other smaller estuaries. These models should include:

- dynamic and spatial considerations of eelgrass such as buffer effect and feedback mechanisms with respect to, for instance, nutrients and resuspension
- additional quality elements than eelgrass (depth limit) such as, for instance, phytoplankton (chlorophyll-*a*)
- sediments and sediment nutrient pools.

If possible, the models should also be able to:

- assess the impact of climate change
- assess the effect of changes in bottom trawling and mussel dredging, effects of establishing stone reefs, mussel farming and seaweed production.

2.2.3 From RBMP 2009-2015 to RBMP 2015-2021

Based on the above recommendations, AU, DHI and the Danish EPA defined a number of required model developments and deliverables. This resulted in a development plan covering:

[3] Danish EPA; part of the Ministry of Environment and Food of Denmark. At that time the Nature Agency (NST) under the Danish Ministry of Environment.

- mechanistic model development including nutrient pools in the sediment and effects from neighbouring waters
- mechanistic model development including dynamic and spatial considerations of eelgrass
- a number of statistical models for assessing ecological effects of nutrient loadings
- development of chlorophyll-*a* targets for all water bodies from Skagen and southwards
- model scenarios and methods for estimating maximum allowable nutrient input (MAI), that will support achievement of good ecological status

The objective of the model development was to improve the scientific basis for the RBMP 2015-2021 and to accommodate the recommendations made by the different working groups. The following chapters include the scientific documentation of the models and methods developed, illustrated by examples from selected water bodies.

The objective of the evaluation is – as DHI and AU understand it – to further qualify this development by identifying improvements to be implemented in the RBMP 2021-2027.

3 Danish marine waters

Danish marine waters can be divided into the inner Danish waters (Danish part of Kattegat, Belt Sea, Western Baltic Sea), the Skagerrak, the North Sea, the Wadden Sea and more than 60 shallow estuaries (in Danish called fjords) (Conley et al. 2000). Most of the estuaries are located in the inner Danish waters, limited by the Kattegat/Skagerrak border.

The inner Danish waters form a transition zone between the Skagerrak and the Baltic Sea. The hydrography of the transition zone is very dynamic, rather complex and greatly influenced by the location between the brackish Baltic Sea (salinity ~8) and the saline North Sea (salinity ~34). The water exchange between the Baltic Sea and Kattegat takes place through the Belt Sea comprising tree straits; the Sound, the Great Belt and the Little Belt (see *Appendix A*). The freshwater inflow from rivers to the Baltic Sea results in a net outflow through the Danish straits of approximately 15,000 $m^3 s^{-1}$ (Skogen et al. 1998; Edelvang et al. 2002) with the Sound discharging about 25% of the total water flow, the Great Belt about 65% and the Little Belt about 10% (Jakobsen & Ottavi 1997). Because of the outflows, the surface waters of Kattegat are heavily affected by the conditions in the Baltic Sea. Besides influencing the salinity, the outflows bring nutrients (inorganic and organic) from the Baltic Proper to the inner Danish waters. Prolonged periods of easterly winds increase the outflow and may in summer result in noticeable transport of phytoplankton (cyanobacteria) to the south-eastern Danish coastal areas.

Denmark has numerous shallow marine estuaries ranging from the Limfjorden, covering about 1,500 km^2, to small bays and inlets comprising only a few hectares. Typically, these coastal areas are shallow (most are <3 m deep, Conley et al. 2000) with soft bottom. In many areas, the water turbidity allows irradiance to reach the seabed, which supports the growth of both benthic microalgae and the important structuring macrophytes, including eelgrass (*Zostera marina*) and macroalgae. Tidal amplitude is small, only 0.1-0.2 m in many areas, except for the Wadden Sea, which means that the water exchange to adjacent, more open areas is low and mainly driven by density currents, wind stress and atmospheric pressure. Rasmussen & Josefson (2000) calculated a hydraulic residence time of <4 months for 65% of the Danish estuaries and a flushing time (volume/freshwater input) of >1 year for 48% of the areas. Flushing time shows high seasonal variability (lowest during winter due to larger runoff and meteorology).

On an area basis, nutrient loading from Danish catchments ranks among the highest around the Baltic Sea (HELCOM 2015) and as a consequence of the high nutrient loading and sensitive recipient water bodies, a number of Danish coastal areas show classical symptoms of eutrophication, such as increased phytoplankton biomass, decreased light penetration, seasonal hypoxia and loss of benthic vegetation.

3.1 WFD water bodies

In connection with the implementation of the WFD, the Danish marine waters have been divided into 119 specific administrative water bodies, see *Figures 2.3* to *2.5*, with 108 located in the inner Danish waters (south of the Skagen-Gothenburg transect) and 11 in the North Sea and the Skagerrak.

These water bodies have been defined by Danish EPA (Bek. nr. 837 2016). According to Bek. nr. 837 (2016), the 119 water bodies are divided into 18 water body types based on physical characteristics as described in Dahl et al. (2005). Water body numbering and naming is included in *Appendix A*.

The 119 water bodies cover a wide range of environments with varying physical and ecological characteristics. They differ in size, water depth, water exchange, nutrient loading etc. The median water body area is 3,300 ha (range between 8 ha and 180,600 ha), with 21 (18%) covering less than 500 ha. The catchments associated with the water bodies have a median size of 14,000 ha (range between 36 ha and 498,300 ha), 61 (51%) comprising between 5,000 and 50,000 ha. This results in a median ratio between catchment area and water body area of 3.4 (25 percentile = 1.3, 75 percentile = 10.4). More details are included in *Figure 3.1*.

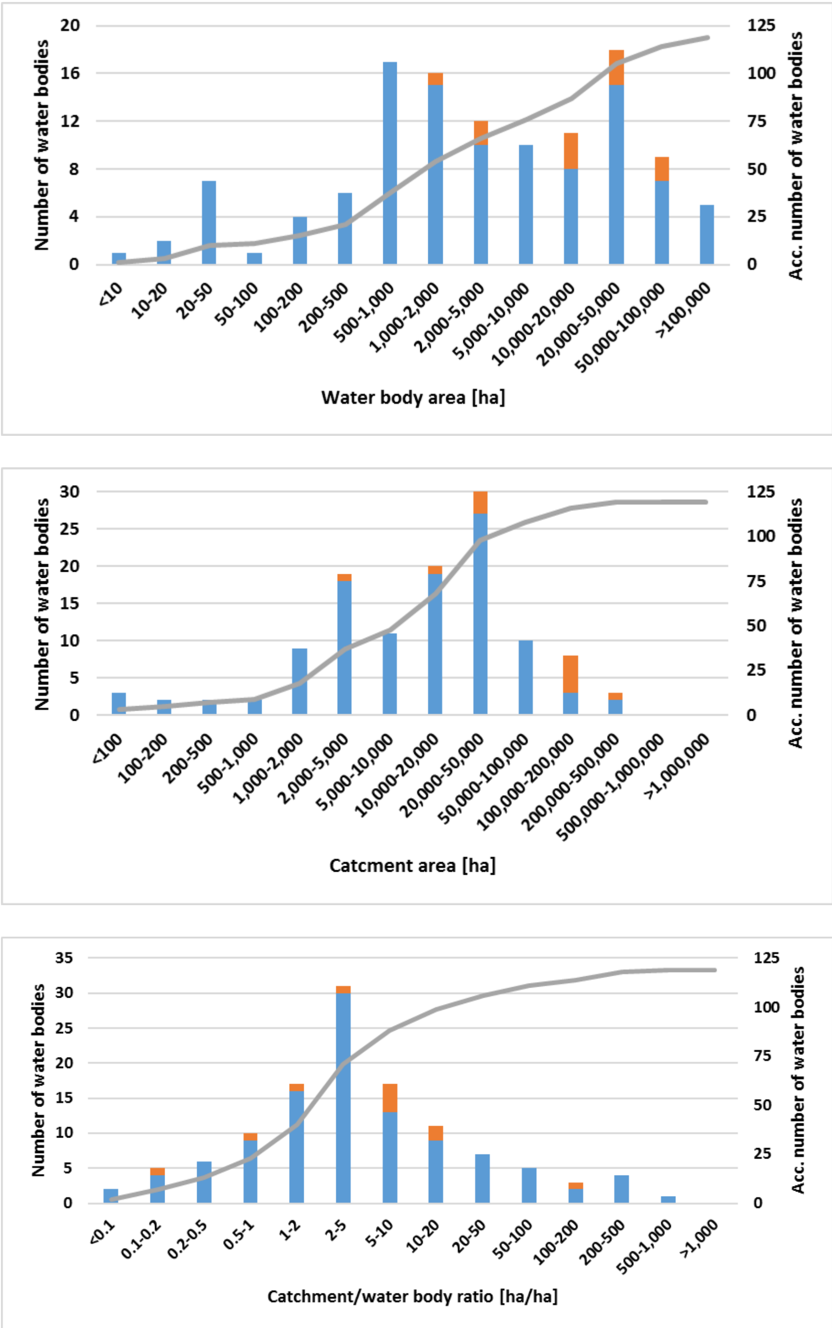


Figure 3.1. Frequency plot of water body area (top panel) and catchment area (middle panel) and the relation between catchment area and water body area (bottom panel). Blue bars represent water bodies of the inner Danish waters and orange bars water bodies in the Skagerrak, North Sea and Wadden Sea areas. Grey line represents accumulated number of water bodies (right y-axis).

3.2 Typology

The Danish WFD water bodies have been categorised into 6 open water types, classified according to bottom water salinity, exposure and tidal range, and 12 estuarine water body types, classified according to stratification, salinity and freshwater influence/residence time (Dahl et al. 2005). The typology was made with main focus on the drivers of benthic vegetation and fauna, thus rendering parameters like bottom water salinity and stratification of crucial importance. However, in our project the typology should support the establishment of chlorophyll-*a* reference conditions and include physical characteristics affecting a water body's sensitivity to anthropogenic pressures, mainly originating from the catchment. Today, more than 90% of the total annual nutrient loading to the Danish coastal zone derives from riverine input (Markager et al. 2006), suggesting that freshwater influence and residence time would be useful proxies for nutrient availability. In addition, the degree of freshwater influence will determine the sensitivity of water bodies to changes in the catchment. In the typology developed by Dahl et al. (2005), the freshwater supply and the residence time is used to estimate the freshwater influence using a Freshwater index F defined as:

$$F = \frac{R}{T}$$

where R is the annual average freshwater supply ($m^3\ sec^{-1}$) and T is the residence time (days) defined as:

$$T = \frac{V}{Q + R} \quad where \quad Q = \frac{\frac{S}{Sm}}{\left(1 - \frac{S}{Sm}\right)} R$$

V is the estuary volume in km^3, Q is the saltwater supply, S is surface salinity in the estuary and Sm is the salinity at the mouth of the estuary.

By focusing on the influence of freshwater, which includes freshwater discharge and residence time in combination with salinity, the original 12 estuarine water body types have been reduced to 4 as outlined in *Table 3.1* and in *Figure 3.2*. As to the inner Danish waters, 4 sub-types were defined based on Baltic Sea influence and upwelling.

Table 3.1. Applied typology for Danish WFD water bodies (modified after Dahl et al. 2005). The water body types are classified according to main type (open water, estuarine or sluice), salinity and freshwater influence (including freshwater runoff and residence time).

Applied typology	Type NS	Type 1**	Type 2	Type 3	Type 4	Type 5
Main type	North Sea	Open water	Estuarine	Estuarine	Estuarine	Sluice
Typology according to Dahl et al. (2005)	OW4-5	OW1-3	M1-2;P1-2	M3-4; P3-4	O3-4	Sluice
Salinity		5-30	>5	> 5	< 5	na
Freshwater influence	Low	Low	Low, F<0.1	High, F>0.1	High, F>0.1	na
Number*	7	29	48	23	5	5

* Two water bodies are not included in the typology, see *section 8.1* for explanation and *Figure 3.2* for location.

** Type 1 consists of 4 sub-types.

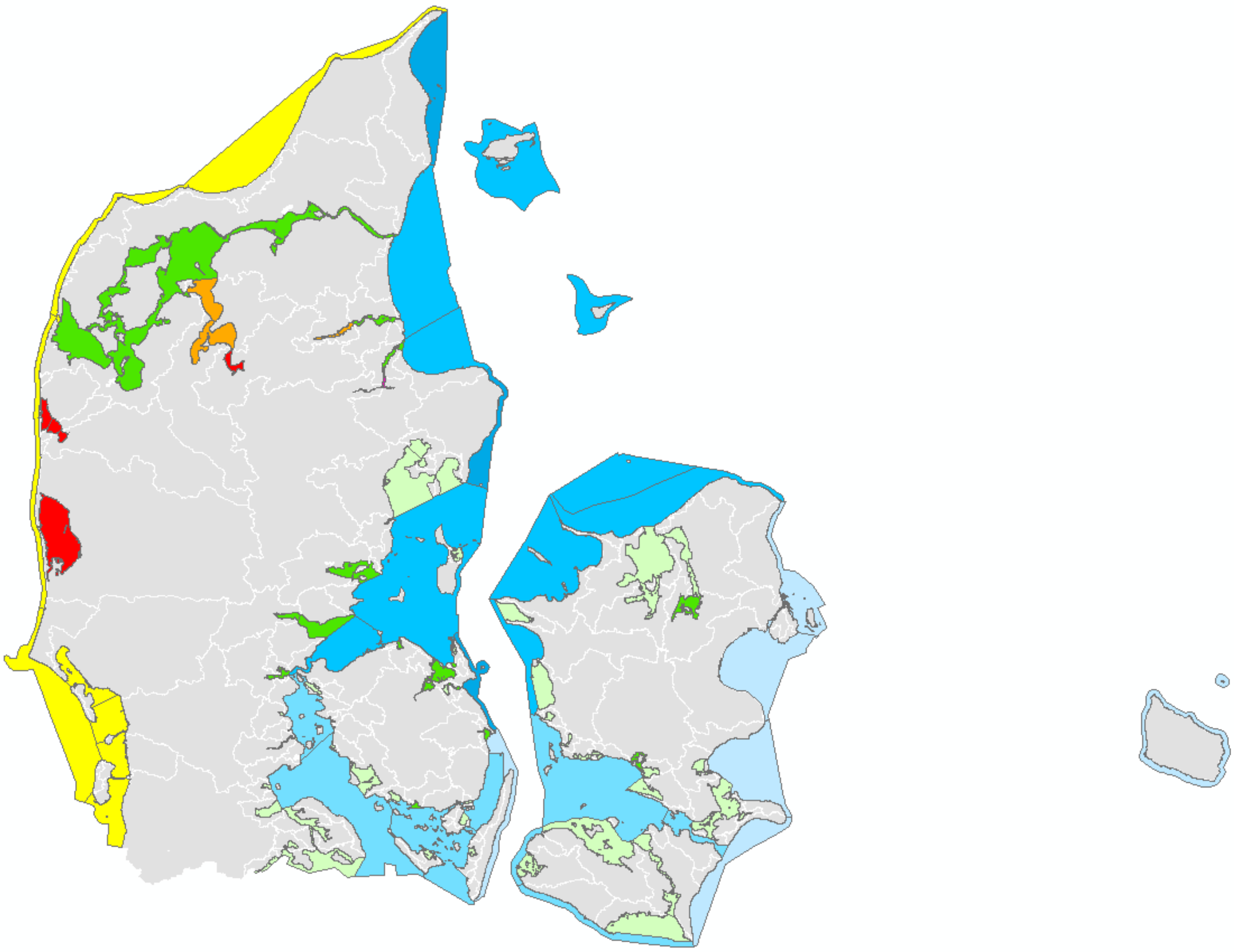


Figure 3.2. Applied typology for Danish WFD water bodies (modified from Dahl et al. 2005). Type 1 (blue – the four different blue colours indicate sub-types), Type 2 (light green), Type 3 (dark green), Type 4 (pink[4]), Type 5 (red), original North Sea types[5] (yellow) and areas not included in the typology[6] (orange). The types are characterised in *Table 3.1*.

As we move from open waters towards more coastal and semi-enclosed waters, water depth decreases and the catchment to water body area ratio increases (*Figure 3.3*). A decreasing depth combined with an increasing catchment to water body area ratio imply an increasing pressure from Danish loadings on the coastal areas compared to the more open water bodies. This is supported by analysis of total N (TN) concentrations, total P (TP) concentrations, light attenuation (K_d) and chlorophyll-*a* concentrations within the different types. Type 1 shows much lower concentrations of TN, TP and chlorophyll-*a* as well as lower K_d values compared with the more coastal water bodies, see *Figure 3.4*.

[4] Not visible in the figure
[5] The typology of North Sea water bodies has not been assessed in this study; thus, the types are defined according to the original division in Dahl et al. (2005).
[6] Two water bodies are considered 'out of category' due to anticipated natural oxygen depletion, see section 8.1 for more details.

Figure 3.3. Physical characteristics of Danish coastal areas and estuaries covered by this work. A) Mean depth of the five different water body types. B) Catchment area to water body area ratio. The boxplots show the median and the 25th (Q1) to the 75th (Q3) percentile. Whiskers represent the distance from Q1 - (Q3-Q1)*1.5 to Q3 + (Q3-Q1)*1.5 and outliers (shown as dots) are data points outside the range of the whiskers.

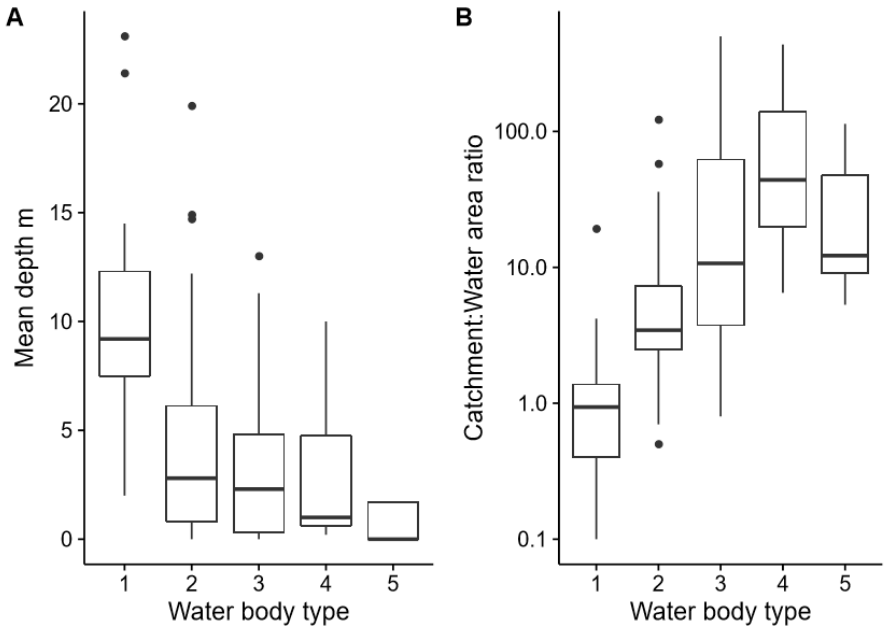
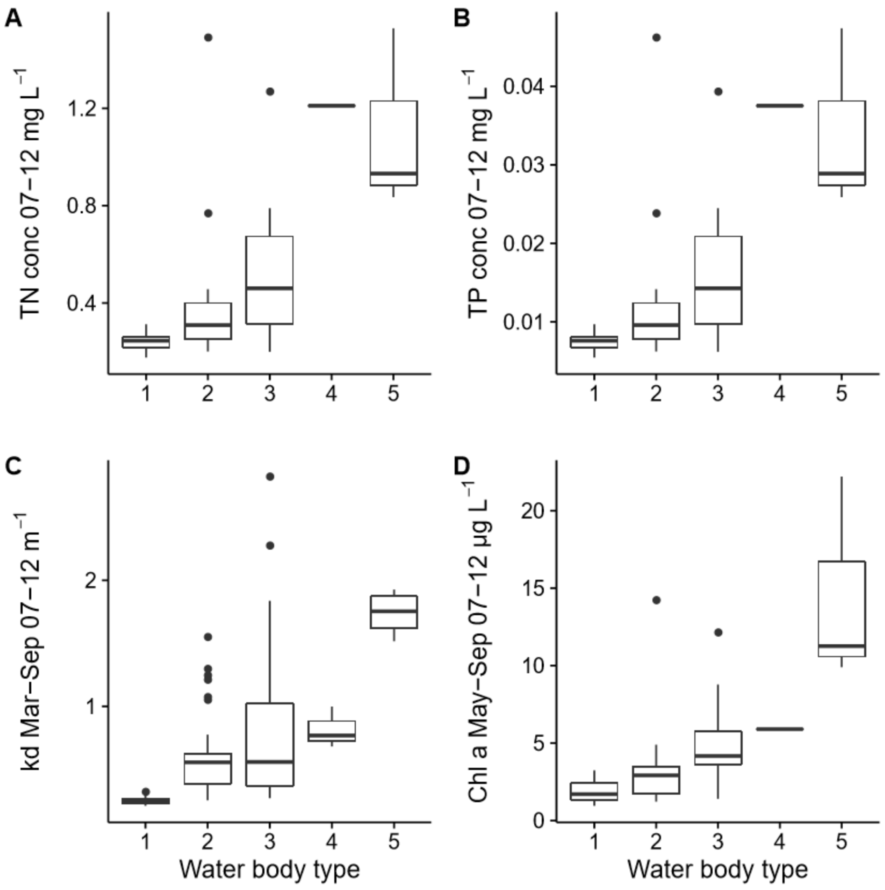


Figure 3.4. Statistics of the physical/chemical and biological state of the different water body types based on observations during 2007-2012 from available monitoring stations. A) Total nitrogen (TN) concentration in mg L^{-1}, B) total phosphorus (TP) concentration in mg L^{-1}, C) light attenuation coefficient K_d (m^{-1}) from March to September, D) chlorophyll-*a* concentration (µg L^{-1}) from May to September.



4 Danish monitoring data DNAMAP (NOVANA)

The Danish National Aquatic Monitoring and Assessment Programme (DNAMAP) was initiated in October 1988 with the purpose of sampling data for assessment of nutrient loadings from various sources to Danish waters (fresh and marine) and the physical, chemical and biological status of the receiving waters as well as to document the effects of action plans on the aquatic environment. The programme has been adjusted several times during the last decades (1992, 1997, 2003, 2010 and 2016), partly to meet the requirements of the WFD and partly to comply with other international obligations (HELCOM and OSPAR). Originally, it was probably one of the most comprehensive programmes in the world (Conley et al. 2002).

4.1 Marine monitoring data

The marine part of the current (2011-2016) national monitoring program contains >90 water chemistry monitoring stations distributed along the Danish coast and in Danish open waters (*Figure 4.1*).

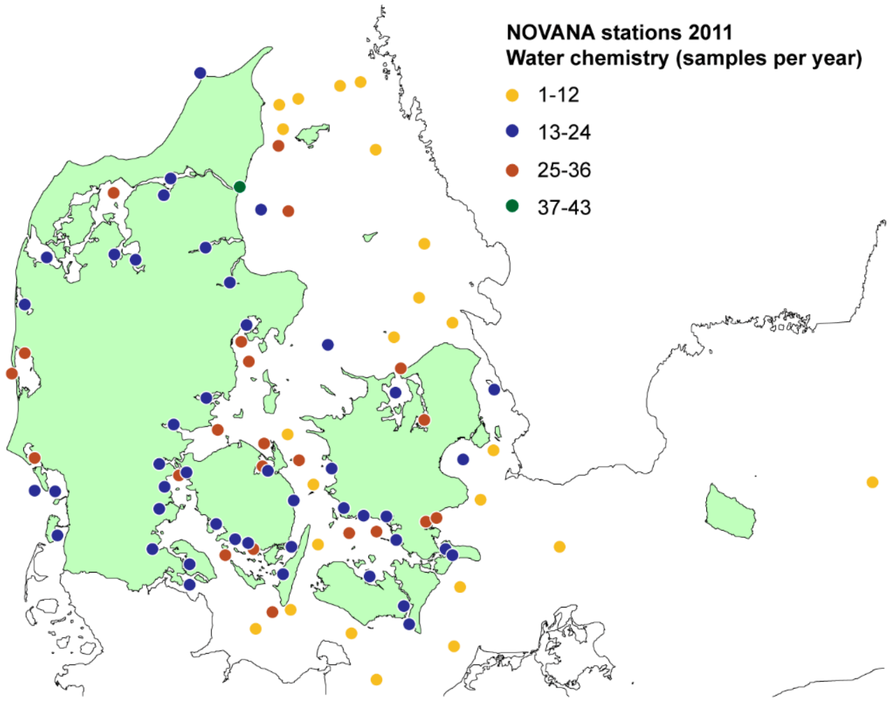


Figure 4.1. Marine stations for monitoring of water chemistry, salinity, light attenuation, chlorophyll-*a* and fluorescence in 2011. Location of monitoring stations and sampling frequency may change slightly between years

The sampling frequency differs between the monitoring stations, ranging from biweekly sampling at most coastal stations to less than 5 samples per year at most open water stations. A suite of physical and chemical parameters is measured, including inorganic nitrogen (DIN), total nitrogen (TN), inorganic phosphorous (DIP), total phosphorus (TP), dissolved oxygen and chlorophyll-*a*. At the same stations, CTD measurements are taken to determine salinity, temperature and depth. In a fully mixed water column, one sample is taken, while in a stratified water column two samples are taken from the top and bottom layer, respectively. CTD measurements are made continuously through the water column. Data acquisition is performed in accordance with the technical guidelines for DNAMAP marine monitoring

(Kaas & Markager 1998; Jakobsen & Fossing 2015; Markager & Fossing 2015; Fossing et al. 2015, Fossing & Hansen 2015).

4.2 Land-based N and P loadings

Land-based loadings of N and P constitute an important input to the development of both the statistical and the mechanistic models. The Danish load data applied in this study are provided by Danish EPA via DCE/AU, Department of Bioscience. The data are part of the annual national inventory elaborated by AU (Windolf et al. 2013). They cover the period from 1990 to 2012 and are similar to the reporting to HELCOM.

The statistical models are based on nutrient loadings over the entire 23-year period, whereas the development of mechanistic models encompasses 10 years of data (2002-2011). In addition to the Danish land-based loadings, the mechanistic models also include N and P loadings at a regional scale, i.e. loadings to the entire Baltic Sea, and atmospheric deposition, see chapter 7.

Since the 1990s, Danish land-based nutrient loadings have declined significantly, and today the N and P loadings constitute approximately 50% and 30% of the 1990 loads, respectively (Carstensen et al. 2006; Riemann et al. 2016). The reduction in P loads is mainly due to improved treatment of urban and industrial wastewater. In contrast, the diffuse P loads from agricultural land are more or less unchanged (Riemann et al. 2016). For nitrogen, the measures initiated to reduce loadings mainly target at diffuse sources. Changes in nitrogen concentrations have occurred more recently and are partly masked by large interannual variations in freshwater discharge (Carstensen et al. 2006). Despite the efforts to reduce the diffuse loads, Danish agriculture remains the major source of both N (80%) and P (50%) in Danish streams, lakes and coastal waters (Kronvang et al. 2005).

As highlighted by Carstensen et al. (2006), the P reductions date back to the late 1980s, and since 1998 the loadings have been more or less constant, see *Figure 2.2* in the introduction. The N reductions declined over the entire 23-year period, but after 2003 loading has been rather constant with large interannual variations in actual loads not normalised according to freshwater discharge.

The total national nitrogen loadings demonstrate substantial local variability. *Figure 4.2* shows the development of N loadings since 1990, calculated as actual loadings and flow normalised loadings based on 1990. While the average of all catchments corresponds to the total national trend (black line), N loadings for the individual catchments exhibit large variability and cover reductions of 20% to more than 60%. In *Figure 4.2*, we have highlighted the decline in the loads to the Sound where treatment of discharge from wastewater treatment plants (WWTP) from Copenhagen improved in the first quarter of the period. In Hjarbæk Fjord and Nissum Bredning, Thisted Bredning, Kås Bredning, Løgstør Bredning, Nibe Bredning and Langerak (all part of the Limfjorden), (flow-normalised) changes in nitrogen loadings are minor from 1990 to 2011, partly because wastewater treatment in the Limfjorden was implemented in 1988, i.e. before the start of the time series (as shown in *Figure 4.2*).

In 2011 the Sound accounted for 2% of the national N loadings to the inner Danish waters (including the Limfjorden), whereas Hjarbæk Fjord represented 4%, and Nissum Bredning, Thisted Bredning, Kås Bredning, Løgstør Bredning, Nibe Bredning, Langerak together accounted for 21%.

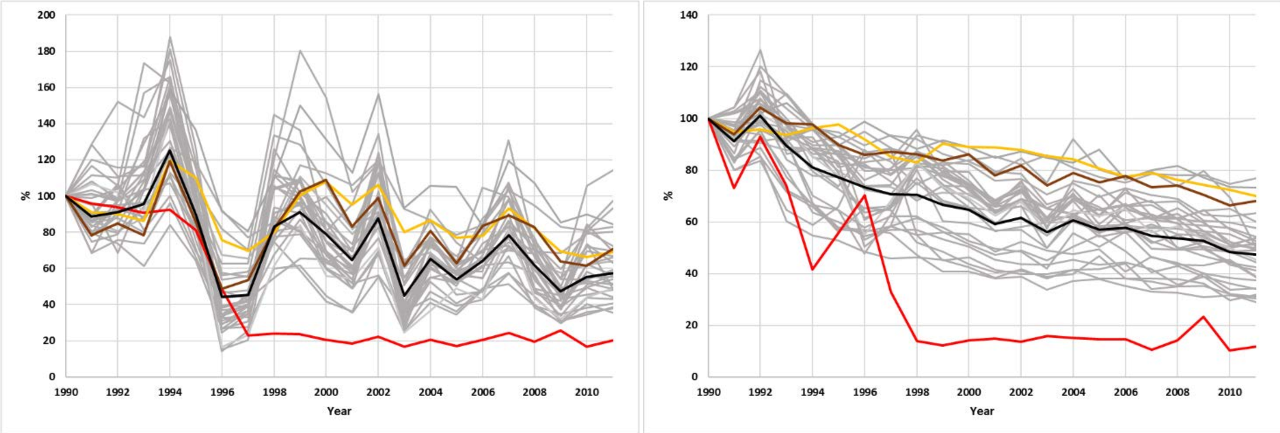


Figure 4.2. Changes in N loadings from the catchments accounting for 80% of total loadings to the inner Danish waters. Calculated as percentages of 1990 loadings based on actual loadings (left) and flow-normalised loadings (right). Black line = average of all catchments. Grey lines = individual catchments. Red line = changes in the Sound (water body no. 6). Orange line = changes in Hjarbæk Fjord, part of the Limfjorden (water body no. 158). Brown line = changes in Nissum Bredning, Thisted Bredning, Kås Bredning, Løgstør Bredning, Nibe Bredning and Langerak (water body no. 156), also part of the Limfjorden. In *Appendix A* water body naming and numbering are included.

A similar analysis of P loadings (see *Figure 4.3*) also demonstrates large variability between water bodies. Contrary to the decline observed in the total national P loading, some water bodies show almost no changes (water body no. 222: Kattegat, Ålborg Bugt (brown line) and water body no. 156: Bjørnholms Bugt, Riisgårde Bredning, Skive Fjord and Lovns Bredning (blue line)). The Sound (water body no. 6 (red line)) shows a large reduction due to enhanced wastewater treatment, and the load to the open part of Smålandsfarvandet (water body no. 206 (yellow line)) increases due to the aquaculture site located here.

In 2011, the Sound accounted for 7% of the national P loadings to the inner Danish waters, whereas the open part of Smålandsfarvandet represented 1%, Nissum Bredning, Thisted Bredning, Kås Bredning, Løgstør Bredning, Nibe Bredning and Langerak for 4% and Kattegat, Ålborg Bugt for 2%.

In *Figure 4.4*, average 2007-2011 N and P loadings are shown for all water bodies. The figure shows that most absolute loadings varied from less than 1 ton N year^{-1} and 1 ton P year^{-1} to more than 5,000 tons N year^{-1} and 5,000 tons P year^{-1}. The catchment area-based loadings constituted the majority of N and P loadings above 14 kgN ha^{-1} year^{-1} and 0.4 kgP ha^{-1} year^{-1}, corresponding to water body area loadings above 50 kgN ha^{-1} year^{-1} and 2 kgP ha^{-1} year^{-1} for the majority of the water bodies.

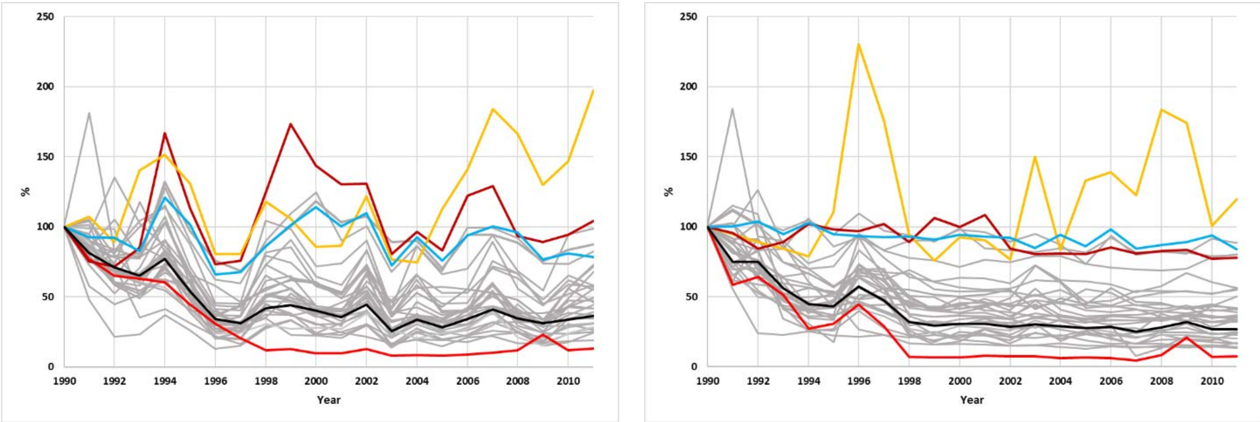


Figure 4.3. Changes in P loadings from the catchments accounting for 80% of the total loadings to the inner Danish waters. Calculated as percentages of the 1990 loadings based on actual loadings (left) and flow-normalised loadings (right). Black line = average of all catchments. Grey lines = individual catchments. Red line = changes in the Sound (water body n. 6). Yellow line = changes in the open part of Smålandsfarvandet (water body no. 206). Brown line = changes in Kattegat, Ålborg Bugt (water body no 222). Blue line = changes in Bjørnholms Bugt, Riisgårde Bredning, Skive Fjord and Lovns Bredning (water body no. 157. In *Appendix A* water body naming and numbering is included.

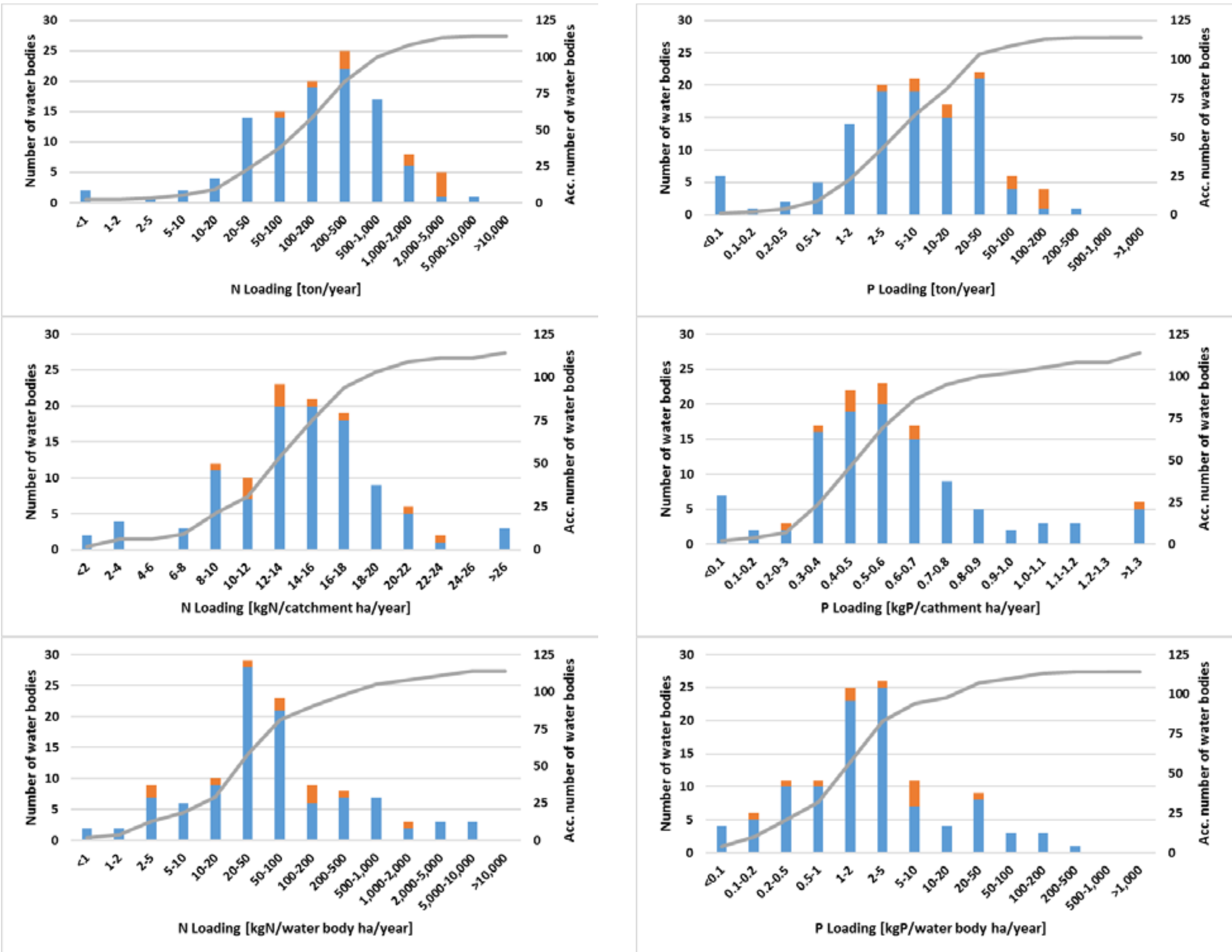


Figure 4.4. Average nutrient loadings for the period 2007-2011 divided into water bodies. Left panels show N loadings and right panels P loadings. Top panels are the yearly loadings per water body, middle panels include the areal loadings per catchment, and the bottom panels represent the areal loadings per water body. Blue columns are loadings to the inner Danish waters and orange columns loadings to the North Sea and Skagerrak. Grey line represents accumulated number of water bodies (right y-axis).

4.3 Nutrient deposition

In addition to the land-based loadings of N and P, the mechanistic models require input of atmospheric deposition of N, which is an important additional source of nitrogen to the marine ecosystems. Data on atmospheric N deposition are provided by AU, Department of Environmental Science. The data form part of the national inventory elaborated annually by AU (Geels et al. 2012; Ellermann et al. 2012) and prepared using deposition modelling covering the period 2002-2011.

5 Overview of WFD tool development in a Danish context

5.1 Modelling approaches

The starting point of the toolbox development was the recommendation given by the Eelgrass Working Group II (see section 2.2). The group explicitly expressed that the toolbox development for the Danish RBMP 2015-2021 should focus on mechanistic models covering the Danish coastal waters and selected estuaries, supplemented with more simple statistical tools for other smaller estuaries.

Taking both budgets and the time schedule into account, we adopted an approach involving development of four mechanistic biogeochemical models (covering 45 water bodies) and statistical models based on data from 29 monitoring stations (covering 22 water bodies). Of the 45 and 22 water bodies coved by models, 11 water bodies were covered by both mechanistic and statistical models and for these water bodies uncertainty estimates have been made.

Since it was not possible to include all the 119 Danish water bodies under the WFD in the models (some water bodies being too small, and only very limited data are available), a meta model approach was applied, including an additional 31 water bodies.

Overall, the models cover approximately 90% of the Danish water body area – equivalent to 70% of the entire Danish catchment area. Areas not covered by any type of model are primarily water bodies without observations usable for defining the present ecological status. Water bodies covered by models are shown in *Figure 5.1*, and the model developed includes:

- Mechanistic biogeochemical models: Together with the Danish EPA, four models were prioritised covering the Inner Danish Waters (IDW model) and the three estuaries: Roskilde Fjord, Odense Fjord and the Limfjorden. An attempt was made to develop a biogeochemical model covering the North Sea, but the time schedule did not allow its finalisation. A hydrodynamic North Sea model was completed, and this model was partly applied in the meta model approach. See chapter 7 for details on model development and evaluation.

- Statistical models: The development of statistical models was governed by the availability of observations within the specific water body. To be able to develop statistical models, time series of chlorophyll-*a*, TN, TP and K_d for no less than 15 years are needed. See section 6 for details on model development and evaluation.

- Meta models: In this context, meta models cover models based on information extracted from the mechanistic and statistical models, which is used for assessment of water bodies not covered by any of the two above model approaches. Details are given in section 8.6.

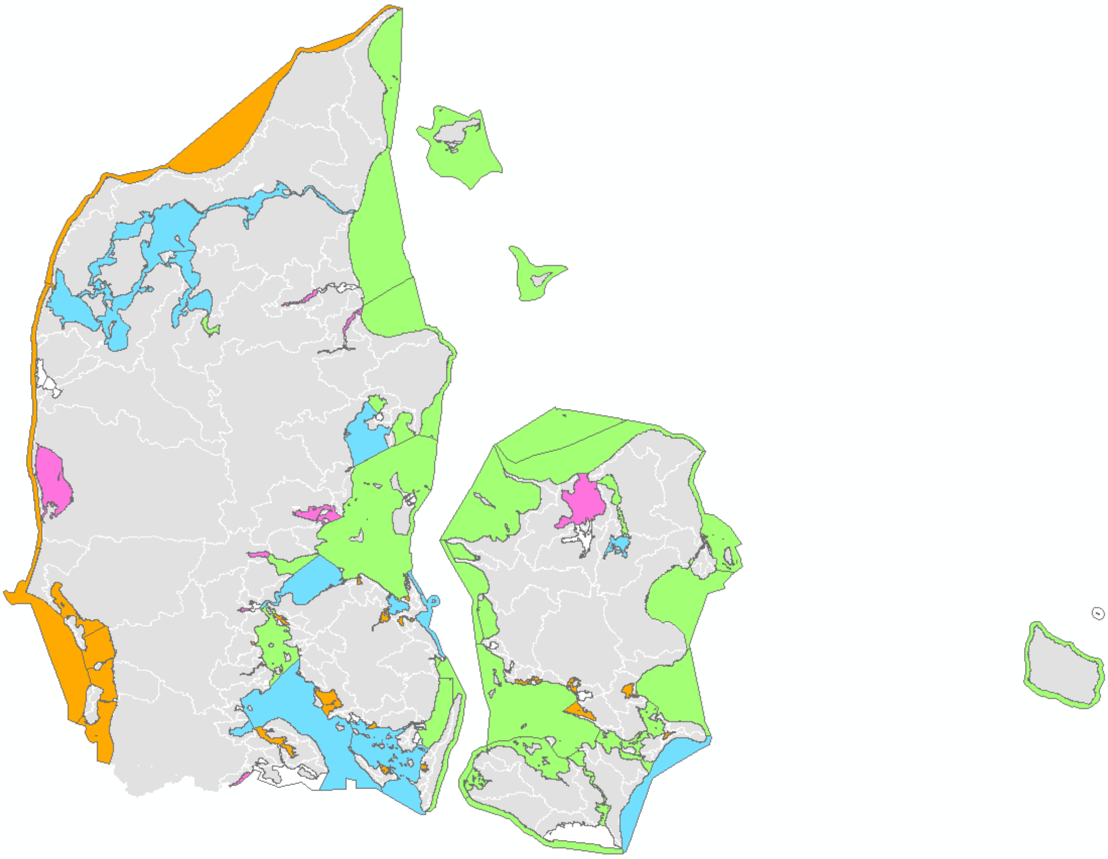


Figure 5.1. Water bodies covered by the different models developed for calculating nutrient reduction requirement and corresponding MAI to obtain GES. Pink colour indicates water bodies for which only statistical models have been developed, green colour indicates water bodies for which only mechanistic models have been developed, blue colour indicates water bodies with model overlap and orange colour indicates water bodies covered by meta models. Blank areas show water bodies not covered by any type of model.

5.2 Model indicators

As mentioned in section 2.2, Denmark operates with three indicators: chlorophyll-*a*, eelgrass depth limit and a fauna index (DKI). However, not all of these indicators can be linked to the model toolbox and some adjustments were therefore made:

Chlorophyll-a: The model development specifically includes chlorophyll-*a* concentrations, this indictor is therefore directly derived from the models (more details in chapter 6 and 7)

Eelgrass depth limit: This indicator cannot be derived directly from the models developed. To account for this, light attenuation, K_d, was adopted as a proxy-indicator[7] for eelgrass depth limit. K_d is directly derived from the models developed (see chapter 6 and 7), and K_d is here assumed to represent the potential depth limit. As described in, for instance, Flint et al. (2016) and Canal-Vergés et al. (2016), other factors than merely light availability determine eelgrass distribution, but light is an indispensable prerequisite for the capability of eelgrass to grow at the depth limits defined as WFD targets.

Hence, based on Duarte (1991) and Sand-Jensen et al. (1994) we assume that an average of 14% surface light should be available during the growth sea-

[7] In the present report we refer to K_d as an indicator although it's a proxy for eelgrass depth limit.

son (March to September), and this average is then used for transferring target values for eelgrass depth limits to corresponding K_d targets.

Danish Quality Index (DKI): DKI is a measure of benthic biodiversity. None of the models developed include biodiversity, and this indicator is thus not assessed further in this report.

6 Statistical model development

6.1 Application of statistical models in environmental studies

Statistical linear models with multiple predictors (MLR, mixed models, PLS etc.) have previously been applied in several studies of marine eutrophication published in international peer-reviewed journals (Conley et al. 2007; Forrest et al. 2012; Hinsby et al. 2012; Timmermann et al. 2013; Carstensen et al. 2014; Lyngsgaard et al. 2014a), and the methods have also been used in several reports dating back to 1999 (Markager & Storm 2003a, 2003b; Markager et al. 2006, 2008, 2010). As for all statistical modelling tools, the data requirements are rather comprehensive and even though the Danish monitoring program is one of the most extensive and dates back to 1989, it is only within recent years that the length of the time series has become sufficient to develop robust statistical models. In the present work and mainly due to the nature of our predictors that are high in number, relatively short data series (from a statistical point of view) and often intercorrelated, we have assessed that the PLS regression models are an appropriate tool.

PLS models have been widely used in chemometrics (Wold et al. 2001) where data are often characterised by large amounts of highly correlated predicting variables. In recent years, PLS regression has been applied in all kinds of disciplines and has been suggested as a convenient tool as an alternative to MLR in certain ecological studies (Carrascal et al. 2009). Recently, PLS regression has also been applied in marine ecology (Karle et al. 2007; Møhlenberg et al. 2007; Rasheed & Unsworth 2011; Lyngsgaard et al. 2014b). It is therefore a natural step to extend earlier work based on MLR (Markager et al. 2006, 2008) with PLS regression models. In this analysis, we couple nutrient loadings from land and a selection of physical and climatic variables with variables representing the environmental status of marine systems.

6.2 Development of statistical models

PLS models were developed with the main purpose of quantifying the relationship between nutrient loadings and the selected response variables chlorophyll-*a*, light attenuation, total nitrogen concentration (TN) and total phosphorus concentration (TP), used as indictors of water quality in the Danish coastal areas covered by the Water Framework Directive (WFD).

The data used for model development consist: i) data describing the environmental status (response variable), i.e. chlorophyll-*a* concentration, light attenuation, TP and TN concentrations; ii) data describing the conditions assumingly affecting the coastal environmental status, i.e. nutrient loadings, climate and water exchange (predictor variables). Following the data compilation, the modelling approach is described.

6.3 Response variables

To assess the water quality in Danish coastal waters, four responding variables (chlorophyll-*a*, light attenuation, TN and TP) were chosen as environmental indicators due to their well-documented response to nutrient enrichment (*Table 6.1*). Moreover, these indicators have been measured frequently in Danish waters as part of the Danish marine monitoring programme (DNAMAP, Conley et al. 2002).

Chlorophyll-*a* is an indicator of phytoplankton biomass and is one of the biological quality elements requiring assessment according to the WFD. The chlorophyll-*a* indicator (average chlorophyll-*a* concentration from surface to 10 m during the months May to September) has been intercalibrated and is widely used especially in the Baltic Sea region. Data used to estimate the chlorophyll-*a* indicator were obtained from the Danish national environmental monitoring database called NOVANA or DNAMAP and pre-processed as described below.

Light attenuation is an important indicator of ecosystem functioning and serves as a proxy for potential eelgrass depth distribution, which is an intercalibrated indicator for the biological quality elements “angiosperms”. Several factors affect the eelgrass distribution, but light availability is usually the principal factor limiting the depth distribution of seagrass (Duarte 1991; Olesen & Sand-Jensen 1993; Gallegos & Kenworthy 1996; Olesen 1996; Lee et al. 2007; Krause-Jensen et al. 2011). Light attenuation is measured either indirectly as Secchi depth or, after 1998, directly as light attenuation with a 4πK PAR sensor mounted on a CTD-sonde.

The concentrations of TN and TP are regarded as supporting indictors in the WFD and are known to respond rapidly to changes in riverine inputs (Riemann et al. 2016).

All data on environmental status come from the Danish National Aquatic Monitoring and Assessment Programme database (DNAMAP). The data on nutrient concentrations are expressed as average value of the discrete samples between 0 and 10 m. Where depth-integrated samples are available, the values of 0-10 m samples are used. Data are publically available (https://oda.dk/main.aspx).

Table 6.1. Overview of response variables (indicators) used in the modelling process

Response variable	Period		Samples
Total nitrogen (TN indicator)	Month 1-12	TN (PON+ DIN + DON) [µM]	Discrete or composite water samples 0-10 m
Total phosphorus (TP indicator)	Month 1-12	TP (POP + DIP + DOP) [µM]	Discrete or composite water samples 0-10 m
Chlorophyll-*a* (Chl. indicator)	Month 5-9	Mg m^{-3}	Discrete or composite water samples 0-10 m
Light attenuation kd1 indicator kd2 indicator	 Month 3-6 Month 7-10	Diffuse light attenuation coefficient for PAR, K_d (metres^{-1})	PAR measurements every 0.2 m or measurements of Secchi depth (site- and month-specific conversion factor = K_d*Zs)

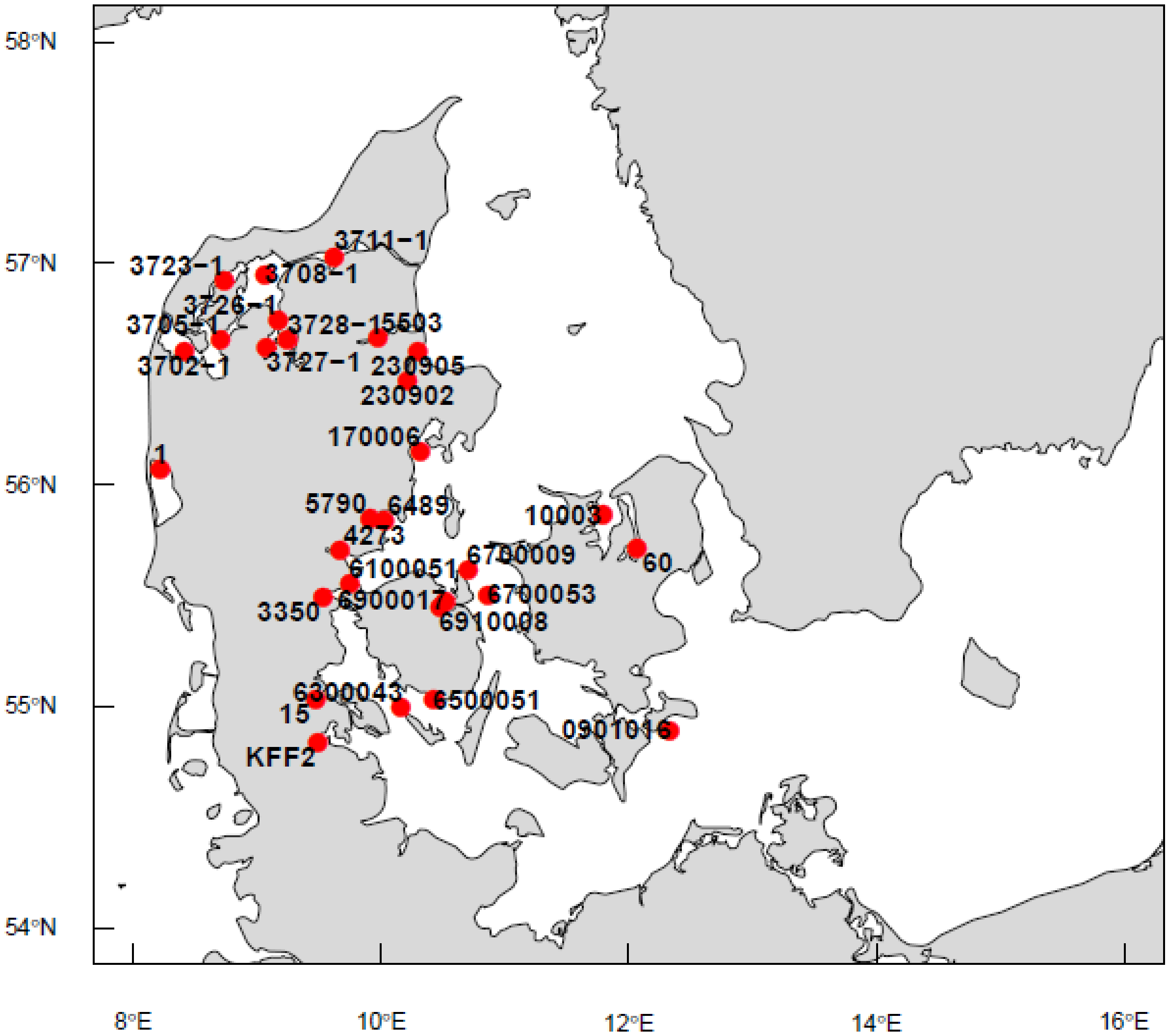


Figure 6.1. Marine monitoring stations used in the development of statistical models. The numbers refer to official station identification numbers.

6.3.1 Predictor variables

Data used to establish time series of the predictor variables: salinity, sea surface temperature and buoyancy frequency were also obtained from DNAMAP. We only used monitoring stations within the zone of the WFD and data series with at least 15 years of data during the period 1990 to 2012 with a minimum of one bimonthly observation (*Figure 6.1*).

The main purpose of the regression models is not to test the hypothesis that for instance chlorophyll-*a* concentration is dependent on the nutrient loadings but to quantify the relationship between the responding variable and the predictor variables especially the nutrient loading which can be managed. Therefore the predictor variables consisted of a suite of physical and chemical factors selected due to their known ability to act as forcing factors on the indicators (responding variables) (*Table 6.2*).

Table 6.2. Overview of selected predictor variables and their potential effects

Predictor variable	Responding variables	Effect	Mechanistic effect*	Selected references
Nitrogen loading (tonne y^{-1})	Chlorophyll-*a* K_d (eelgrass) TN conc.	+ + +	Increased phytoplankton growth/biomass and enhanced production of DOM, reducing water clarity.	(Nixon 1995; Pinckney et al. 2001; Greening & Janicki 2006; Elser et al. 2007; Latimer & Rego 2010)
Phosphorus loading (tonne y^{-1})	Chlorophyll-*a* K_d (eelgrass) TP	+ + +	Increased phytoplankton growth/biomass and enhanced production of DOM, reducing water clarity.	(Dennison et al. 1993; Elser et al. 2007)
Freshwater discharge (m^3 y^{-1})	Chlorophyll-*a* K_d (eelgrass) TN conc. TP conc.	+/- +/- +/- +/-	Nutrient and DOM transport from the catchment, affecting nutrient concentrations, light and chl. *a* conc. May also affect water column stability and hydraulic residence time with indirect effects on responding variables.	(Moran et al.; 1991; Brand 2001)
Wind energy (cubed wind speed)	Chlorophyll-*a* K_d TN conc. TP conc.	+/- +/- +/- +/-	May induce nutrient transport from deeper layers to the photic zone. May increase resuspension of particles incl. benthic microalgae, increasing light attenuation.	(Olesen 1996; Lawson et al. 2007)
Irradiance	Chlorophyll-*a* K_d	+/- +/-	Supports phytoplankton growth and increases decay of coloured DOM (CDOM).	(Hernes, 2003; Vodack et al. 1997)
Sea surface temperature	TN TP	+/- +/-	Increased remineralisation and release from sediments but also increased uptake. A stratified water column, resulting in decreased nutrient concentrations in surface water. A higher temperature may increase denitrification and thereby lower nitrogen concentrations.	(Behrenfeld et al. 2006; Bopp et al. 2001; Boyd & Doney 2002; Epply 1972; Nowicki 1994; Nowicki et al. 1997; Plattner et al. 2001; Taucher & Oschlies 2011; Taucher et al. 2012)
Salinity	Chlorophyll-*a* K_d (eelgrass) TN conc TP conc	+/- +/- +/- +/-	Salinity can be a proxy of exchange with oceanic water, which is often more nutrient poor, contains less CDOM and suspended solids, potentially influencing all response variables.	Siegel & Michaels 1996; Ferrari & Dowell 1998
Buoyancy frequency	Chlorophyll-*a* K_d (eelgrass) TN conc TP conc	+/- +/- +/- +/-	Water column stability affects the vertical transport of particles, nutrients and oxygen, potentially affecting all response variables.	(Cloern, 1984; Pingree et al. 1976, 1978)

* Short summary of mechanistic effects that have earlier been reported in technical reports or peer reviewed papers. This is not considered as the final answers, but we find any relationship that may come up in the regression models more likely if a mechanistic effect has already been described elsewhere.

Weather stations (irradiance and wind)

Data on solar radiation are a combination of data from two different sites in Copenhagen and a station on Sprogø (central part of the Great Belt, see *Figure 6.2*).

Wind speed is daily measured values collected by the Danish Meteorological Institute from land-based meteorological stations (*Figure 6.2*). Data from the meteorological station closest to the marine/estuarine monitoring station were used.

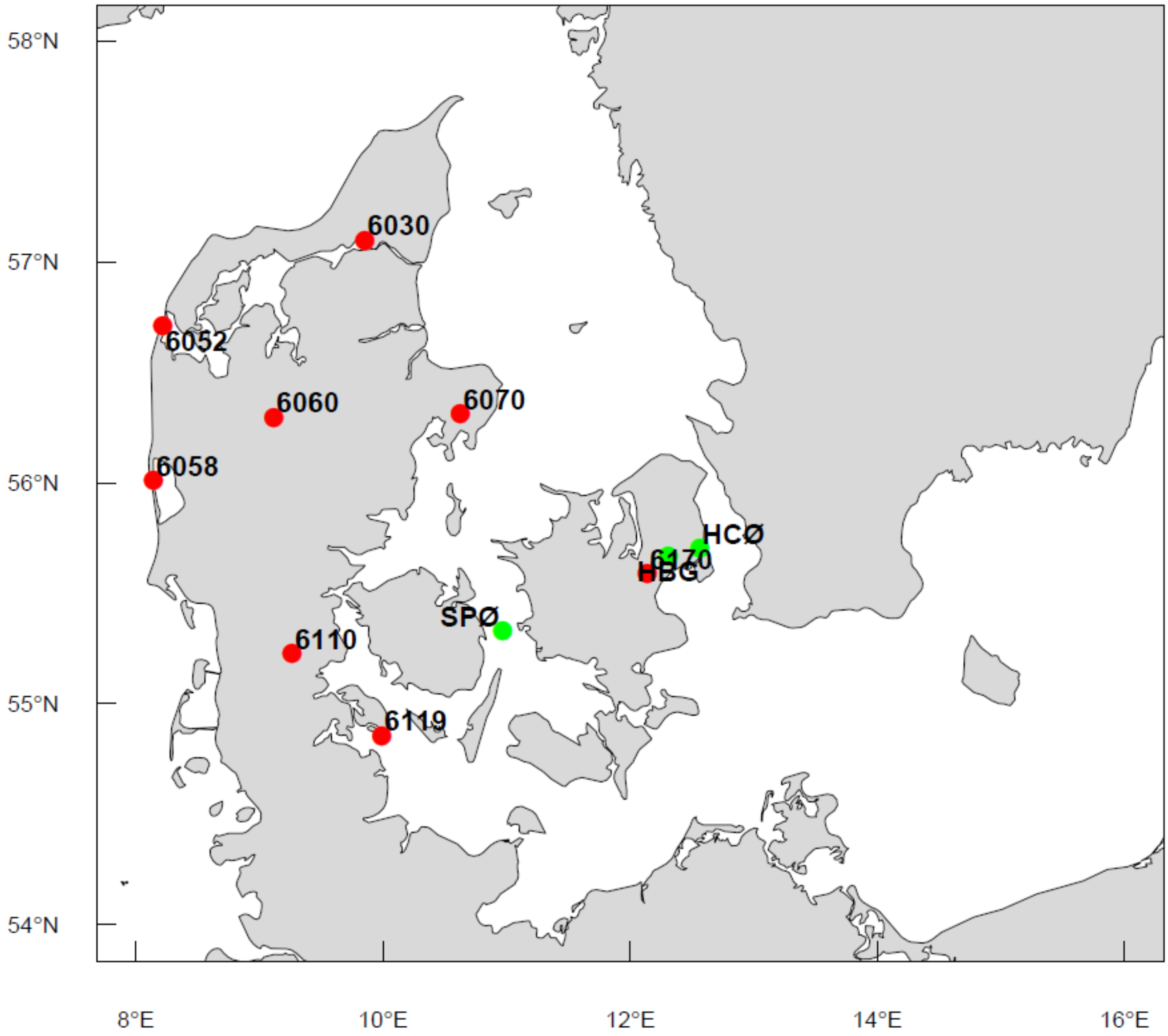


Figure 6.2. Location of meteorological stations providing wind data (red dots) or irradiance data (green dots).

6.3.2 Data processing

Sampling frequency

In general, the marine monitoring data were sampled at weekly to bi-weekly intervals (24-35 samples year^{-1}). Minimum bimonthly sampling frequency was required before applying a one-year data series, and only time series with a minimum of 15 years were used.

Filtering and visual inspection

The data in the database has been quality assessed and are as such ready to use.

Before the data were used in the model development, they were visually inspected for extreme values, and all the values above the annual 98 percentile were removed from each year's data set. This procedure was chosen to avoid extremely large values that would not necessarily represent average/normal conditions. As an example, very high nutrient concentrations are observed in a freshwater layer underneath the ice in cold winters. Such values can be 10-100 times above the normal values and may therefore affect the annual value, in this case due to physical conditions rather than factors related to environmental status. The method is relatively crude and ideally all data series should have been analysed for outliers individually and with more sophisticated tools; however, the timeframe and resources of our project did not allow for a complete data quality procedure. We considered whether to log transform some of the data (e.g. chlorophyll-*a*) but refrained from doing so to avoid excessive emphasis on low values. As to nutrient concentrations, log transformation would shift the focus towards summer where periods with low values occur, which may not reliably reflect overall nutrient levels.

Interpolation

To obtain monthly averages weighted relative to the frequency of observations, the data were filtered and interpolated linearly between the observations using the expand procedure in SAS®
(https://support.sas.com/documentation/onlinedoc/ets/132/expand.pdf).
The daily values gained from the interpolation were then used to construct monthly average values.

Predictor variables

Buoyancy frequency was calculated as the Brunt-Väisälä buoyancy frequency (N) based on the difference between surface (0-1 m) and bottom density (1 m above bottom):

$$N = \sqrt{-\frac{g}{p_0}\frac{dp}{dz}} \qquad \text{Eq. 6.1}$$

where g is the regional gravitational constant (9.82 ms^{-2}), p_0 is the potential density (surface density – kg m^{-3}), dp is the difference between bottom and surface density (kg m^{-3}), dz is the depth difference between bottom and surface (m) and N is the buoyancy frequency (s^{-1}) .

The mechanical force of wind on the water surface is proportional to the cubed wind speed (Alexander et al. 2000) and we therefore cubed the wind speed to obtain a relative measure on the wind energy delivered to the sea surface.

The irradiance data were obtained as half hourly values of global irradiance from 1990 to 2012 and these were then converted to PAR values based on an algorithm from the Danish Agricultural University. Data gaps are filled with data from Sprogø (SPØ) after adjustment of the level based on maximum level of irradiance (0.96 of the level measured in Copenhagen). Data from the two sites in Copenhagen (HCØ and HBG) have the same level and slope, and the final unit is µmol photons m^{-2} s^{-1} calculated from global irradiance (W m^{-2}) (*Figure 6.2*).

Data from 1990 to 1993 were hourly data, which were interpolated linearly to obtain 30 min intervals using the "Proc Expand" procedure in SAS. For some years, values were adjusted (dark values subtracted) due to a significant sensor offset. In addition, all values below 2 µmol photons m^{-2} s^{-1} were set to zero due to low sensitivity within that range and problems in some years with a dark offset. This is significant in some winter months where a dark offset may constitute a significant part of the daily sum. Finally, the data were translated to monthly mean values. The remaining gaps were filled with average values for the same day and time from other years.

Salinity and water temperature were calculated as monthly means of the mean salinity or temperature of the layer of the water column above 10 m depth.

For all the semi-enclosed waterbodies, all nutrient loadings were calculated as monthly sums of nutrients entering the water body directly from the local catchments. In the relatively open coastal areas, a larger but still local catchment was used.

Climate variables, salinity, sea surface temperature and water column stability were all detrended over time by fitting a linear regression model to the variable and then calculating the detrended values as:

$$Xvar\_detr_i = Xvar_i - (\alpha + \beta \cdot year_i) \qquad \text{Eq. 6.2}$$

where *Xvar_detr* is detrended predictor variable, *Xvar* is predictor variable, α is the intercept of the *Xvar* vs. *year* regression line and β is the slope. i is the i'th year.

Conversion from Secchi depth to light attenuation

In periods where only Secchi depths were measured, K_d values were estimated using site- and month-specific values for the factor K_d*Zs (Murray 2015). This was done using periods where both Secchi depth and light attenuation were measured. For each station in each month, the station- and month-specific conversion factors between Secchi depth and light attenuation were found using this relationship:

$$-\ln\left(\frac{L_z}{L_0}\right) = Kd \cdot Zs \qquad \text{Eq. 6.3}$$

where K_d is light attenuation coefficient (m^{-1}) and Zs is Secchi depth (m). L_z is light at depth z and L_0 is surface irradiance. $-\ln(L_z/L_0)$ is the site- and month-specific conversion factor.

Time periods of the predictor variables

Time lags between causes and effects are well known in ecological systems; for instance, nutrients exported from land will have an effect in the ecosystem for a period of time after the N or P leaves the river mouth and some macrophytes may use several years or decades to re-colonise areas after removal (Duarte 1995). The existence of time lags in ecological systems poses a challenge in statistical modelling, further augmented by the difference in time lag duration among systems depending on retention time, depth, temperature and other parameters. In order to handle potential time lags in the statistical models, we tested different time lags for each predictor variable. However, use of multiple predictor variables increases the risk of overfitting the model, resulting in spurious correlations. To balance these two aspects, we specified that the predictor variables should not start earlier than the year before the responding variable, and we defined the following rules for predictor variables:

1. Each of the eight predictors can only occur once.
2. Time periods are 4, 8 and 12 months.
3. Final month of the periods should be earlier than or correspond to the final month of the response variable.

4. Starting month of the 4-month period should be no later than September the previous year and is moved forward at one-month intervals. The latest possible start is defined by rule #3.
5. The starting months of the 8- and 12-month periods range from January the previous year to the latest possible start conforming to rule #3 and are moved forward at one-month intervals.

(See example in *Figure 6.3*)

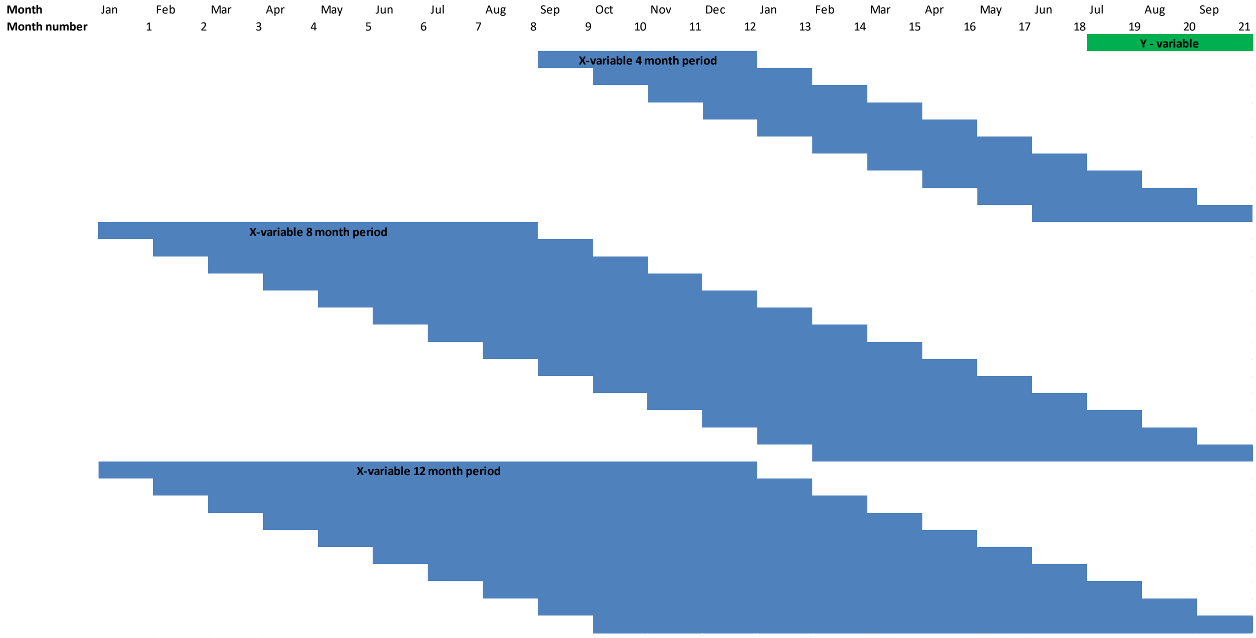


Figure 6.3. Example of predictor variable time periods based on a 3-month response variable period.

Normalisation

Both predictor and response variables were normalised to a mean of zero and a standard deviation of one in accordance with the processing requirements prior to any MLR and PLS regression. This was done by subtracting the mean from each observation and dividing with the standard deviation.

$$\acute{x} = \frac{x - \bar{x}}{Stdev_x} \; and \; \acute{y} = \frac{y - \bar{y}}{Stdev_y} \qquad \text{Eq. 6.4}$$

where $\acute{x}$ is the normalised x value, $\acute{y}$ is the normalized y value, x and y are the observed values and $\bar{x}$ and $\bar{y}$ the mean of all the observed values. $Stdev_x$ or $Stdev_y$ is the standard deviation of all the observed x and y values respectively.

To enhance applicability, the normalised values were rescaled after the modelling, for instance the modelling results, by multiplying with the standard deviation and adding the mean. The regression coefficients were rescaled by dividing the standard deviation of y with the standard deviation of x and multiplying with the coefficient. This normalisation ensured that all predictor variables had the same weight in the regression. An additional advantage is that the relative importance of a predictor variable can be interpreted directly from the normalised coefficients.

6.3.3 Modelling

Cross-validation and variable selection

Objective, iterative and automated variable selection was done by a cross-validated multiple linear regression (MLR) technique in combination with stepwise exclusion of the predictor variable that gave the lowest model error in the form of "Root Mean Square Error of Cross Validation" (RMSECV). The cross-validation was used to predict each value in the data set from a regression between response and predictor variables in the remaining data set (leave-one-out cross-validation), this was one of the steps undertaken to reduce the risk of overfitting the model. This method is a variation of a method described as "forward selection" by Broadhurst et al. (1997). (Martens & Dardenne 1998) recommend cross-validation as an appropriate method, assuming that the data sets are representative of the X-Y relationship in the population.

The normalised MLR model follows the formula:

$$\acute{y} = \acute{a} + \acute{b}_1\acute{x}_1 + b_2\acute{x}_2 + b_3\acute{x}_3 \ldots \qquad \text{Eq. 6.5}$$

where $\acute{y}$ is the normalised value of the dependent variable y, a is the intercept and b are coefficients, and $\acute{x}_j$ are normalised values for the predictor variable, included in the model.

In the first iteration, the predictor variable with the lowest RMSECV, i.e. the predictor variable explaining the largest proportion of the variation in the y variable, was chosen. This variable was then excluded from the data set, and among the remaining predictor variables, the next variable with the lowest RMSECV was selected. A condition for the selection of the second and onward predictor variable was that only one variable from each class of the predictor variables (e.g. N input, wind speed and salinity) may be selected. The entire set of predictor variables was tested in this iterative manner until eight predictor variables were selected. This was done for each combination of years (leave-one-out cross-validation) and the eight variables that gave the best solution with respect to RMSECV were selected. In this way, a number of model solutions corresponding to the number of years available, multiplied by the potential number of selected variables (i.e. 8), were produced.

The various model solutions provided the opportunity to assess the explanatory power of the individual x variables from their selection frequency and developments in RMSECV as a function of the number of selected variables.

The entire procedure was conducted using a "free" unconstrained solution, i.e. the procedure could choose between all types of predictor variables. Subsequently, the unconstrained model was compared with a constrained solution where either N or P was selected as first variable. This gave us the opportunity to compare "nutrient-driven" models with the unconstrained models when these were not identical.

Additionally, four evenly distributed years were excluded from the selection process, providing an independent basis for evaluating the predictive power of the model after the variable selection and calibration. Due to the relatively short time series, the validation years were included in the final parameterisation of the model (*Figure 6.4*).

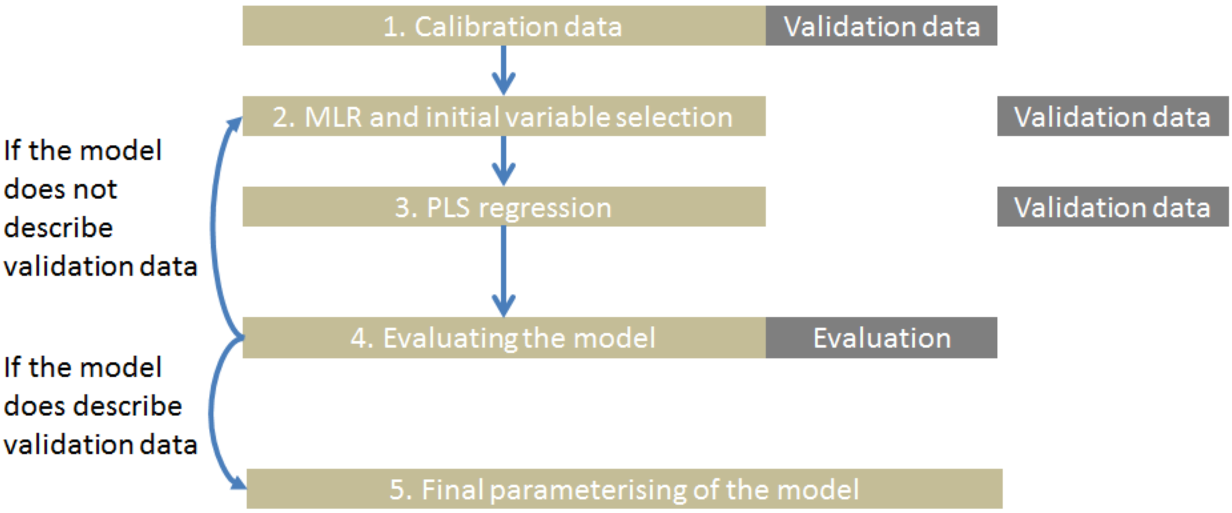


Figure 6.4. Schematic diagram of model development used in this work. For the development of each model we went through a calibration and a validation stage to do so one must: 1) divide the data set in a calibration set and a validation set (25% of the data set); 2) select a suite of possible explanatory variables through MLR and cross-validation; 3) make a PLS regression using selected variables and choose the number of predictor variables based on the reduction of the RMSE that each "new" variable produces; 4) evaluate the predictive power of the model on the validation set. If the model is able to describe the variation in the validation set (based on RMSEP), continue to step 5; otherwise return to step 2. 5. Use the whole data set to make the final parameterisation of the model.

All the predictor variables were tested for intercorrelation. According to Tabachnick et al. (2001), independent variables with a bivariate correlation of more than 0.7 should be omitted in multiple regression analysis since they cannot be considered truly independent. Thus, it is desirable to avoid variables in the model that are too intercorrelated. In principle, the PLS regression technique should be able to deal with intercorrelated predictor variables given its latent variable structure. However, we experienced that the parameters (PLS coefficients) were still sensitive to small variations in the data set when highly intercorrelated predictors ($r > 0.9$) were used, making use of highly correlated data sets problematic even in PLS regressions. Often N and P loadings were correlated and, therefore, we generally avoided including both variables in the same model even though it could be relevant from an ecological perspective for some recipients (see below for the technique used to quantify limitation). The selection process for chlorophyll-*a* and light attenuation models included an assessment of which nutrient had the greatest potential to govern the modelled response variable based on the limiting nutrient in the period covered by the response variable (see *Table 6.1*). Nutrient limitation plots (examples shown in *Figures 6.5* and *6.6*) were used to select the most limiting nutrient in situations with high intercorrelation. Estimation of potential nutrient limitation was based on the concentration of either DIN or DIP and the half saturation constant (Ks) from the literature. The potential limitation ranged from 0 to 1 where 0 is no nutrient limitation (infinite nutrient resources) and 1 is full nutrient limitation (DIN or DIP concentration = 0). The potential limitation (Pot_{lim}) was calculated as 1- the Monod growth constant (Monod 1949):

$$Pot_{lim} = 1 - \frac{[S]}{Ks + [S]} \qquad \text{Eq. 6.6}$$

where [S] is the concentration of the substrate and Ks is the half saturation constant.

The potential nutrient limitation was then estimated for each month during the year based on all years in the time series for each monitoring station and plotted over time. Two examples are given in *Figures 6.5* and *6.6*. We then evaluated in which months the nutrient limitation peaked and how that fitted with the period of the responding variable, for example chlorophyll-*a*. If there was a clear discrepancy between the periods (*Figure 6.6*), we chose the nutrient with the most pronounced overlap with the period of the responding variable. In cases where potential nutrient limitation from both N and P both overlapped with the period of the responding variable, the model with the best fit was chosen.

The half saturation coefficients (Ks) for phosphorus limitation and nitrogen limitation were chosen to be 0.2 µM and 2 µM, respectively, based on values from other studies (Eppley et al. 1969; MacIsaac & Dugdale 1969; Klausmeier et al. 2004). The exact *Ks* for N and P is not possible to define and may differ between sites and over time, which means that the exact level of limitation involves a certain level of uncertainty; thus, we mainly evaluate the effect of nutrient limitation on the seasonal pattern and to a minor degree the level.

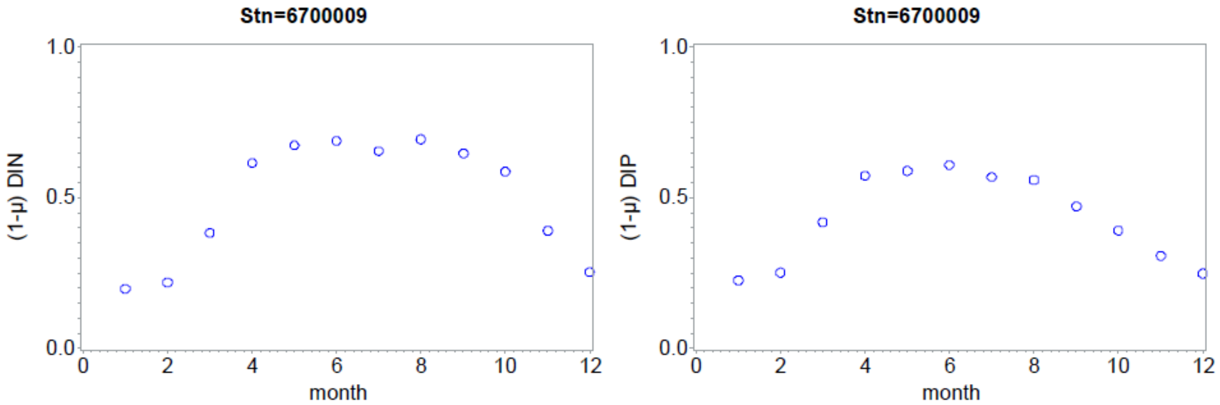


Figure 6.5. Potential nutrient limitation for each month at station 6700009 in the northern part of the Great Belt (Fynshoved). Left = potential limitation of nitrogen, right = potential limitation of phosphorus. 0 is no nutrient limitation and 1 is maximum nutrient limitation (no dissolved inorganic nutrients). This station is an example of an almost identical seasonal pattern where the only difference is a slightly earlier decline in P than N limitation in autumn.

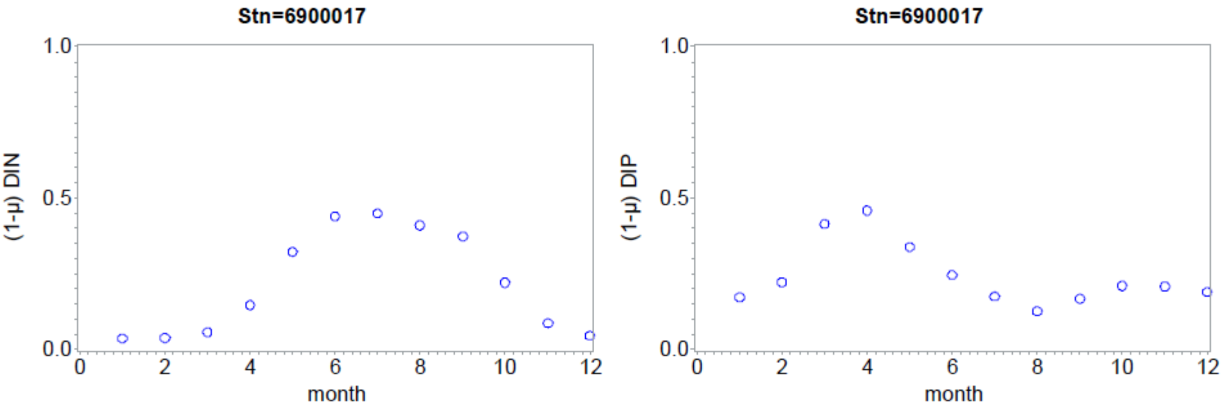


Figure 6.6. Potential nutrient limitation for each month at station 6900017 in Odense Fjord. Left = potential limitation of nitrogen, right = potential limitation of phosphorus. 0 is no nutrient limitation and 1 is maximum nutrient limitation (no dissolved inorganic nutrient) This station is an example of a clear shift in the limiting nutrient where P is limiting in spring (March to June) and N is limiting from May to September.

In cases where different sets of predictor variables described the selected responding variable almost equally well, additional analyses were used to identify the most likely variable. One method was, as mentioned above, to quantify the potential most limiting nutrient. Another method was to plot R^2 values between a predictor variable and the responding variable over time and for different periods (4, 8 or 12 months, see *Figure 6.3*). This technique can elucidate if a high correlation between predictor and response variables was an isolated "event" or part of a more consistent pattern. A consistent pattern with a wide peak when plotted over time was considered more likely to represent causality. *Figure 6.7* shows two examples of this, one where a four-month average of cubed wind speed from September to December the year before is highly correlated with the average chlorophyll-*a* concentration from May to September the year before. This is in contrast to the nitrogen load where a four-month period of nitrogen load increases almost monotonically in correlation with the concentration of chlorophyll-*a* when moving closer to the period in which the chlorophyll-*a* concentration was calculated. Another example of the effect of the selected period for a predictor variable (N loading) can be found in (Lyngsgaard et al. 2014).

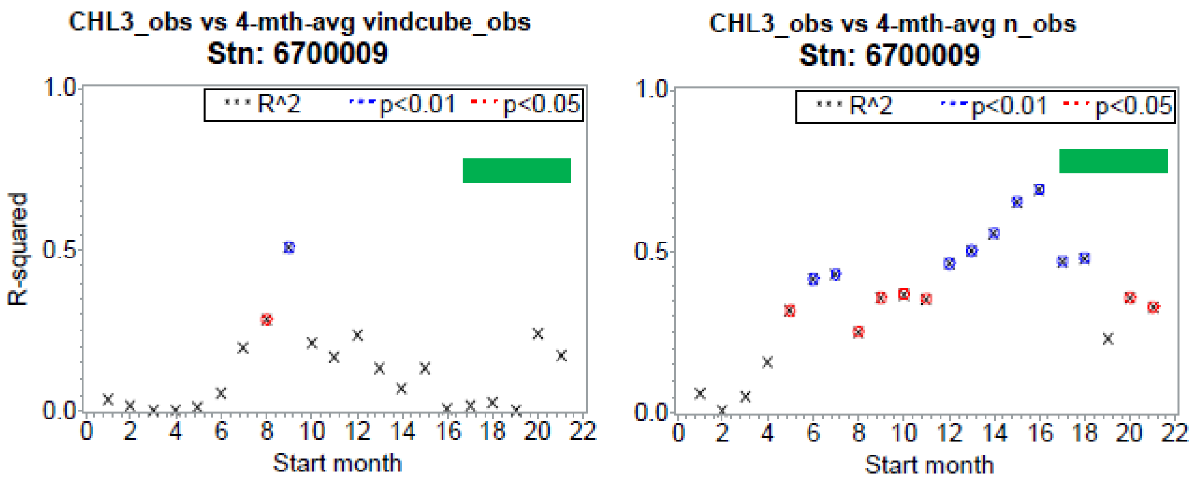


Figure 6.7. R-squared plotted over the starting month of the predictor variables for the responding variable Chl3 (average chl. a month 5-9). The predictor variables are four-month average of vindcube (cubed wind speed) to the left and n-obs (N loading) to the right. Month 1 is January a year before the year of the responding variable, i.e. Chl3, beginning in month 17 and ending in month 21, marked with a green bar. Therefore, periods of the predictor variable starting from month 16 and onwards will include months after the period of the responding variable. Station 6700009 is in the southern Great Belt.

Generally, several time periods of the predictor value were significantly related to the response variable, and the time period with the strongest correlation was chosen in order to get the best estimate of the correlation coefficient. However, the implication is that other time periods of, for example, nutrient loadings also affect the response variable (e.g. chlorophyll-*a* concentration).

PLS regression and validation

The variables selected through MLR and cross-validation were used to produce a number of PLS regression models, and based on their ability to predict the validation data a final model was chosen. In contrast to MLR, the PLS regression is based on regression between latent variables (Abdi 2010) and as such an indirect modelling approach. The latent variables consist of a factor matrix (*T*) for the predictor variables (X) and a factor matrix (*U*) for the response variable (Y) based on the scores that give the highest covariance between the two latent variables. The scores (in T and U) can be converted back to the original variables using a loading matrix (P and Q):

$$X = TP^T + E \qquad \text{Eq. 6.7}$$

and

$$Y = UQ^T + F \qquad \text{Eq. 6.8}$$

where E and F are the error terms and T and U are optimised to covariate the most.

PLS regression is usually preferred when the number of predictor variables is greater than the number of observations and/or when intercorrelation exists between the predictor variables as was often the case in the present analysis.

During the validation process, it was evaluated for every additional variable in the model how much the variable reduced the error, measured as root mean squared error, of both the calibration data and the validation data sets (*Figure 6.8*). Variables that increased the error in the prediction of the validation data were omitted. The model should also be able to describe the validation set unbiased and with approximately the same error (MSRE) as the calibration set.

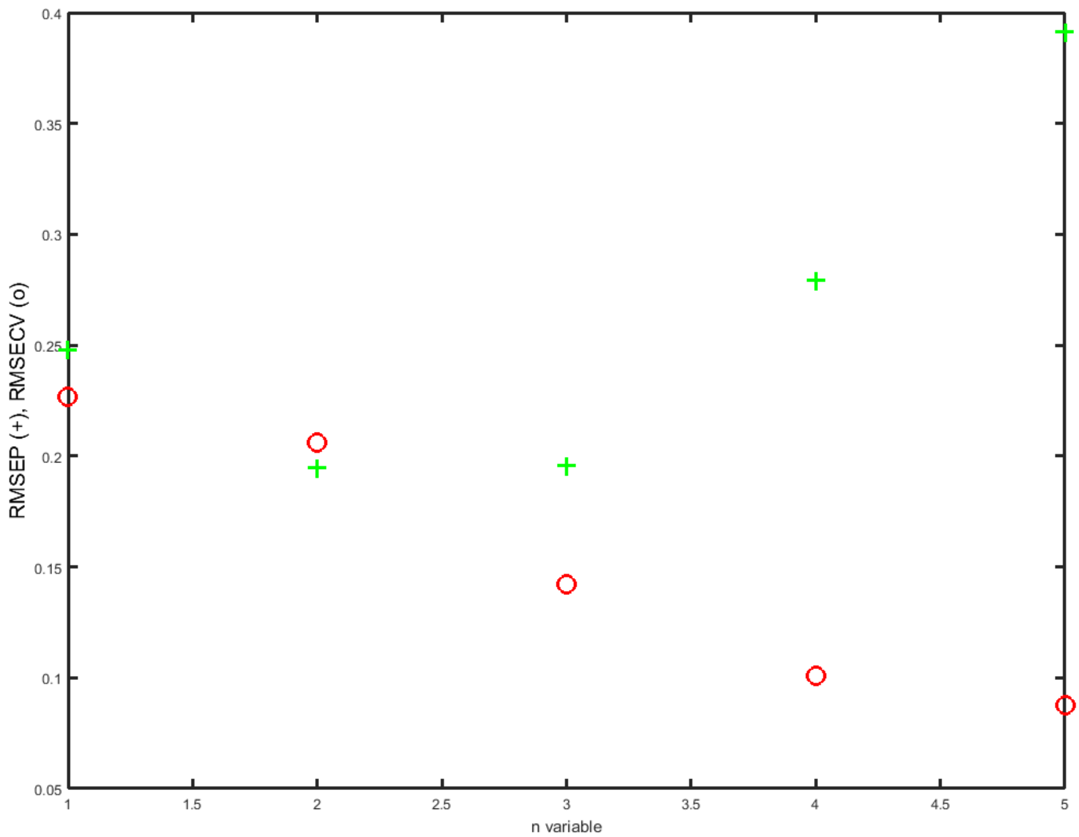


Figure 6.8. An example (from Isefjord) of RMSECV and RMSEP plotted against the number of variables in the PLS regression. In this case, the number of variables would be three since this is the point where the independent validation data start to increase in error RMSEP (+), and after three variables the obtained error reduction of the calibration decreases.

Summary of the full variable selection process

To summarise the overall model development process, the procedure can be considered a two-step process. The first part is an initial objective process undertaken to select the most likely variables predicting the development of the responding variable. From the initial suite of possible predictor variables, one or more PLS regressions were developed. Different combinations of the pre-selected variables were tested, and during this process we evaluated the combination and the number of variables based on RMSECV and RMSEP (*Figure 6.8*) (prediction of the validation data set) and simultaneously checked that the chosen variables and corresponding coefficients were in accordance with the known causality (*Table 6.2*) or otherwise very convincing and/or reoccurring in many systems and temporally persistent (see, for instance, *Figure 6.7*). Overall, the selection process was a combination of variable selection based on objective criteria and ecosystem understanding.

After the final selection process, the model was recalibrated as a PLS regression using the whole data set. In some cases, several models were produced and post-evaluated by assessment of the model's ability to describe peaks and/or average conditions based on plots and on residuals that were tested for autocorrelation and normality.

Outliers

Extreme values in one or more responding variables occurred in some of the time series (*Appendix B*). They were always included in the initial models, but in some cases a final regression model was mainly able to describe this extreme value and not the dynamics of the remaining time series. This was evaluated on the statistical leverage of each year calculated by the PLS package from eigenvector® and on visual inspection of the model. In cases where only extreme values were described, the model developing procedure was repeated without the potential outlier and the results were evaluated in accordance with the aforementioned process.

The PLS regression analysis was done in MATLAB® using a PLS program package from Eigenvector®, while the selection process was mainly undertaken by the authors.

6.3.4 Models and scenarios

Coefficients of de-normalisation

As earlier mentioned, all models were based on data that were normalised to a mean of zero and a standard deviation of one. For scenario purposes (e.g. nutrient reduction scenarios), the normalised coefficients were converted to coefficients with the original units to quantify the importance of a change in, for example, annual nitrogen loading.

All scenarios are designed for a situation of ”normal climate”, i.e. we assume that all climate variables in a scenario are similar to their average value in the model calibration period. In principle, it is possible to create scenarios that also take into account the variation in climate, but temperature is the only climate variable for which we have relatively sound long-term forecasts, and as seen in *Table 6.3* temperature is not consistently included in the models (16 out of 89 models).

As a result of the normalisation of both predictor and response variables, the intercepts in MLR and PLSR equal 0, and all stages of xb resulting from climate variables will also be equal to 0 when climate is assumed to be equal to its mean. Therefore, the effect of climate will be equal to 0 when it is assumed to be a "normal" climate or equal to its mean values in the modelled period. This applies to the other predictor variables as well, and in the scenarios we will assume that all other variables than nutrients are equal to their mean value of the calibration period.

Under this assumption, the regressions can be reduced to:

$$\acute{y} = \acute{b}_1 \acute{x}_1 \qquad \text{Eq. 6.9}$$

where b´₁ is the normalised coefficient for supply of nitrogen or phosphorus and x´₁ the normalised supply calculated as an average for the months, as seen in model. By inserting equation 6.4, the final equation is:

$$b = \acute{b}_1 \frac{Stdev_y}{Stdev_x} \; and \; a = \bar{y} - b\bar{x} \qquad \text{Eq. 6.10}$$

where b is the coefficient for the effect of a change in the flow in absolute units, and a is an intercept. The value of b in equation 6.10 will be calculated from the average of the monthly amount for the period of the model inputs. In the reduction scenarios, we want to use the annual supply, and b must be scaled to the annual transfers:

$$\hat{}b = \acute{b}_1 \frac{Stdev_y}{Stdev_x} \frac{x}{L} \qquad \text{Eq. 6.11}$$

L is the mean value of the annual inputs during the period. The coefficient ^b is thus the unit (unit for response variable/tons annual input) and can be directly used in the reduction scenarios.

6.4 Model evaluation

The statistical models were evaluated individually for each predicted indicator and water body by comparing the modelled response variables (i.e. yearly values for TN, TP, chlorophyll-*a* and K_d) with the corresponding observations. The overall performance of the models was assessed using a suite of statistical measures in addition to visual inspection. The following evaluation tools were used:

- RMSEP and RMSECV, which are used as a measure of model accuracy for predicting both the validation data (RMSEP) and the cross-validated data (RMSECV) – RMSE is an empirical estimate of the standard deviation of the model.

- R^2 describes the part of the variance in the data that is explained by the model and is used to evaluate if the model captures the year-to-year variation in the observed data.

- Normality test of residuals (Shapiro-Wilk) is used to test if the residuals between model and observations are normally distributed.

- Autocorrelation test of residuals (Box Pierce) is used as an indicator of autocorrelations in the dataset.

- Availability of data points (number of years). The amount of data points (in this case the length of the time series) is crucial for the development of statistical models. We have used a tentative minimum of 15 years as a guideline.

- Visual inspection of time series, residuals and scatter plots (*Appendix B*) with focus on identification of systematic deviations and the ability to capturing year-to-year variation.

As a result of the model evaluation process, several models were excluded from further analysis, mainly because they failed to meet the data requirement criteria. The models used to perform nutrient load reduction scenarios were considered reliable for this purpose if potential model weaknesses did not influence the capability of the models to capture overall system behavior. The reliability of the model predictions will, of course, decrease when moving away from situations covered by data used in the model calibration. Due to substantial variations in year-to-year loadings, ranging from 42.700 in 2003 (a dry year) to 127.000 in 1994 (a wet year) (total nitrogen loadings from Danish land), the statistical models were evaluated relative to a wide range of load conditions. From a statistical point of view, it is important to note that the lowest values almost encompass the estimated national MAI of 42 ktons nitrogen pr. year (see section 8.7.2). Thus, on the overall level the statistical models are not used far outside the data range.

For most of the models, there was an overall good agreement between the predicted and the observed parameters, indicating that the models were able to describe the system behavior. However, especially the TN and TP models showed a tendency to systematic deviations where 8 out of 23 and 3 out of 22 models, respectively, showed significant autocorrelation for the residuals. These resulted in an underestimation of high nutrient concentrations (in the 90s) and overestimation of lower nutrient concentrations (in present time). This unexplained systematic variance was attributed to pools of nutrients in the systems, particular in the sediments (Jørgensen et al. 2014), causing an autocorrelation where nutrient loadings (and nutrient concentrations) from previous years affect nutrient concentrations in several subsequent years. A closer autocorrelation analysis revealed that the historical signal for TN had a half-life between approximately 0.2 (Randers estuary) up to 8 years (western part of the Limfjorden), indicating that the timespan for some systems to reach a steady state between nutrient loadings and nutrient concentrations could be two to three decades. The quantification of the time lag was, however, very uncertain due to the relatively short time series, especially for estuaries with a long residence time. In consequence of the uncertain quantification of autocorrelation, this effect was not included in the models. When longer time series become available, inclusion of autocorrelation might improve the next generation of models.

The K_d models generally did not capture the year-to-year variation as well as the rest of the models as seen, for instance, by the generally lower R^2 values ($R^2 = 0.50 \pm 0.19$ mean ± SD). This is probably due to K_d being influenced by light absorption of DOM and detritus and scattering of light by particles, this probably being affected by nutrients on a longer time scale.

The predictor variables, which best explain the response variables, are listed in *Table 6.3*. Not surprisingly, nitrogen and phosphorus loadings control TN (100% of the models) and TP (91% of the models) concentrations, respectively, at most of the stations. One exception is Skive Fjord and Lovns Bredning where water temperature was selected instead of P loading and where internal loading dominates the system (see also *Figure 8.5*, section 8.3.5). We attribute this to the seasonal anoxia in these areas, inducing release of phosphorus from the sediments. This implies that the phosphorus concentration is less controlled by riverine loadings. Nitrogen loadings were most often selected as predictor variable for the chlorophyll-*a* indicator (~ 70% of the stations), likely because nitrogen often is limiting primary production during summer and autumn. In spring, however, phosphorus is often the limiting nutrient and analyses have revealed (results not shown) that phosphorus loading often controls chlorophyll-*a* concentrations during this period. For the K_d indicator, nitrogen loadings were selected as predictor variables at approximately 70% of the examined stations, and especially physical parameters (e.g. wind and salinity) appear to have more influence on the K_d indicator than the other response variables.

For the response variables where N loading is selected as predictor variable, a linear relation between N loadings and the response variable can be derived. The slope of this cause and effect relation is a measure of how sensitive the response variable is towards changes in N loading.

From a management perspective, a challenge arise when nitrogen loadings is not selected as predictor variable. In this case, two conclusions are offered: first, that nitrogen loadings are insignificant for the status of the ecosystem when considering the specific combination of area and response variable or, secondly, that the combination of rather few data points (short time series), uncertainties in data and systematic effects from other predictor variables masks the effect of nitrogen loadings. In case of the latter, it is still possible that nitrogen affects the status of the ecosystem, but the effect cannot be directly quantified for the specific combination of water body and response variable.

The first conclusion would lead to omission of nitrogen loadings as a management tool for that specific area. The other implies that nitrogen loadings might affect the status of the ecosystem. We assumed that the latter is the case and therefore used an average response for nitrogen loadings versus the response variable obtained from similar areas (the so-called meta model approach, see section 8.6). This approach should be used carefully and is only valid when nitrogen loadings are found to be significant in the majority of the cases. The other choice would imply that nitrogen loadings do not significantly affect marine ecosystems, which may be the situation in, for instance, tidal areas where turbidity leads to severe light limitation, where phosphorous is the main limiting nutrient (as in spring in several Danish estuaries) or where other pressures are dominant.

Table 6.3. Summarised model performance. Station name is the name of the monitoring station from which data were used to develop the models. The response variable is the variable that the model is developed to predict (chl3 is mean chlorophyll-*a* concentration from May to September; kd2 is mean light attenuation coefficient from July to September; TN is annual mean concentration of total nitrogen and TP is the annual mean concentration of total phosphorus). Variable number is the number of the predictor variable, and for each variable number column the predictors in the PLS regression model are given(N load and P load are total nitrogen and total phosphorus loadings from land, wind^3 is cubed wind speed, SST is sea surface temperature, BV-BF is Brunt-Väisälä buoyancy frequency for the whole water column, irradiance is incoming PAR radiation, salinity is salinity in the water surface (upper 10 m) and discharge is the freshwater discharge from the local catchments. RMSECV is the root mean squared error of the cross-validated model, and RMSEp is the root mean squared error of the predicted values. R^2 is the coefficient of determination for the relationship. Shapiro-normality is the test result for the residuals. NS means that the residuals are not significantly different from normal distributed residuals, 'yes' means that the residuals are significantly different from normal distributed residuals ($p < 0.05$). AC Box-Pierce is a test of autocorrelation in the residuals, NS implies not significantly auto correlated, while 'yes' means significantly auto correlated ($p < 0.05$).

Station	Response variable	1. Predictor variable	2. Predictor variable	3. Predictor variable	4. Predictor variable	RMSECV	RMSEp	R^2	Shapiro normality	AC Box-Pierce
Flensborg	chl3	N load	wind^3			0.34	0.49	0.54	NS	NS
Flensborg	kd2	wind^3	SST			0.08	0.35	0.65	NS	NS
Flensborg	TN	N load	BV-BF			0.17	0.55	0.45	NS	NS
Flensborg	TP	P load				0.11	0.19	0.81	NS	NS
Horsens inner	chl3	N load	irradiance			0.16	0.35	0.66	NS	NS
Horsens inner	kd2	N load	salinity	BV-BF	wind^3	0.09	0.35	0.65	NS	NS
Horsens inner	TN	N load	SST			0.11	0.28	0.72	NS	NS
Horsens inner	TP	P load	BV-BF			0.12	0.42	0.58	NS	NS
Horsens outer	chl3	N load	P load	salinity		0.20	0.57	0.43	NS	NS
Horsens outer	kd2	N load				0.15	0.86	0.14	NS	NS
Horsens outer	TN	N load				0.07	0.25	0.75	NS	Yes
Horsens outer	TP	P load	SST			0.08	0.38	0.62	NS	Yes
Isefjord	chl3	N load	wind^3			0.37	0.62	0.41	NS	NS
Isefjord	kd2	N load	wind^3	irradiance		0.12	0.57	0.43	NS	NS
Isefjord	TN	N load				0.11	0.48	0.53	NS	Yes
Isefjord	TP	P load	irradiance			0.16	0.42	0.59	NS	Yes
Kolding	chl3	salinity	irradiance	irradiance		0.20	0.23	0.79	NS	NS
Kolding	kd2	salinity	wind^3			0.11	0.29	0.71	NS	NS
Kås	chl3	N load	irradiance	SST		0.13	0.43	0.57	NS	NS
Kås	kd2	N load	salinity			0.19	0.88	0.13	NS	Yes
Kås	TN	N load				0.19	0.70	0.31	NS	Yes
Kås	TP	P load	irradiance			0.08	0.35	0.65	NS	NS
Lillebælt	chl3	N load	wind^3			0.16	0.41	0.60	NS	NS
Lillebælt	kd2	N load	irradiance	SST		0.07	0.33	0.67	NS	NS
Lillebælt	TN	N load				0.07	0.31	0.69	NS	NS
Lillebælt	TP	P load				0.07	0.17	0.83	NS	NS
Lovns	chl3	N load	wind^3			0.27	0.48	0.54	NS	NS
Lovns	kd2	SST	wind^3	salinity		0.14	0.56	0.44	NS	Yes
Lovns	TN	N load				0.16	0.50	0.50	NS	Yes
Lovns	TP	SST	wind^3			0.08	0.17	0.83	NS	NS
Løgstør	chl3	N load	irradiance			0.34	0.72	0.30	NS	NS
Løgstør	kd2	N load	salinity			0.16	0.61	0.40	NS	NS
Løgstør	TN	N load	SST			0.22	0.77	0.24	NS	Yes
Løgstør	TP	P load	SST	irradiance		0.11	0.43	0.57	NS	NS

Station	Response variable	1. Predictor variable	2. Predictor variable	3. Predictor variable	4. Predictor variable	RMSECV	RMSEp	R^2	Shapiro normality	AC Box-Pierce
Nissum	chl3	N load				0.25	0.57	0.44	NS	NS
Nissum	kd2	N load	BV-BF			0.11	0.73	0.27	Yes	NS
Nissum	TN	N load	irradiance			0.28	0.69	0.32	NS	NS
Nissum	TP	P load				0.15	0.81	0.20	NS	NS
Odense outer	chl3	N load	salinity			0.18	0.80	0.20	NS	NS
Odense outer	kd2	N load	wind^3			0.12	0.67	0.33	NS	NS
Odense outer	TN	N load	wind^3			0.13	0.29	0.72	NS	Yes
Odense outer	TP	P load	irradiance			0.21	0.44	0.57	NS	NS
Odense inner	chl3	P load	SST	salinity		0.19	0.34	0.67	NS	NS
Odense inner	TN	N load	irradiance			0.07	0.11	0.89	NS	NS
Odense inner	TP	P load	salinity			0.27	0.44	0.58	Yes	NS
Randers inner	chl3	P load	SST	wind^3		0.23	0.28	0.74	NS	NS
Randers inner	kd2	N load	BV-BF	irradiance		0.09	0.18	0.82	Yes	NS
Randers inner	TN	N load				0.09	0.26	0.74	NS	NS
Randers inner	TP	P load	BV-BF			0.06	0.34	0.66	NS	NS
Randers outer	chl3	wind^3	discharge			0.27	0.58	0.44	NS	NS
Randers outer	kd2	N load	salinity			0.20	0.70	0.31	NS	NS
Randers outer	TN	N load	wind^3			0.09	0.14	0.86	NS	NS
Randers outer	TP	P load	wind^3			0.09	0.28	0.72	NS	NS
Riisgaarde	chl3	wind^3				0.26	0.48	0.54	NS	NS
Riisgaarde	kd2	salinity	wind^3			0.14	0.41	0.59	Yes	NS
Riisgaarde	TN	N load	salinity	irradiance		0.08	0.32	0.68	NS	NS
Riisgaarde	TP	P load	irradiance			0.05	0.09	0.92	NS	NS
Ringkøbing	chl3	N load				0.33	0.88	0.12	NS	NS
Ringkøbing	kd2	N load	wind^3	SST		0.11	0.32	0.69	NS	NS
Ringkøbing	TN	N load	wind^3	BV-BF		0.03	0.04	0.96	NS	NS
Ringkøbing	TP	P load				0.14	0.39	0.61	NS	NS
Roskilde	chl3	N load	wind^3			0.24	0.25	0.77	NS	NS
Roskilde	kd2	N load	salinity	irradiance		0.15	0.51	0.50	NS	NS
Roskilde	TN	N load	SST			0.10	0.57	0.43	NS	NS
Roskilde	TP	P load	salinity			0.17	0.12	0.89	NS	NS
Skive	chl3	N load	irradiance	wind^3		0.43	0.76	0.27	Yes	NS
Skive	kd2	N load	salinity	irradiance		0.15	0.58	0.42	NS	NS
Skive	TN	N load	irradiance	BV-BF		0.17	0.46	0.55	NS	Yes
Skive	TP	SST	wind^3			0.14	0.37	0.64	NS	NS
St.bælt Fynshoved	chl3	N load				0.12	0.31	0.70	NS	NS
St.bælt Fynshoved	kd2	N load	wind^3			0.05	0.29	0.71	NS	NS
St.bælt Fynshoved	TN	N load	wind^3			0.06	0.36	0.64	NS	NS
St.bælt Fynshoved	TP	P load	salinity			0.09	0.24	0.76	NS	NS
St.bælt Romsø	chl3	N load	salinity	BV-BF		0.09	0.38	0.62	NS	NS
St.bælt Romsø	kd2	N load	wind^3			0.09	0.56	0.44	NS	NS

Station	Response variable	1. Predictor variable	2. Predictor variable	3. Predictor variable	4. Predictor variable	RMSECV	RMSEp	R^2	Shapiro normality	AC Box-Pierce
St.bælt Romsø	TN	N load				0.09	0.48	0.52	NS	NS
St.bælt Romsø	TP	P load				0.06	0.11	0.89	Yes	NS
Thisted	chl3	N load	BV-BF			0.48	0.56	0.49	NS	NS
Thisted	TN	N load				0.15	0.61	0.39	NS	NS
Vejle	chl3	P load	wind^3	irradiance		0.24	0.31	0.71	Yes	NS
Vejle	kd2	irradiance	BV-BF			0.11	0.47	0.53	Yes	NS
Vejle	TN	N load				0.14	0.48	0.52	NS	NS
Vejle	TP	P load	BV-BF			0.03	0.03	0.97	NS	NS
Åbenrå	chl3	P load	irradiance			0.21	0.47	0.54	NS	NS
Åbenrå	kd2	N load	wind^3	irradiance		0.10	0.53	0.47	NS	NS
Åbenrå	TN	N load	SST			0.20	0.50	0.51	NS	NS
Åbenrå	TP	P load	salinity			0.16	0.52	0.49	NS	Yes
Århus	chl3	P load	salinity			0.19	0.53	0.48	Yes	NS
Århus	kd2	P load	irradiance	BV-BF		0.04	0.28	0.72	NS	NS
Århus	TN	N load	salinity			0.09	0.56	0.44	NS	Yes
Århus	TP	P load	wind^3			0.09	0.65	0.35	NS	NS

6.5 Reflections about the concept and perspective for next WFD-period

As mentioned above, the aim of the project is to provide a model-based management tools for estimating maximum allowable loadings (MAI) for each of the 119 marine water bodies covered by the WFD in Denmark. The concept behind the statistical approach, as described above, was designed to optimise the results, i.e. the estimated values for the required load reduction to reach GES. It is important to distinguish this from a scientific analysis testing the hypothesis that nutrient loadings affect the environmental conditions in a water body. In the latter case, it would be important to test the effect of e.g. de-trending and to test the significance of the number of times when each predictor variable was selected. Given the large number of models tested, there will be cases of spurious correlations, i.e. where the model suggests a relationship that is not based on a causal mechanism but is a coincidence. On the other hand, as mentioned above, the opposite situation will also occur. In this project, based on overwhelming evidence in the scientific literature, we have assumed that nutrient loadings do have an impact on the selected response variables and we therefore designed the method to provide the most likely coefficient for this response.

The usefulness of the approach is clearly highest when predicting the response of a change in loadings within or close to the range covered by data. As for all statistical and mechanistic models, their robustness and trustworthiness will decrease when moving away from the range against which the models are validated. The original idea with the concept was to predict responses of marine ecosystems to changes in loadings within or slightly outside the range for which we have data. We believe that the methods applied fulfil this aim. However, when the target is far away, i.e. large reductions are necessary to achieve GES, the predictions will be uncertain and merely reflect the fact that large reductions are needed.

To obtain more certain MAI estimates, it is important continuously to monitor the ecosystems as they approach GES and to update the models accordingly. Thus, the statistical models should be regarded as a tool that, together with a comprehensive monitoring effort, gradually can fine-tune the MAI estimates. Over longer time spans like decades, other pressures, not least climate change, will call for re-evaluation of GES values and the concurrent MAI values. Thus, we see the concept as a starting point for a continuous effort to achieve balance between society and the surrounding environment.

Overall, we believe that the concept and the results are valid and represent a major step forward from the previous WFD plan period. However, there is room for improvement of the concept in a number of ways for the next plan period. These are briefly mentioned below.

Time lag

The time lag between load reductions and responses is considerable and this should be addressed in a quantitative way. A major constraint is the number of years available, a problem that will gradually decline as the time series become longer. On the other hand, reductions in the monitoring program mean that the number of stations with sufficient data declines, limiting the possibility of undertaking in-depth investigations.

Dual effect of nitrogen and phosphorous

This issue is briefly discussed in later sections (e.g. 8.3). As mentioned there, this represents a major reorganisation of the indicators. The current intercalibrated indicator for chlorophyll-*a*, covering the months May to September, should be split into a spring (e.g. March-June) and late summer (e.g. July-October) indicator, and additional indicators that are more responsive to phosphorus loadings than the current indicators would be necessary. Such indicators could be chlorophyll-*a* spring peak, spring primary production, the share of phosphorus sensitive phytoplankton species or a nutrient limitation indicator analogous to the nitrogen limitation indicator (see section 8.3); preferably a parameter that is already included the existing monitoring program.

Hierarchical models and Bayesian statistics

In many ways the simplicity of the PLS regressions is an advantage when the causal relationship between the response variable and the predictor variable is already known from other studies or if the causal relationship is of minor importance. As long as a more or less linear response can be expected the PLS regressions can be a good tool to predict the level of the response variable from the values of the predictor variables. However, the relatively simple regressions have a drawback. Thus, it is difficult quantitatively to include prior knowledge and documented feedback mechanisms in the model, which implies that caution must be taken when making extrapolations far outside the calibration range. Furthermore, no formal theoretical methods to deal with model uncertainty exist.

The next step in the model development process could be to combine Bayesian network with hierarchical modelling. This approach have been applied in e.g. lake water quality modelling (Malve & Qian 2006) and combines some of the advantages of the regression models with the mechanistic models, allowing inclusion of feedback mechanisms and built-in parameter constraints based on a prior probability space. This could also provide managers and policy makers with estimated probabilities of achieving the desired result under different future nutrient scenarios (Borsuk et al. 2004; Malve & Qian 2006). The network approach also permits meta data (morphometry, catchment characteristics, hydraulic retention time etc.) to be built into the model in a more parametric way than in the meta models developed in this study (see section 8.6). This may potentially improve the estimated MAIs for areas that are only extensively monitored.

7 Mechanistic model development

7.1 Application of mechanistic models in environmental studies

Dynamic mechanistic models focusing on water quality have for more than four decades provided important support for the management of lakes, estuaries and coastal waters (Fath et al. 2011; Janssen et al. 2015). Since the early start in the 1970s, the models have evolved continuously, and with the steadily increasing CPU capacity, spatial resolution, spatial coverage and/or biogeochemical details have been expanded. The earliest mechanistic models focused on plankton growth, the associated nutrient dynamics and dissolved oxygen (e.g. Di Toro et al. 1971; Thomann et al. 1974; Nyholm 1977; 1978). Later, driven by the ambitions to mimic "real nature", water quality models gradually turned into aquatic ecosystem models (AEMs) by introducing several phytoplankton groups, zooplankton, biotic benthic state variables such as filter feeders and deposit feeders, and carbon and nutrient pools as well as dynamics in the sediment, etc. (e.g. Baretta et al. 1995; Butenschön et al. 2016). While the pitfalls posed by overly complex models (i.e. overfitting resulting in reduced reliability) is recognised by some modellers, for instance Friedrichs et al. (2007), quantitative evaluation of the appropriate model complexity is rare. As a rule of thumb, models should only be as complex as needed to address specific questions. Furthermore, inclusion of new state variables without firm knowledge of their role in the ecosystem and associated processes and rates should be avoided. Therefore, excess CPU capacity may be better used on increasing the spatial resolution and the coverage as well as temporal resolution than on enhancing the complexity of the ecosystem description. Higher temporal resolution is, for example, more beneficial when transport time scale dominates over the scale of the biogeochemical processes, implying that advection plays an important role in distributing soluble and planktonic state variables (Fraysse et al. 2013).

Today, AEMs are applied as management tools to evaluate the efficiency of eutrophication mitigation strategies (Thieu et al. 2010) and other actions related to, for instance, climate change (Neumann 2010; Meier et al. 2011). In a recent survey, about 80% of the respondents (environmental managers and decision-makers) representing 25 states across the USA replied that they used models (statistical and/or mechanistic) or model results in their management of aquatic ecosystems (Fitzpatrick et al. 2016).

Results from detailed science-based mechanistic models can be difficult to incorporate in a management framework and such issues might be the reason for reluctance of managers in several EU countries to adopt mechanistic modelling as a general tool. A way forward was demonstrated by Nobre et al. (2005) who combined mechanistic modelling to estimate current and pristine status of chemical and biological components not quantified and fed the model output - and monitoring data - into a screening tool (Assets) to grade eutrophication status into five classes: High, Good, Moderate, Poor and Bad.

In the development of the model toolbox for the Danish RBMP 2015-2021, the key question was how to calculate the maximum allowable input (MAI) of nutrients supporting maintenance/achievement of good ecological status (GES) of the coastal water bodies governed by the WFD. In Denmark, attempts have never been made before to develop a mechanistic model toolbox applicable to a large part of the Danish WFD water bodies. Collective modelling efforts, especially by the Baltic NEST Institute (BNI) and HELCOM, have paved the way for the Baltic Sea Action Plan[8] in which MAIs are set for the regional water bodies of the Baltic Sea, but the spatial resolution of the BNI model is too crude to resolve the coastal zones that directly receive the land-based nutrient loads.

Numerous model studies have been carried out to clarify the impacts of nutrient loads on the environmental status of different water bodies in Europe (Neumann & Schernewski 2004; Carstensen et al. 2011; Lenhart et al. 2010; Thieu et al. 2011; Meier et al. 2011). Thorough studies have also been made of individual Danish estuaries (Kuusemäe et al. 2016), and some of these model studies form the basis for the development of the marine mechanistic model RBMP toolbox. In the following sections, the models developed and their application are described in detail.

7.1.1 Nutrient loads and circulations

As described in chapter 5, the model indicators adopted for this study are summer chlorophyll-*a* and the indicator summer-K_d. Chlorophyll-*a* is an indicator of phytoplankton biomass and K_d is used as a proxy for the potential abundance of eelgrass. The major factors governing the development of phytoplankton in offshore Danish waters are nutrient loadings and hydrography, both of which are tightly coupled with meteorology. In inlets and shallow estuaries, the biomass of benthic filter feeders filtering the bottom water also has a decisive influence on phytoplankton concentrations (Møhlenberg 1995). Total nutrient loads from Danish land to Danish marine waters averaged 61.2 kton N and 2.5 kton P per year for the period 2007-2011. Loads to in the inner Danish waters (Kattegat south of the Skagen-Gothenburg cross-section, the Belt Sea and the Western Baltic Sea) averaged 43.4 kton N and 1.8 kton P, and the remaining loads were discharged to the North Sea/Skagerrak area. Other important external nutrient sources for the inner Danish waters are atmospheric deposition, nutrients in inflowing water from the Skagerrak and Baltic Sea and direct runoff from Sweden and Germany to the area. Along with the land-based loads and exchanges with adjacent seas, nutrients are continuously recycled in the water column and sediments by heterotrophic activity (bacteria, zooplankton and higher trophic levels).

Two field studies carried out in the Bay of Aarhus (1990-1991) and in the southeastern Kattegat (1988) estimated that the yearly nitrogen remineralisation rates (primarily in the form of NH_3-N) ranged from 40 to 50 g N m^{-2} (Kaas et al. 1990; Jørgensen et al. 1994), suggesting that the recycling of nitrogen was at least a magnitude higher than the present-day land-based Danish nitrogen load. Both runoff and mineralisation show pronounced seasonal variation, but with a mirroring pattern; runoff peaks during wet winter months when the soil is saturated with water and crop growth is light limited, while the remineralisation in the water column and the sediment grossly follows temperature and peaks during late summer and autumn. Therefore, land-based runoff is important during early spring as it fuels the algae spring bloom, while remineralised nutrients are the main driver of summer and autumn production. However, winter run-off also constitute a

[8] An ambitious programme to restore the good ecological status of the Baltic Sea by implementing various measures including nutrient load reductions by 2021.

proportion of the mineralisation of nutrients later in the season, relative to total nutrient inputs and the standing stock of bioavailable nutrients in the system.

7.2 Model description

All dynamic mechanistic models were set up using DHI's model software; for hydrodynamic modelling MIKE 3 HD Flexible Mesh (FM) (DHI 2013a) was applied and for biogeochemical modelling the numerical MIKE solver ECO Lab model was employed (DHI 2013b).

Four separate mechanistic biogeochemical models were developed: one model covering the Baltic Sea from the Bothnia Bay to the Skagerrak (IDW model), and three models covering three different coastal estuaries: Odense Fjord, Roskilde Fjord and the Limfjorden. With regard to the Baltic Sea model, focus was on simulating the environmental condition of the Danish part of the Baltic Sea, collectively termed the "Inner Danish Waters", comprising Kattegat, the Belt Sea and the western Baltic Sea (see chapter 5). The model was therefore called "Inner Danish Waters", abbreviated IDW in the text below. In addition, a hydrodynamic model covering the Danish part of the Wadden Sea and the west coast of Jutland was developed.

7.2.1 Hydrodynamic model

Hydrodynamic MIKE 3 HD Flexible Mesh (FM) models were set up to represent the specific hydrodynamic conditions of the specific study areas.

MIKE 3 HD FM is applicable for the study of a wide range of physical phenomena, for instance: tidal exchange and currents, including stratified flows, salinity and heat transfer.

MIKE 3 HD FM solves the time-dependent conservation equations of mass and momentum in three dimensions, the so-called Reynolds-averaged Navier-Stokes equations. The flow field and pressure variation are computed in response to a variety of forcing functions when provided with bathymetry, bed resistance, wind and atmospheric pressure field, hydrographical boundary conditions, etc. MIKE 3 HD FM uses the UNESCO equation for the state of seawater as the relation between salinity, temperature and density (UNESCO 1981).

The unstructured flexible mesh and finite volume solution technique of MIKE 3 FM allow for a variation of the horizontal resolution of the model grid mesh within the model area to obtain a finer resolution of selected subareas. In *Figure 7.1*, a zoom of the model extent of the IDW model is included, and *Figure 7.2* shows an example of one of the estuary models.

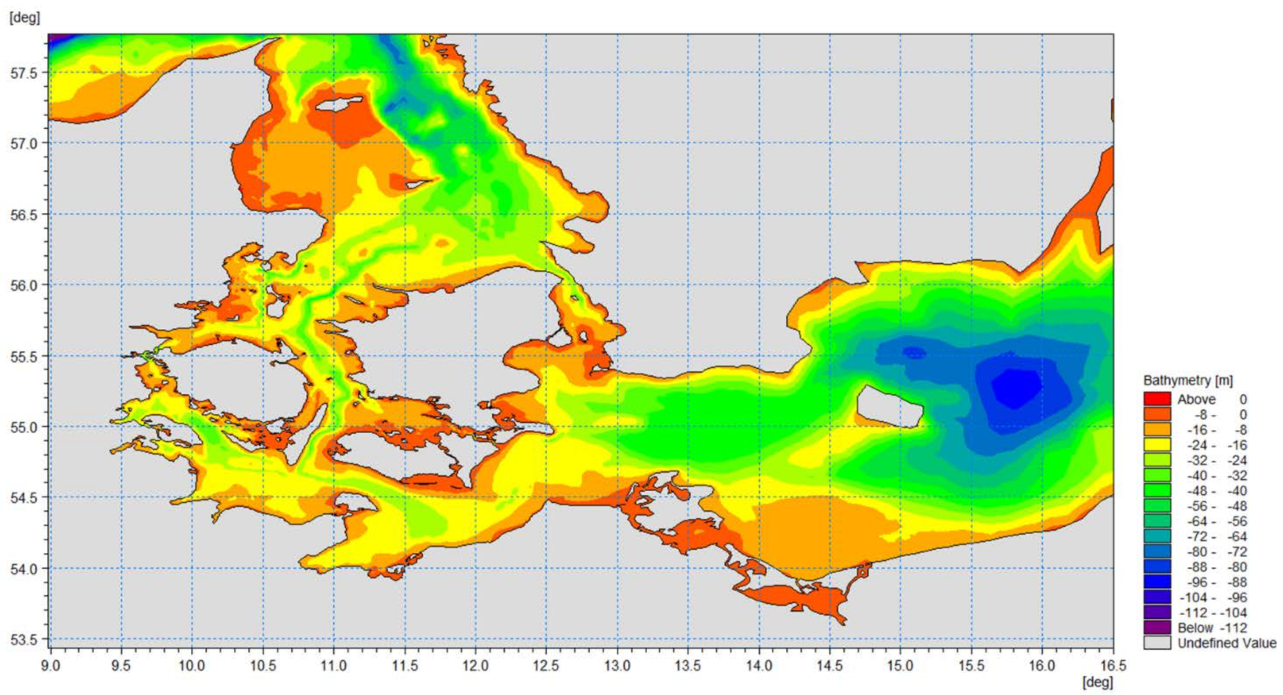


Figure 7.1. Model zoom of the inner Danish waters from the regional IDW model. The model covers the waters from the Skagerrak in the north-west to the Bothnian Bay in the east. Colours indicate water depths relative to mean sea level.

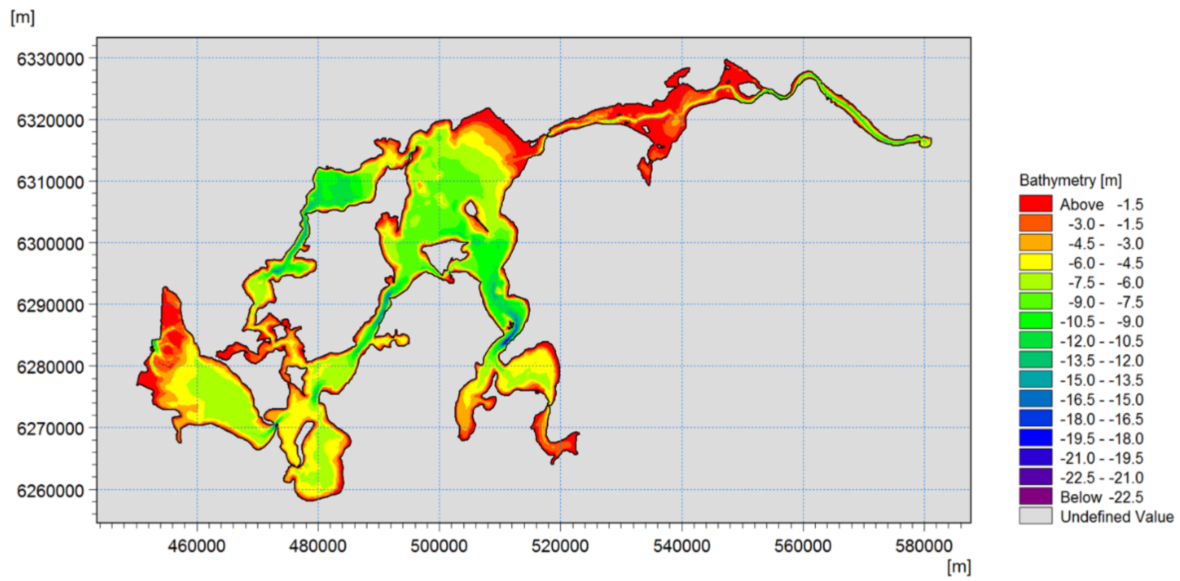


Figure 7.2. The model covering the Limfjorden extending from Thyborøn Canal in the west to Hals in the east. Colours indicate water depths relative to mean sea level.

7.2.2 Biogeochemical model

The numerical MIKE solver ECO Lab comprises a set of standard models (templates) that is used as basis for describing the processes relevant for a particular biogeochemical model. In the RBMP model development process, two well-established templates were adapted into the biogeochemical models – one, which simulated the ecosystems covered by the IDW model, and one, slightly different, which was implemented in the three estuary models (estuary model). The basic structure of the two models is identical. They both include interactions between the pelagic and the benthic compartments, but in the estuary models more details are included to describe the pelagic-benthic interactions. As an example, the estuary models include a two-layer sediment model, whereas the IDW only includes one layer, but the IDW includes more algae groups. In the following, we briefly describe the basic characteristics of the biogeochemical models, and in *Table 7.1* the similarities and differences between the models are highlighted.

The general biogeochemical model adapted for the RBMP toolbox describes the relationships and interactions between nutrients and primary producers. The biogeochemical model consists of two major sub-modules: the pelagic system and the benthic system. A schematic representation of the pelagic cycling of carbon (C), nitrogen (N) and phosphorous (P) is given in *Figure 7.3*.

Biogeochemical model – pelagic compartment
Most biogeochemical AEMs include two to three (or more) phytoplankton functional groups in order to simulate the seasonal variations in phytoplankton biomass and composition (see *Box 7.1*). Three typical variables included are i) a diatom state variable to represent a non-motile, silicate-dependent trait having low light requirements and relying on turbulence to prevent early sedimentation; ii) a flagellate variable to allow for neutrally buoyant cells; iii) a colony-forming cyanobacteria variable to represent N_2 fixing species inhabiting brackish waters (< 10-12 psu) that have an ability to aggregate in surface waters during calm periods and exhibit a steep growth response with increasing temperature.

Phytoplankton growth is the result of primary production minus losses. Where production mainly is controlled by nutrient and light availability as well as temperature, losses comprising respiration, grazing and sedimentation. The nutrient dependency of phytoplankton is described by a two-step process. Firstly, the inorganic nutrients are taken up into an internal pool of the algal cells, following the Michaelis-Menten kinetics for nutrient uptake as a function of the ambient nutrient concentration. Following the uptake, algal growth is described according to the Droop quota model (Droop 1968) for growth as a function of the intracellular nutrient concentration (Morel 1987; Haney & Jackson 1996; Erichsen & Rasch 2001). Details on how these processes are solved mathematically are available in Lessin & Raudsepp (2006), DHI (2013b) and DHI (2014).

The nutrients in the pelagic compartment originate from external sources (river and direct discharges, atmospheric deposition, etc.), pelagic recycling, as described in section 7.1, and "internal loading" from the sediments driven by mineralisation of organic matter produced in the water column (or by benthic plants harvesting inorganic nutrients from the water column).

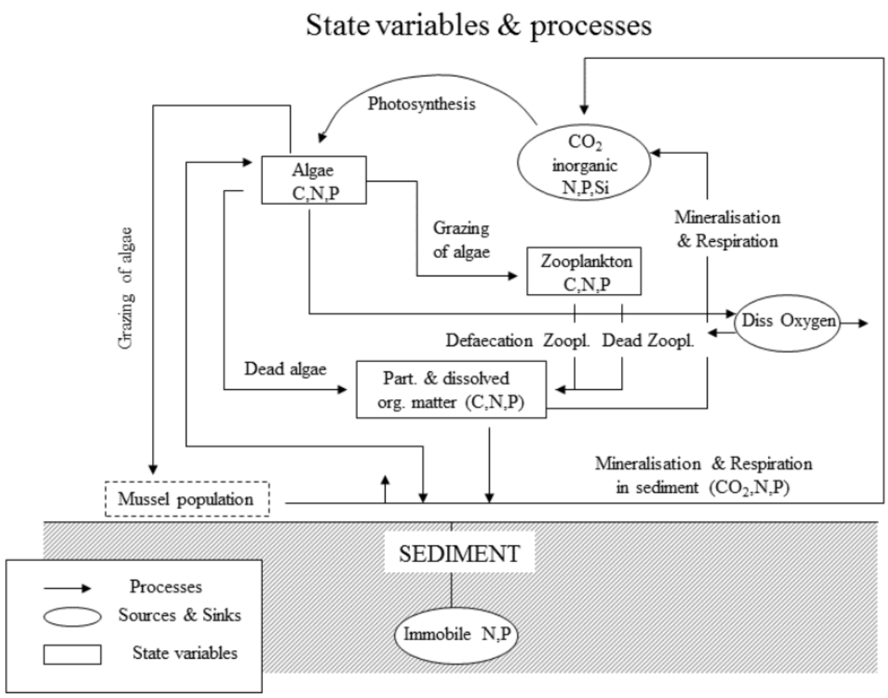
In the IDW model, nitrogen is also made available through nitrogen fixation by cyanobacteria. N_2 fixation is an important source of N to the Baltic Sea, and some estimates suggest that N fixation is comparable to the total land-based N load to the Baltic Sea (Neumann & Schernewski 2008). Since cyanobacteria are rare in the three estuary systems, this process is not included in the models. Hence, based on experience from numerous model implementations, DHI decided to use three functional groups to represent the phytoplankton dynamics of open waters and one phytoplankton state variable for eutrophic estuaries.

Box 7.1.

Inclusion of phytoplankton diversity in AEM models

Generally, the phytoplankton community in the sea consists of numerous species exhibiting differences in morphology, size and density, affinity to nutrient uptake, capability to store nutrients, N_2 fixation, light requirement, buoyancy regulation, maximum growth rate, mixotrophy and susceptibility to grazing loss (Litchman & Klausmeier 2008; Klais et al. 2017). Such plankton diversity cannot be represented in mechanistic models, which may question the results if nutrient (or hydrographic, light) conditions are imposed that lay outside the range used for model calibration. Generally, small-sized individuals and taxa are most competitive at low nutrient concentrations (say at historic "reference conditions"), while larger individuals and taxa (e.g. diatoms, dinoflagellates, chlorophytes) benefit from higher and pulsed nutrient inputs to surface waters (Edwards et al. 2012; Litchman et al. 2015). In the Baltic Sea, seasonal variation in phytoplankton traits roughly reflects the variation in meteorology and hydrography, Kattegat being an exception as diatoms dominate here through all seasons (Klais et al. 2017). Recent model exercises predict lower phytoplankton concentrations and reduced cell size in a future warmer ocean characterised by more oligotrophic surface waters (Morán et al. 2010; Acevedo-Trejos et al. 2014), reflecting the allometric scaling of nutrient uptake efficiency that is part of such models (Ward et al. 2013). As an alternative or a supplement to a trait-based approach with several phytoplankton state variables, implementing plasticity in phytoplankton processes, for example temporally varying nutrient loads and light affinity for growth, may allow tracking of seasonal variation in phytoplankton biomass using only one state variable (Lefevre et al. 2003).

Figure 7.3. Simplified structure of the pelagic ecological module. For details on the sediment module, see Rasmussen et al. (2009).



Grazing and decomposition "transform" phytoplankton to zooplankton and detritus (particulate and dissolved organic matter, including C, N and P), respectively. The fate of detritus includes sedimentation and mineralisation in the "microbial loop", which consists of mainly bacterially driven processes, leading to the remineralisation of dissolved and particulate organic matter that (re-)supply N and P to phytoplankton. Several AEMs explicitly represent members in the microbial loop (bacteria → heterotrophic flagellates → ciliates) and allow bacteria to compete with phytoplankton for nutrients (e.g. ERSEM; Vichi et al. 2007). Other AEMs take a simpler approach whereby a proportion of the detritus (at a temperature-dependent rate) is directly remineralised to inorganic nutrients (e.g. Yool et al. 2011). The two types of ECO Lab models adopted in this study use the latter approach, the mineralisation of organic matter by bacteria is parameterised (mainly based on temperature) without explicitly including the bacterial biomass as a state variable.

In the Baltic Sea and the estuary systems, substantial amounts of dissolved organic matter (DOM) are discharged from the rivers and contribute to the turnover of organic matter and influence light conditions. Two fractions of dissolved organic matter are represented in the model: labile dissolved organic matter (LDOM) and coloured dissolved organic matter (CDOM). Each of the three states of organic matter (detritus, LDOM and CDOM) is represented by three dynamic state variables (OC, ON and OP).

Grazing on phytoplankton by zooplankton (micro- and mesozooplankton) may have a regulating effect on the phytoplankton biomass, and the intensity of grazing can be a determinant reason for development of an algal bloom. Mesozooplankton, representing copepods, consists of a lumped biomass encompassing all 12 active grazing stages. Growth efficiencies and dependence on phytoplankton concentrations are based on the energy budget for *Acartia tonsa* (Kiørboe et al. 1985) and stage duration/growth rate dependence on temperature comes from Hirst & Sheader (1997). Regular zooplankton monitoring data on Danish waters after 1997 are sparse and formal model calibration is thus not possible. However, our modelling results showing seasonal variation with a peak in microzooplankton about a month after the spring bloom and a peak in mesozooplankton in July (30-60 µg carbon L^{-1}) are consistent with earlier studies from Kattegat (Kiørboe & Nielsen 1994; Nielsen and Kiørboe 1994). The model does not include heterotrophs (predators) at higher trophic levels than zooplankton and loss by predation is integrated in the death rate of the zooplankton. Hence, predation on mesozooplankton – whether by fish or jelly fish etc. – is handled by an unnaturally high zooplankton death rate, described by a quadratic function of zooplankton biomass.

In summary, the pelagic ecosystem model computes the concentration of phytoplankton (as carbon and chlorophyll-*a*), zooplankton, detritus and dissolved organic matter as well as the nutrient and dissolved oxygen content of the water phase. For more details, DHI (2013b) and DHI (2014) may be consulted.

Biogeochemical model – sediment compartment

Benthic-pelagic coupling encompasses numerous processes (sedimentation, filtration, nutrient uptake in benthic plants, bioturbation, mineralisation, resuspension, predation) that drive the exchange of solutes, particles and organisms between the pelagic and benthic compartments (Griffiths et al. 2017). The nutrients in the pelagic compartment originate, among other sources, from “internal loading” from the sediments due to mineralisation of organic matter. The internal sediment loading varies according to the size of the biogeochemically available pools of C, N and P in the sediment together with bottom oxygen concentrations, water temperature and the bottom water exchange.

The applied ECO Lab model seamlessly integrates the pelagic and benthic compartments. The estuary models include two sediment layers, while the open water model includes one vertically integrated layer. *Sensu* the Soetaert et al. (2000) definition, the estuary models are characterised as "level 4-3½" and the IDW model as "level 3".

The degradation of the organic C, N and P pools of the sediment (utilising oxygen or nitrate as electron acceptors) releases N and P to the sediment pore water. The rate of degradation depends on the availability of oxygen (or NO_3) and the C:N ratio in the sediment. A minor fraction of the organic pool (C, N and P) is immobilised depending on the C:N ratio in the sediment (the fraction increases with increasing C:N ratio). Nitrate in the pore water can be denitrified to N_2. Inorganic pore water P can bind to oxidised iron (Fe^{+++}) when the sediment is oxidised, and when the sediment is "reduced" (Fe^{++} being the dominant form) the inorganic P is released to the pore water again. The inorganic nutrients in the sediment exchange with nutrients in the water phase and, hence, the sediment may either act as a sink or a source of inorganic nutrients to the water above the sediment. For details of the sediment model, we refer to Rasmussen et al. (2009).

Biogeochemical model – benthic production

All four AEMs include a description of the benthic primary producers. Four different benthic primary producers are included: perennial ("brown") macroalgae typified by fucoid species, annual macroalgae (e.g. filamentous brown algae and *Ulva* sp.), benthic microalgae and the flowering plant eelgrass (*Zostera marina*).

As for the pelagic primary production, the benthic primary production depends on water temperature, nutrient availability and availability of photosynthetic active light, but the relations between the different factors and growth differ between groups. Besides growth regulation factors, perennial macroalgae need hard substrate to attach to, and eelgrass requires appropriate sediment (appropriate grain size and organic carbon below 4%). An important distinctive characteristic is that macroalgae (annual and perennial) can only utilise inorganic nutrients from the water phase, whereas eelgrass can take up nutrients from the sediment pore water as well. Hence, eelgrass can grow in areas and seasons with low nutrient concentrations in the water if pore water nutrient concentrations are sufficiently high. Analogously, two nutrient sources are available to benthic microalgae growing on sediments, namely pore water nutrients in sediment and nutrients in overlaying water. As for phytoplankton, the internal pools of N and P drive the growth of the different groups and the pools are described explicitly for eelgrass, benthic microalgae and macroalgae. Accumulation of internal nutrients (resulting in low C:N and C:P-ratios) during winter and spring allows continued growth when external nutrients become depleted in surface waters (Pedersen & Borum 1996).

Finally, the light dependency varies between the different benthic groups as do losses. Losses include respiration, decay, grazing and loss of parts of plants. Dead organic material is partly returned to the water phase and partly to the organic pools of the sediment. In this way, the benthic primary producers contribute to the organic and inorganic nutrient pools in the model. More details on the benthic primary production model can be found in Kuusemäe et al. (2016).

Distinctions between IDW and estuary models

An overview of the similarities and differences between the two types of models developed, the IDW and the estuary models, is given in *Table 7.1*.

Table 7.1. Overall similarities and differences between the open water model and the estuary models.

	Model components	**Open and coastal water IDW model**	**Estuary models**
Pelagic compartment model	Phytoplankton C, N and P	Three algae groups: diatoms, flagellates and cyanobacteria. Growth rates, light dependency, sedimentation/buoyancy etc. differ between groups. Cyanobacteria capable of N fixation.	One lumped phytoplankton group. Growth rates and light dependency change between spring and summer/autumn.
	Zooplankton, C	Two groups of zooplankton: micro- and mesozooplankton.	One lumped zooplankton group.
	Benthic filter feeders	Not modelled explicitly, but effects by filter feeders are included.	Not modelled explicitly, but effects by filter feeders are included.
	Inorganic nutrients, NH_4, NO_x, PO_4, Si	Si included.	Si not included.
	Organic matter, C, N, P and Si	Si included.	Si not included.
	Inorganic sediment	Inorganic sediment not included. In areas with resuspension, an empirical relation between modelled shear stress and turbidity is developed. The relation is based on measurements from the western Baltic Sea and is a further development of the relation described in FEMA (2013).	An empirical relation is implemented linking modelled shear stress (wave and current generated) at the seabed, 2D-maps of inorganic and organic sediment pools and concentrations of particulate matter in the water phase to light regime.
	Dissolved oxygen	No structural differences.	
	Hydrogen sulphide	No structural differences.	
Benthic compartment model	Primary producers C, N and P Macroalgae Microbenthic algae Eelgrass	Loss of ignition and seed-burrowing effects of the lug worm *Arenicola* not included.	Sediment loss of ignition part of the possibilities for eelgrass to recolonise as well as part of a description of losses based on *Arenicola* burrowing activity. Processes of loss are described in Kuusemäe et al. (2016).
	Organic matter	Si included.	Si not included.
	Inorganic nutrients	Si included.	Si not included.
	Inorganic sediment	Not included.	An empirical relation is implemented linking modelled shear stress (wave and current generated) at the seabed, 2D-maps of inorganic and organic sediment pools and concentrations of particulate matter in the water phase to light regime.

7.3 Model setup

The hydrodynamic model behind IDW was originally set up as part of the EIA for the fixed link between Denmark and Germany (The Fehmarn Belt Fixed Link) (FEHY 2013). In the present IDW model, the mesh along the Danish coastline has been refined to resolve as many Danish coastal water bodies as possible. A more detailed description of the input to the hydrodynamic model is found in FEHY (2013).

Both the Odense Fjord model and the Roskilde Fjord model were developed and described by Kuusemäe et al. (2016). The Limfjorden model was set up specifically for the RBMP toolbox, applying the same biogeochemical model used for the two other estuaries. All models were set up and executed for the period 2002-2011. Details on the different models are given in *Table 7.2*.

Table 7.2. Overview of the four different model setups.

Model specifications		**Odense Fjord**	**Roskilde Fjord**	**The Limfjorden**	**Inner Danish Waters[a]**
Horizontal resolution (minimum / maximum) (average / median)		22 m / 804 m 275 m / 243 m	39 m / 663 m 155 m / 122 m	85 m / 1167 m 381 m / 317 m	299 m / 6330 m 1480 m / 1363 m
Vertical resolution[b]	Sigma layers	3 layers, ≤ 1 m	2 layers, ≤ 1 m	5 layers, ≤ 1 m	10 layers, ≤ 1 m
	z-layer	18 layers, 1 m	20 layers, 0.5 m	15 layers, 1 m	130 layers, 1-2 m
Model time step		300 sec.	300 sec.	3600 sec.	1800 sec.
Model period		2002-2011			
Initial fields	Hydrodynamic Biogeochemical	Measured S and T profiles Measured water chemistry profiles			
Boundaries	Hydrodynamic Biogeochemical	Measured S and T profiles, water levels from IDW Measured water chemistry profiles		Regional model data Regional model data	
Forcings	10 m wind 2 m air temp.	Measured wind and air temp		Meteorological model data	
Solar radiation		Measured global radiation		Meteorological model data	
Sediment maps (2D maps of nutrient pools in the sediment)		Based on extrapolated DNAMAP sediment data			
Fresh water and nutrient sources		15 point sources split into WWTP and diffuse loads	27 point sources split into WWTP and diffuse loads	42 point sources representing accumulated WWTP and diffuse loads	340 Danish point sources and 70 Baltic Sea point sources[c]

[a] The IDW model covers the entire Baltic Sea, but here we only highlight the resolution in the area from Skagen-Gotenburg and Rügen-Trelleborg (along 13° east).

[b] For all models, a sigma-z layer approach is applied, see DHI (2013a) for details.

[c] Baltic Sea point sources other than Danish point sources.

7.3.1 Nutrient loadings

An important input to the setup of the mechanistic models is the external supply of nutrients. Apart from Danish land-based nutrient loadings, the mechanistic models include nutrient input to the Baltic Sea from other countries and atmospheric deposition. In section 4.2, Danish land-based nutrient loadings and atmospheric deposition are described, both based on data from the Danish monitoring programme DNAMAP. The nutrient data included in the mechanistic modelling are summarised in *Table 7.3.*

Table 7.3. Nutrient loadings for the years 2002-2011 adopted in the mechanistic models

Loadings	**Source**
Danish land-based nutrient loadings	Delivered by AU on request from Danish EPA (Windolf et al. 2013)
Baltic Sea nutrient loadings	HELCOM (HELCOM 2011) combined with SMHI model data
Atmospheric N deposition	AU, Geels et al. (2012) and Ellermann et al. (2012)

In the following, the different nutrient loadings and adaptation to the mechanistic models are briefly described.

Danish land-based nutrient loadings

Data on Danish land-based nutrient loadings were provided by Danish EPA via DCE/AU, Department of Bioscience, see section 4.2 for more details. The data are part of the national inventory elaborated by AU every year (Windolf et al. 2013), which is similar to the reporting to HELCOM. However, an important difference between the national data and the data adopted by AU for the mechanistic modelling is the resolution in time. Whereas the national data are reported on an annual basis, the data used for the modelling were provided on a daily basis, both for water discharges and nutrient loadings.

The data covered the period from 1990 to 2011[9] and were distributed into 4th order marine waters, which are even more detailed than the 119 WFD water bodies. The mechanistic modelling applied data for the period 2002-2011, with distribution of the loadings into approximately 340 Danish freshwater and nutrient sources.

The loadings were estimated as discharges of total nitrogen (TN) and total phosphorus (TP). Since the mechanistic models differentiate between the different chemical forms (inorganic/organic, dissolved/particulate, nitrogen and phosphorous species), the data were subsequently transformed into nutrient forms required by the modelling. Through an assessment of available observations on nutrients in water discharged from Danish catchments, monthly relations between inorganic and organic nutrients were developed and applied to split TN and TP into an inorganic and an organic fraction. By combining TOC and COD/BOD observations, the organic part was further split to separate the organic nutrients into the three forms adopted in the modelling process.

Baltic Sea land-based nutrient loadings

Data on TN and TP loading from countries surrounding the Baltic Sea other than Denmark (including Norway) were extracted from HELCOM's Baltic Sea Pollution Load Compilation (HELCOM 2011). Hence, the data are those officially reported by the various countries. Differentiation of TN and TP loadings was done according to Stepanauskas et al. (2002).

As part of the model work carried out by Baltic Nest Institute (BNI), Savchuk et al. (2012a) evaluated the Baltic Sea Pollution Load Compilation elaborated in 2011/2012. The evaluation questioned some of the loads. Especially, the authors suggested that the P load to the Gulf of Riga was significantly underestimated. In addition, total loads from Russia to the Gulf of Finland and loads from Kaliningrad were considered uncertain. As Danish waters are located in the transition zone between the Baltic Sea and the North Sea and are heavily influenced by the outflow from the central Baltic Sea, N and P loadings to the entire Baltic Sea play a role for the Danish waters. Hence, deviations between reported data and actual loading data could potentially affect the overall performance of the models. In our modelling work with focus on the inner Danish waters, we did not, however, experience any systematic errors and therefore concluded that the official data on loadings were valid for the purpose of the modelling.

A brief analysis of the data from the Baltic Sea Pollution Load Compilation (HELCOM 2011; HELCOM 2015) showed that the general loadings to the

[9] The model development was initiated in December 2012. However, data on nutrient loadings were not available until late 2013, implying that the development of the mechanistic models and their subsequent application include only data from 2007 to 2011.

Baltic Sea declined in the period from 1994 to 2010. Both N and P loadings decreased, but the decrease was not evenly distributed. During the period, the actual N and P loads from Denmark declined by, respectively, 57% and 54% (1994 compared with 2010). In Germany, the figures were 45% N and 38% P and in Sweden 19% N and 13% P. In contrast, in Poland loadings increased by 2% N and 11% P. This increase is caused by the 2010 data – when averaging the data from the four-year period 2007 to 2011, a decrease of 29% N and 19% P emerges. Overall, the loadings to the Baltic Sea have decreased by 20% N and 12% P.

Figure 7.4 shows the relationship between the annual N loadings from Denmark and the corresponding loadings from Poland, Sweden and Germany. As can be seen, there is a strong correlation between especially the Danish and the German N loads, but also a rather strong correlation between the Danish and the Swedish loads. The correlation with the Polish loads is weaker, but the trend line indicates, though, that Danish and Polish loads decline simultaneously. The P loads display a similar trends (not shown), although the correlations with Swedish and Polish loads are weaker.

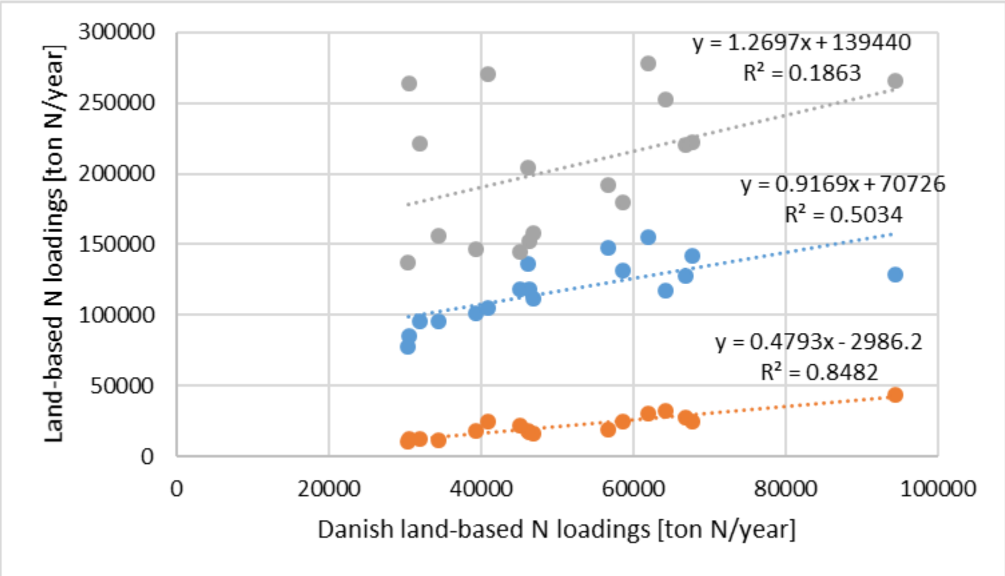


Figure 7.4. The relation between Danish annual loadings (tons) in the period 1994-2010 and the corresponding loadings from Germany (orange line), Sweden (blue line) and Poland (grey line).

7.3.2 Observations applied for evaluation of model skills

All observations used in the evaluation of the mechanistic models skills originate from the Danish National Aquatic Monitoring and Assessment Programme (DNAMAP). To evaluate model skills, we selected a number of monitoring stations representing the different model domains and the different water body types covered by the mechanistic models. The monitoring stations applied for the evaluation of model skills are shown in *Figure 7.5*. They all have at least 5 years of biogeochemical-data in the period 2002-2011 and in general they are visited 2-4 times per month.

The developed mechanistic models covers five of the water body types (four types and one with no category) adopted for this project, and to condensate the mechanistic model skills the modelled and measured data were lumped according to the typology described in section 3.2: Type 1 (open waters), Type 2 (estuarine), Type 3 (estuarine), Type 5 (sluice fjords) and No Category (no Type 4 water bodies are covered by a mechanistic model).

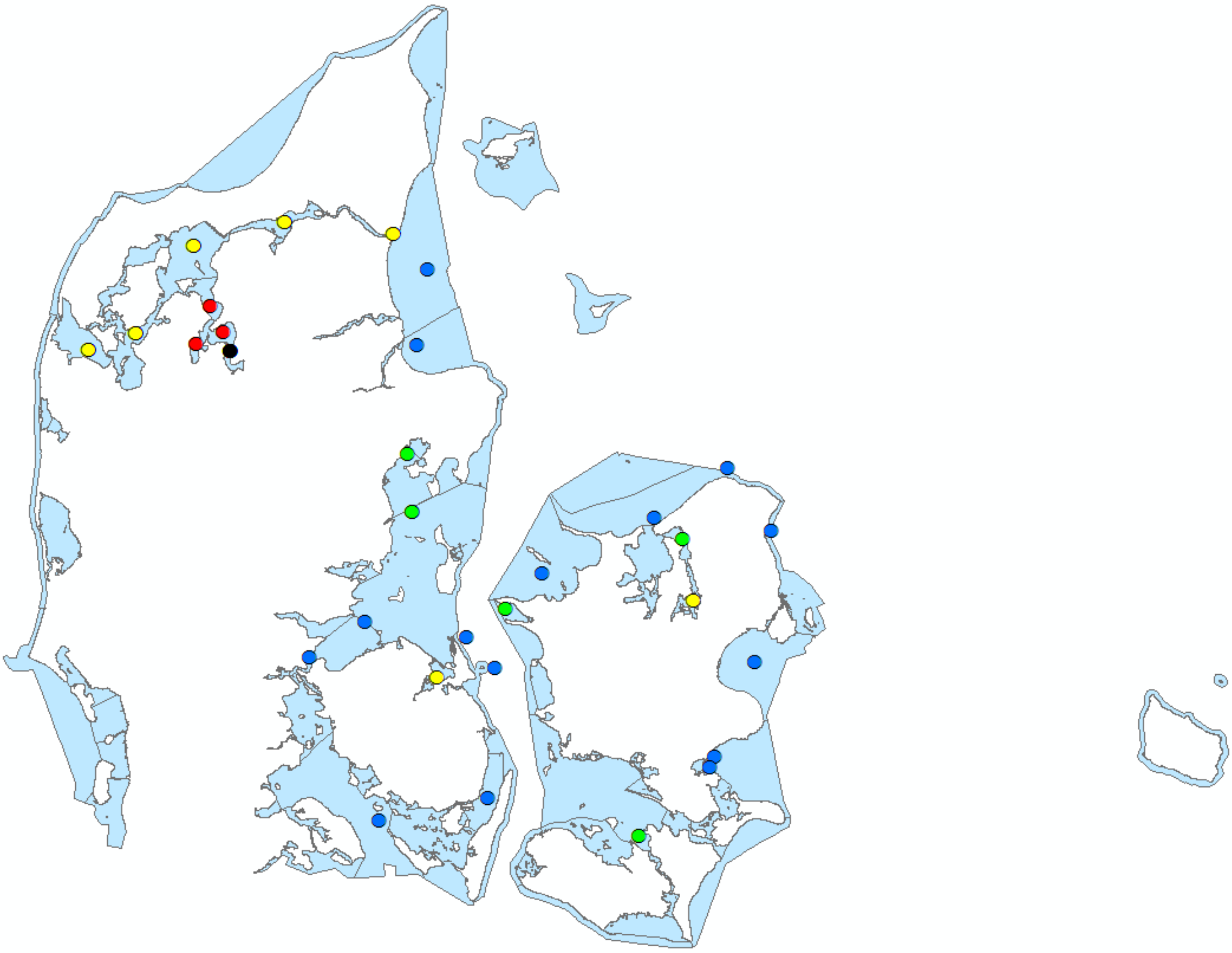


Figure 7.5. Water bodies and monitoring network applied for assessment of the model skills. The different colours of the dots indicate monitoring stations representing different water body types. Black dot in the most inner part of the Limfjorden represent a sluice-fjord (Type 5), yellow dots in the central part of the Limfjorden, the Odense Fjord and the Roskilde Fjord represent Type 3, green dots in coastal areas and the outer part of the Roskilde Fjord represent Type 2, blue dots represent Type 1. Red dots in the inner part of the Limfjorden is a water body out of category.

7.4 Skills of developed models

AEMs are increasingly used in management and decision-making (e.g. Fitzpatrick et al. 2016). To continue and expand the use of AEMs as management tools, decision-makers must have confidence in the model outputs, while understanding and accepting their limitations. Most frequently, the degree of confidence that can be placed in the results of a model is determined via various assessments of model skills, ranging from visual comparison of observed and modelled time series through seasons to numeric metrics (indices) that calculate deviation of model outputs from corresponding observational data. By building on indices in the assessment of model skills, potential inconsistencies arising from personal judgement will be minimised.

7.4.1 Skills of hydrodynamic models

Hydrodynamic models formed the basis of the biogeochemical models. For the model development, water levels, salinity and water temperature were verified (data not shown). To present a statistical summary of the performance of the model in modelling salinity and water temperature, three measures were used: BIAS, RMSE and R^2. For the estuary models, all available monitoring stations were used in the verification of model performance, and for the evaluation of the IDW model performance a large number of coastal and open waters stations were applied.

When calibrating the four different models, we endeavoured to reach statistical acceptance criteria corresponding to BIAS ≤ 1psu/1°C and RMSE ≤ 2psu/2°C for a minimum of 80% of all surface and bottom measurements at the open water stations and BIAS ≤ 2psu/2°C and RMSE ≤ 4psu/4°C at the coastal stations. For salinity, the acceptance criteria were compared with salinity values of about 8-30psu (lower in river outlets) in the Inner Danish Waters, intra-annual standard deviations of 3-4psu and typical vertical differences of 10psu, implying that the accepted BIAS and RMSE were limited. For temperature, the criteria were compared with a temporal standard deviation in the Inner Danish Waters of 4-6°C, the criteria values thus corresponding to about 20% and 40%, respectively. The specific acceptance criteria were lower for the coastal areas and enclosed water bodies as specific bathymetric details and local conditions become increasingly important here.

For the inner Danish water model (IDW), the criteria were fulfilled. The criteria were also met for the Limfjorden and the Roskilde Fjord models, whereas the salinity criteria for Odense Fjord were not fulfilled. The salinities in the bottom of Odense Fjord depend strongly on cooling water intake and outlet from the Odense power plant "Fynsværket", and we applied averaged intake/outlet values and not actual data (data not available), which influenced the performance of the model for the innermost part of the estuary. Temperatures in Odense Fjord ranged within the specified criteria.

7.4.2 Skills of biogeochemical models applied

Four biogeochemical models were developed, each covering a wide range of water body types. Here we focus the evaluation of the biogeochemical model skills on water body type level, and hence, evaluate the model skills over the variability of the different water body types. In *Table 7.4* the types covered by the different models are listed.

Table 7.4. Overview of models applied for the different water body types covered by a mechanistic model. The colours of the dots refer to *Figure 7.5*.

Model	Type 1 Open waters (Blue dots)	Type 2 Estuarine (Green dots)	Type 3 Esutarine (Yellow dots)	Type 5 Sluice fjord (Red dots)	No category (Black dot)
IDW	X	X			
Limfjorden			X	X	X
Roskilde Fjord		X	X		
Odense Fjord			X		

For the final application of the mechanistic models the focus is on summer chlorophyll-*a* and the indicator summer K_d averages for 2007-2011, but for the evaluation we include all seasons and all years (2002-2011). The parameters evaluated are focused on chlorophyll-*a* and K_d, but we also include NO_x[10], TN, DIP and TP.

[10] We use NO_x as not all stations included both NH_4 and NO_x.

Comparison of modelled and measured concentration of chlorophyll-*a* and light extinction coefficient (K_d) is depicted in *Figure 7.6* as monthly averages (±StDev). Summary statistics of data used in the comparison are shown in *Table 7.5.*

Table 7.5. Summary statistics of number of water bodies, number of stations per water body, number of samples per water body per year, and number of years included in comparison of measured and modelled chlorophyll-*a* and K_d (K_d observations in Type 5, however, being much less frequent, see *Figure 7.6).* Division of water bodies according to the defined typology, see section 3.2.

	Type 1 Open water	**Type 2 Estuarine**	**Type 3 Estuarine**	**Type 5: Sluice fjord**	**No category**
Number of water bodies	12	5	3	1	1
Average number of stations/WB	1.3	1.0	2.3	1.0	3.0
Average number of samples/WB/year	22	22	74	40	95
Years with samples	10	10	10	9	10

In the Type 1 and Type 2 water bodies, the seasonal succession in phytoplankton follows the well-known bimodal pattern for northern temporal waters: a spring bloom of diatoms develops in February-March and usually empties the nutrient pool within a couple of weeks (Andersson & Rydberg 1988). This is succeeded by a mixed, low-biomass summer assemblage comprising diatoms, dinoflagellates, mixotrophic organisms and nanoflagellates that are grazed intensively and fuelled by remineralized nutrients. The repletion of the nutrient pool in autumn increases the amount of algae and blooms of diatoms and/or large dinoflagellates may develop. In the Baltic Sea and eastern parts of the Danish waters there are recurring events of cyanobacterial blooms during summer and early autumn. The chlorophyll-*a* level usually varies between 1 and 3 $\mu g\ l^{-1}$ during summer and winter, and between 5 and 20 $\mu g\ l^{-1}$ during bloom situations in spring and autumn, or in connection with cyanobacterial blooms. As can be seen from *Figure 7.6*, the model tracks the yearly phytoplankton biomass expressed as chlorophyll-*a*, but underestimate chlorophyll-*a* in October-December (see the two lower panels), these months, however, not being part of the period used for the assessment of the chlorophyll-*a* indicator.

Seasonal variations in K_d in the Type 1 and Type 2 water bodies are less pronounced because the semi-inert dissolved organic matter (partly originating from the Baltic Sea (Stedmon et al. 2000; Babin et al. 2003) dominate light extinction. A small increase in K_d during spring bloom can be observed (two bottom panels in *Figure 7.6*) as well as a slight increase towards the coastal waters, the difference between the two bottom panels. Whereas the model reproduces K_d in the Type 2 water bodies well, K_d seems to be slightly overestimated in the Type 1 water bodies.

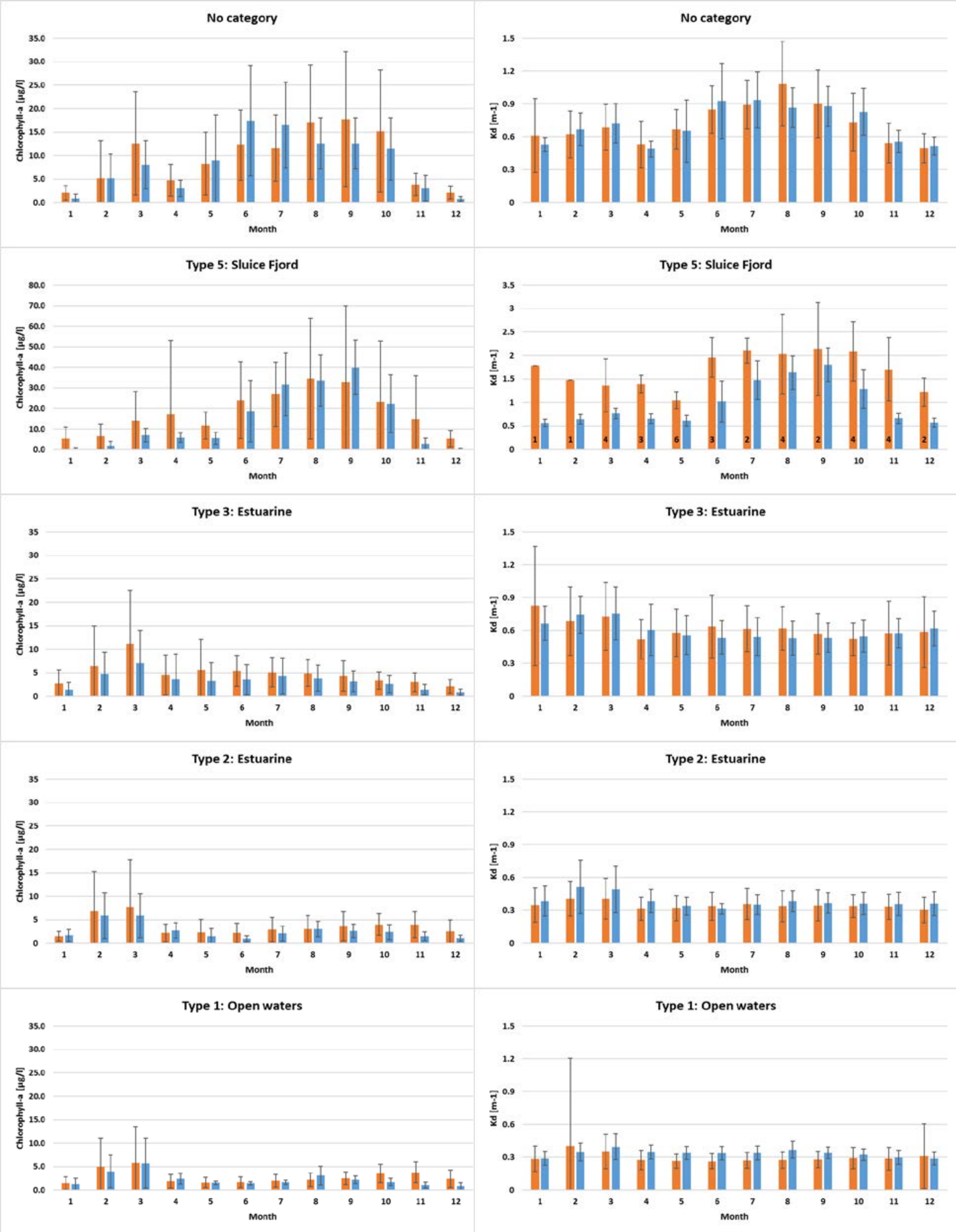


Figure 7.6. Average monthly measured (orange bars) and modelled (blue bars) concentrations of surface chlorophyll-*a* (left column) and K_d (right column). Error bars represent one standard deviation. From top panel to bottom panel the different water body categories are included: No category, Type 5, Type 3, Type 2 and Type 1. Notice that Type 5 have different y-axis than the other panels. Numbers in bars for the Type 5 water body indicate number of observations included in the analysis.

In the near-shore zone, including Type 3 and Type 5 water bodies and the No category water body, where the estuaries are affected by freshwater run-off, the chlorophyll-*a* levels are higher (Conley et al. 2000) and summer blooms are more common than in offshore waters, especially in areas where large pools of nutrients are released from the sediment during late summer when temperature peaks and oxygen in bottom water drops. Average summer levels vary between 5 and 20 µg l^{-1} with maximums up to 50-100 µg l^{-1}, see top three panels in *Figure 7.6.* This variation is reflected in both measurements and model results.

Especially, in Type 5 water body and the No category water body (the No category being a part of the Limfjorden) the K_d increases during summer coinciding with release of sediment nutrients and high concentrations of chlorophyll-*a* and dissolved organic matter. In the Type 3 water bodies (middle panel in *Figure 7.6*), K_d seems to drop during summer, which may be due to the low seasonal nutrient run-off, increased filtration from benthic filter feeders and/or reduced wind-driven resuspension. The seasonality in K_d and K_d levels are captured by the models for all three types, with larger deviation in the Type 5 water body. The deviations between the modelled K_d and the measured K_d in the Type 5 water body probably can be explained by a very low amount of measurements (in total 36 observations between 2002 and 2011, see number of observations per month indicated in the bars).

As a consequence of seasonal variations in nutrient loads, pelagic-benthic interactions and the phytoplankton biomass, the inorganic nutrients and total nutrients also display strong seasonality. Overall, these variations are captured by the model (data included in *Appendix C*). A tendency of the models to overestimate NO_x (as nitrite and nitrate) in April/May may be due to underestimation of phosphorus release from the sediment in the same period. This spring deficit is somewhat accounted for in the autumn, where DIP and TP are overestimated.

Quantitative indices are based on yearly resolved (2002-2011), monthly averaged measured and modelled concentrations of nutrients, chlorophyll-*a* in surface (0-1 m) and K_d (see *Table 7.6* above) for each of the five types of water bodies. Modelled values are harvested from model output data corresponding to sample positions and time. Two different metrics are used to quantify model skills: the Pearson Correlation Coefficient (*R*) and the Cost Function (*CF*).

The correlation coefficient *R* expresses the strength of a linear relationship between modelled and observed data:

$$R = \frac{\sum_{i=1}^{N}\left(M_i - \overline{M}\right)\bullet\left(O_i - \overline{O}\right)}{\left[\sqrt{\sum_{i=1}^{N}\left(M_i - \overline{M}\right)^2}\right]\left[\sqrt{\sum_{i=1}^{N}\left(O_i - \overline{O}\right)^2}\right]} \qquad \text{Eq. 7.1}$$

where M denotes modelled value, O denotes observed value, N is the number of paired values considered, while $M^{overbar}$ and $O^{overbar}$ denote the mean of modelled and observed values, respectively. R values range from -1 to +1 and a value of +1 corresponds to all data pairs lying on a straight line with positive slope in a scatter diagram. A value of R close to zero indicates an absence of linear correlation between the variables. Squared R (R^2) describes the proportion of the variance in measured data explained by the model. R^2 ranges between 0 and 1, with high values indicating low error variance, and values above 0.5 are considered acceptable in hydrodynamic models (van Liew et al. 2003). In evaluation of coupled hydrodynamic-ecosystem models R^2 values above 0.65 is considered "high", values between 0.65 and 0.35 is considered "moderate" and values below 0.35 is considered "low" in model skill assessment (Allen et al. 2007). R and R^2 have been widely used for model evaluation, but both are very sensitive to extreme values and insensitive to off-set of levels between model predictions and measured data (Legates & McCabe 1999).

The Cost Function (CF) is a measure of the misfit between the observed (Oi) and the predicted (Pi) variable values:

$$CF = \frac{1}{N}\sum_{1}^{N}\frac{|O_i - M_i|}{SD_O} \qquad \text{Eq. 7.2}$$

where SD_O is the standard deviation within observed data. The closer CF-value is to zero, the better the model. The difference between model and observation is related to the inherent variation in the field observations; hence a CF value of 0.5 means that the model error on average is 50% of the standard deviation of observations. CF can be seen as a complementary index to R and R^2, as it focuses on concentration levels (or "goodness-of-fit") rather than pairwise evolution in time or space of model predictions and observed data. Interpretation of CF values in terms of model skill assessment varies in the literature. We have used a conservative assessment scale applied by Radach and Moll (2006), where CF values below 1 characterizes a "very good" model, values between 1 and 2 a "good" model, values between 2 and 3 a "reasonable" model, and CF-values higher than 3 a "poor" model.

A summary of assessment statistics is shown in *Table 7.6*. Overall based on the metric scores, the models applied must be characterized as "good". Based on regression coefficients, models score higher than based on CF, indicating that models are "good" to "very good" to simulate seasonal variation in inorganic nutrients and chlorophyll-*a* and with the exception of Type 1 water bodies capable to simulate seasonal variation in K_d. The Cost Function values suggest lower model skill than the R^2-values, as the deviation between modelled and observed values on average is on level with or higher than the natural variation (StDev) in observations. However, applying the assessment scale for CF suggested by Radach & Moll (2006), models for all water body groups can be characterized as "good" to "very good". Less that "good" scores were found only on two occasions; PO_4-P in Type 3 water bodies and for K_d in the Type 5 water body (*Table 7.6*).

Table 7.6. Water quality skill evaluation of aquatic ecosystem models applied to 5 different water body types. Assessment based on regression coefficient (R^2) and Cost Function (CF). Background colour indicates assessment level: dark green: very good model; light green: good/moderate; yellow: reasonable-low.

Water body type	**Chlorophyll-*a***		**NO_x**		**PO_4-P**		**K_d**	
	R^2	**CF**	**R^2**	**CF**	**R^2**	**CF**	**R^2**	**CF**
No category	0.68	1.41	0.84	0.77	0.95	0.98	0.78	1.05
Type 5: Sluice Fjord	0.90	1.74	0.78	1.07	0.68	0.58	0.66	2.68
Type 3: Estuarine	0.89	1.66	0.89	1.28	0.98	2.05	0.40	1.34
Type 2: Estuarine	0.79	1.58	0.98	0.89	0.49	1.01	0.70	1.43
Type 1: Open waters	0.54	0.98	0.86	0.99	0.75	1.72	0.07	1.77

7.4.3 Primary production and retention

Assessment of model skills for simulation primary production has been carried out slightly different compared to nutrients, chlorophyll-*a* and K_d because number of monitoring stations and number of observations are much lower.

The skill assessment of primary production builds on data from 6 stations in Type 1 water bodies (including 2 station located in MSFD water bodies, and hence, not included in the overall evaluation), 3 stations representing the Limfjord model (Type 3 water bodies and No category), 2 stations representing the Roskilde Fjord (Type 2 and 3 water bodies) model and 1 station representing the Odense Fjord model (Type 3 water body). Besides, the sampling frequency varies substantially over the period from 2002 to 2011, and only few of the 12 stations have measurement covering the entire period.

As shown in *Figure 7.7* the modelled primary production clearly track the seasonal variation in measurements with peak in daily production in late summer when water temperature is highest and nutrient turn-over at its maximum. Averaged over 12 stations, months and years, the model underestimates the "measured" primary production by 18%. In the Danish monitoring programme, primary production is estimated from short-term (2 h) ^{14}C-assimilation incubations carried out in a light gradient, and subsequent integration of rates over depth and daytime. The modelled primary production is calculated from the summed growth of phytoplankton, and thus represents net production. In contrast, it is unclear and has been debated for more than 50 years, whether the ^{14}C method measures gross production (respiration not subtracted), net production, or something in-between (Nielsen 1955; Barber et al. 2002; Marra 2009). Hence, the modelled primary production most likely will be lower than the measured production, but we would not expect a deviation larger than 10%.

Month 5 and 6 display the largest deviations between observations and model. That corresponds to the months where we see the largest deviations between observed and modelled NO_x, see *Appendix C*. Hence, this also points to a lower modelled production in the late spring compared to observations. The deviations in primary production in not reflected in chlorophyll-*a* and K_d.

Based on regression coefficients, models score higher than based on CF indicating that models are very good to simulate seasonal variation in primary production. The Cost Function value suggests lower model skill than the R^2-value, but is still rate as a good model, see *Table 7.7*.

Table 7.7. Primary production skill evaluation of aquatic ecosystem models applied. Assessment based on regression coefficient (R^2) and Cost Function (CF). Background colour indicates assessment level: dark green: very good model; light green: good/moderate.

	Primary production	
	R^2	CF
12 stations	0.95	1.29

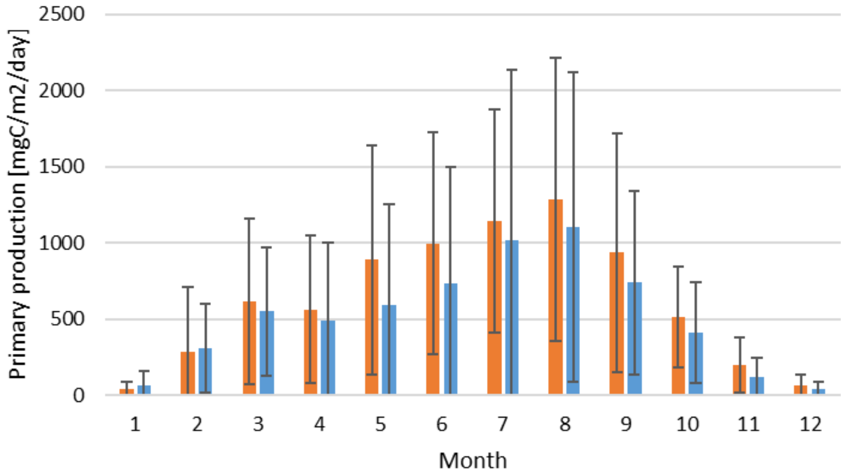


Figure 7.7. Average monthly measured (orange bars) and modelled (blue bars) primary production. Error bars included represent one standard deviation. The evaluation covers the years 2002-2011.

Finally, we have evaluated the modelled N- and P-retentions within the different water bodies covered by models. The modelled retentions (pelagic and benthic retention lumped) include denitrification and immobilization (burial in sediments) and for P immobilization/burial.

At local and regional scales, denitrification removes N that would otherwise be available for primary production or microbial assimilation. In oligotrophic systems, denitrification contributes to N limitation by further decreasing N concentrations and by reducing the N:P ratio of inorganic nutrients. In systems highly enriched with N from anthropogenic sources, removal of fixed N by denitrification reduces the export of N, and thus reduces eutrophication of downstream ecosystems (Seitzinger et al. 2006).

Modelled retentions for the 5 different water body types are listed in *Table 7.8*. Local N- and P-retention is governed by a number of different factors including loadings, water depth, water exchange (freshwater residence time), oxygen conditions etc. Seitzinger et al. (2006) suggest retentions between 5.6 to 8.4 gN m^{-2} $year^{-1}$ averaged over a number of estuaries and continental shelfs. Similarly, Lomstein & Blackburn (1992) reported retention 5.0 gN m^{-2} $year^{-1}$ in Aarhus Bay, and Nielsen et al. (1995) estimated 4.4 gN m^{-2} $year^{-1}$ in Norsminde Fjord. The data from Seitzinger et al. (2006) should be compared to Type 1 and Type 2 in *Table 7.8* whereas Aarhus Bay and Norsminde Fjord "belong" to Type 2 and Type 3 water bodies. For Randers Fjord, Nielsen et al. (2001) found retentions of 18 gN m^{-2} $year^{-1}$. Randers Fjord is highly affected by fresh water and with year-around occurrence of NO_3-N in the mg l^{-1} range and in that sense similar to Hjarbæk Fjord the Type 5 water body in *Table 7.8*. Finally, Ærtebjerg et al. (2004) reported N-retentions in Limfjorden, Odense Fjord and Roskilde Fjord of 6.6, 3.6 and 8.6 gN m^{-2} $year^{-1}$, these estuaries representing Type 2 and 3 water bodies.

Overall, we conclude that the modelled N-retentions are very similar to the data previously reported for the different water body types in Denmark.

Larger variations are found for P-retention, because P-retentions also strongly depend on past history of loads, the present P-pool in the sediment, variation in Fe-content in sediments and occasional P-release during anoxic events (Cloern 2001). For Randers Fjord Nielsen et al. (2001) estimated retention at 0.6 gP m^{-2} $year^{-1}$, and in Ærtebjerg et al. (2004) P-retentions in the

Limfjord, Odense Fjord and Roskilde Fjord of 0.21, -0.18 and 0.44 gP m^{-2} $year^{-1}$ were reported.

Overall, the modelled P-retentions are larger than previous estimates which may be due to reduction in the mobile pool in sediment due to burial or "wash-out" of previously excess concentration originating from input from sewer plants.

Table 7.8. Modelled N- and P-retention within the different water bodies types included in the mechanistic models. Values represent yearly retention for the period 2002-2011 averaged over water bodies (Type 1-3). Values in brackets represent one StDev.

	No category	**Type 5 Sluice Fjord**	**Type 3 Estuarine**	**Type 2 Estuarine**	**Type 1 Open waters**
Number of water bodies included	1	1	4	12	27
Average N retention [gN m^{-2} $year^{-1}$]	2.3 (-)	17.3 (-)	5.0 (3.5)	4.5 (1.6)	6.9 (4.1)
Average P retention [gP m^{-2} $year^{-1}$]	0.01 (-)	0.71 (-)	0.36 (0.84)	0.65 (0.25)	0.83 (0.48)

8 Model application

In the previous two sections, the developed statistical and mechanistic models are presented. The main purpose of the models are to support the implementation of the WFD in Denmark by providing tools for quantification of the maximum allowable nutrient input (MAI) for each single Danish water body. In addition, it was requested to develop a methodology for establishment of chlorophyll-*a* target values for all water bodies within the inner Danish waters (from the Skagen-Gothenburg transect and southwards) applying the developed models.

This section is about the methods developed for application of the models. Firstly, the application for setting chlorophyll-*a* target is documented. Secondly, the specific methods adopted for the statistical and mechanistic models are presented followed by the approach for meta modelling. The final sections document the ensemble approach merging the model results and transposing them into the Danish MAI supplemented by assessments of sensitivity and uncertainty.

8.1 Chlorophyll-*a* reference and corresponding target values

According to the WFD, good ecological status (GES) is defined as "The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions" (Annex V, Directive 2000/60/EC). Hence, GES is defined relative to a reference condition, which describes a situation with no or only minor disturbance from human activity. The establishment of reliable reference conditions and determination of "acceptable deviation" from the reference condition for all WFD water bodies is then essential for defining environmental targets and corresponding effort needed to obtain these targets. Based on WFD guidelines, the reference condition should be determined for each type of water body either from i) observations from existing undisturbed sites ii) historical data, iii) modelling or iiii) expert judgement in prioritized order (Guidance Document No. 5). All of these approaches have been used to develop reference values for several biological elements in different marine waters throughout Europe (Basset et al. 2013; Borja et al. 2012; Muxika et al. 2007; Bennion et al. 2004; Krause-Jensen et al. 2005; Schernewski et al. 2015). The meaning and quantification of "slight deviation", which is essential for the definition of GES, is considered as part of the intercalibration exercise. This exercise is performed by the EC member states with the aim to ensure consistency and comparability of boundary values between the classes of high and good status and between good and moderate status (Guidance Document No. 14 2005).

As described in section 2.2.1, one of the three indicators applied in the Danish WFD ecological status assessment is chlorophyll-*a* being an indicator of phytoplankton biomass. Phytoplankton is one of three biological quality elements to be used for assessing the ecological status in coastal waters according to the WFD. Although several sub-elements (phytoplankton composition, abundance and biomass) are described in the WFD, at present only chlorophyll-*a* concentration has been part of the WFD phytoplankton intercalibration (i.e. target harmonizing) process in the Baltic Sea region. The chlorophyll-*a* indicator used in the Baltic Sea region (south-western Baltic

Sea) is defined as the average chlorophyll-*a* concentration from May to September, and the intercalibration process resulted in a common agreement of an acceptable deviation from the reference condition, measured as an Environmental Quality Ratio (EQR) (see Bek. no. 1001 2016).

For the North Sea, the chlorophyll-*a* EQR and target values have been intercalibrated with Germany for the Wadden Sea and these values have been extrapolated to national values covering the west coast of Jutland and Skagerrak. For the North Sea water bodies, the chlorophyll-*a* indicator is based on the 90-percentile of the March to September chlorophyll-*a* concentrations (Bek. no.1001 2016). For the present work, values for the chlorophyll-*a* indicator for the North Sea region is provided by Danish EPA, and no further evaluation of that indicator is included in this report.

For the Danish Baltic Sea region, i.e. inner Danish waters, the present project has developed a methodology for establishing chlorophyll-*a* reference conditions and corresponding WFD target values applicable to all Danish WFD water bodies located south of Skagen. The methodology includes model estimation of a reference condition and subsequent transformation to GES values using the intercalibrated EQR. The estimation of reference conditions utilize the statistical and mechanistic models described in chapter 6 and 7.

Modelling of a reference situation is subjected to uncertainties related to both model quality and to the extensive but necessary model extrapolation, i.e. the hind-cast ability. In order to reduce (some of) the uncertainties, we have applied a typological approach where site-specific model results were used to establish robust type-specific reference and target values transferable to Danish water bodies. Whenever possible, we have applied a model ensemble approach where hind-cast results from two independent models (statistical and mechanistic) were used in order to reduce the impact of potential model bias. Based on previous work on WFD implementation in Denmark, the year 1900 was chosen as the historical reference condition. This reference period is mainly founded on historical observations documenting that light penetration and eelgrass depth distribution was still high during this time period (Krause-Jensen and Rasmussen, 2009; Henriksen, 2009) indicating little influence of anthropogenic activity. Further, this period coincide with periods used as reference in Germany (around year 1880, Schernewski et al. 2015) and in the BSAP (Gustafsson et al. 2012; HELCOM 2013a).

8.1.1 Applied typology

In order to obtain robust estimates of the chlorophyll-*a* reference (year 1900) concentrations we use the type-specific approach where water bodies of the same type are given the same chlorophyll-*a* reference concentration. This approach is in accordance with the WFD guidelines stating that "the purpose of typology is to enable type specific reference conditions to be established" (Guidance Document No. 5 2003). The applied typology is presented in section 3.2. Briefly, the typology is a modified version of the typology for Danish waters developed by Dahl et al. 2005. It contains 3 estuarine types, a sluice type and an open water type. The estuarine types are categorized according to salinity and degree of freshwater influence (expressed by the F-index) taking into account the freshwater runoff and residence time. These factors are assumed to be useful proxies for nutrient availability and thus important drivers for chlorophyll-*a* concentration even under reference conditions.

Two estuarine water bodies (the inner part of Mariager Fjord and the southern part of Limfjorden) are currently affected by anoxia, resulting in additional release of nutrients from the sediments with implications for summer chlorophyll-*a* concentrations. Although the anoxic area has increased due to eutrophication and increased temperatures (Bendtsen and Hansen 2013), historic measurements suggest that anoxia could be a natural occurring phenomena in these areas (Christensen et al. 2004) potentially affecting chlorophyll-*a* concentrations even under reference conditions. Therefore, estuaries with suspected significant "natural hypoxia" have not been categorised according to the typology.

As described in section 3.2, one type of water body is the open water type. For the study of chlorophyll-*a* reference and GES values, this water body type is sub-divided based on knowledge of freshwater influence as well as hydrographic conditions (currents, upwelling, exposure and salinity). As the importance of upwelling to availability of inorganic nutrients as well as the influence from the North Sea and Baltic Sea waters (concentrations of dissolved organic matter (DOM), transparency etc.), respectively, varies in the open waters, reference chlorophyll-*a* is assumed to differ between those different open water bodies. The categories used for the open waters is described in *Table 8.1*.

Table 8.1. Open water typology.

Type	Description
1.1	Open seas, coasts and bays largely affected by Baltic Sea water, salinities ranging between 5-18 psu
1.2	Open seas, coasts and bays protected from main fluxes out of the Baltic Sea, generally less affected by Baltic Sea water, salinities ranging between 5-18 psu
1.3	Open seas, coasts and bays largely affected by North Sea water, some direct freshwater discharges and salinities ranging between 18-30 psu
1.4	Open seas, coasts and bays facing east and affected by upwelling and North Sea water, salinities ranging between 18-30 psu

8.1.2 Statistical modelling

Statistical models used to calculate chlorophyll-*a* concentration in a reference situation was developed as described in chapter 6. Briefly, chlorophyll-*a* concentration from May to September was chosen as response variable and estimated based on monitoring data from 1990 to 2012. The bulk suite of explanatory variables consisted of site-specific estimations of nutrient (N and P) loading, freshwater discharge, solar radiation, temperature, salinity, buoyancy and wind[11]. Explanatory variables for each coastal site were selected using MLR and PLS regression and the final site-specific models were used to simulate the summer chlorophyll-*a* concentration in a situation with year 1900 N loadings.

8.1.3 Mechanistic modelling

The mechanistic models used to calculate chlorophyll-*a* concentration in a reference situation are described in chapter 7. These models entail the inner Danish water model (IDW) and the three estuary models Odense Fjord, Roskilde Fjord and the Limfjorden. In brief, these models (IDW, Odense

[11] For the statistical models, only N and P loadings have been changed to reference conditions. Remaining explanatory variables are similar to the model development (1990-2012).

Fjord, Roskilde Fjord and the Limfjorden models) were forced with reference N and P loadings, reference boundaries and reference N depositions to account for reference conditions. In addition, the N and P sediment pools were also adjusted for the IDW model. Model forcings (other than N and P loadings, reference boundaries and reference N depositions) were identical to the model development (2002-2011), meaning that meteorological and physical forcings are identical to the present day (status) modelling. Chlorophyll-*a* data was extracted for the last 5 years.

8.1.4 Nutrient loadings under reference conditions (year 1900)

As described earlier, the statistical modelling applies the Danish land-based nutrient loadings as explanatory variable while mechanistic modelling requires additional data on loading from other countries around the greater Baltic Sea (including Norway) and atmospheric deposition. Furthermore, the boundary conditions have to imitate the reference condition.

Danish land-based reference nutrient loadings

In order to simulate the chlorophyll-*a* concentration in year 1900, both the statistical and mechanistic models have to be forced with year 1900 nutrient loadings from Danish catchments. Since data on nutrient concentrations from around year 1900 are very scares and unsuitable for this analysis, year 1900 nutrient loadings were estimated from a) background concentrations of TN, TP, dissolved nitrogen and dissolved phosphorus and b) present day's freshwater discharges. The reference riverine nutrient concentrations were provided by Danish EPA via DCE/AU, Department of Bioscience and the methodology is described in Bøgestrand et al. (2014b). Briefly, Bøgestrand et al. (2014b) used present background loadings as proxy for nutrient loadings around year 1900. To estimate the background loadings, stream monitoring data from catchments with low anthropogenic influence were transposed into area weighted average values of the nitrogen concentration in streams. For nitrogen concentration it was possible to establish average concentrations on the scale of water bodies, whereas larger catchments (geo-regions) were applied for phosphorus.

An overview of the reference loadings as well as present day (2007-2012) loadings to the water bodies in the inner Danish waters is shown in *Figure 8.1*. As can be seen from this figure, reference N and P loads are generally significantly lower than present day loadings expressed both as total N and total P loading per year and per area. The accumulated N loadings in a reference condition calculates at 17 kton N year^{-1} from all Danish catchments and 12 kton N year^{-1} when only considering loadings to inner Danish waters. This should be compared to the present day loadings of 61.2 kton N year^{-1} and 43.7 kton N year^{-1}, respectively.

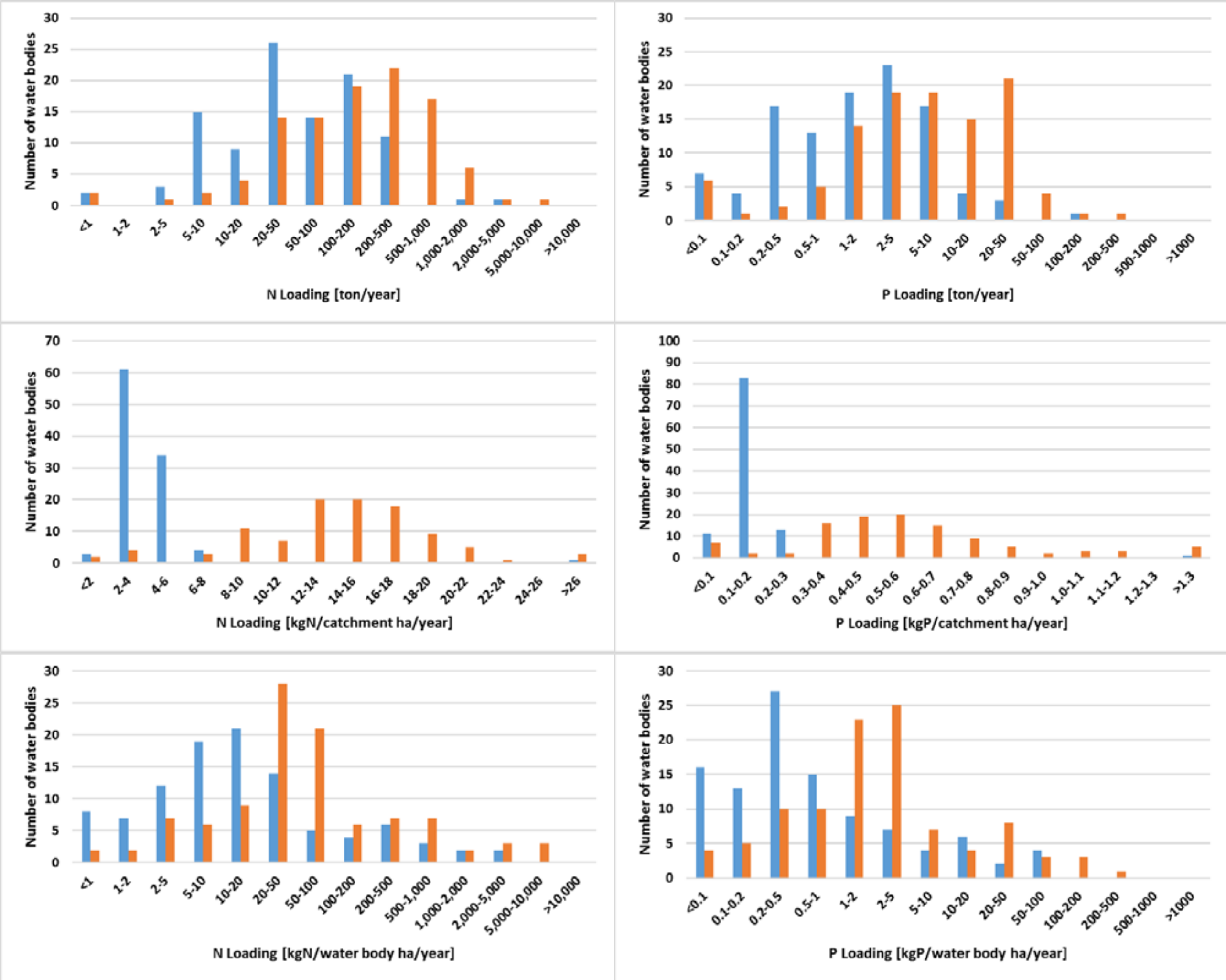


Figure 8.1. Statistics based on average yearly nutrient loadings to each of the 119 Danish WFD water bodies under reference (year 1900; blue columns) and present (2007-2011; orange columns) conditions. The water bodies are divided according to the grouping indicated by the x-axes. Left vertical panel shows N loadings and right panels P loadings. Top panel presents the distribution according to loadings per water body, middle panel according to loadings per Danish catchment area, and bottom panel according to loadings per water body area

Compared to the present situation, loadings per catchment area show limited variation in the reference situation (most loads are within 4-6 kg N ha^{-1} $year^{-1}$). Reference loadings per water body area do on the other hand have the same extent as present loadings but with the majority of the water bodies having loads less than 50 kg N ha^{-1} $year^{-1}$ while most water bodies show higher loads under present conditions.

For the mechanistic model development, the nutrient loadings need to be split into different nutrient species, see chapter 7. For the reference scenario, we have adopted the same ratios between the different species as developed for the present day modelling. Most likely, the ratios between e.g. NO_x, PON and DON have changed due to changes in land-use, crops etc. However, we do not have data to support different ratios than those developed for the present day modelling. The data were provided as uniform concentrations (no seasonal variations) for the different water body catchments. From present day analysis, seasonal differences in concentrations exist over the entire study area, but no information exists on the seasonality in a reference situation why we keep the concentrations constant over the year.

Baltic Sea reference nutrient loadings

The mechanistic IDW model is in addition to the Danish loads forced with nutrient loadings from the non-Danish Baltic Sea catchments. The model development is based on HELCOM's Baltic Sea Pollution Load Compilation (HELCOM 2011), but this data set does not include reference loadings. Instead the Baltic Sea nutrient loadings from 1850-2006 reconstructed by Gustafsson et al. (2012) and Savchuk et al. (2012b) were used relatively together with the HELCOM (2011) data. The relative difference within the different sub-basins defined in Gustafsson et al. (2012) and Savchuk et al. (2012b) between the present day (average 2000-2006) and historical (average 1890-1910) data from the reconstruction, see example in *Figure 8.2,* were inflicted on the HELCOM (2011) data to provide a reference loading dataset applicable in the mechanistic model.

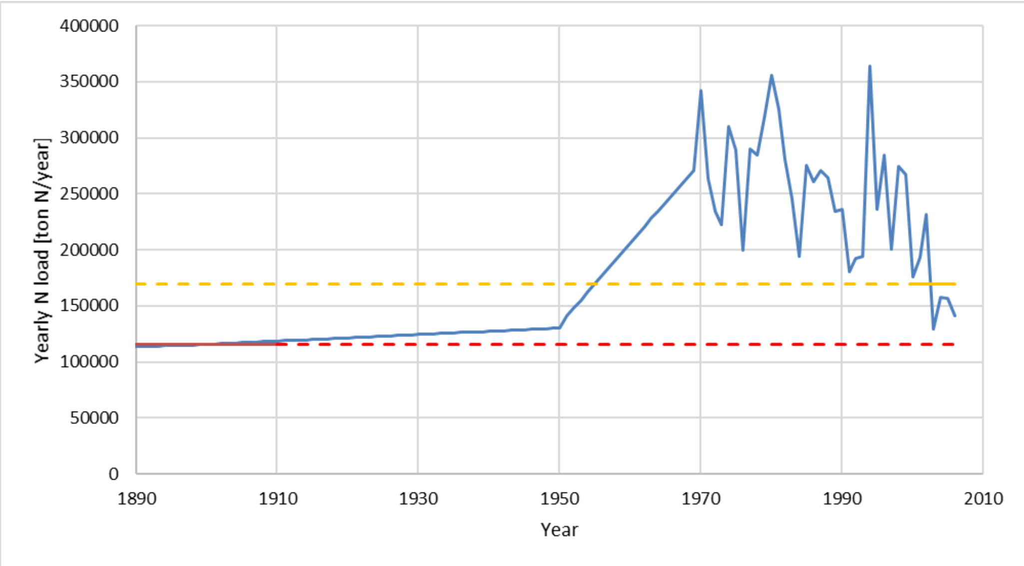


Figure 8.2. Yearly riverine N loadings to the Gotland Basin (basin no. 9), modified from Savchuk et al. (2012a). Red line indicates average 1890-1910 N loadings whereas orange line indicates average 2000-2006 N loadings.

Atmospheric reference N deposition

Model data on atmospheric nitrogen deposition under reference conditions (i.e. around year 1900) are provided by AU, Department of Environmental Science. The year 1900 simulation is conducted with an atmospheric model describing transport, chemical reactions and deposition of various chemical species including NO_X and NH_4 (Geels et al, 2012). The atmospheric model is forced with historical emissions provided by IIASA, "Representative Concentration Pathways" (RCPs; from http://tntcat.iiasa.ac.at:8787/RcpDb/dsd?Action=htmlpage&page=welcome) while the meteorological forcing corresponds to present days (2002-2011). Hence, the latter is coherent with the mechanistic modelling meteorological forcings (see below). The resulting N deposition data is provided as monthly means (from a 10-year simulation) with a spatial resolution of 5 × 5 km^2 and used without any post processing in the marine mechanistic models. More detailed descriptions can be found in Geels et al. (2012) and Ellermann et al. (2013).

Marine reference boundary conditions

All the mechanistic models have to be forced by proper boundary conditions describing the state variables at the boundary of the model domain. The IDW model has one open boundary located in the central part of Skagerrak.

No historical observations exist for this boundary, why the reference boundary conditions are deduced based on the literature. Since the 1950ies, the nutrient concentrations in the central part of the North Sea and in the surface waters of Skagerrak have been stable (Radach & Patsch 1997), even though the nutrient concentrations in the rivers discharging to the German Bight have been reduced by approximately a factor of 4 (Topcu et al. 2011). This indicates that the reference concentrations in Skagerrak have not been significantly different from today's concentrations. This is supported by model results from Savchuk et al. (2008) who modelled the historical Baltic Sea concentrations and estimated reference concentrations equivalent to 85% of today's (2007-2011) at the Skagerrak boundary. All boundary concentrations including C, N or P have therefore accordingly been reduced by 15%.

Boundaries for the estuary models are based on reference condition results from the IDW model by reducing the measured present day's boundary conditions with a factor derived from the relative difference between status and reference model results adjacent to the estuaries.

8.1.5 Initial conditions (sediment nutrient pools)

The methods for defining the reference sediment pools of nutrients differ between the IDW model and the three estuary models due to differences in residence time. Compared to the open waters, and especially the deeper parts of the Baltic Sea, the sediments in the three estuaries respond much faster due to a much shorter residence time. Where the Baltic Sea is known to have a residence time of approximately 30 years (Wulff et al. 1990), the residence time in Odense Fjord, the Limfjorden and Roskilde Fjord is < 1year (Kuusemäe et al. 2016). Hence, determination of the estuarine sediment pools in a reference situation is done using the same 10-year simulation period as for the status model runs, but where the models are forced with reference loadings (land-based, atmospheric depositions and boundaries).

The model results (not shown) reveal that the largest reduction in sediment pools (top 10 cm) occur within the first 5 years of the 10-year model period. After 5 years, the sediments almost reach a new equilibrium (steady-state) and re-running an additional 10 year period does not change the pools. Correspondingly, after the first 5 years, the last 5 years chlorophyll-*a* results do not change significantly, why one 10-year model period is assumed as appropriate for defining reference chlorophyll-*a* concentrations.

For parts of the open Danish waters and the Baltic Sea sediment pools of organic bound nutrients are based on analysis of historical sediment concentrations (Almroth & Skogen 2010; Andersen et al. 2011; Carman & Cederwall 2001; Savchuk et al. 2008). Based on the analyses the present day sediment pools are reduced using the following reduction key: organic N is reduced by 55%, organic C is reduced by 34%, organic P is reduced by 34% and the iron-bound P is reduced by 34%, to mimic the pools around year 1900.

8.1.6 Calculating target values from reference values

Setting the value for the border between the good and moderate classes (the GM target) is one of the most critical issues in the implementation of the WFD as this border determines whether management action is necessary (Guidance doc 5). Hence, besides determination of the reference condition for the indicators, the ecological quality ratio (EQR, Guidance doc 14) is paramount to set this border. For the Baltic Sea, the intercalibration process has defined an EQR for the good-moderate boundary at 0.6 (see *Table 8.2*; Bek. 1001 2016) following this EQR definition for the boundary between class *i* and *j*:

$$EQR_{chl}^{i,j} = \frac{Reference\ value\ for\ chlorofyll{-}a}{Target\ i,j\ chlorophyll{-}a\ concentration} \quad \text{Eq. 8.1}$$

The EQR values for the 4 boundaries separating the 5 classes are listed in *Table 8.2*.

Table 8.2. EQR values for the applied summer chlorophyll-*a* indicator (Bek. 1001 2016). The EQR values define the borders between the WFD classes.

Boundaries	High-Good	Good-Moderate	Moderate-Poor	Poor-Bad
EQR value	0.8*	0.6*	0.4	0.2

* EQR values for High-Good and Good-Moderate borders have been intercalibrated.

To establish the required basis for estimation of MAI, the approach in this study was therefore to set the good-moderate target values for the chlorophyll-*a* indicator in the different types of Danish water bodies by combining the intercalibrated EQR value of 0.6 with the estimated type-specific reference conditions as:

$$Target\ value_{chlorophyll-a\ indicator} = \frac{\text{Reference value for chlorofyll-}a}{0.6} \quad \text{Eq. 8.2}$$

Similarly, chlorophyll-*a* concentrations at the remaining borders can be calculated using the corresponding EQR values stipulated in *Table 8.2*.

8.1.7 Changes in N input since year 1900

Nitrogen loading from Danish catchments discharging to the inner Danish waters (south of Skagen) in year 1900 was estimated to 12 kton year^{-1}, indicating an increase in nitrogen loading to the current (2007-2012) loading of 44 kton year^{-1} of nearly a factor of 4 and an increase of more than a factor of 5 to the nitrogen loadings in the 1990 (*Figure 8.3*). The overall national increase in nitrogen loadings since year 1900 does, however, include huge variations between individual water bodies. N loadings to Limfjorden, the largest estuary in Denmark, have increased by more than a factor of 6 between year 1900 and the 1990s whereas the increase for Randers Fjord, the third largest catchment, is between 3-4 fold. The increase in the Northern Sound show a 35-fold increase from 1900 to 1990 due to the waste water outlet from Copenhagen, but the introduction of waste water treatment in Copenhagen has reduced this load by 80% still leaving this water body with a present days load almost 8 times the load in a reference situation.

Also, the atmospheric N deposition to Danish coastal waters as well as nitrogen loading to the Baltic Sea has increased since year 1900. On average, the atmospheric N deposition to the Danish waters has increased by a factor of 4 since year 1900 whereas the nitrogen loading to the different basins in the Baltic Sea (from Bornholm Basin in eastwards) has increased with factors between 2.1 (Gulf of Riga) and 1.4 (Bothnian Bay).

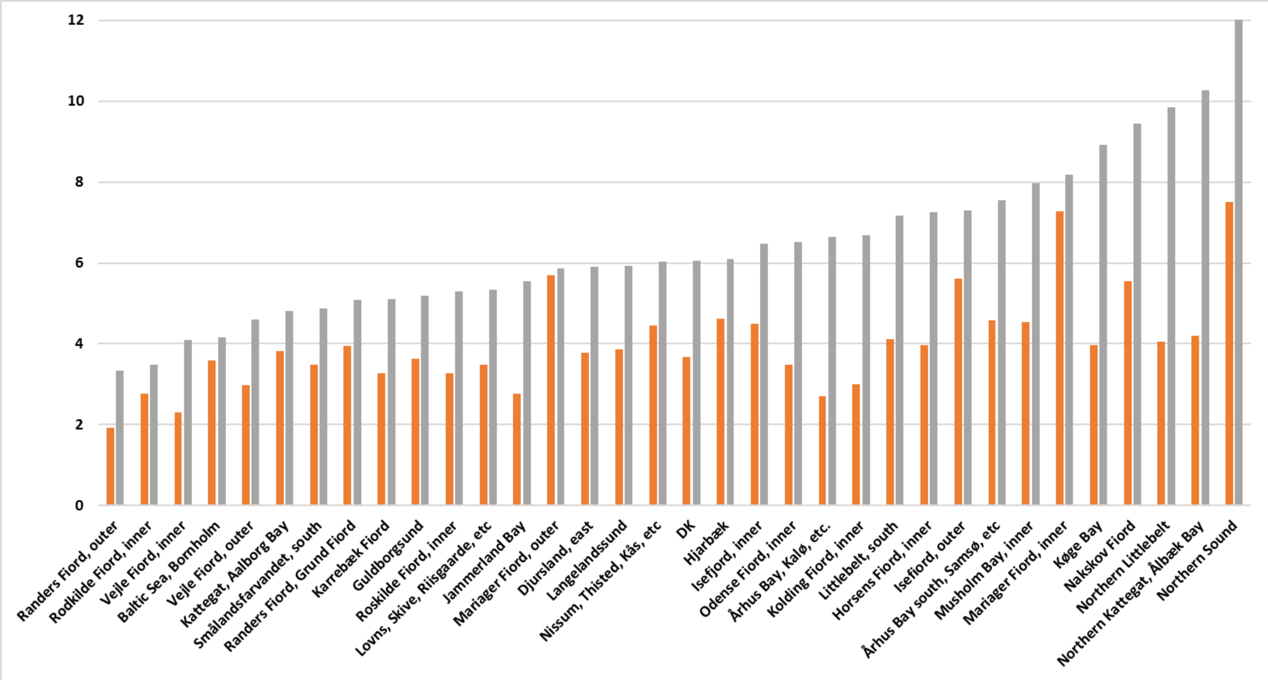


Figure 8.3. Relative increase in nitrogen loadings (catchments accounting for 80% of present day land-based N loading) to Danish water bodies from reference load (approx. year 1900) to present day (2007-2012) loadings (orange) and the 1990s (grey).The y-axis has been cut at a factor of 12, but for the Northern Sound the factor is 35 from reference loading to the loadings in 1990.

The models used in this project have been developed using loadings and corresponding chlorophyll-*a* values from a eutrophic time period and the increase in total nitrogen input from both Danish catchments, atmospheric deposition and the Baltic Sea since the reference period (year 1900) induces high uncertainty in the year 1900 model predictions. In order to reduce the uncertainty and make the year 1900 predictions more robust, we have applied an ensemble modelling approach for the estuarine water types combined with a typology approach for both estuarine and open water types.

8.1.8 Estuarine type-specific reference and target values

For both estuarine and open water types a typology approach was used to establish chlorophyll-*a* reference concentrations. For the estuarine types we also applied an ensemble modelling approach involving results from statistic and mechanistic modelling to further increase robustness of the estimates and reduce the influence of potential model bias.

Ensemble modelling of reference chlorophyll-*a* concentration was possible for water bodies belonging to Type 2 (estuaries with low freshwater influence) and Type 3 (estuaries with high freshwater influence). The resulting type-specific reference and good-moderate target value for Type 2 water bodies was estimated to 1.3 µg l^{-1} (reference) and 2.1 µg l^{-1} (good-moderate target), respectively (*Table 8.3*). For the Type 3 water bodies, the corresponding values were 2.2 µg l^{-1} (reference) and 3.6 µg l^{-1} (GM target).

Table 8.3. Ensemble modelled chlorophyll-*a* reference and good-moderate target (GM target) values for Type 2 and 3 water bodies. Site-specific reference values are estimated using a mechanistic model (mech) or a statistic model (stat), respectively. The values are rounded to one decimal. WB = water body.

WB name	WB number	WB Type	Site-specific reference value		Type-specific reference value [µg l^{-1}]	Type-specific GM target value [µg l^{-1}]
			Mech	Stat		
Isefjord, outer	165	2		2.0	1.3	2.1
Roskilde Fjord, outer	1	2	1.0			
Åbenrå Fjord	102	2	1.0	1.1		
Århus Bugt	147	2	1.0	1.3		
Roskilde Fjord, inner	2	3	2.5	2.2	2.2	3.6
Odense Fjord, Outer	92	3	1.3	4.1		
Vejle Fjord	123	3	1.4	1.7		
Løgstør Broad	156	3	1.8	2.4		

One water body belonging to Type 3 was not included in the setting of Type 3 reference and GM target value although models were developed for this water body. The water body no. 157, Bjørnsholm Bugt-Lovns Bredning-Skive Fjord, located in the southern part of the Limfjorden, has probably always been affected by hypoxia (Christensen et al. 2004) and it is likely that also under reference conditions not only riverine nutrient inputs but also hypoxia induced nutrient release from the sediments affects the chlorophyll-*a* concentration. This water body was therefore analysed separately, using ensemble modelling. The estimated reference and good-moderate target values are presented in *Table 8.4*.

Table 8.4. Ensemble modelled chlorophyll-*a* reference and good-moderate target (GM target) value for water body no.157 in the southern part of the Limfjorden. Site-specific reference values are estimated using a mechanistic model (mech) or a statistic model (stat), respectively. WB = water body.

WB name	WB number	WB Type	Site-specific reference value		Ensemble reference value [µg l^{-1}]	GM target value [µg l^{-1}]
			Mech	Stat		
Bjørnholms Bugt, Lovns Bredning, Skive Fjord	157	3	4.2	2.9	3.5	6

Also, the inner part of Mariager Fjord was not considered as part of the typology due to suspected occurrence of anoxia even in a reference situation. However, chlorophyll-*a* models were not developed for this water body (do to faulty chlorophyll-*a* data), hence site-specific reference and GM target values were not established and the estimated reference and GM value for the southern part of Limfjorden was tentatively applied for this water body.

For Type 4 (water bodies with low salinity and high freshwater influence) it has not been possible to estimate type-specific reference values due to lack of available ensemble models for the water bodies belonging to this category. In addition, the two water bodies belonging to the sluice-type (Type 5) are quite different with respect to salinity, freshwater influence, depth etc. Therefore, individual site-specific values are defined for these water bodies using the type of models available. The derived site-specific reference and

GM target values for the two sluice fjords, Ringkøbing Fjord and Hjarbæk Fjord, and the type of model applied can be seen in *Table 8.5*.

Table 8.5. Model estimated chlorophyll-*a* reference and good-moderate target (GM target) value for the sluice water bodies (Type 5). Site-specific reference values are estimated using either a mechanistic model (mech) or a statistic model (stat). Wb = water body.

WB name	WB number	WB Type	Site-specific reference value		Site-specific GM target value	
			Mech	Stat	Mech	Stat
Ringkøbing Fjord	132	sluice		4.7		8
Hjarbæk Fjord	158	sluice	5.1		9	

The ensemble modelled reference chlorophyll-*a* concentrations were tested with a two-way ANOVA and we did not find a significant difference between the two modelling approaches ($p>0.05$) even though the least squared (LS) mean of the mechanistic models were a bit lower than the LS mean of the statistical models. Further, we did not find any interaction between model and typology. There was, however, a significant difference between Type 2 and 3 (the only typologies that were tested) as they have a LS mean at 1.3 ± 0.3 µg chlorophyll-*a* l^{-1} (LS.mean ± SEM) and 2.1 ± 0.3 µg chlorophyll-*a* l^{-1} (LS.mean ± SEM), respectively.

Open water type-specific reference and target values

For the open waters, the type-specific reference and GM target values are based on the results from the mechanistic model covering the inner Danish waters (IDW). As described earlier the open waters water bodies were subdivided into four categories. For each category, the reference value is estimated as the arithmetic mean of the values for each water body belonging to it and subsequently transformed into GM target value using the EQR of 0.6, see *Table 8.6*.

Table 8.6. GM target chlorophyll-*a* values for Type 1, open waters. The target values are solely based on mechanistic modelling. WB = water body.

WB name	WB number	WB Type	Site-specific reference value	Site-specific GM target value	Type-specific GM target value [µg l^{-1}]
Northern Sound	6	1.1	1.1	1.8	1.7
Hjelm Bay	44		1.0	1.6	
Køge Bay	201		1.1	1.8	
Fakse Bay	46		1.0	1.6	
Baltic Sea, Bornholm	56		1.0	1.7	
Great Belt, SW	95		1.1	1.8	
Langelands Belt, east	41	1.2	0.9	1.5	1.5
Femer Belt	208		0.9	1.5	
Grønsund	45		0.7	1.2	
Langelandssund	90		1.0	1.6	
Smålandsfarvandet, open part	206		0.8	1.4	
South Funen Sea, open part	214		0.8	1.3	
Little Belt, south	216		0.9	1.5	
Litte Belt, Bredningen	217		0.9	1.5	
Kattegat, Northern Zealand > 20m	205		0.9	1.5	1.6
Anholt	139		0.8	1.4	
Kattegat, Læsø	154		0.8	1.4	
Northern Little Belt	224		1.0	1.6	
Kattegat, Northern Zealand	200		1.0	1.6	
Sejerøbugt	28		0.9	1.5	
Hevring Bay	138		1.0	1.7	
Århus Bay, south, Samsø and northern Belt Sea	219		1.0	1.7	
Kattegat, Aalborg Bay	222		1.0	1.7	
Northern Kattegat - Ålbæk Bay	225		1.1	1.9	1.9
Djursland east	140		1.1	1.8	
Great Belt, NW	96		1.1	1.9	

8.1.9 Discussion

The ambitious objective of the WFD is that European waters hold at least good ecological status (GES), i.e. aquatic ecosystems are deviating only slightly from undisturbed conditions. Since the WFD adaptation by the EU in 2000, managers and scientists around Europe have been struggling to transform the political intentions and normative definitions to quantitative goals and operational managerial frameworks. One of the main scientific challenges is to establish solid reference conditions reflecting an "undisturbed condition".

The present methodology developed for establishment of reference and GM target values for chlorophyll-*a* in the Inner Danish waters relies on statistical and mechanistic models (applied to estuaries, and estuaries and open waters, respectively). According to the WFD guidance (Guidance Document No. 5 2003), the reference conditions should be determined for each type of water body either from i) observations from existing undisturbed sites, ii) historical data, iii) modelling or iv) expert judgement in prioritized order. With respect to the chlorophyll-*a* indicator, neither strategy i) nor ii) is possible. No Danish water bodies can be considered undisturbed and no historical observations exist to support determination of reference chlorophyll-*a* values. Expert judgement should be the last option to choose, since it is exposed to subjectivity, and very hard to quantify and to document (i.e. to secure transparency). Hence, the most feasible way to establish the reference and GM targets is to apply quantitative modelling. Different modelling approaches have been applied to both Danish waters (Carstensen & Henriksen 2009; Henriksen 2009) and other regions of the Baltic Sea area (Schernewski et al. 2015; Gustafsson et al. 2012; Schernewski & Neumann 2005). The present study is the first attempt to define targets for all inner Danish waters – estuaries as well as open waters.

In agreement with the recommendation of Guidance Document No. 5, our establishment of reference values follows a type-specific approach. Site-specific reference values may, however, be preferable since each estuary, bay, lagoon etc. has its own characteristics in terms of e.g. hydrodynamic conditions and distance to nutrient sources influencing not only present day chlorophyll-*a* concentrations, but also most likely results in chlorophyll-*a* reference conditions differing between sites. Modelling of site-specific reference chlorophyll-*a* concentration is, however, restricted by the availability and the quality of hind-cast models, and of historical forcing data. For Danish marine WFD water bodies, availability of suitable site-specific hind-cast models is limited to the models developed during the present project making a type-specific approach preferable.

Although it is not possible to evaluate the "true" variation between sites belonging to the same water body type, the results from the present study do indicate that the variation within each type is significantly smaller than between types. However, for the unassessed water bodies, the water body type may not necessarily be a good representative. The differentiation could likely be increased by adjusting the applied typology, e.g. by taking diversity of freshwater influence, residence time and water exchange more into account and by including physical characteristics important for benthic filtration, which in some situations may influence the pelagic chlorophyll-*a* concentration (Petersen et al. 2008). A more differentiated typology, however, requires an increased amount of data and number of site-specific models for each type in order to obtain sufficient power to estimate robust type-specific reference values.

The type-specific approach is applied to the open water body types as well as two of the estuarine water body types (Type 2 and 3). However, for the sluice estuaries and the estuaries potentially affected by "natural anoxia", site-specific reference values are defined since the estuaries are too distinctive for the typology. At the same time, it is not possible to make ensemble modelling of these waters bodies because "overlapping" models are not available. Despite the higher uncertainty caused by the limited data and model base, it was considered that site-specific model values would provide the best possible estimates of the reference chlorophyll-*a* concentrations for these water bodies.

Model prediction of reference conditions is inherently uncertain for more reasons. The two major (but related) reasons are that the use of models outside the calibration area inevitable induce higher uncertainties and that ignorance of condition prevailing under the more pristine conditions with e.g. more wide spread eelgrass beds providing important ecosystem services may induce a bias in the model predictions.

The use of statistical models outside the calibration range area is problematic due to the lack of explicit description of mechanisms and feedback processes. The statistical models applied in the present study are developed using data from eutrophic conditions. However due to the substantial variation in year-to-year N loadings including "dry years", the models have been evaluated under a wide range of load conditions. Transient low load situations in an otherwise eutrophic situation is however not directly comparable to a more stable low load situation especially due to nutrient pools in the sediments. Hence, considerable uncertainties are associated with modelling of a reference situation using statistical models. The mechanistic models are less sensitive to the extrapolation outside the calibration range as they include mechanistic process descriptions and feedback mechanisms and operate with e.g. reduced sediment pools. However, the mechanistic modelling of a reference situation is also associated with considerable uncertainties. These are mainly related to the model parameterizations and uncertainties in historic input data necessary for mimicking conditions prevailing under more pristine conditions, e.g. the data base behind the implemented reference sediment flux and pore water pools is largely absent.

Extracting model results to simulate a stable reference condition by changing e.g. nutrient loadings will indicate some plausible changes, but the changes might be more abrupt if ecosystems undergo regime shifts (Duarte et al. 2009). Abrupt shift between states might especially occur if shallow waters dominated by pelagic primary production changes into ecosystems dominated by benthic production (Orth et al. 2012; McGlathery et al. 2013; van der Heide et al. 2011). If the recovery trajectory for Danish marine ecosystems includes regime shifts, these will not be captured by the statistical models and probably not captured by the mechanistic models.

Comparison of the reference scenario results of the type of models applied indicate a tendency for the statistic models to predict elevated reference concentrations compared to the mechanistic models. A likely cause for the statistical models to "overshoot" the chlorophyll-*a* reference concentration might be related to the lack of feedback mechanisms especially related to the sediments. Sediment plays an important role as nutrient sources and sinks, and hence, affects the status of all water bodies, although to different extends. When changing nutrient loadings, the interaction with the sediment pools will hamper immediate effect – both when increasing and when decreasing loadings. For some of the deeper parts of the Baltic Sea the time scale of the effects is expected to be at least decades whereas shallow water bodies with low retention time will react within few years.

The statistical models indicate time delays between loadings and nutrient concentrations, which might be attributed to sediment effects. Hence, the sensitivity to land-based loadings might be under-estimated in these models why statistical models might overestimate reference concentrations.

Similarly, the mechanistic models show delayed effects when reducing land-based nutrient loadings significantly, as when applying reference loadings rather than present day loadings. In Odense Fjord and Roskilde Fjord the models have been run for a 10 year period – and then re-run for an additional 10 year period applying the end of first run as initial field for the second run. Estimating the reference concentrations as the average of the last 5 years model results, and evaluating the last 5 years of the first run and the last 5 years of the second run, almost no difference were observed. This indicates that for Odense Fjord and Roskilde Fjord the active sediment layer of the model is adjusted to a new steady-state during the first 5 years. For the inner parts of the Limfjorden, the response time is a bit longer and some minor differences between the two runs is observed, suggesting a new steady-state within 5-6 years.

To account for the long residence time in parts of the Baltic Sea the sediments in the IDW model were adjusted. No re-runs were made due to time constraints, but mass budgets for the sediments revealed that sediment pools did not reach a new steady-state. In the Kattegat area, the manual sediment adjustments resulted in some accumulation during the reference modelling suggesting a too large reduction of nutrient pools in the sediment in the reference modelling.

Despite uncertainties, the model ensemble analysis revealed fairly similar results between the two model approaches and although, as previously described, there was a tendency for the statistical models to predict higher reference values compared to the mechanistic models, we did not detect any significant difference between the model approaches. The predicted reference values showed a clear gradient from the open waters with low freshwater influence towards the coastal and semi-enclosed water bodies closer to the nutrient sources, which is consistent with the expectations.

In order to reduce the influence of model bias, we have used as far as possible an ensemble model approach for the estuarine water body types by combining the statistical model results with the mechanistic model results. The ensemble model approach has previously been used in both marine hindcast (Eilola et al. 2011; Meier et al. 2014; HELCOM 2013) and forecast (Meier et al. 2011; Meier et al. 2012) model studies in order to reduce model bias and provide more reliable results. HELCOM (2013) combined three different mechanistic models forced with estimated year 1900 nutrient loadings. They observed differences between the models, and concluded that the most robust chlorophyll-*a* estimates were achieved using the ensemble approach. In a model evaluation study, Eiola et. al. (2011) conclude that no model shows outstanding performance in all aspects, and that ensemble means provide better or as good results as any of the individual models. In our study, the ensemble approach was applied for two (Type 2 and 3) of the five defined water body types and for the southern part of Limfjorden.

Quality assessments of the inventories of loadings and deposition have not been part of this study as the precondition has been to use data from the national records and HELCOM reporting. Besides the uncertainty related to model capabilities, uncertainties do exist mainly related to input and forcing data. To the mechanistic modelling, this involves nutrient loadings, depositions, boundaries and sediments. The applied N depositions are based on other models and hold the uncertainties associated with these models. The loadings from both Danish catchments and neighbouring Baltic Sea catchments are subject to discussions. This goes for both present loadings, where Savchuk et al. (2012b) suggest modifications to the loadings reported to HELCOM during preparation of PLC-5 (HELCOM 2011), and for Danish reference loadings where some stakeholders have suggested modifications to the loading applied for this study.

According to the WFD (Guidance Document No. 5), historical data are preferable to model results when defining reference conditions and corresponding targets. However, a number of parameters affect the indicators: Changes in wind-patterns impact re-suspension and light regime, changes in precipitation impacts run-off and nutrient loadings, changes in temperature increases nutrient turn-over and respiration rates of e.g. benthic vegetation, etc. Since 1875 precipitation has increased up to 26% (from 800 mm), and water temperature have increased up to 1.3 °C (Grøndahl et al. 2014). Here we apply reference TN and TP concentrations together with present day meteorology and run-off. The estimated reference chlorophyll-*a* concentrations then corresponds to the present day meteorological conditions and potential challenges with shifting baselines (Duarte et al. 2009) coursed by climate changes are avoided.

The GM boundary values were estimated from reference values using an intercalibrated EQR value of 0.6 resulting in GM boundary values between 1.5 (open waters) and 9 (Hjarbæk Fjord). By applying a completely different approach for estimating reference and GM boundary values for chlorophyll-*a* (May-Sept), Carstensen and Henriksen (2009) obtained GM boundary values for selected Danish coastal sites ranging between 1.2 (Karrebæksminde Bugt) and 9.9 (Skive Fjord) and where the majority (approx. 70%) of the estimated boundary values for estuarine areas were between 2 and 5 µg l^{-1}. Although the values for estuarine areas obtained by Carstensen & Henriksen (2009) generally appear slightly higher than in the present study the overall good agreement further consolidate the chlorophyll-*a* GM values obtained in this study. For Flensborg Fjord, which is partly Danish and partly German, we estimated a GM chlorophyll-*a* target value of 2.1 µg l^{-1} (type-specific) for both the inner and outer part of the estuary. This is in accordance with the German target values of 1.9 µg l^{-1} adopted for RBMP 2009-2015 (Bundesministerium 2014), but a new target value of 6.1 µg l^{-1} for the inner part has been suggested (Schernewski et al. 2015).

Finally, we compare the modelled open water targets (based on reference modelling with the IDW model) with targets defined for comparable areas in other studies. As part of RBMP 2009-2015 a chlorophyll-*a* target of 1.9 µg l^{-1} for the western Baltic Sea was adopted by both Germany and Denmark.

The similar targets based on the reference model results from this study and the intercalibrated EQR resulted in chlorophyll-*a* targets of 1.7 µg l^{-1} in the western Baltic Sea, 1.5 µg l^{-1} in the Little Belt area, 1.6 µg l^{-1} in Kattegat and 1.9 µg l^{-1} in a few upwelling zones. Hence, the targets defined through this study results in a target for the western Baltic Sea, which is 20% lower than the earlier target. This is very similar to data reported in Bundesministerium (2014). Here the earlier target of 1.9 µg l^{-1} was also adopted for RBMP 2009-2015, but going through the modelled targets from Bundesministerium (2014), it is clear that new German targets are also lower: Along the more open water of the German coast the new targets vary between 1.0-1.8 µg l^{-1} whereas the targets for the open bays (Kieler Bay and Bay of Mecklenburg) are even as low as 1.2 µg l^{-1} and 1.3 µg l^{-1}, respectively.

Similar targets reported in Andersen et al. (2015) are 1.8 µg l^{-1} for Arkona Basin and Bornholm Basin, 1.6 µg l^{-1} for the Danish Straits, and 1.5 µg l^{-1} for the Kattegat, and hence, very similar to the findings in this study.

8.2 Estimation of required nutrient reductions

The following sections (8.3 to 8.7) concern the methodologies developed for application of the models described in chapters 6 and 7 for estimation of the maximum Danish land-based loadings allowing for good ecological status of the Danish WFD water bodies.

As mentioned in section 5.1, three different models constitute the basis for the quantification of the maximum allowable nutrient input (MAI) to each of the 119 Danish WFD water bodies: statistical models, dynamic mechanistic models and meta models. The most significant stepping stones to estimate MAI are illustrated in *Figure 8.4*. With the developed statistical and mechanistic models, cause-effect relationships (slopes) are established between Danish land-based N loadings and each of the indicators. The meta models build on the outcome of the statistical and mechanistic modelling. Subsequently, the derived slopes are used, together with the distance between the present status and the GES target value to calculate required N load reduction for each indicator. The resulting load reduction for each water body is calculated from the required load reduction for each indicator and finally, MAI is estimated by subtracting the required load reduction from the present loading.

Details on the three modelling approaches are provided in sections 8.3, 8.4 and 8.6, while section 8.7 documents the methodology used to integrate all model results into estimates of the load reductions required to support GES. Finally, considerations regarding uncertainty and sensitivity are presented in section 8.8.

Below, a short description is given of the central concepts included in the methodologies.

Figure 8.4. Overall approach for calculating the specific water body reductions in land-based N loads.

Status

Status values for the applied indictors are based on DNAMAP observations from the period 2007-2012.

For calculations based on mechanistic models, status values are converted into water body averages by relating the observed status to the model results at the actual observation point and applying the ratio between the two (model and observation) to correct the modelled water body average. Hence, we apply real observations but distribute these relative to the model results. Since status data (i.e. monitoring data) on some of the open water bodies belonging to the IDW model domain are absent or insufficient, values are extrapolated from observations from neighbouring water bodies with similar characteristics in order to obtain a status value.

For calculations based on statistical models, the monitored status is used directly since the models are based on individual monitoring stations and not on whole water bodies. Models are generally only developed for monitoring stations with sufficient data. However, for a few stations, monitoring was terminated in 2003 and for these modelled status values were applied instead.

For water bodies covered by meta models status, chlorophyll-*a*- values are based on observations from DNAMAP and provided by Danish EPA, whereas K_d status is estimated as part of the present study. To estimate K_d status, we define at least one year of observations and not less than eight observation. If this was not obtainable, no status was estimated.

Distance between status and GES

Following the status evaluation, we estimate the distance between indicator status and GES target (in percentage) as:

$$Distance = \frac{Status - Target}{Status} \times 100\% \qquad \text{Eq. 8.3}$$

Before translating "distance" into required load reductions, some adjustments are made. For the mechanistic models, "distance" is first corrected for the effect of regional treaties i.e. BSAP and GP and then adjusted to express only the proportion governed by Danish controllable N loads. For the statistical models, the approach is somewhat different and no corrections are made; instead additional supporting parameters (occurrence of hypoxia, ecological effects of hypoxia and N limitation) are included. Details on these model-specific adjustments are presented in sections 8.3 and 8.4.

In cases where one of the indicators meets GES, and distance becomes negative, we set the distance to zero.

Required load reduction for each indicator

The required N load reduction to achieve the defined GM target value is estimated as:

$$Required\ load\ reduction\ (\%) = \frac{Distance\ (\%)}{Slope\ (\%/\%)} \qquad \text{Eq. 8.4}$$

where the slope represents the sensitivity of the indicator to land-based N loads (%-change in indicator value per %-change in N-load). The slope is derived from either a statistical model (section 8.3), a mechanistic model (section 8.4) or a meta model (section 8.6). The meta-slopes are extracted from the statistical and mechanistic models.

As indicated in *Figure 8.4*, a few specific calculations are included in this step.

Resulting load reduction required for each water body

Following the above process, the "resulting load reduction required" for a specific water body is calculated by averaging the "required load reduction" estimated for the individual indicators. For mechanistic models, it is the two indicators chlorophyll-a and K_d, for statistical models additional indicators, besides K_d and chlorophyll-a, are included. For meta models, the included indicators depend on the model(s) on which the meta-analysis is based.

The purpose of averaging the "required load reduction" estimated for the different indicators is to reduce uncertainty. This concept contrasts somewhat the 'one-out-all-out' principle, which will be addressed in the final discussion.

Maximum allowable input (MAI)

The maximum allowable nutrient input, MAI, supporting GES, is derived from the estimated "resulting load reduction required" and status (2007-2012) loadings as

$$MAI = (1 - resulting\ load\ reduction\ required) \times status\ loading \qquad \text{Eq. 8.4}$$

As we assess the needed reductions based on 2007-2012 status observations, we simply apply the resulting reductions needed to the status load for the same period to estimate the MAI. Whereas the "required load reduction" in % depends on the time period used in the calculations, MAI is theoretically independent of the starting point, assuming that all other factors, for instance climate, remain unchanged compared to present conditions.

8.3 Statistical model approach

The key result of the project is the estimated targets for maximum allowable loads for each water body. As mentioned above (chapter 6 and 7), the estimates are derived using two types of models – dynamic mechanistic and statistical models. The dynamic-mechanistic model approach covers the open marine areas and the estuaries Odense Fjord, Roskilde Fjord and the Limfjorden, and the target estimates are based on the two indicators "light attenuation" and "chlorophyll-*a*". The statistical modelling approach is based on data from 29 stations covering 22 estuaries and coastal areas. Just as the dynamic mechanistic models, it uses light attenuation and chlorophyll-*a* as indicators but in addition supporting indicators describing the occurrence of anoxia, ecological signs of anoxia and the degree of nitrogen limitation for phytoplankton during the growth season. As described in section 8.2 a prerequisite for calculating maximum allowable loads is that an ecological target value is set for each indicator. The approach used in the WFD for establishing ecological target values is to define the reference condition and then combine this with an EQR value for, for instance, good-moderate status. However, this implies a restriction to indicators for which a reference condition and an EQR value for good-moderate status have been established. Based on our knowledge of the characteristics of Danish estuaries, for example shallow water columns, soft bottom and eelgrass as a dominant component; we have introduced four supporting indicators to obtain a more comprehensive base for evaluating environmental status. The supporting indicators are not intercalibrated. The four supporting indicators are; occurrence of hypoxia, signs of hypoxia in the seasonal patterns of chlorophyll-*a* and dissolved inorganic phosphate concentrations, as well as nitrogen limitation of phytoplankton growth (*Table 8.7*).

The indicator "light attenuation" is a proxy for the indicator "depth limit for eelgrass" for which observations are available from about 1900 (see section 8.3.3). No data on chlorophyll-*a* concentrations are available this far back in time, and establishment of a reference condition for chlorophyll-*a* is consequently more problematic. A separate task within the project was therefore to estimate a reference condition for chlorophyll-*a* concentration for each type of water bodies based on estimated loads in the year 1900 and to model a likely chlorophyll-*a* concentration. Section 8.1 describes this process.

8.3.1 Targets for maximum allowable loads based on statistical models results

The basic concept for setting targets for maximum allowable loads is to estimate the necessary load reductions in percent (percent load reductions; PLR) for the present load (L_p) for each of the six indicators and then calculate the target for maximum allowable loads (L_t) from a weighted average:

$$PLR = (PLR_1 * w_1 + PLR_2 * w_2 + PLR_3 * w_3 + PLR_4 * w_4 + PLR_5 * w_5 + PLR_6 * w_6) / \sum w_1 - w_6 \quad \text{Eq. 8.5}$$

where PLR_n is PLR for each indicator and w_n is the associated weight

and

$$L_t = L_p - (PLR/100 * L_p) \quad \text{Eq. 8.6}$$

The main principles behind this approach are (1) use of observed data to describe the present state, (2) use of empirical coefficients to describe the relative change in state per percent change in pressures at the present time, (3) use of a holistic approach to describe the state of each water body by integrating the response of as many ecologically relevant indicators as possible and 4) combination of the results into one value that becomes the best possible estimate of the target load.

Weighting or one-out-all-out

In the WFD, the "one-out-all-out" principle is used to assess the ecological status of a water body. This principle is sound when the ecological quality of a water body is affected by different pressure factors as it ensures that low values of one pressure factor cannot compensate for high values by other factors. From a management assessment perspective, this is a sturdy principle ensuring that significant environmental problems are not overlooked in the overall assessments. However, this project considers only one pressure factor (nutrient loadings) but makes use of several indicators to describe the effects of this pressure factor. Our assumption is that a weighted average approach will provide a more correct estimate of the maximum allowable load, making it less susceptible to random variation in the data parameters. When used repeatedly in the future WFD plan periods, it will allow us to gradually obtain loadings that can ensure GES. A weighted average approach minimises the risk of "overshooting", i.e. estimation of loadings lower than necessary to obtain GES. In summary, the aim of this project was to provide the most reliable estimate for nitrogen loadings that, over time and given the current situation for other pressure factors, will decrease the pressure on the marine systems to a level where GES conditions may potentially be obtained.

Usefulness of each indicator

Ideally, an indicator should respond to changes in the pressure factor without a time delay, be easy to measure, have a clearly defined reference condition and/or an objectively defined threshold as well as be ecologically relevant. Unfortunately, no indicator meets all these criteria, which is another important argument for using several indicators and a weighting procedure. Since each indicator has its own characteristics and associated shortcomings, some modifications or adjustments are necessary. Moreover, the weighting in Eq. 8.5 reflects the suitability of each indicator based on an evaluation of these aspects.

For some indicators, a long time lag and potential non-linearity between a change in loadings and the subsequent ecosystem response cause problems. Examples are the depth limit of eelgrass, light attenuation and occurrence of anoxia that all have a long response time. In order to overcome these problems, we have used proxies (e.g. light attenuation instead of depth limits for eelgrass) and categorised values for PLR instead of the estimated output from the models and, thirdly, transformed indications of eutrophication, for example widespread anoxia, into a required lowering of the total nitrogen concentration. *Table 8.7* gives an overview of the use of each of the six indicators, and arguments for the modifications and the values of the constant involved are given below.

8.3.2 The indicator "chlorophyll-*a* concentration"

Chlorophyll-*a*, depth limits for eelgrass and the DKI (section 8.3.7) are the three indicators that are intercalibrated. Therefore, the chlorophyll-*a* indicator and the K_d indicator (used as proxy for eelgrass depth limit) are assigned double weight in the overall assessment of environmental state in order to give more weight to indicators that have been through the comprehensive inter-calibration process. This choice is based on our wish from a management perspective to emphasise intercalibrated indicators and has no scientific basis.

In 17 out of 28[12] cases covered by the statistical models, a significant N coefficient was found. The coefficients varied between 0.10 and 1.08%/%[13] with a mean value of 0.65%/% (a coefficient of 2.2%/% from one station was discarded as unlikely as the nutrient reduction to the water body was negligible and the reductions were obviously driven by the adjacent water bodies). In eight cases, phosphorous loadings were selected as predictor variable, and in two cases none of the two nutrient loadings were selected as predictor variables, i.e. the best model contained only climatic variables. Thus, for 93% of the stations, nutrient loadings, most frequently nitrogen, were found to be the best predictor variable. The indicator thus fulfils the requirement of response.

The reference conditions for chlorophyll-*a* in each typology of water body were calculated as explained in section 8.1. The reference values for chlorophyll-*a* concentrations are associated with considerable uncertainty since we do not have any historical measurements. Here, chlorophyll-*a* concentrations differ from, for example, eelgrass, for which depth limits have been measured since around 1900, and from anoxia where sediment cores can be used to determine whether an area has suffered from anoxia or not. Therefore, reference conditions for chlorophyll-*a* can only be modelled by forcing the models with background nutrient loadings. This is a weakness from scientific point of view, as the models are used outside the area for which they are developed and for which we have validation data. Yet, as an indicator, chlorophyll-*a* has the advantage that it is easy to measure, and from a management perspective, it is therefore important to develop reference values for chlorophyll-*a*. Mechanistic and statistical models both provide results for the reference chlorophyll-*a* concentrations that are not significantly different from each other, which give some support to the results. Accordingly, we have included chlorophyll-*a* concentrations as an indicator because it is an already well established indicator for eutrophication and intercalibrated within the WFD. There is substantial scientific evidence that chlorophyll-*a* responds to changes in nutrient loadings, confirming its applicability as an indicator for changes in environmental state. However, determination of the exact threshold is a challenging task.

Since chlorophyll-*a* is always present in natural water, no objective criteria exist for setting a threshold. In this sense, chlorophyll-*a* differs from the other indicators for which we have either observations (depth limits for eelgrass), objective arguments for ecological effects as for anoxia (oxy, oxy-chl, oxy-dip) and lack of nutrient limitation (nlim), albeit the thresholds we have set for these indicators also are open for discussion (see *Table 8.7*).

A combination of coefficients, current status and thresholds yields PLR values for the chlorophyll-*a* indicator between zero and 134% (mean value 61%). Values above 100%, implying that GES values are not obtained even

[12] Chlorophyll-*a* data on Mariager Fjord were faulty for part of the period.
[13] The unit refers to the percent change in the current (2007-2012) mean value of chlorophyll-*a* as per percent change in the current (2007-2012) mean value of loadings.

with loadings equal to zero, are most frequently found in the open areas of Kattegat and the Belt Sea. The reason may be that nutrients imported from adjacent seas play a role for the more open Danish areas (Jørgensen et al. 2014, section 6.4 and chapter 9). However, since the statistical models are not used for these areas, we have not studied this in further detail.

The ecological relevance of chlorophyll-*a* concentrations as an indicator is obvious. Positive relationships between nutrient loadings and chlorophyll-*a* concentrations are well established for many aquatic systems and are also included in our statistical models. Moreover, a higher chlorophyll-*a* concentration will increase primary production as the fraction of incoming light absorbed by phytoplankton will increase (Markager et al. 2003). Hence, a positive feedback mechanism occurs, enhancing the effect of nutrient loadings. On the other hand, the chlorophyll-*a* concentration is the sum of production versus loss processes where grazing is significant. To the extent that grazers, such as zooplankton or benthic filtrators like blue mussels, also respond positively to nutrient enrichment, the grazing rate will increase and balance the higher primary production so that the net effect on the chlorophyll-*a* concentration is limited. This situation is probably most likely to occur in moderately enriched systems.

8.3.3 The indicator "light attenuation"

The indicator "light attenuation" is used as a proxy for the indicator "eelgrass depth limit", which has been intercalibrated within the WFD. Light availability is one of the main drivers determining the maximum depth distribution of eelgrass. However, since light availability is a necessary, but not sufficient, condition for eelgrass growth, the light attenuation indicator is not a measure of the presence of eelgrass at a given depth but a measure of the potential eelgrass depth distribution. Studies have shown that eelgrass growth requires between 11% and 20% surface radiation (Duarte 1991; Krause-Jensen & Middelboe 2000; Nielsen et al. 2002; Olesen 1996; Short et al. 1995). Using a minimum light requirement of 14%[14] of surface radiation, it is possible to convert reference and target values for the "eelgrass depth limit" indicator to reference and target values for the "light attenuation" indicator using the following equation:

$$Kd\ target = \frac{-\ln(0{,}14)}{eelgrass\ depth\ limit\ target} \qquad \text{Eq. 8.7}$$

Due to the close link between the "light attenuation" indicator and the intercalibrated "eelgrass depth distribution" indicator, the light attenuation indicator has been given double weight in Eq. 1 (see *Table 8.7*). Moreover, light attenuation is essential for the structure of pelagic systems. The presence of a deep chlorophyll-*a* maximum means that oxygen is produced below the pycnocline and it has been shown that the most significant effect of high nutrient levels on primary production is a shift in primary production from within or below the pycnocline to the mixed layer (Lyngsgaard et al. 2014). In shallow areas, where light may potentially reach the bottom, a low K_d value stimulates several types of benthic plants and thereby the oxygen pro-

[14] The value of 14% is based on a laboratory study. We are currently investigating the reasons for the variability in the different estimates for the minimum light requirement of eelgrass. Probably 11-14% represents a physiological minimum. Different factors may enhance this value under field conditions, for example, other loss factors than internal respiration, variability in light levels over the season and unsuitable substrate. If the minimum light requirements are higher, the plants around year 1900 grew at higher light levels, and the reference values and the GES thresholds for K_d will therefore be higher, requiring higher estimates of maximum allowable loads. This is a subject for further investigation for the next round of water plans.

duction at the seabed (Krause-Jensen et al. 2012). This, in turn, inhibits the release of phosphorous and nitrogen from the sediment to the overlying water column (Jørgensen 1996).

The indicator used is the average K_d value for July, August and September. For this late summer period, we obtained significant positive coefficients between N load and K_d for 16 out of 22 stations, and the remaining stations are dependent on temperature, salinity, irradiance, wind speed, water column stability and phosphorus. In spring-early summer (March to June), K_d was mainly governed by phosphorous loading, and phosphorous loading was therefore selected for 14 out of 22 stations; at 4 out of 22 stations N loading was selected, while the remaining 4 stations were best explained by SST (2), wind and irradiance. This result is in accordance with previous findings showing that the environmental state in Danish estuaries is controlled by phosphorous in the spring-early summer period (Markager et al. 2006). However, as explained (section 6.4), inclusion of phosphorous would require additional indicators and we had to focus on nitrogen in order to complete the project within the given period and bearing in mind the available economic resources.

The values for change (%/%) in light attenuation were in general lower than for chlorophyll-*a* (*Figure 8.20*), particularly for water bodies within Type 1 and 2, the average value being 0.21. Thus, the response is, albeit significant, so low that light attenuation will not decrease notably even with large reductions in nitrogen loading. We attribute this to the fact that light attenuation is mainly governed by light absorption by dissolved organic matter, detritus and scattering of light, whereas pigmented particles (phytoplankton) only play a minor role (Markager et al. 2003). If we assume that decades with high nutrient loadings and primary production have caused a general accumulation of organic matter in the systems, the K_d values will respond only slowly to a decrease in loadings, and there will thus be a considerable time lag between load reductions and improvements in K_d. In order to overcome the effects of this time lag on the estimated load reductions, and hence to avoid over-implementation, we have transformed the estimated PLR values into categories when above 25% (see *Table 8.7*). The motivation behind this is that a high PLR value, even though it cannot be used directly, reflects that the current status is far from GES and a large reduction of loadings is therefore necessary.

8.3.4 The indicator "occurrence of hypoxia"

It is well documented that hypoxia or anoxia in the bottom water will accelerate the negative effects of eutrophication, such as loss of macro vegetation, release of both nitrogen and phosphorus from the sediment, fish kills and, ultimately, direct release of hydrogen sulphide to the atmosphere. Although low oxygen concentrations can occur also in undisturbed systems such as Southern Little Belt, it is well documented that the frequency and spatial extension are regulated by the nutrient loadings to the water body. Hypoxia and anoxia are therefore not only of relevance for ecosystem functioning, they are also an indicator of eutrophication.

There is direct evidence for a relationship between nutrient loadings and oxygen concentrations in bottom water (Markager et al. 2006) and the size of hypoxic/anoxic areas (Scavia et al. 2003; Christensen et al. submitted). However, these relationships are complicated by a considerable time lag and a high sensitivity to climate variables like water temperature and wind stress.

Therefore, and due to the time constraints on our project, we chose not to develop models for a direct coupling between nutrient loadings and oxygen concentrations. Instead, we used the occurrence of low oxygen conditions (evaluated from the monitoring data) as a sign of severe eutrophication demanding a decrease in nutrient loadings. This demand was then assigned as a 25% reduction of TN concentrations, which was subsequently translated into a reduction of nitrogen loadings (PLR values) using the coefficient from the statistical models. The limits for occurrence of hypoxia are given in *Table 8.7* and are based on our best judgement as to when hypoxia is severe. Thus, a certain occurrence of hypoxia is allowed in some years without introducing a demand for reductions.

8.3.5 Ecological signs of hypoxia

A previous investigation conducted in the estuary Limfjorden revealed that the seasonal patterns in the concentrations of chlorophyll-*a* and dissolved inorganic phosphorous differed significantly between eight of the basins (*Figure 8.5*), depending on the occurrence of hypoxia (*Figure 8.6*).

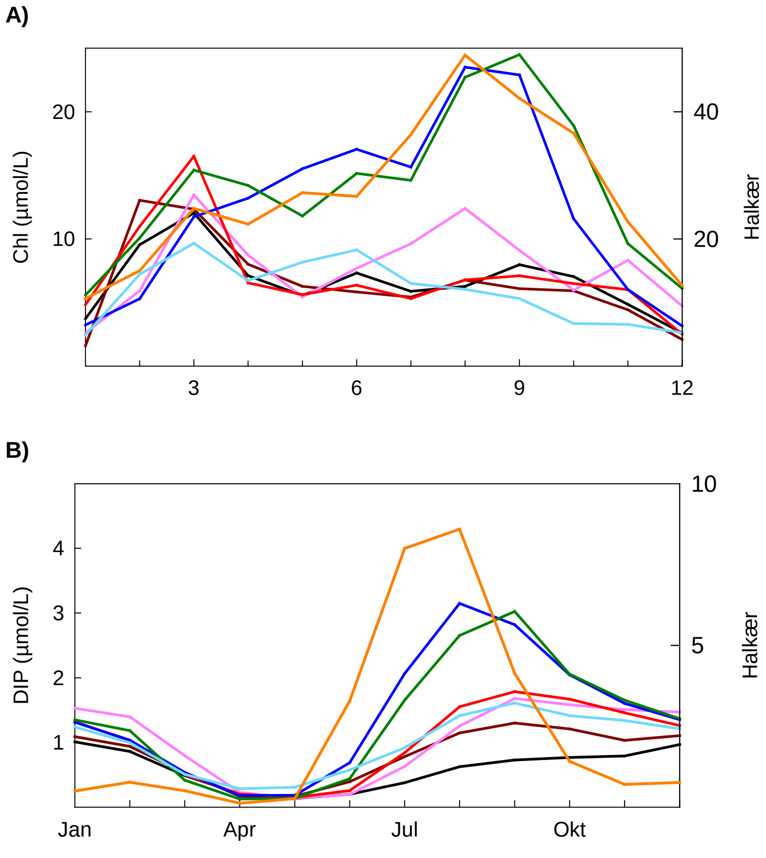


Figure 8.5. **A)** Seasonal patterns in eight basins of the eutrophic estuary the Limfjorden. The corresponding distribution of oxygen depletion is shown in **B)**. The blue and green lines are from the south-eastern basins Skive Fjord and Lovns Broad where anoxia is widespread, frequent shifts occur between stratified and mixed periods, transferring nutrients from the sediment into the photic zone. The yellow line is from the very shallow (mean depth 0.5 m) Halkær Broad with intense sediment-water column contact. The other lines represent the five basins where oxygen depletion is limited or absent or, in the case of Thisted Broad, the water column is permanently stratified and deeper so that nutrients released from the seabed are not directly transported up into the photic zone.

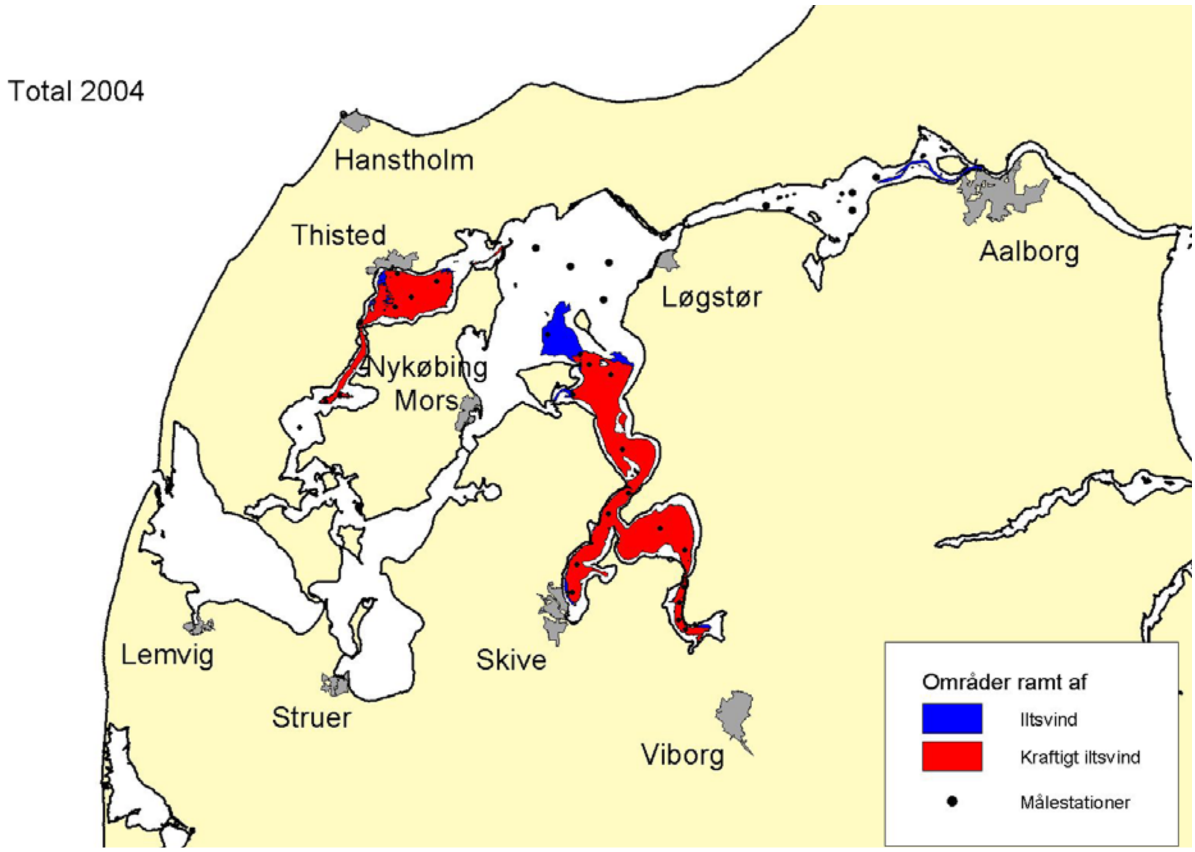


Figure 8.6. The spatial distribution of oxygen depletion in the Limfjorden in 2004. Blue and red zone correspond to concentrations below 4 and 2 mg oxygen l^{-1}, respectively.

The observed patterns are interpreted as the ecological effects of hypoxia as hypoxia is known to cause an out flux of phosphorus from the sediment (Jørgensen 1996), producing both an increase in dissolved inorganic phosphate concentrations in June and onwards and, secondarily, an increase in chlorophyll-*a* concentrations (*Figure 8.5A*). In contrast to oxygen measurements, which are often conducted at the deepest site of the estuary, the ecological effects reflect the damage and feedback mechanisms that hypoxia may create. If the low oxygen concentrations are restricted to a deep hole in an estuary, it may not have a significant impact on the estuary as a whole, whereas comprehensive hypoxia covering a large-sized area will most likely result in notable derived negative effects.

Based on the above, two indicators were defined from the seasonal patterns in chlorophyll-*a* and inorganic phosphorus concentrations. For both indicators, the thresholds were defined as when the ratio of the concentrations in July, August and September divided by the annual mean exceeded 1 (*Table 8.7*). As for "occurrence of hypoxia", for both indicators, values above one trigger a demand for a 25% decrease in TN concentration. However, as the same underlying process stimulate both indicators, they were given a weight of '0.5' in equation 8.5.

8.3.6 Nitrogen limitation of phytoplankton growth

The basic process in eutrophication is stimulation of phytoplankton growth, and the degree of nutrient limitation experienced by phytoplankton is therefore an obvious indicator for eutrophication. In all but the most turbid systems, phytoplankton growth is limited by nutrients from the cease of the spring bloom and onward, and an increase of nutrient limitation will decrease ecosystem productivity and improve the ecological status. Out flux of nutrients from the seabed can, in combination with a fully mixed water column, periodically relax the limitation, but overall a more or less permanent nutrient limitation is the natural situation in Danish estuaries.

The average concentrations of inorganic nutrients are not a good proxy for nutrient limitation as the concentrations may fluctuate substantially over the season. We have therefore developed an indicator based on the number of days where nitrogen is limiting. A similar indicator for phosphate was also considered[15]. Based on studies of phytoplankton growth kinetics (Eppley et al. 1969; Klausmeier et al. 2004; MacIsaac & Dugdale 1969), we chose a half saturation coefficient (Ks) for nitrogen limitation of 2 µM. For each station, we counted the days with concentrations of dissolved inorganic nitrogen. The sum of the days with nutrient concentrations below Ks constitutes a relative measure of the nutrient limitation of the phytoplankton community.

In systems with excess nutrients, phytoplankton biomass can be constrained by limitation in light or intense grazing (Armstrong 1994; Cloern 2001, 1999). In order to identify at which point nutrient limitation is the main limiting factor for the phytoplankton biomass, a piecewise regression analysis of the relationship between mean chlorophyll-*a* concentrations in the period of interest against total number of days with nutrient limitation was conducted.

For estuaries, the analysis showed a strong significant relationship ($r= 0.56$, $p < 0.00001$) between ln(chl-*a*) and days of N or P limitation above 150 days, while there was no significant relationship between ln(chl-*a*) and limitation below 150 days ($r = 0.00$, $p > 0.05$) (*Figure 8.7*).

[15] Overall, the analysis was limited to quantifying the effects of nitrogen loadings since nitrogen has been shown to be the main limiting factor for phytoplankton growth. However, it should be recognised that phosphorous plays a role for eutrophication in Danish estuaries and coastal waters. In general, the importance of phosphorus decreases when moving outward from closed estuaries, over coastal areas to the more open systems like the Belt Sea and Kattegat. Moreover, phosphorus limitation is most important in spring (Timmermann et al. 2010), with nitrogen taking over in late April/May in the Belt Sea and in late May/June in most estuaries. A full inclusion of phosphorous loading would require a definition of new indicators for e.g. chlorophyll-*a* and K_d for spring (March to June) and late summer (July to October) and would be in conflict with the time periods for indicator as used by HELCOM. Moreover, the corresponding models, i.e. four times the present number of models, would be needed. In addition, the results would be considerably more complicated to interpret in a management perspective, as they would be a vector of nitrogen and phosphorous loadings, potentially creating GES conditions. This approach was applied in 2006 for the Limfjorden (Markager et al. 2006) and should be considered for the next WRD plan period but its application in the present project for all the 119 areas was beyond the scope of the present project.

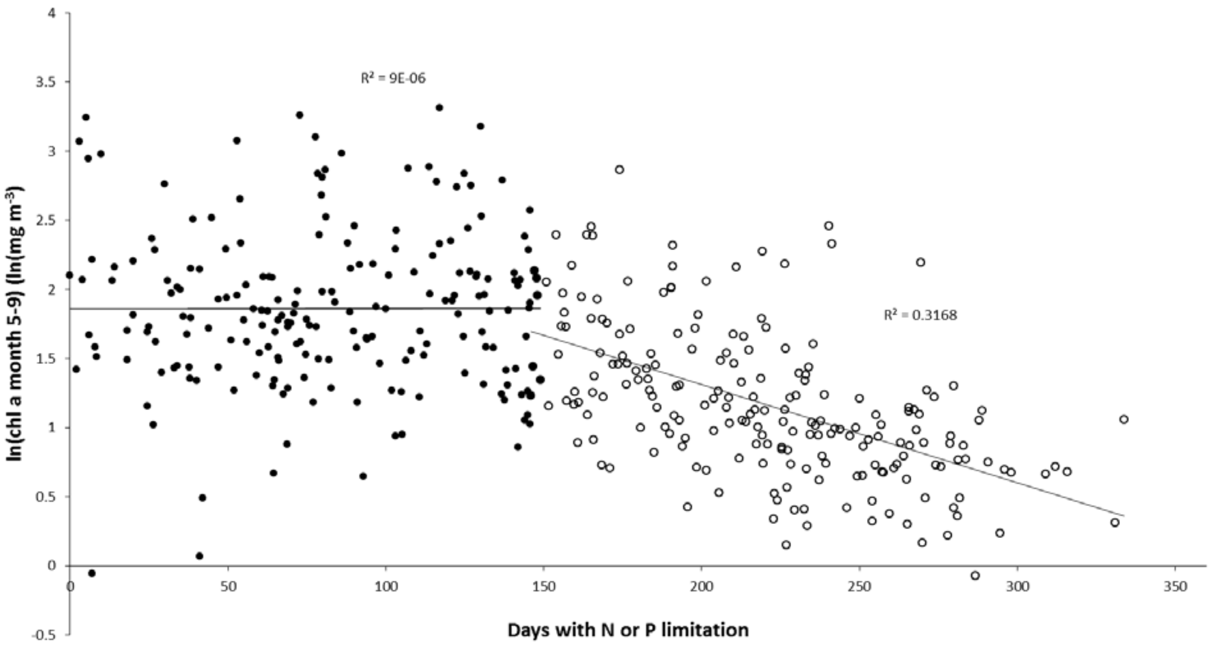


Figure 8.7. The relationship between ln(chl-*a* month 5-9) and the number of days with either N or P limitation. Black dots are observations (station and year) with fewer than 150 days, while the white dots are observations of chlorophyll-*a* concentration in years or at stations with more than 150 days of N or P limitation.

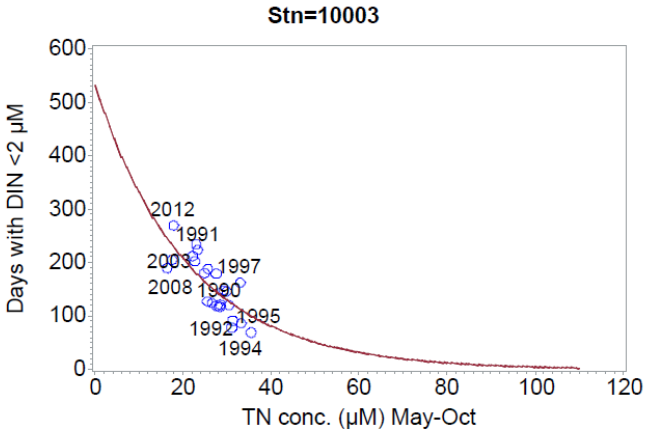
Similar analyses for coastal and open waters revealed threshold values of 200 days and 220 days, respectively.

The relationship between chlorophyll-*a* concentration and nutrient limitation is based on the concentration of bioavailable (i.e. inorganic) nutrients, which only constitutes a minor fraction of the total nutrient concentration (Carstensen & Henriksen 2009; Jørgensen et al. 2014; Stedmon et al. 2006). To set a site-specific goal for the total nutrient level, site-specific relationships between the total nutrient pool and the lability/bioavailability of the total nutrient pool were established for DIN vs TN (*Figure 8.8*) and DIP vs TP. For nitrogen, the equation was:

$$Days\ of\ N\ limitation = 530 \cdot e^{(\alpha_{site} \cdot TN_{May-Oct})} \qquad \text{Eq. 8.8}$$

where *Days of N limitation* is the number of days where DIN is below Ks_{DIN}, a_{site} is the site-specific lability constant and $TN_{May\text{-}Oct}$ is the TN concentration in the period of interest. For each station, we have set a TN goal based on the number of nutrient-limited days aimed for.

Figure 8.8. An example of a site-specific relationship between days of N limitation and the concentration of total nitrogen.



Finally, the indicator is based on a K_m-value of 2 µM DIN and a period of 150, 200 and 220 days depending on the type of water body (*Table 8.7* and chapter 6). Relationships between days with nitrogen limitation and TN-concentrations were established for each station, and PLR values were established from the statistical models between nitrogen loadings and TN concentrations (chapter 6). As for the indicator 'light attenuation', high PLR values were found for some water bodies. The mechanism behind is also assumed to be the same: accumulation of organic matter over decades with high nutrient loadings and primary production. From previous investigation (Knudsen-Leerbeck et al. submitted), we know that remineralisation of dissolved organic nitrogen can constitute about 90% of the total nitrogen supply for phytoplankton growth in summer. Therefore, we used categorisation of the PLR values as demonstrated in *Table 8.7*.

8.3.7 Benthic biodiversity

Indicators for benthic macro fauna were not used in the assessment of maximum allowable load. The benthic macro fauna is one of three intercalibrated indicators for marine areas within the WFD. In Denmark, this indicator is assessed using the multi-metric index "DKI" (abbreviation for "Danish Quality Index") as described in Josefson et al. (2009). This index has been tested in various pressure gradients and intercalibrated against other Scandinavian multi-metric indices. This quality measure has been found to be sensitive to indirect eutrophication effects (secondary effects related to increased ecosystem productivity) such as excess organic matter enrichment of the sediment and bottom water hypoxia. However, no direct link has been established with excess nutrient concentration. Also, it is worth mentioning that other stressors independent of eutrophication likely influence the DKI value and thereby the classification of the water body. Thus, the response of DKI to nutrient loadings is complex and, although DKI is highly relevant for the assessment and classification of overall ecosystem quality, it is not presently integrated in the described process models. However, we suggest that its potential inclusion in the next generation of plans should be discussed.

8.3.8 Holistic framework

For each station, a final target value is calculated from *Eq. 8.5*. For some estuaries, two or more stations are available and in these cases, the final value for the catchment was set to that of the station with the highest PLR values, i.e. with the highest demand for reduction. Finally, the target values are rounded to the nearest value at 5% intervals (see *Figure 8.23)*.

Table 8.7. A summary of the six indicators used in the assessment of required load reductions for Danish marine areas.

Indicator	Chlorophyll-*a* concentration (May-September)	Light attenuation coefficient for PAR (July-September)	Occurrence of hypoxia (July-September)	Ecological signs of hypoxia, pattern of chlorophyll-*a* concentration	Ecological signs of hypoxia, pattern of dissolved inorganic phosphate	Nitrogen limitation
Explanation/ecological relevance	Indicator of phytoplankton biomass.	Proxy for maximum depth distribution of eelgrass and indicator of the light climate.	Occurrence of hypoxia or anoxia is one of the most devastating effects of eutrophication. It directly affects the biodiversity of the benthic fauna, reduces the food availability to benthic fish. In severe cases, it can cause nausea, bad smell and unclear water.	Reduced conditions in the sediment occur in most estuaries at various depths below the sediment surface. However, if the reduced conditions reach the sediment surface, leakage of both ammonium and phosphate can occur. This will typically happen in late summer and produce elevated chlorophyll conc.	This indicator is similar to the previous one. Here the data evaluated is the seasonal pattern in phosphate concentrations, which will show elevated levels in late summer at reduced conditions in the sediment.	Stimulation of phytoplankton growth is the first process in the cascade of processes in marine eutrophication; a logical indicator is the number of days with potential nutrient limitation during the growing season. As nitrogen is most often limiting, focus is on nitrogen limitation.
Abbreviation	Chl	Kd	Oxy	Oxy-Chl	Oxy-DIP	Nlim
EU intercalibrated	Yes	No (but depth distribution is)	No	No	No	No
Reference conditions	Estimated from loads in 1900 and models, see *Table 8.3* to *8.5* for reference conditions for each area type.	Estimated from actual observations around 1900 for eelgrass and transformed into K_d-values.	None	None	None	None
Confidence of reference conditions	Low – based on model estimates extrapolated outside the data domain and somewhat uncertain estimates of historical nutrient loadings.	High – based on direct observations.	-	-	-	-
Ecological Quality Ratio (EQR) value	0.6	0.74 - depth limit for *Zostera marina*.	-	-	-	-

Table 8.7. continued

Threshold value	Reference condition for Chl*EQR-value.	Reference condition for *Zostera**EQR value. Transformed to K_d target values by assuming a min. light requirement of 14% of surface radiation.	'Yes' if > 25% of time <4 mg O_2 l^{-1} or > 10% of time <2 mg O_2 l^{-1}.	'Yes' if mean value of $[Chl]_{7,8,9}/[Chl]_{1-12} > 1$.	'Yes' if mean value of $[DIP]_{7,8,9}/[DIP]_{1-12} > 1$.	K_m for DIN=2 µmol l^{-1}, DIN limitation for 150, 200 and 220 days depending on area type.
Relationship with nitrogen loadings[a]	Yes	Yes, but time lag[b].	No, but TN reduction of 25%.	No, but TN reduction of 25%.	No, but TN reduction of 25%.	Yes, through TN, time lag[b].
Categorised in case of time lag[b]	-	25% (25-100), 50% (100-200), 75% (>200)				25% (25-50, 50% (50-75), 75% (>75)
Weight	2	2	1	0.5[c]	0.5[c]	1

a: 'Yes' means that a significant relationships between land-based loadings of nitrogen and the indicator for the majority of the areas is documented.

b: Due to regime shift or delayed response, the slope for the aforementioned relationship is low causing a need for a large reduction in loadings that may even exceed the present loadings. In this case, the targets for the necessary reduction are most likely overestimated and the actual values for the slope (see section 8.3.1) are therefore transformed into categories.

c: These two indicators represent two facets of the same process, combined they have a weight of 1.

Numbers in subscript refers to month numbers.

8.4 Mechanistic model approach

8.4.1 Estimation of required loading reductions

Following the model development and evaluation, the models have been used for scenario modelling with the aim of providing model input to a following estimation of required load reductions to obtain GES in the Danish water bodies. This section describe the scenarios executed, the model results and the method developed to allow for an overall screening and estimation of the required load reductions based on mechanistic models.

For all model scenarios, the models have been executed for the period 2002-2011, but in the following analysis of model results we only apply data from the period 2007-2011, and hence, the period[16] which corresponds to the RBMP 2015-2021.

8.4.2 Definition of scenarios

An important part of the model development and the subsequent estimation of ecological status is the freshwater and nutrient loads included in the models, as well as nitrogen input from the atmospheric deposition. The loadings included in the development of the models relate to the actual loadings from the model period (2002-2011), see section 4.2 and 7.3. The model results from this period is denoted the 'status' period in the following sections, and covers model results from 2007-2011.

In addition to the load from 2002-2011 a model scenario covering "reference conditions" was developed. Reference conditions is here defined as the period around year 1900 and this scenario forms the background for defining the chlorophyll-*a* targets as described in section 8.1. For the modelling we do only change the loads, boundaries and initial fields in the biogeochemical model according to the reference period, whereas the physical modelling remains unchanged and represents the freshwater runoff, water levels, currents, salinity and temperature from 2002-2011.

In addition to the status and reference condition modelling, a number of N- and P-reduction scenarios were implemented and executed. Developing these scenarios the nutrient reductions are grouped in regional treaties and local (Danish) river run-off reductions.

For all reduction scenarios we assume full implementation of the Baltic Sea Action Plan (BSAP) HELCOM 2007 and HELCOM 2011 as well as full implementation of the Gothenburg Protocol (GP) as a minimum. Some countries already have fully implemented the BSAP, and some even reduced more than required, and for those we assume no changes to status loads. Hence, we do not regard BSAP and GP as end goals but as minimum requirements.

With respect to local N-load run-off scenarios, we assume three national scenarios corresponding to a 15%, 30% and 60% reduction in all Danish land-based N loads. In these three scenarios the freshwater run-off remains the same as for the status (corresponding to year 2002-2011) and P loads re-

[16] Year 2012 should have been included to cover the entire WFD period being assessed but the timing of the model development, and project schedule, did not allow for year 2012 to be included in the model development.

mains unchanged corresponding to a situation where only the N load reductions are assessed.

In addition to the three scenarios where only N loads are reduced, the same N load reduction scenarios are combined with some P reductions, ranging between 10-20%. These reductions vary from source to source and the reductions are estimated assuming increased P treatment in wastewater treatment plants (WWTP) and combining with the existing P originating from diffuse loadings. These reduction specifications were provided by the Danish EPA.

All together the four models are executed for 8 different combination of load scenarios, of which one corresponds to the model development (the status scenario).

8.4.3 Evaluation of status

Despite a careful model calibration and validation, some deviations between model results and parameter values at specific monitoring stations remain. Hence, when determining the status of a specific water body, we still adopt the monitoring data as 'true' values for that specific water body. However, looking at a specific monitoring station, the station does not necessary optimally represent the entire water body.

Most Danish water bodies are rather dynamic with strong gradients going from the brackish Baltic Sea to the saline North Sea as well as from an inner estuary (inner fjord) and/or estuary to the open waters.

Evaluating the location of the monitoring stations included in this study as the percentage difference between the area-averaged modelled water body indicator value and the modelled value at the location of the monitoring stations, reveals large variability. As an example the average difference for all water bodies are 17.2% (StDev = 23.6%) for chlorophyll-*a* and 6.5% (StDev = 7.5%) for K_d. Hence, changing the location of the monitoring station to a more representative location potentially could change the observed concentration with up to 17.2% respectively 6.5% in average.

These figures differs between the different types and in the no category estuary we find average differences in chlorophyll-*a* figures of 35.6% (StDev = 46.2%) respectively K_d figures of 13.1% (StDev = 14.6%), whereas similar figures for the Type 1 water bodies are 4.7% (StDev = 6.4%) respectively 4.7% (StDev = 6.5%). Not surprisingly, the closer we approach to the location of the fresh water and nutrient sources, the stronger is the difference between monitoring location and area average conditions.

In this study we handle such off-sets by normalizing the modelled water body average (over its entire area) to match the observations in the specific monitoring station location: If the model is 10% off in the specific location compared to the measurements, the water body average is corrected by 10%. With this approach we evaluate the water body as a whole, but maintain a strong link to the "true" observations. Also, in water bodies where no specific observations exist, we adopt the same approach, using a nearby monitoring station and using the modelled gradients to extrapolate the observations into the water body without observations. We assume this to be a valid approach as the overall calibration seems strong.

In the following we only regard the water body averages and not the average indicator values in the single point of observations.

8.4.4 Sensitivity to Danish N load

As mentioned earlier 119 specific (administrative) water bodies are defined in Denmark. The mechanistic biogeochemical models applied in this study covers 45 of these 119 water bodies, and for each of the 45 water bodies a specific analysis have been developed. First, we analyse the sensitivity to loadings, and especially the response to Danish land-based loadings. In total 8 scenarios have been conducted, but as 6 of the 8 scenarios differ between Danish N reductions and Danish N reduction combined with Danish P reductions, we have grouped them into two basis scenarios (status and reference), and 2 × 3 loading scenarios (three N scenarios and three N scenarios combined with P reductions). In practice we do not see any significant difference between the Danish N reductions and Danish N reduction combined with Danish P reductions when analysing the model indicators defining GES. Hence, we do not include results from the P scenarios in this particular part of the discussion, but we will briefly get back to the findings on P reduction in section 8.4.7.

In the following the model results from a subset of water bodies are exemplified and discussed in more detail:

- The water body *Bjørnholms Bay, Riisgårde Broad, Skive Fjord, Lovns Broad* (No. 157) located in the central part of the Limfjorden, see *Figure 8.9*. This water body is a no category water body.

- The water body *Little Belt Broad* (No. 217) located just south of the narrowing in the Little Belt, see *Figure 8.9*. This water body belongs to the Type 1 water bodies.

Figure 8.9. The two water bodies included as examples in the data analysis (purple colour). The northern water body is Bjørnholms Bay, Riisgårde Broad, Skive Fjord, Lovns Broad (No. 157) whereas the southern water body is Little Belt Broad (No. 217).

The two water bodies differ in several aspects; water body No. 157 is strongly affected by land-based nutrient inputs. It suffers from oxygen depletion, experience extensive internal loads (seasonal release of nutrients from sediments) and minor water exchange. In contrast, water body No. 217 is to a much larger extent affected by external loads as well as water and nutrient exchanges with neighbouring water bodies. Water body No. 217 is also frequently suffering from oxygen depletion.

As highlighted in section 5.2 we include two model indicators for the final analysis of effort needed to obtain GES:

- Chlorophyll-*a* concentrations (1 m depth) (1 May – 30 September)
- K_d (1 March – 30 September)

Hence, in this section we discuss the model results with an off-set in these two model indicators. However, in this section we also include the yearly average of TN concentrations. No specific targets exists in Denmark with respect to TN, but for the sake of sensitivity to land-based loadings and to understand the dose-response to N loadings we include it here anyway.

For each model indicator the water body average is estimated, average covering both the spatial average as well as the temporal average. As mentioned above we have 2 basis scenarios and 2 × 3 loading scenarios. In addition to this, we have executed the models and extracted model results for the 5 years 2007-2011. Hence, we have 25 single data sets where loadings, freshwater input, meteorology etc. differ.

In *Figure 8.10* respectively *Figure 8.11* some scenario results are shown for water body No. 217 respectively No. 157. For water body No. 217 we clearly see that TN and the indicators chlorophyll-*a* and K_d does not respond very strongly to changes in Danish land-based N loadings. When analysing all scenarios and all years (left panels) some response in chlorophyll-*a* and TN are observed, whereas for K_d, the trend line is almost horizontal, with a slight negative correlation to N loadings. This, however, covers over year-to-year variations and the indicator (incl. TN) responses are stronger within the specific years, see *Figure 8.10* right panels. Analysing year-by-year the average trend line slopes are 1.0×10^{-3}, 4.8×10^{-5}, 1.7×10^{-1} for chlorophyll-*a*, K_d respectively TN, and these should be compared to the trend line slopes in *Figure 8.10* left panels of 6.4×10^{-4}, -1.5×10^{-5} respectively 3.3×10^{-2}. Hence, averaging all years and all loadings the slopes becomes less pronounced as compared to the trends of the individual years. This indicate that for this specific water body, the year-to-year variations are similar or stronger than the Danish N loadings alone can explain. In section 8.4.5 we use the average slopes (right panels in *Figure 8.10*) and not the slopes derived from lumping all data and all years.

This particular water body is affected strongly by changes in e.g. the Baltic Sea, why we also see a clear drop in TN and indicator values at reference conditions (orange dots in the figure), and these reference data is not included in the trend lines.

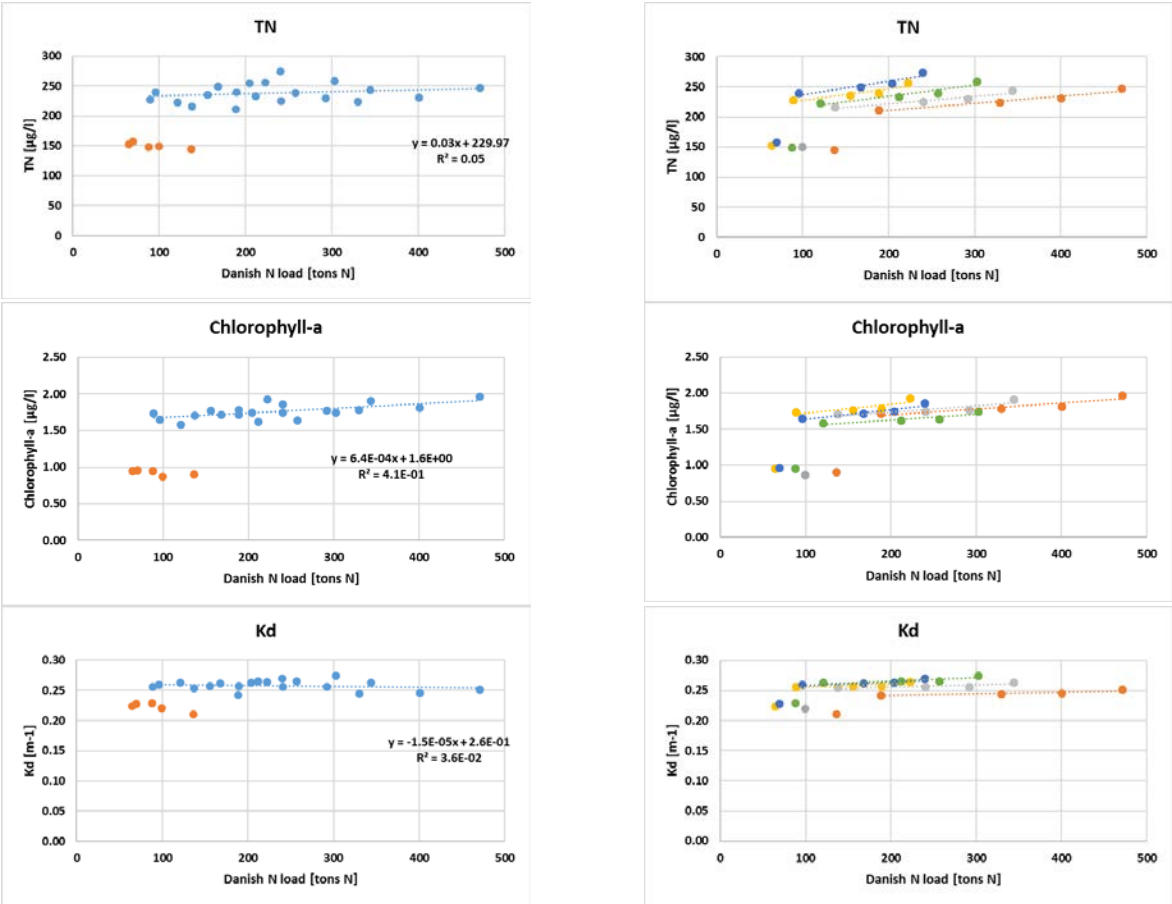


Figure 8.10. Indicator sensitivity to Danish land-based N loadings for water body No. 217. Top panel show TN concentrations, middle panel show chlorophyll-*a* concentrations and bottom panel show K_d. Blue dotes in left panels originate from different scenarios and different years, whereas orange dots originates from reference scenario and different years. In right panel the scenario results have been split into years: Orange is 2007, grey is 2008, yellow is 2009, blue is 2010 and green is 2011. In right panels trend lines for the different scenarios (except the reference scenario) are included.

Water body No. 157 is much more affected by Danish land-based N loadings, see *Figure 8.11* compared to water body No. 217. The correlation between N loading and TN is nearly linear, with some year-to-year variability, and with a slope of 0.38 and R^2=0.93. At increasing N load the TN concentrations increase, and within the scope of the model we see no sign of an upper limits.

We see a similar strong correlation between Danish land-based N loading and chlorophyll-*a*, but the correlation is not linear for this specific water body. The Limfjorden is eutrophied and strongly affected by Danish land-based N loading why we are closer to a Sigmoid curve fit, suggesting self-shading of phytoplankton (Edwards & Bees 2001). A Sigmoid function is a mathematical function having an "S" shaped curve (Sigmoid curve) and with the theoretical formulation:

$$S(t) = \frac{1}{1+e^{-t}} \qquad \text{Eq. 8.9}$$

Applying a least squares method we can transform this shape into this formulation which is plotted in the middel panel in *Figure 8.11*:

$$chlorophyll - a = \frac{10.4}{1+0.99 \times e^{-0.0032 \times Load + 1.71}} \qquad \text{Eq. 8.10}$$

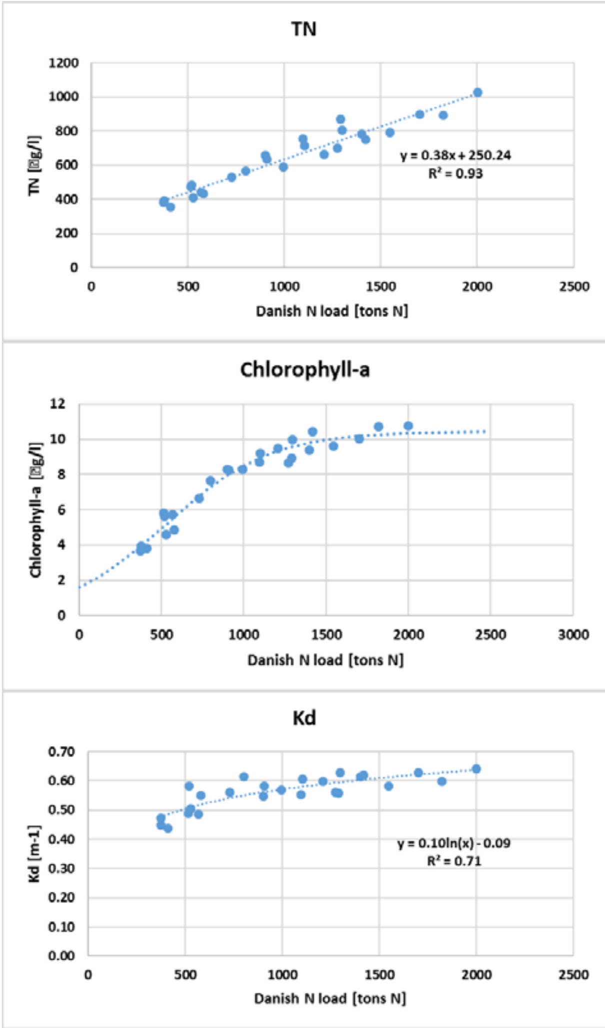
Where chlorophyll-*a* is the concentration in µg l^{-1} and Load corresponds to the Danish land-based N load for that specific water body.

It is important to mention that this Sigmoid approximation is only truth when assuming a stable ecosystem – meaning it will not change dramatically even though loadings changes dramatically. However, this is not necessarily the case and when reducing loadings we may face a more abrupt change at a certain level, corresponding to an actual regime shift where e.g. eelgrass cover or other benthic vegetation becomes much more abundant compared to present situation. For this specific study we do, however, assume no regime shifts and linearity within the scope of the model.

The theory behind the Sigmoid approximation is that at increasing N loadings the chlorophyll-*a* concentration increases until we reach a level where more N loadings does not result in higher chlorophyll-*a* concentrations. This correlation is explained mainly with self-shading effects at high nutrient inputs. Lowering the N loads decreases the chlorophyll-*a* concentrations but not at the same rate. At the lowest N loads the rate of change is smaller (e.g. < 500 tons N year^{-1}) than at higher loadings (between 500-1000 tons N year^{-1}). This is a classic shape and reported by McCauley et al. (1989) and Edwards & Bees (2001). In the Sigmoid approximation we have included the reference conditions. As indicated above the summer chlorophyll-*a* concentrations are strongly dependent on the Danish land-based N load in this water body. However, in the reference conditions also large reductions in P load is included, why the response in summer chlorophyll-*a* potentially is affected by this reductions as well.

The correlation curve for K_d is less strong than for TN and chlorophyll-*a*. As can be seen from the figure reducing N load does decreases K_d but obviously not as strong as for both chlorophyll-*a* and TN. K_d is a much more complex parameter where especially re-suspension, loadings of dissolved organic matter (DOM) etc. plays a vital role. Chlorophyll-*a* concentrations also impacts K_d, but only a smaller part of it, and year-to-year variability becomes more important, as for water body No. 217.

Figure 8.11. Indicator sensitivity to Danish land-based N loadings for water body No. 157. Top panel show TN concentrations, middle panel show chlorophyll-*a* concentrations and bottom panel show K_d. Blue dots originate from different scenarios and different years.



Similar model results, as shown in *Figure 8.10* and *8.11*, are developed for all 45 water bodies covered by the mechanistic models, and these data form the backbone of the continued development of a method for assessing the need for reduction in Danish land-based N loads to obtain GES.

8.4.5 Developing surrogate models

As described in section 8.2.2 the Danish land-based N loading is distributed in more than 300 specific sources within the different model domains, and having more than 300 sources of nutrient input and 119 water bodies it very early in the project was clear that we would never be able to cover that potential myriad of scenarios or combinations of scenarios that could evolve. Hence, we decided to develop a surrogate model approach that could feed into a screening tool, and this tools would be used for setting the final reduction targets.

First we rearrange the data presented in *Figure 8.10* and *Figure 8.11*. In those two figures we presented spatial averages for the different years, 2007 to 2011. Now we also average the indicators over the 2007-2011 resulting in only one indicator value per scenario for each water body covered by a mechanistic model.

For each water body we develop one specific surrogate model where we estimate:

- One spatio-temporal average indicator value at status loads: Present Danish land-based N&P loadings, present regional land-based N&P loadings and present atmospheric N deposition. This value corresponds to present modelled status of that specific water body.

- One spatio-temporal average indicator value at reference conditions: Reference Danish land-based N&P loadings, reference regional land-based N&P loadings and reference atmospheric N deposition, see section 8.1.4 for information on reference loadings and depositions. This value corresponds to a modelled situation around year 1900 of that specific water body (but assuming present day climate).

- Three spatio-temporal indicator values estimated from the three N reduction scenarios: 15%, 30% and 60% reductions in Danish land-based N loads, present Danish land-based P loads[17], BSAP implemented as a minimum for regional N&P loads, and GP implemented for atmospheric N deposition.

- The slope of the trend line of the three scenario indicator values.

- The intercept between the trend line and the 100% Danish land-based N loads, corresponding to the indicator value for present day Danish land-based N loads with fully implemented BSAP and GP.

In *Figure 8.12* the principles behind the specific surrogate models are shown.

As indicated in *Figure 8.12* we assume linearity between the changes in Danish land-based N loadings and the response to the specific indicator value. Within the scope of this project, and only applying changes in the Danish land-based loadings, this is also the case in the majority of the water bodies covered by the mechanistic models.

However, as we showed in *Figure 8.11*, the ecosystem in reality reacts in a more complex matter. When moving from a highly eutrophied situation towards a more oligotrophied situation, changes in feedback mechanisms, changes in species composition etc. impacts the response-curve. Out of the 45 water bodies covered by mechanistic models we see non-linear responses in a very few water bodies, mainly in the Limfjorden. For the surrogate model development we assume linearity, and consider the implications small within the scope of the model use.

[17] Here we only show the principles for the N reduction scenarios without including P reductions. Principles are however similar between the N reduction scenarios and the N reduction scenarios supplemented by P reductions.

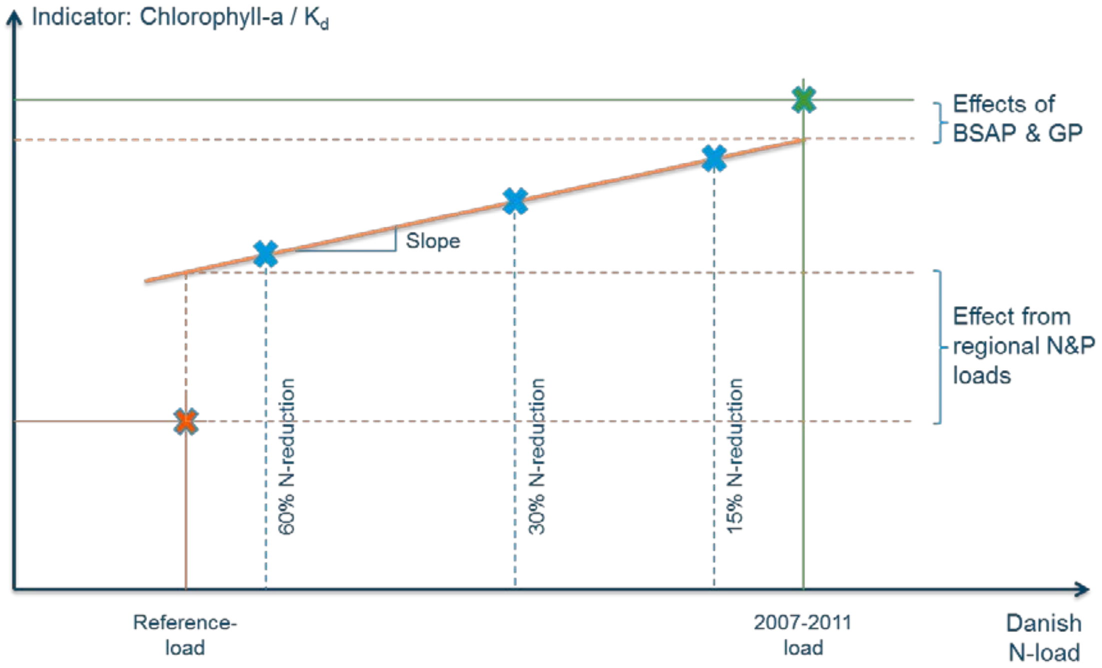


Figure 8.12. Schematic illustration of the surrogate models developed for each single water body covered by a mechanistic model. The y-axis show the value of the average indicator value (spatio-temporal average) covering the period 2007-2011. The x-axis show the Danish land-based N load. Loadings from other countries and atmosphere are also included in the model scenarios but not included in this this illustration (could be seen as a z-axis). The effects from regional N&P reductions according to reference conditions corresponds to the difference between reference condition and the trend line. Green mark show the status indicator value, red mark show modelled indicator value at reference loadings, including both Danish reference loads as well as regional reference load data. Blue markings show indicator values at 15%, 30% and 60% Danish land-based N reductions including also BSAP and GP implementation). Orange line show the local sensitivity to Danish land-based N loads and the difference between the green mark and the intercept between 2007-2011 loads and the trend line indicate the direct effects of the regional treaties BSAP and GP for that specific water body.

In *Figure 8.13* the developed surrogate models for TN, chlorophyll-*a* and K_d for water body No. 157 and 217 are shown. Notice that all years have now been averaged into one specific value for each scenario and each indicator for the specific water body addressed, and the average trend line applied, as described section 8.4.4. Furthermore, it is clear that within the scope of the models the assumption of linearity seems correct for water body no 217, whereas we see a minor discrepancy assuming linearity for chlorophyll-*a* in water body No. 157, due to the reasons just described. The waist majority of water bodies behave like water body No. 217, whereas less than a handful show similar features as for water body No. 157.

What is also evident in *Figure 8.13* is that for some water bodies the regional treaties do make an impact (the difference between status and intercept between trend line and 100% Danish land-based N loads). In water body No. 217 the trend line of both chlorophyll-*a*, K_d and TN does not intercept with the status line, and the difference between status and the indicator value at 100% Danish N load reveals the impact of BSAP and the Gothenburg Protocol in that specific water body, see schematic illustration in *Figure 8.12* and left panels in *Figure 8.13*.

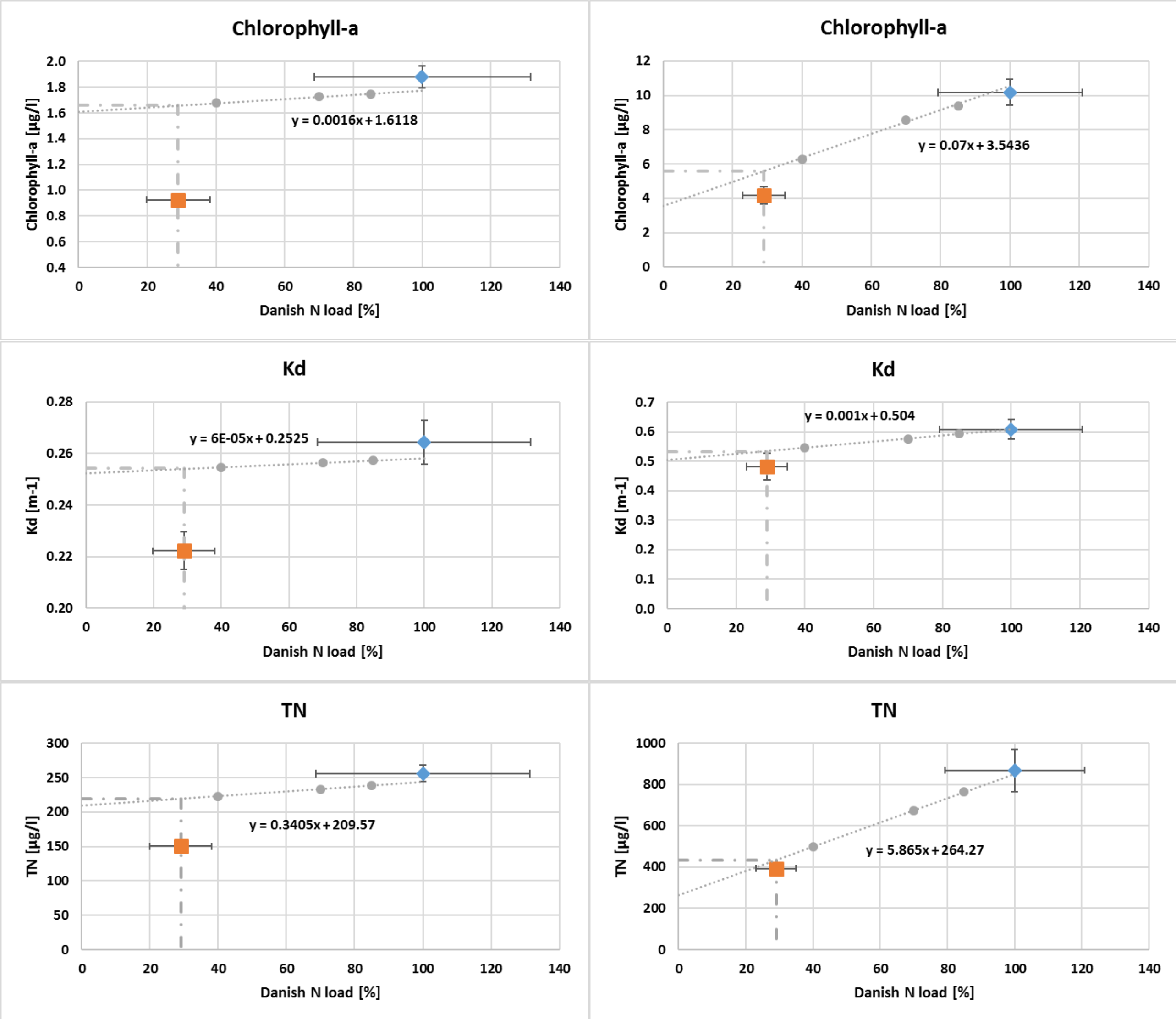


Figure 8.13. Impact correlations between Danish N loadings (StDev show the variability in loadings for the specific scenarios) and the two indicators chlorophyll-*a* (top panel) and K_d (middle panels) and the supporting parameter TN (bottom panels). Left figures are based on model results from water body No. 217 and right figures are based on results from water body No. 157. See *Figure 8.12* for detailed descriptions of the different values and symbols. Notice the differences in y-axis and intercept with x-axis.

In *Figure 8.13* we have included the intercept between Danish reference load conditions and the trend line from the three scenarios. In water body No. 157 (right panels) we see an intercept that is very close to the reference indicator values and reference TN concentrations (orange squares), indicating that the Danish land-based N load alone does explain a dominant part of the two indicators and TN in this water body. It does not alone explain the entire response, why we conclude that at reference conditions also P land-based loads and atmospheric N deposition impact the response.

However, in water body No. 217 we see much less of the modelled reference values explained by the Danish land-based N loads alone. The indicator values and TN concentrations in this water body is to a larger extent also impacted by nutrient loads originating from e.g. Germany, and other Baltic countries, as well as atmospheric N depositions, why the direct land-based impacts from Denmark is less pronounced.

This is of course important to keep in mind when assessing reduction targets to meet GES. In some water bodies one single country might not be in a position to ensure GES alone, and relays on nutrient reduction in neighbouring countries.

8.4.6 Proportion of GES explained by Danish land-based N loadings

As mentioned earlier, see section 8.2 we need a status, a target and a slope (or response) due to Danish land-based N reductions to be able to evaluate the reduction needed to obtain GES within each water body. The status is evaluated as the present situation, meaning the average of the indicator value over the years 2007-2012, and from the previous section we showed how to obtain the slope. Also, we have the targets, as described in section 8.1. However, as can be observed from especially the left panel in *Figure 8.13* it is not at all possible to obtain GES alone by reducing Danish land-based N loads for all water bodies. Even if we regulate the Danish loadings to reference condition this will not be sufficient to obtain GES, according to the model-tools applied. However, as the indicator values extracted from the reference model simulation (orange squares in *Figure 8.13*) indicate we do predict a much lower indicator value at reference conditions, when including reference conditions in all N and P sources around the Baltic Sea, including boundaries and atmospheric N depositions. Hence, the discrepancy between the slope intercept at reference loadings then illustrates that part of the modelled indicator values that is governed by land-based N and P loads from neighbouring countries, boundaries and atmospheric N deposition.

The mechanistic models includes – and responds to – nutrient loadings from all sources, but as the objective of this project was to develop reduction targets for Danish land-based N loadings we need to translate the model findings into a local context.

To do this we need to rearrange the data and model results presented in the previous section. The status, which corresponds to the model calibration, the modelled reference values and the slope estimated from *Figure 8.12* are kept unchanged, see *Figure 8.14*, and are therefore similar to the findings in *Figure 8.12*.

However, the trend line from *Figure 8.12* as been moved in parallel now having an intercept with the status indicator value. In *Figure 8.14* the original modelled indicator values at 15%, 30% and 60% reductions are included, but the trend line is moved.

We now work with slope intercepts at reference loads respectively status loads (status loads being 100% load). The corresponding indicator values now shows how much of the status (present days) indicator values that, according to the models, can be regulated by Danish land-based N loads, assuming we cannot regulate to less than reference loadings, see *Figure 8.14*. The remaining part of the indicator values then is account for by other nutrient sources and/ or other factors, like climate change.

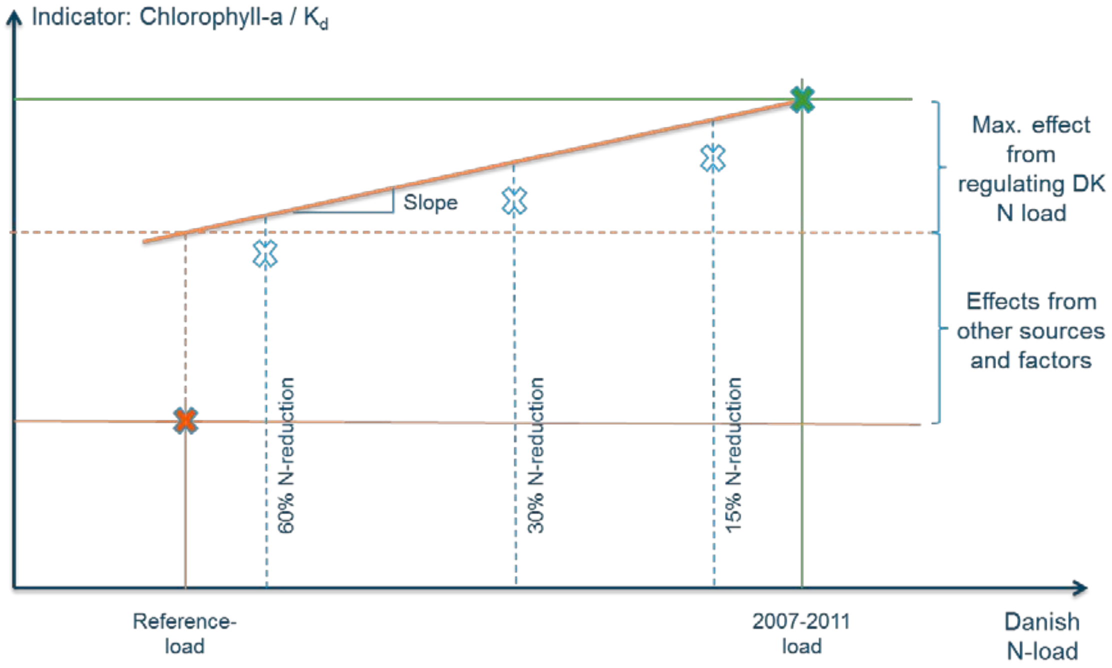


Figure 8.14. Schematic illustration of the surrogate models developed for each single water body describing the maximum effect achievable on the individual indicators from regulating Danish land-based N loadings alone. Orange line show the local sensitivity to Danish land-based N loads and is similar to the line in *Figure 8.12.* However, the orange line has been moved in parallel to intercept at status-load and status indicator value.

Target reference chlorophyll-*a* value versus modelled value

In *Figure 8.14* the modelled reference value (orange square) coincide with the water body type reference value (orange vertical line). However, this is not always the case. From section 8.1 the reference chlorophyll-*a* value used for setting the different target values (Type 1, Type 2 and Type 3) is estimated based on the average of several mechanistic and statistical modelled reference values, why the modelled reference value for the specific water body will not necessarily coincide exactly with the water body type value.

This is illustrated in *Figure 8.15*. In this figure we show the status value, the modelled reference value, the trend line intercepting with the status indicator value, and the water body type specific reference indicator value, as defined in section 8.1. Top panel show chlorophyll-*a* for the two water bodies No. 157 and No. 217. From the figure, it is obvious that the modelled indicator reference value and the water body reference value are not identical for the two water bodies, although they may be close. As the models are used to define the reference chlorophyll-*a* values (see section 8.1), this is expected, but for K_d, reference values originates from historical observations why larger discrepancy is expected here.

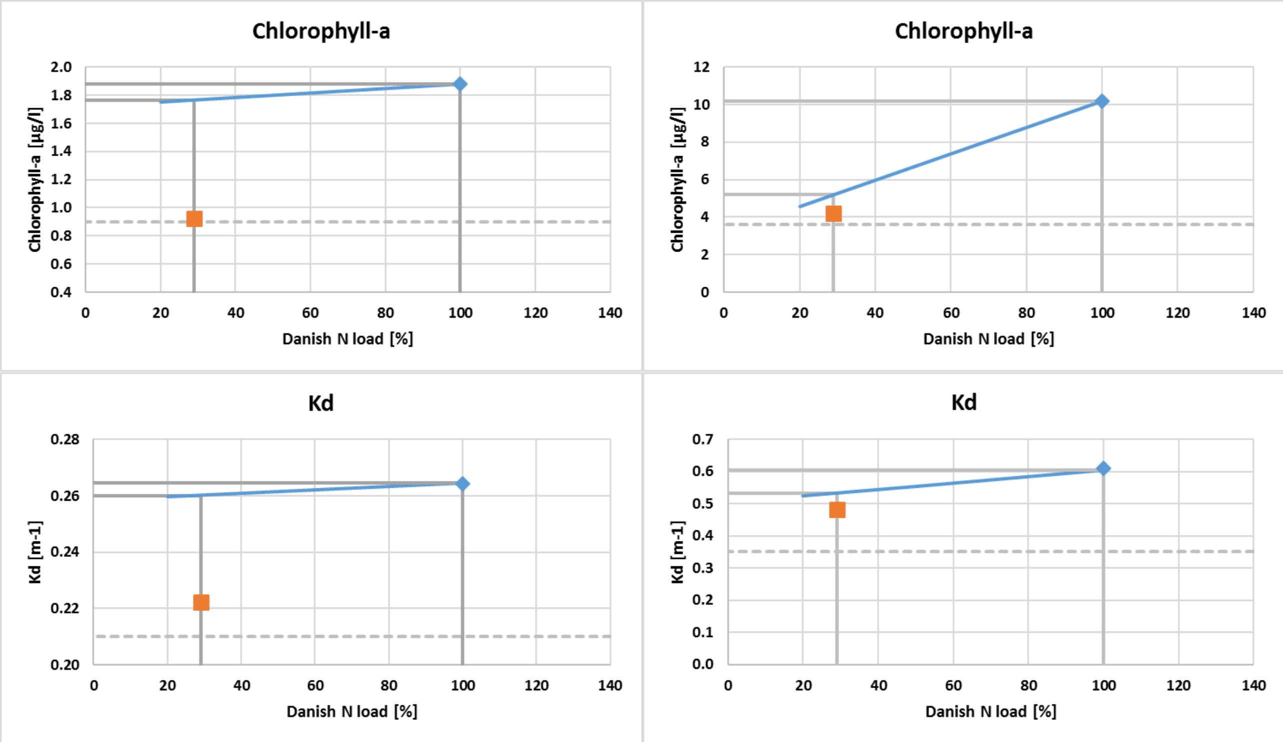


Figure 8.15. Impact correlations between Danish N loads and the two indicators chlorophyll-*a* (top panel) and K_d (bottom panels). Left figures are based on model results from water body No. 217 and right figures are based on results from water body No. 157. Blue dot corresponds to the status indicator value (present day model run), orange square correspond to the modelled reference indicator value. Blue line show the response to Danish land-based N loadings (the trend line) and the dotted horizontal line show the type specific reference value for the specific water body. Notice that y-axis vary between figures.

Target reference K_d value versus modelled value

Similarly, we see difference between modelled K_d at reference conditions and target reference values. However, the differences between the modelled and the targets reference values are generally larger than for chlorophyll-*a*, see bottom panels in *Figure 8.15*. In water body No. 217 we see a clear drop in K_d when modelling the reference value from more than 0.26 m^{-1} to approximately 0.22 m^{-1}. However, the target reference value is closer to 0.21 m^{-1}. In water body No. 157 the modelled drop in K_d is less pronounced and not very aligned with the target value.

In *Figure 8.15* target reference values are lower than the modelled values in both examples. This is, however, not the case for all water bodies. In general the modelled reference values are not that low as the target values, but for the water bodies covered by mechanistic models the target reference values are in average 0.23 m^{-1} (StDev=0.08 m^{-1}) and the corresponding modelled reference values are 0.25 m^{-1} (StDev = 0.06 m^{-1}).

In *Figure 8.16* the target reference K_d values are compared to the modelled reference K_d values. In the upper range of K_d values the model seems to under estimate, whereas the model slightly overestimate in the lower part of the K_d range. However, we should remember, that the target reference values are derived from observations of eelgrass depth limits, and not directly observations of K_d.

The average reference and modelled reference K_d values should be compared to an average modelled K_d status value of 0.33 m^{-1} (StDev = 0.13 m^{-1}) from the same water bodies. Hence, we see a rather large impact in K_d values, moving from status loads to reference loads, both in models and in the historical derived K_d parameter.

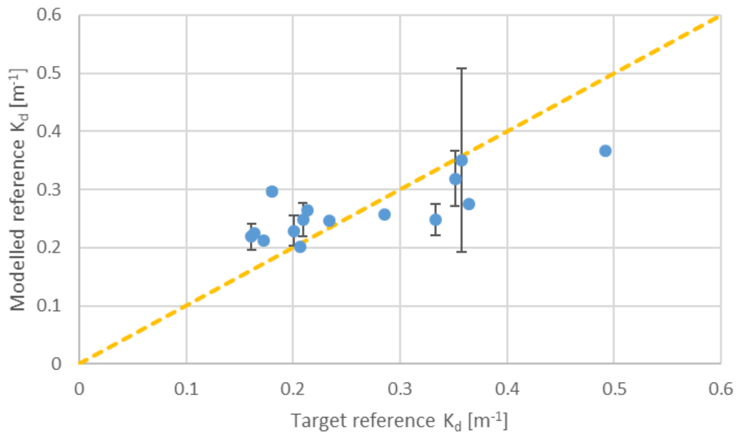


Figure 8.16. Reference K_d values derived from historical observations of eelgrass depth limits and corresponding modelled reference K_d values. Where more model results exist these values are averaged and include error bars of ± 1 StDev. Orange line show the 1:1 line.

K_d used for reference values are derived from historical data on eelgrass depth limit transformed to K_d, from the beginning of the 20ties century, or more than 100 years ago. Also, we compare water body average K_d values from the model to K_d values derived from observed eelgrass depth limits, and these depth limits do not necessarily represent the average water body depth limit but a value which is the maximum (depth limit or minimum K_d value) within that specific area.

However, the WFD target is still defined by the derived K_d values, and even though modelled and target reference values in average are close differences exists, why we use the target reference values when evaluating the part of GES that is explained by other nutrient sources and/or other factors than Danish land-based N loadings, see *Figure 8.14*.

Defining GES explained by Danish land-based N loadings

Applying the methodology described in the beginning of this section and summarized in *Figure 8.14*, and combining with the reference values from section 8.1, the part of the individual indicator that can be regulated from Danish land-based N loadings alone (down to reference load) can be estimated. In *Figure 8.17* the results are exemplified by chlorophyll-*a* in the water bodies covered by mechanistic models.

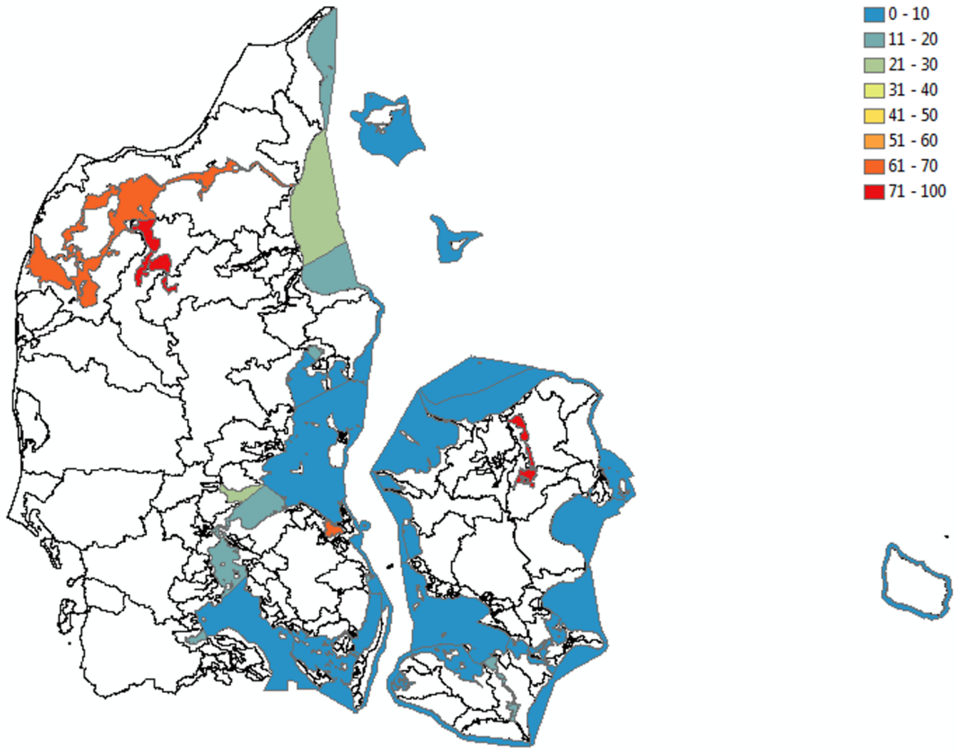


Figure 8.17. Proportion (in %) of chlorophyll-a, that can be regulated by Danish land-based N loads alone.

8.4.7 Sensitivity to Danish P load

As mentioned in section 8.4.4 the N reductions in Danish land-based N loads have been combined with 10-20% reduction in land-based P loads. The P reductions were estimated assuming improved P removal in wastewater treatment plans (WWTP) in the different catchments areas. The improved P removal was then included in the total land-based P loads and resulted in a 10-20% reductions in the local Danish P loads.

The sensitivity of summer-chlorophyll-*a* and summer-K_d to Danish P loads was then estimated applying same method as in section 8.4.5. The results (not shown) did not change slopes and intercepts within the different water bodies significantly, why we conclude that summer-chlorophyll-*a* and summer-K_d is controlled by N loads and P reductions within the range of 10-20% reductions does not impact the indicator values.

8.4.8 Estimation of target reductions from mechanistic models to obtain GES

Based on the results from the above descriptions we estimate reduction targets for the specific water bodies covered by the mechanistic models. The step-by-step method is illustrated in *Figure 8.18*. Basically, it contains 5 steps: i) Estimation of the observed status as described earlier (5-year average), ii) determination of the distance between the status and the target values for each of the two indicators, chlorophyll-*a* respectively K_d, iii) subtraction of the effect from regional treaties, if any (see *Figure 8.12* and left panels in *Figure 8.13*), iv) calculation of the part of the missing distance to the WFD target that can be regulated by Danish N load, and v) using the slopes from *Figure 8.12* to estimate the reduction targets for each indicator and for each individual water body covered by the models.

As we consider that some uncertainties exists in both method, status and WFD target for each of the water bodies and each of the indicators, we use the average of the chlorophyll-*a* and the K_d reduction targets to set the resulting reduction targets. Examples from water body No. 157 and No. 217 are included in *Table 8.8*. The numbers in *Table 8.8* originates from combining the distance between status and target values for chlorophyll-*a* and K_d with the slope from *Figure 8.13* and the proportion of the distance that can be regulated by Danish land-based N loads (exemplified in *Figure 8.17*).

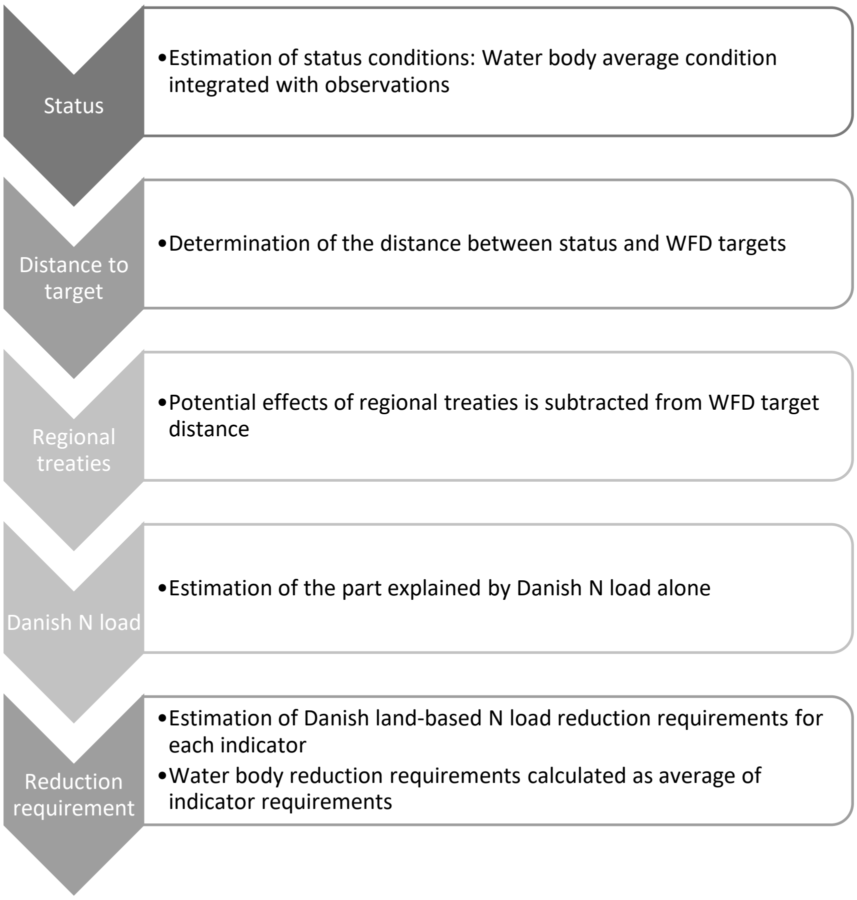


Figure 8.18. Step-by-step method used for calculating reduction targets for each specific water body covered by the mechanistic models.

Table 8.8. Examples of reduction targets calculated for each indicator as well as the resulting reduction targets, estimated for each water body covered by the models

Water body number	Reduction targets		Resulting reduction targets
	Chlorophyll-*a* based	K_d based	
157	66%	39%	52%
217	57%	20%	38%

8.4.9 Screening tool

To account for interactions between water bodies we have decided to lump reductions from the individual water bodies into common areal-reductions. Some variation in local need for reductions were estimated and most likely due to differences in estimations of status. However, to overcome these variations reductions were averaged within local areas, see *Figure 8.19*.

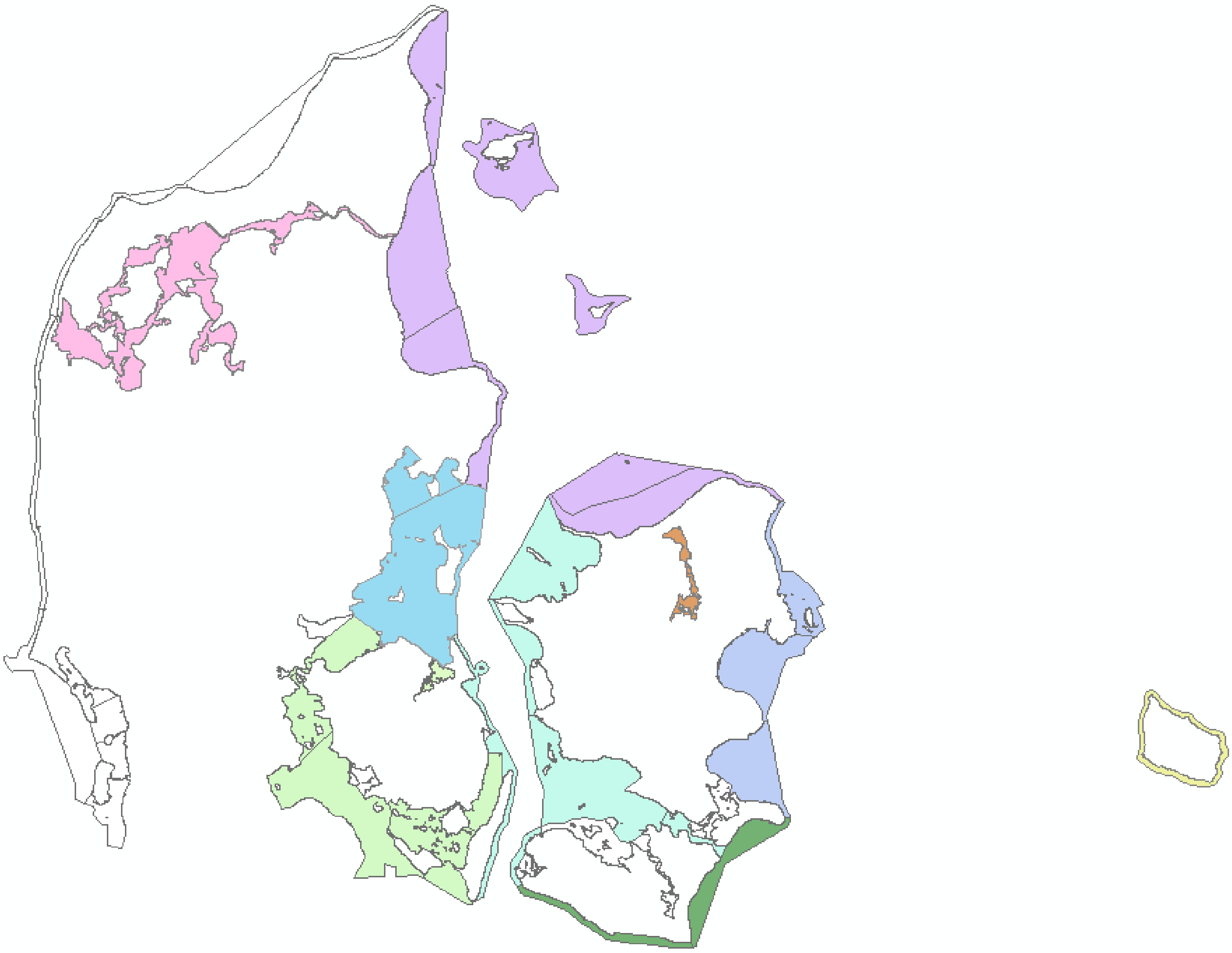


Figure 8.19. Map indicating local areas were individual estimated needs for reductions have been averaged.

8.5 Cause and effects between N loadings and indicators

In the previous sections, the development and application of statistical models and mechanistic models were presented. The two types of models are very different. Thus, statistical models are "black-box" models with a direct link to observations but without any descriptions of causal links. Mechanistic models are "white-box" models describing governing processes – to the best of our knowledge – and are linked to observations through calibration.

From the model development and model application, some of the most important results are the sensitivity of each indicator to changes in N loadings within each specific water body. This sensitivity is being expressed and quantified by the slope of the relation between N loading and an indicator (site-specific cause-effect relationships).

Although the mechanistic approach includes a Danish proportion of the controllable N loadings, the slope still represents the sensitivity of an indicator to changes in N loadings and we therefore tested the slopes (between N load and TN, Chlorophyll-*a*, and K_d) derived from both model types within the five different water body types (*Figure 8.20*). The slopes were tested using two-way ANOVA (model type and water body type) and the results showed that the slopes within each of the five water body types did not differ significantly ($p > 0.05$), suggesting no significant differences between model types, whereas the slopes varied significantly between the water body types ($p < 0.001$). However, especially in the open waters (Type 1) there was a tendency for the statistical models to predict higher slope values than the mechanistic models. This might be related to variation in loadings to the Baltic Sea, which cannot be separated from variations in the Danish loads in the statistical models (see *figure 7.4* for co-variation in loadings) and/or assumption regarding remineralisation of organics matter in the mechanistic models.

The results are in line with the results from the test carried out for reference chlorophyll-*a* values that did not differ significantly within each water body type. Hence, even though the nature of the model types differs pronouncedly, the slopes are very similar, which supports both the use of models for defining MAI and the application of water body types.

Figure 8.20 displays the slopes derived from the statistical and the mechanical modelling for each of the five water body types.

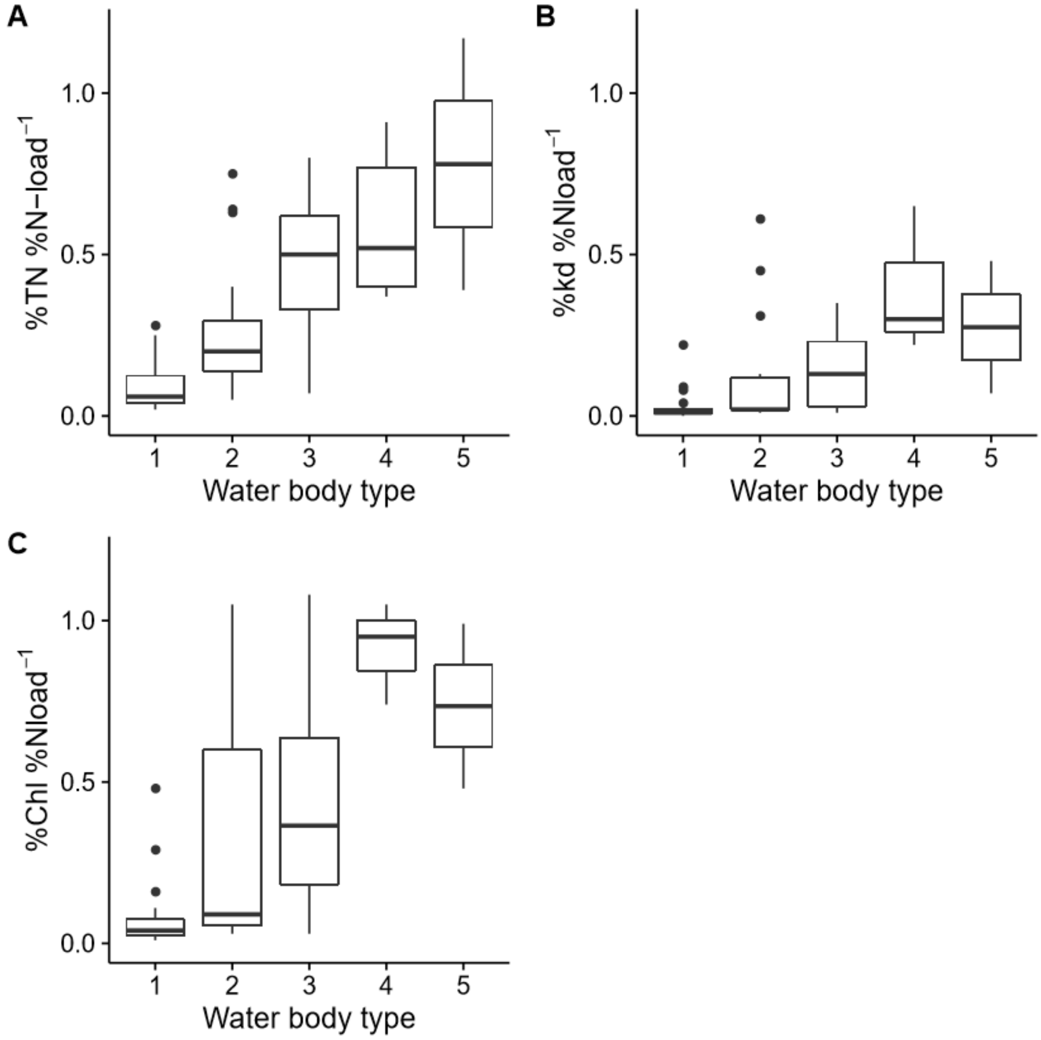


Figure 8.20. Slopes derived from statistical and the mechanistic models for each of the five water body types. A: relative change in TN concentration as a function of the relative change in nitrogen load from Danish catchments; B: relative change in light attenuation as a function of the relative change in nitrogen load from Danish catchments; C: relative change in chlorophyll-*a* concentration as a function of the relative change in nitrogen load from Danish catchments.

8.6 Meta models

The "meta model" approach is based on the assumption that water areas with the same characteristics will respond similarly to changes in, for example, nutrient loadings.

Use of meta models is very common within areas such as lake management (Janse et al. 2008) and can be the preferred choice if either data availability is not sufficient for development of site-specific models and/or if site-specific models are considered to be too uncertain.

For approximately 30, mainly smaller, water bodies, the monitoring data were too sparse to allow development of site-specific models or the IDW model was too coarse to resolve them. For these water bodies, we apply a meta model approach using the cause-effect relationship obtained for areas with similar characteristics (i.e. the same water body type). Thus, the basic assumption is that water bodies of the same type will respond similarly to changes in nutrient loadings and therefore type-specific cause-effect relations are applicable. From the findings described in the previous section, we concluded that the slopes derived from the different models did not vary significantly within the five water body types, whereas the slopes varied significantly between the five types. This consequently supports the assumption.

In order to calculate nutrient reduction targets, knowledge of the current status is necessary. Hence, an important prerequisite for applying the meta model approach is that sufficient observations exist for the specific water body to estimate a status value for the two indicators: chlorophyll-*a* and K_d. The statistical approach includes additional indicators and these are used if sufficient site-specific observations are available. Although the data requirement for obtaining status values is far less extensive than for the development of site-specific models, there are water bodies for which status values for the K_d and/or chlorophyll-*a* indicator cannot be established, and in these cases no site-specific reduction needs have been estimated.

For the water bodies where K_d and chlorophyll-*a* values can be established, nutrient reduction requirements are calculated using current status values based on monitoring data, type-specific environmental targets and meta models founded on statistical and mechanistic models. Depending on the type of model supplying the necessary data for a meta model, different approaches are applied (see sections below). The "resulting load reductions requirement" for each water body is calculated as an average of the results from the statistical and the mechanistic model approach.

The principles of the meta model methodology are:

- Estimation of status values for the indicators chlorophyll-*a* and K_d based on observations from 2007-2012. At least one year of sufficient observations within the period should be available.

- If the amount of observations is insufficient to determine status values for both the chlorophyll-*a* and the K_d indicator, no water body-specific reductions are estimated. If status values for additional indicators can be set, these will be included (statistical approach); however, additional indicators are not a prerequisite.

- As described earlier, we use K_d as indicator, where K_d targets are derived from eelgrass depth limit targets. In some of the meta model water bodies, water depth is shallower than the depth limit, or the observed eelgrass depth limit is larger than expected based on K_d observations. In these cases, we either transform actual water depth into corresponding K_d targets or assume no additional (K_d) reductions.

- Statistical and mechanistic meta model approaches are applied to estimate load reduction required per indicator, and the resulting load reduction required for each water body is calculated as an average of the results from the statistical and mechanistic approach.

8.6.1 Meta models based on the statistical model approach

The method for calculating nutrient reduction targets using statistical models builds on site-specific cause-effect relationships (slopes) between N loadings and indictors established from long-term time-series of monitoring data, as described in chapter 6. As the meta model water bodies all belong to Type 2 (semi-enclosed water bodies with low freshwater influence) or Type 3 (semi-enclosed water bodies with high freshwater influence), type-specific cause-effect relationships for these two categories were estimated as an average of slopes derived from statistical models developed for water bodies of the same type. The statistics on the resulting type-specific slopes between N loadings and the different indicators are shown in *Figure 8.21*.

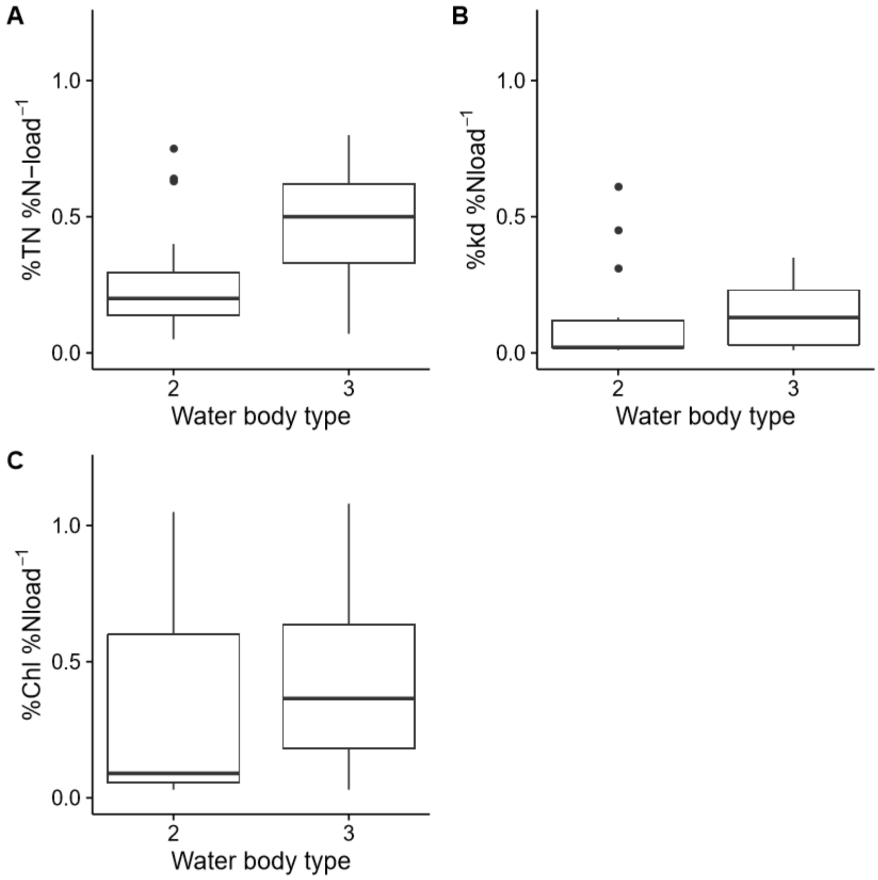


Figure 8.21. Statistics on the type-specific slopes for Type 2 and Type 3 water bodies between relative change in N loading and relative change in A) the yearly TN concentration, B) the K_d indicator and C) the chlorophyll-*a* indicator used in the meta models.

The type-specific slopes are used together with site-specific status and GM target values for each indicator to calculate the nutrient reduction needed to obtain GES for each indicator:

$$\text{Required load reduction}(\%) = 100 \cdot \left(\frac{\text{Status-GM target}}{\text{Status}}\right) \cdot \left(\frac{1}{\text{type-specific slope}}\right) \quad \text{Eq. 8.11}$$

This approach is similar to that used for water bodies with site-specific statistical models with the exception that the site-specific slope is replaced with a type-specific slope. The statistical model approach includes additional indicators besides K_d and chlorophyll-*a* to estimate the overall N reduction requirement for each water body as described in section 8.3. For the meta models, K_d and chlorophyll-*a* indicators were defined as a prerequisite for calculating nutrient reduction targets for meta model water bodies; however, for most meta-areas it was possible to include the indicator "occurrence of hypoxia" (section 8.3). The oxygen depletion indicator requires frequent observations to detect especially transitory oxygen depletion events. It was, however, assumed that hypoxia did not occur if there were no indications of low (< 4 mg l^{-1}) oxygen concentrations in the oxygen depletion period (July-Oct) during 2007-2012, even though the data requirement for the indicator was not always met.

If the limits for occurrence of hypoxia (see *Table 8.7*) were exceeded, this resulted in a required reduction of 25% in TN concentrations as described in section 8.3. This was translated into a reduction of N loadings by the use of type-specific relations between N load and TN concentrations.

For the remaining indicators (i.e. "ecological signs of hypoxia" and "N limitation") applied in the statistical approach (*Table 8.7*), the amount of data did not suffice to establish reliable indicator values in the meta-areas, and these indicators were therefore not included in the calculations.

The overall reduction needed for each water body is then calculated as a weighted average of the reduction needed for each of the three applied indicators, similar to the description in section 8.3 using the same weight as described in *Table 8.7*.

8.6.2 Meta models based on mechanistic models

The method developed applying mechanistic models builds on the established slopes (see section 8.4.5 and 8.4.6) and the proportion of the specific indicator that can be regulated by Danish land-based N loadings. These data are derived from the model scenarios described in section 8.4.4. The indicators involved in the meta models based on mechanistic modelling are chlorophyll-*a* and K_d.

When assessing the need for reduction in meta model water bodies, the site-specific data need to be transformed into general cause-effect data applicable for the meta model water bodies. Section 8.2 describes the overall method for estimation of the "required reduction needed" when applying mechanistic models. As explained, the immediate defined slopes (i.e. relations between load and status) have to be corrected for the proportion of the indicator that can be regulated by Danish land-based N loadings before application of the load/indicator slope to the specific water body. For the water bodies with site-specific models, a correction factor was estimated for each water body. For the meta models, common correction factors were defined for the two indicators based on an analysis of established slopes and correction factors for the water bodies covered by mechanistic models.

Figure 8.22 shows the slope (based on N scenarios) for chlorophyll-*a* and K_d as a function of the proportion (in %) of the indicator that can be regulated by Danish land-based N loadings. For chlorophyll-*a*, the data seem to be divided into two groups depending on the proportion of the indicator that can be regulated by Danish land-based N loadings – a group of water bodies with a proportion less than 40% and a group with a proportion larger than 40%. For K_d, a similar division into groups did not appear, see *Figure 8.22*.

By extracting the trend line slopes in *Figure 8.22*, we get three uniform meta-slopes, two for chlorophyll-*a* and one for K_d:

- Chlorophyll-*a* (<40% DK proportion): 0.84% chlorophyll-*a* per %N load (StDev = 0.30% chlorophyll a per %TN load)

- Chlorophyll-*a* (>40% DK proportion): 1.00% chlorophyll-*a* per % N load (StDev = 0.25% chlorophyll-*a* per %TN load).

- K_d: 0.74% K_d per %TN load (StDev = 0.43% K_d per %N load).

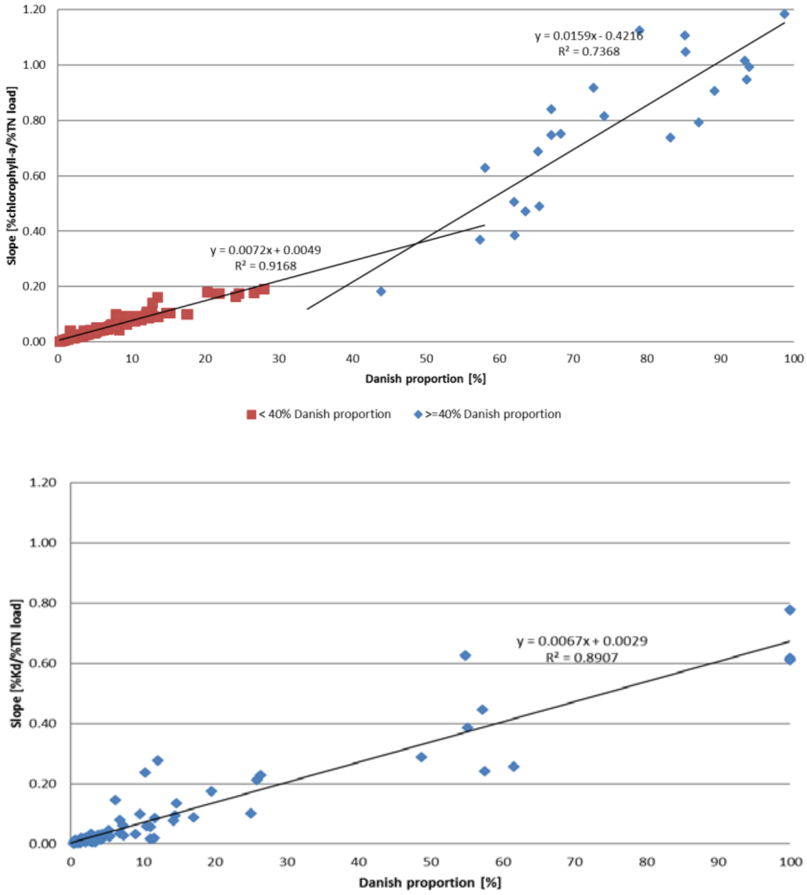


Figure 8.22. Slopes (based on N scenarios) for chlorophyll-*a* (top panel) and K_d (bottom panel), respectively, as a function of the proportion (in %) of the indicator that can be regulated by Danish land-based N loadings. Data are from water bodies covered by a mechanistic model.

8.6.3 North Sea meta model analysis

With respect to the North Sea water bodies, the data basis does not support the methodology described for mechanistic model-based meta model since biogeochemical modelling was not included in the study. However, GES has not been reached in any of the Danish water bodies in the North Sea and Skagerrak, and an approach taking limitation and differences into account has therefore been developed. Since the GM target for eelgrass depth limit is not defined for the North Sea water bodies, the methodology only includes the chlorophyll-*a* indicator. Furthermore, the chlorophyll-*a* indicator is defined differently for the North Sea waters (90% percentile of March to September chlorophyll-*a*).

In this approach, North Sea slopes were established assuming that the correction of slopes to take the proportion (in %) of the indicator that can be regulated by Danish land-based N loadings into account is applicable to both the North Sea and the inner Danish waters. However, as the chlorophyll-*a* meta-slope differs relative to the proportion that can be regulated by Danish land-based N loadings, estimates of the proportion are required.

Here, we use the hydrodynamic model and an advection-dispersion model to trace the different sources of loading. The model tracers are divided into four fractions: Danish land-based rivers, other rivers, initial values and boundaries. Following this, the different tracers are rated relative to the concentration of inorganic nitrogen, Danish land-based N loadings accordingly being 100% bioavailable, other rivers being 80%, initial values 0.8% and boundaries 0.4%. Based on these assumptions and tracer simulations, estimates of the proportion of the chlorophyll-*a* indicator that can be regulated by Danish land-based N loadings are made.

The described approach is subject to uncertainty. The turnover varies and the bioavailable fraction of nitrogen originating from the different sources is by nature difficult to estimate without a biogeochemical model. Furthermore, it was assumed that cause-effect relationships are similar for the North Sea water bodies and the water bodies of the inner Danish waters. Especially for the Wadden Sea, these assumptions are very rough. It may also be of importance that the correlation factors (%) for the water bodies of the inner Danish waters (*Figure 8.22*) are developed for summer averages (May-September), whereas the indicator for the North Sea and Skagerrak is defined as the 90% percentile of the March to September chlorophyll-*a* concentrations. Finally, we included only one indicator, whereas the resulting load reduction for the water bodies outside the North Sea and Skagerrak is based on both chlorophyll-*a* and K_d.

Generally, the estimates of resulting N load reductions for this part of the Danish water bodies are relatively uncertain. Looking ahead, the estimated resulting reductions needed in the North Sea and Skagerrak should be revised and updated in connection with the implementation of RBMP 2021-2027.

8.7 Integration of results

8.7.1 Overall principles

The above sections describes the methods for calculating reductions needed in land-based N loadings to each of the Danish WFD water bodies based on a mechanistic model and/or a statistical model approach; or a meta model approach.

Since the Danish water bodies are all more or less connected, the reduction needed for a single water body cannot be assessed in isolation. In addition, it is necessary to consider the load reduction requirement estimated for nearby water bodies. To make these considerations transparent, a set of rules was defined to guide the determination of final reduction needs and integrating the results into a nationwide reduction need:

1. For water bodies covered by a mechanistic model, the adopted reduction needed is the result of this model approach.
2. For upstream estuarine water bodies[18] without mechanistic models, the reduction needed is estimated using statistical models if such exist. Otherwise, a meta model approach is used.
3. For upstream water bodies for which a lower reduction is needed than for downstream water bodies, downstream reductions are applied (downstream approach).
4. For upstream estuarine water bodies without sufficient observations or models, the reduction needed for downstream water bodies is adopted (neighbour approach).

[18] Some estuaries are divided into two water bodies, a so-called inner (upstream) water body and an outer (downstream) water body.

Table 8.9. Examples of application of the developed rules for integrating the results.

Name and no. of water body	Model used for defining the reduction needed	Reduction
Kattegat, Aalborg Bugt (no. 222)	Mechanistic model approach	A mechanistic model exists and the resulting reduction needed is estimated to 7%.
Horsens Fjord, upstream (no. 128)	Statistical model approach	A statistical model exists and the resulting reduction needed is estimated to 50%.
Bjørnholms Bugt, Riisgårde Bredning, Skive Fjord og Lovns Bredning (no. 157)	Mechanistic model approach	Both mechanistic and statistical model exist. According to the mechanistic model, the estimated resulting load reduction required is 48%.
Vejle Fjord, downstream (no. 122)	Downstream approach	According to the statistical model, a reduction of 15% is needed. However, the downstream water body, Nordlige Lillebælt (no. 224), requires a 39% reduction, and the resulting load reduction required is thus set to 39% for Vejle Fjord.
Karrebæk Fjord (no. 35)	Meta model approach	No mechanistic or statistical model exist, and the load reduction is therefore estimated to 38% by averaging the results of mechanistic (32%) and statistical (45%) meta models.
Lillestrand (no. 62)	No status, no models, Neighbour approach	Reductions needed were determined to 11% for the downstream water body Århus Bugt syd, Samsø, and Nordlige Bælthav (no. 219), and the load reduction required for Lillestrand is therefore set to 11%

In the Limfjorden, the inner parts of the estuary requires larger % reductions than the outer parts. As the flow in the inner part is predominantly outgoing and loadings are significant, the absolute reductions in the inner parts actually diminish the absolute need for reductions in the outer parts. Hence, taking this into account, the reductions have been optimised for the Limfjorden.

Applying the developed toolbox and methodologies, final reductions needed were determined for all 119 Danish WFD water bodies and consequently the demand for nitrogen reduction measures in their catchments. *Figure 8.23* shows the water bodies (grey areas) and their catchments with indications of the demands for nitrogen reductions from the catchments to the water bodies. The overall pattern is that the most intensive efforts are required for catchments discharging into the estuaries with the lowest water exchange. Especially for catchments discharging to the inner part of the Limfjorden, Mariager Fjord, and a number of the estuaries on the east coast of Jutland and Odense Fjord (all the dark green areas), the estimated required load reduction leads to a strong demands for reductions in nitrogen discharges (40-75%). On Zealand, in the eastern part of Denmark, the demands are generally lower. The catchments of Karrebæk Fjord and Præstø Fjord have the largest reduction requirement, calculated to 20-30%, and most catchments on Zealand have a load reduction demand <10%.

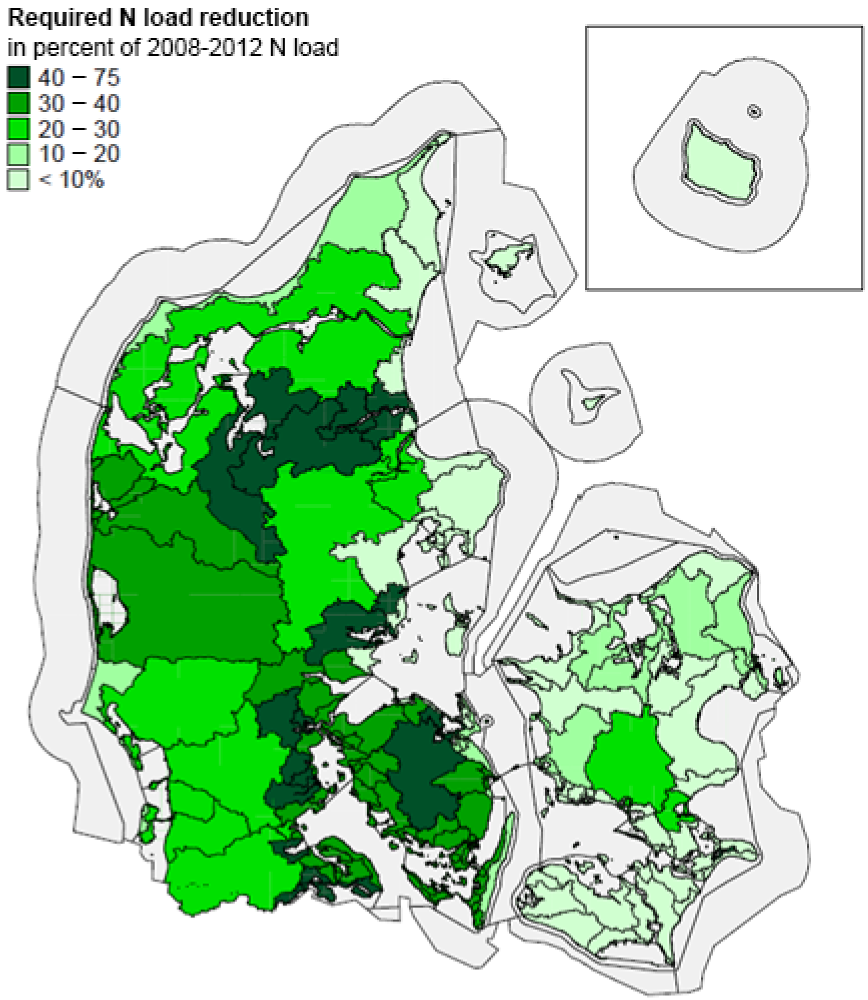


Figure 8.23. Resulting reduction needed based on the models and methods described in the present report.

8.7.2 From estimated load reductions to MAI

The central result of the model applications is the load reduction percentages estimated for the 119 Danish WFD water bodies. The difference between status and targets and the casual relationship between status of indicators and nitrogen loads derived from the developed statistical, mechanistic and meta models with the developed methodologies for post-processing of model data are crucial for these estimates.

However, from an administrative point of view, it is the maximum allowable loads (MAI) that define the efforts required in the catchments. Consequently, the estimated reduction needs have been transferred into MAI:

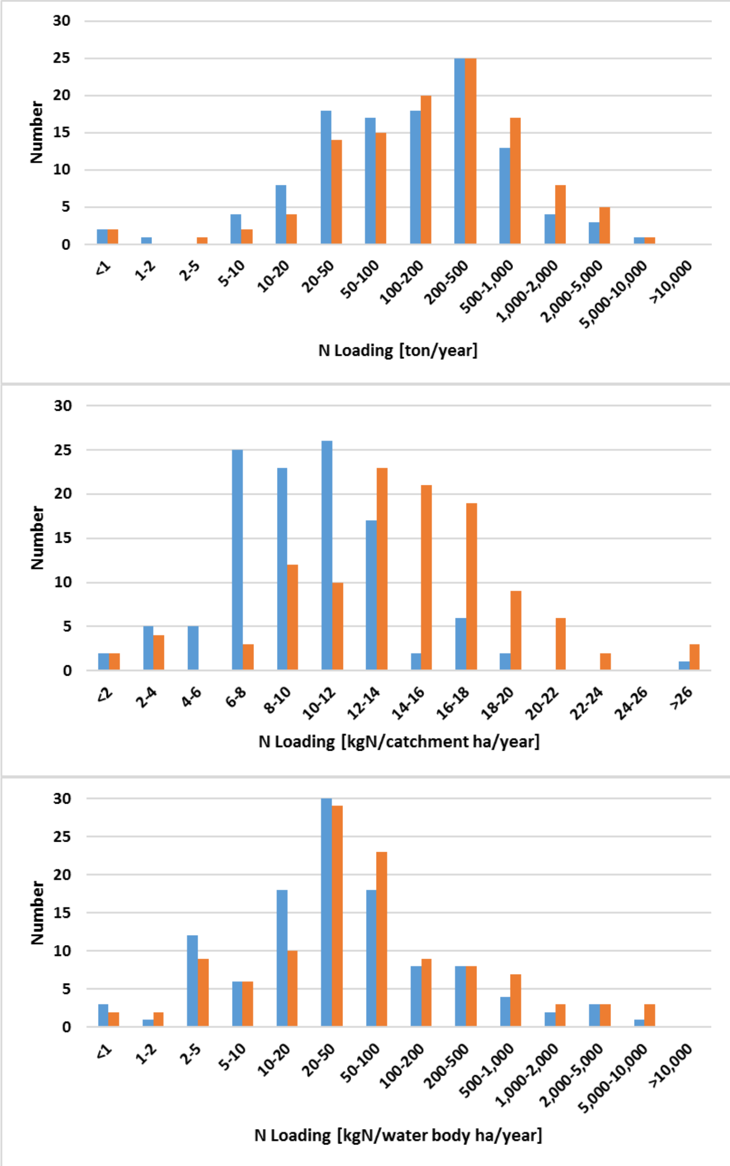
$$MAI\left(\frac{ton\ N}{year}\right) = \left(1 - \frac{Reduction\ needed\ (in\ \%)}{100}\right) * Status\ load \qquad \text{Eq. 8.12}$$

where the status N load is defined as the average load over the period 2007 to 2012. The use of a 6-year average provides a more robust estimate as it incorporates the year-to-year variation in meteorological conditions. In theory, MAI is independent of the actual loadings; however, this is only the case if we assume e.g. an unchanged climate.

In *Figure 8.24*, the distribution of MAIs is compared with the status N loads. The most obvious differences can be observed in the middle panel showing (catchment-based) areal loadings. The impact from the estimated reductions is clear, and the vast majority of catchments have a MAI between 8 and 12 kg N ha^{-1}, whereas most current loadings range between 14 and 18 kg N ha^{-1}.

Similarly, MAIs per water body area show trends towards lower kg N ha^{-1}, although the pattern is less clear, and the variety in both catchment-based loadings and water body-based loadings remains.

Figure 8.24. Average MAI (blue bars) and status (2007-2012) loadings (orange bars) divided into water bodies. Top panel is the yearly loadings per water body, middle panel is the areal loadings per catchment and bottom panel the areal loadings per water body.



Adding up the MAIs for the individual water bodies, results in a total MAI for Danish catchments of 42 kton N year^{-1}. When this is compared with the average Danish N loading for the years 2007 to 2012 (61 kton N year^{-1}), it corresponds to a total reduction in nitrogen loadings of 19 kton N year^{-1}.

8.8 Model uncertainty and sensitivity analysis

8.8.1 Quantification of model uncertainty

Two model approaches, mechanistic and statistical models, have been developed and applied to estimate nutrient reduction requirements to fulfil GES in all 119 Danish WFD water bodies. The mechanistic models were mainly applied in the open waters whereas the statistical models were mainly used in estuaries and coastal areas. However, for a few water bodies (11) both modelling approaches were applied making it possible to compare the two model results as part of an internal evaluation as well as to obtain a measure of the uncertainty level in the model predictions (*Table 8.10*).

Since both model approaches utilize the same primary data for estimating status and target values in the model predictions, the "uncertainty analysis" only account for the model associated uncertainty and not uncertainties related to status values or target values.

To assess the results from the two approaches we compared the predicted N-reduction requirement (in % of the current loading, see *Table 8.10*), since these figures have equal variance which is a prerequisite for the statistical analysis.

Table 8.10. Required Load reduction in percentage of current nitrogen loadings for the 11 water bodies where both modelling approaches were used.

Water body no.	Type	Reduction % Mechanistic	Reduction % Statistical	Mean	Abs. difference
2	3	4	11	8	7
44	1	0	18	9	18
92	3	23	26	25	3
96	1	34	44	39	10
102	2	41	50	46	9
147	1	7	2	5	5
156	3	37	31	34	6
157	3	52	60	56	8
214	1	30	40	35	10
216	1	36	32	34	4
224	1	56	58	57	2

The effect of model approach and water body, respectively, was tested using a paired t-test which showed no significant difference between the model approaches ($p = 0.06$). The mean difference is 4.7 ± 5.0% reduction (mean ± 95% CI, df =10). The mean for each of the models are 29.1% reduction and 33.8% reduction for the mechanistic and statistical approach, respectively (*Figure 8.25*).

A mixed model was used to test the effect of model approach (fixed effect) and water body (random effect) or type (random effect), but due to lack of significant effect ($p = 0.06$) of approach, the model was reduced to a one-way ANOVA where each model result was interpreted as an independent sample on each station, and equal variance was assumed. Given these assumptions we would apply the residual standard error at 6.06% reduction to all the waterbody areas. With 10 degrees of freedom this would result in a 95% confidence interval at ± 13.5% reduction for each water body.

The estimated 95% confidence interval of ± 13.3% should be considered as a lower limit since the assumption of independency might not be fulfilled. Especially the neighbouring water bodies (water body 156 and 157) might be correlated.

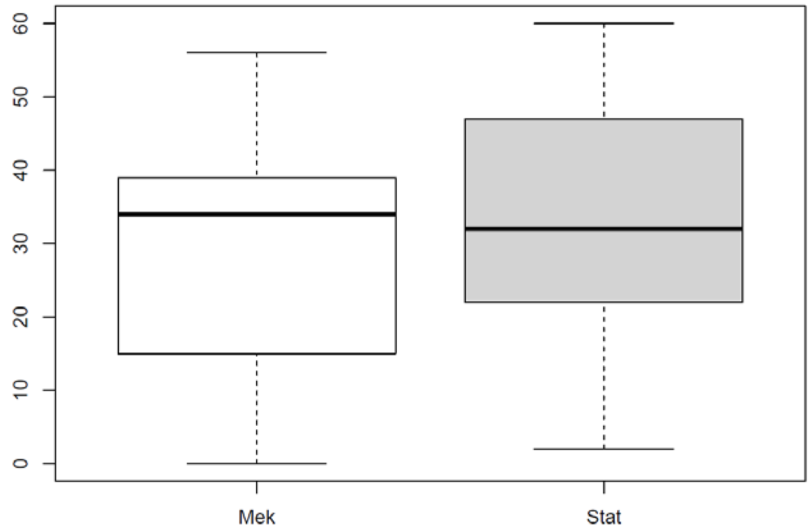


Figure 8.25. Boxplot of the load reduction results for the ensemble modelled water bodies estimated using the mechanistic model approach (Mek) and the statistical model approach (Stat).

Although the two approaches used to calculate N reductions are very different both in terms of model type as well as in the post-processing procedure, the predicted N-reduction results are not significantly different based on the 11 ensemble-modelled areas. Also, the results from the two model approaches seem to coincide very well at water body level. The largest difference was observed in area 44 – a large 360 km^2 open water body type – with the smallest catchment-to-water body area ratio of the 11 water bodies, which indirectly implies that area 44 receives most of its water and nutrients from the Baltic Sea. In contrast to the statistical approach, the mechanistic modelling also incorporate future N and P reductions resulting from the BSAP and GP. Therefore, the largest differences in the mechanical and statistical approaches is expected in the open water types (Type 1) most affected by reductions in N loadings to the Baltic Sea.

8.8.2 Sensitivity to status, targets and slopes

To be able to estimate the nutrient reductions needed to fulfil GES in all Danish water bodies, basically three input parameters are needed: i) The status of each indicator in every specific water body, ii) the corresponding targets and iii) the indicator sensitivity to Danish land-based N-loadings (i.e. the slope between N-load and indicator). These three input parameters define the need for reduction for each indicator, as described in section 8.2. Briefly, we want to examine (a) how sensitive the estimated load reduction is to changes in individual parameter values; and (b) which parameters have most influence on specific output variables

To evaluate the contribution of input parameters to the level of nutrient reduction required, a range of sensitivity tests were carried out encompassing all water bodies covered by the a mechanistic or a statistical model. Tests were carried out by changing the status, targets and the slopes by ±10% and observing the response in the %-wise nutrient reductions. Sensitivity tests were carried out for the chlorophyll-*a* and K_d indicators.

Assessing the sensitivity over the broad range of water bodies reveals that the final N-load reductions generally are most sensitive to variations in target and status values and less sensitive to variations in slopes.

Varying the slopes by ±10% lead to changes in N-load reductions of ±2-3% on average, whereas changes in status and target values of ±10% lead to changes in load reductions of ± 10-11% on average.

When estimating need for reductions we based the status primarily on observations, hence uncertainties in observations will be transferred to the load reduction estimates.

With respect to targets, the N-load reductions were sensitive to the targets. The K_d targets were based on historical observations of eelgrass depth limit and have not been assessed further in this report. The chlorophyll-*a* targets were derived from models developed in this study, and hence, they indirectly depend on slopes.

9 Discussion

The overarching objective of the model development project initiated in 2012 was to support the implementation of the WFD in Danish coastal waters by providing modelling tools and methods for calculation of the maximum allowable nutrient input (MAI), which would ensure the maintenance/achievement of Good Ecological Status (GES) as required by the WFD.

The objective was divided into the following sub-objectives:

- To develop a toolbox for improving the Danish River Basin Management Plan 2015-2021 (RBMP), including development of indicators for biological quality elements and supporting indicators.

- To ensure maximum coverage of Danish water bodies, including areas with no or few observations.

- To base the development on state-of-the-art knowledge.

The scientific basis is documented by this report and the expert panel performing the evaluation has been instructed to evaluate the tool development and application as well as the specific use for setting chlorophyll-*a* targets and calculating the load reduction requirements from Danish catchments. Furthermore, the evaluation panel shall assess other relevant pressures such as phosphorous loads, fisheries activity etc.

9.1 Environmental pressures

As stated in the introduction coastal ecosystems worldwide have been under extensive anthropogenic pressure during the last half of the 20th-century and the list of potential pressures is substantial and include nutrient enrichments, exploitation of coastal resources, overfishing, destruction of habitats and chemical pollution (Boesch 2002). The relative importance of these pressures do, however, vary in both time and space and assessments of the impacts require a suite of indicators targeting the different pressures.

The EU-water directives (WFD, descriptor 5 in MSFD, Nitrate Directive) are addressing eutrophication as a main pressure for marine ecosystems, and marine WFD indicators are expected to respond to eutrophication and hence nutrient loadings as such. Analogously, D5 of the MSFD is developed to quantify degradation of ecosystems due to eutrophication and thus applicable to the management of these ecosystems. A number of other pressures, important to the overall ecosystem and ecosystem functioning are handled by other descriptors (besides D5) included in the MSFD.

In Denmark, three indicators have been developed to monitor the progress towards GES within the Danish WFD water bodies: Chlorophyll-*a*, eelgrass depth limit and a benthic fauna index (DKI), which are all expected to respond to eutrophication. However, other pressures are known also to influence especially eelgrass and DKI.

For the model indicators i.e. summer chlorophyll-*a* and the summer K_d, our analyses show, that N is the most important nutrient pressure and the RBMP

2015-2021 assesses N reduction needs and corresponding MAI do not include other pressures.

However, in WFD marine water bodies other pressures exist potentially affecting the WFD indicators. Pressures considered most relevant for the WFD indicators will briefly be discussed below.

9.1.1 Nutrients

Numerous studies have highlighted nutrient enrichment as the main driver/trigger of the ecosystem degradation, and the nutrient enrichment is reported as having altered a number of coastal ecosystems of developed nations in Europe, North America, Asia and Oceania (Boesch 2002; Conley et al. 2009a; Cloern 2001; Kemp et al. 2005). However, the limiting nutrient differs between water bodies and between seasons. In general, the importance of phosphorus decreases when moving from typically P-limited freshwater systems towards N-limited marine systems with estuaries, and coastal waters positioned at the transition experience varying degrees of both N and P limitation (Seitzinger et al. 2006).

Nitrogen

Today, there is a strong consensus within the scientific community that N is the primary cause of eutrophication in many coastal ecosystems (e.g. Howarth & Marino 2006; Seitzinger et al. 2006), but Howarth & Marino (2006) also pointed out that optimal management of coastal eutrophication should include control of both N and P, because P may limit primary production in some systems and during specific periods of the year.

Nitrogen fixation by bloom-forming filamentous cyanobacteria is important for both N cycling and N budget in the Baltic Sea (Bianchi et al. 2000), but N-fixation rarely occurs at salinities above 10-12 psu (Howarth & Marino 2006). As the inner Danish waters are located in the transition zone between two large water bodies, the brackish Baltic Sea (salinity ≈ 7-8 psu) and the saline North Sea (salinity ≈ 34 psu), large parts of the Danish water bodies have salinities above 12 psu and (see *Figure 9.1*) thus do not support the growth of and N-fixation by the bloom-forming filamentous cyanobacteria that are abundant in the Baltic Proper. From *Figure 9.1* it is evident that only few water bodies have salinities where N fixation potentially might occur during summer. Besides the water body around Bornholm, only few eastern water bodies in the Fehmarn Belt and the Sound reaches average salinities below 12 psu.

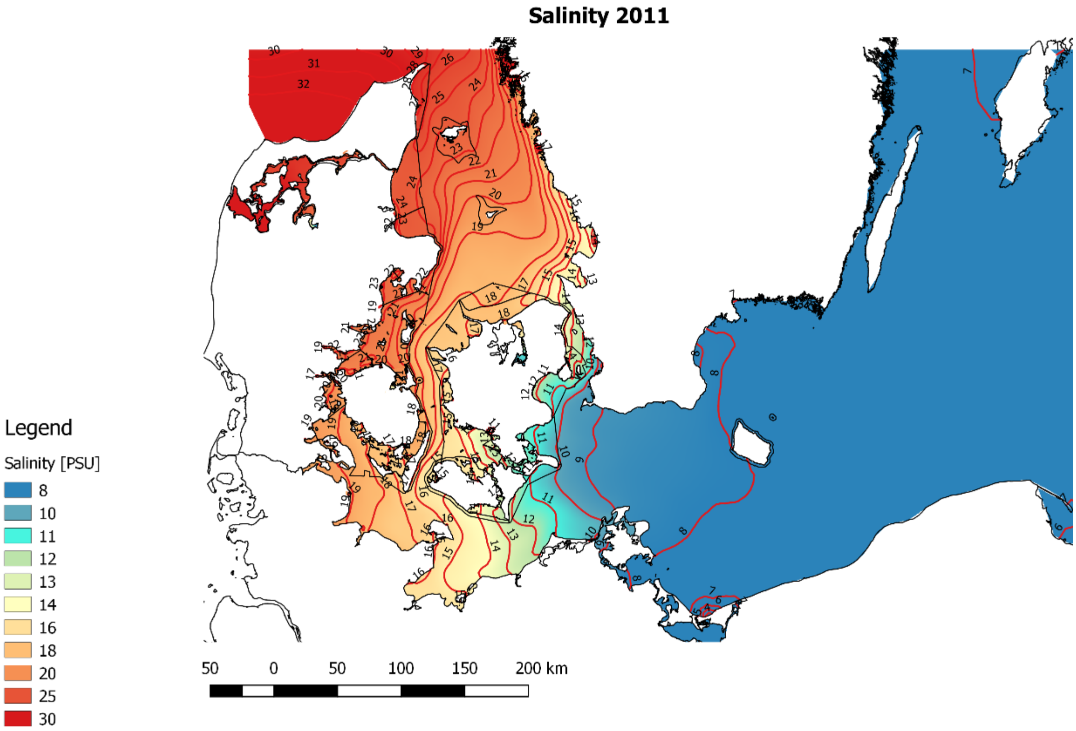


Figure 9.1. Modelled (IDW model) average yearly surface salinity, year 2011.

Phosphorus

In most Danish coastal and estuarine areas, P is limiting primary production in the spring, whereas N is the most limiting nutrient during summer and autumn. The intercalibrated chlorophyll-*a* indicator is a measure of summer (i.e. May through September) chlorophyll-*a* concentration, thus coinciding with the N-limited period. This is also evident from the modelling where N and not P turns out to be the most important nutrient. Although the intercalibrated WFD indicators in most areas do not appear to be sensitive towards P loadings, the importance of P for coastal eutrophication should be addressed and quantified. P loadings will most likely affect spring primary production with direct and indirect effects on eutrophication levels in Danish estuaries. In order to quantify the importance of P and the potential interactions with N loadings, additional indicators which are sensitive toward P and P loadings have to be developed and included in future WFD assessments.

9.1.2 Fishery

Fishing activities affects marine ecosystems both directly by removal of fish and physical disturbance as well as indirectly through trophic cascade effects and deterioration of benthic habitats resulting in e.g. loss of benthic biodiversity, reduced benthic vegetation and increased resuspension (Botsford et al. 1997).

Several studies, some from Danish areas, have documented that fishery affects stock size of target fish and that overfishing can lead to population collapse and that the effects of fish removal may propagate through the food chain (Polis et al. 2000) affecting the lower trophic levels including phytoplankton biomass. A recent modelling study covering the open inner Danish waters suggest that changes in zooplankton grazing pressure (e.g. as a result of fishery) may have top down effects on summer chlorophyll-*a* concentrations (Maar 2014; Petersen et al. 2017). It has not been assessed if coastal fishery (after mainly blue mussels and finfish) is able to induce trophic cascade effects.

Dredging (mainly for blue mussels) and bottom trawling occurs in Danish marine WFD areas (*Figure 9.2*) with potential impact on benthic habitats including eelgrass (Erftemeijer & Lewis 2006) and benthic fauna (Thrush et al. 1998; Tillin et al. 2006).

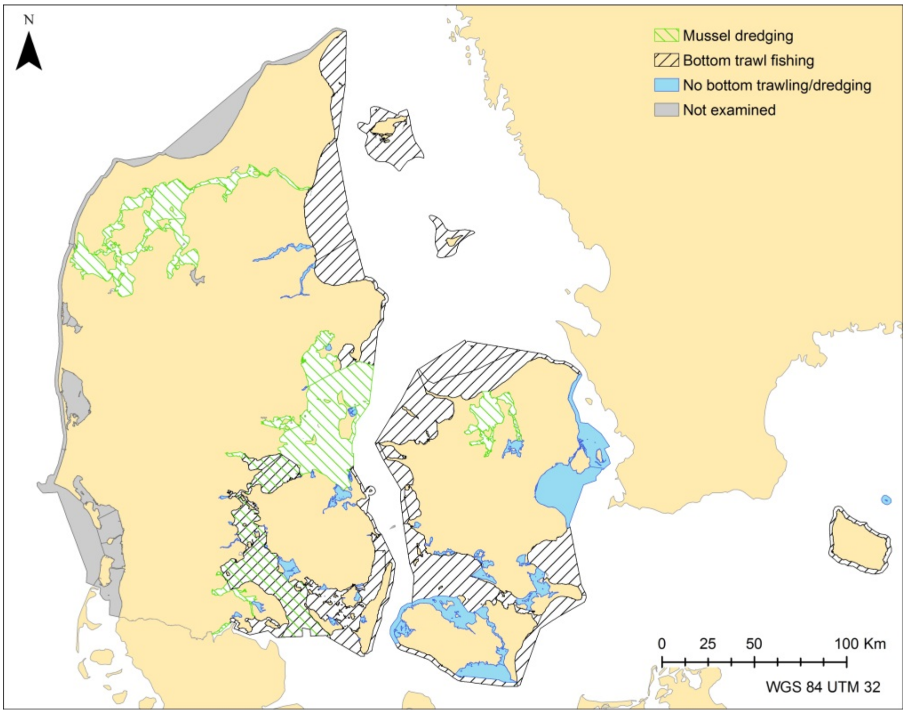


Figure 9.2. Danish WFD water bodies affected by mussel dredging and bottom trawl fishing.

Although eelgrass and benthic fauna are not directly included in the estimation of MAI for Danish WFD areas, they are important components of the Danish WFD assessment of environmental quality, hence identification of pressures such as dredging and quantification of their impact on benthic organisms is important for the Danish WFD implementation.

The influence of dredging on eelgrass distribution in Danish WFD areas was examined in a habitat GIS modelling study based on observations of dredging activity (during 2006-2013) and potential eelgrass habitats. The results revealed that in 10 of the examined water bodies there was a physical overlap between dredging areas and potential eelgrass habitats. Although the overlapping area was small compared to the area of potential eelgrass habitats, dredging might have impacted eelgrass in these areas. However, in the vast majority of locations, factors controlling eelgrass habitats (e.g. light availability and sediment characteristics) is limiting the spatial distribution of eelgrass and not dredging (Timmermann et al. 2015).

The indirect effects of dredging i.e. increased resuspension resulting in increased light attenuation and subsequent potential decreased light availability for eelgrass growth (Neckles et al. 2005), are expected to vary with dredging activity and sediment characteristics and have not yet been quantified for Danish WFD areas.

Dredging and trawling is known to be an important stressor for benthic organisms with both short and long term effects on benthic diversity, biomass and community compositions. In the deeper parts of Kattegat, which have been intensively trawled for the last 80 years (Pommer et al. 2016), a recent analysis suggests that benthic biodiversity is affected by trawling at least in the deeper and muddy part of Kattegat and that the WFD indicator DKI is sensitive towards effects of trawling (Hansen et al. 2015). DKI is currently being explicitly tested for use in the open water under the MSFD. A significant negative correlation between DKI and fishing intensity has also been established for shallow water habitats resembling those occurring in the WFD-areas (Hansen et al. 2015). However, the shallow areas in the Kattegat cover several ecological gradients that are not easily disentangled from effect of bottom trawling partly due to the low spatial resolution of available fishing intensity maps. It is, however, not clear to what extent trawling has an effect on the quantitative ecological status classification in terms of DKI in WFD areas. WFD areas are generally shallower and more exposed to wave action and resuspension compared to the deeper parts of Kattegat and benthic communities in WFD areas are presumably more adapted to physical disturbance making them less sensitive toward trawling as observed in the North Sea (Van Denderen et al. 2014).

9.1.3 Climate change – future and past

Climate changes are also considered to be a pressure that potentially may influence the ecological status of the different coastal waters. Especially, changes in freshwater input due to changes in precipitation and changes in water temperature are expected to be important future pressures.

The impact that future changes in climate and other anthropogenic drivers together will have on the biogeochemical cycles in the Baltic Sea is unclear (HELCOM, 2013), but modelling studies suggest that in a predicted future climate water quality, characterized by ecological quality indicators (e.g. summer bottom oxygen and annual mean phytoplankton concentration), will be deteriorated compared to present conditions (Meier et al. 2012). The goal of the WFD is to obtain GES within the coming 4 to 10 years and assessing climate changes on these short time scales becomes speculative, and hence we have not addressed climate changes as a pressure in the development of the tools for the Danish RBMP 2015-2021.

Climate changes have occurred over the past 100 years and these changes might interfere with the possibilities of ecosystems to return to an acceptable deviation from a pre-existing reference condition (Duarte et al. 2009) as implicitly assumed and required by e.g. the WFD.

In Denmark significant changes have been reported since the beginning of the last century. According to (Grøndahl et al. 2014) the most significant changes are observed in the western part of Denmark where precipitation has increased by 26% (from ~800 mm year^{-1}) and yearly average temperatures have risen by 1.3 °C from 1875 to 2010.

The assessment of GES in Denmark and the calculations of MAI partly builds on observations of eelgrass depth limit from the around year 1900 (Krause-Jensen et al. 2005; Krause-Jensen & Rasmussen 2009) and the historical changes in e.g. sea temperature, precipitation and run-off most likely hamper the recovery of eelgrass as well as the capacity for eelgrass to return towards the reference situation. It is, however, unclear how effects of historical changes in e.g. climate should be handled within the WFD and we have not addressed the issue in the present project.

9.1.4 Other pressures

Do other pressures exist? This was also a question for the Nitrogen Working Group back in 2012. From the working group outcome (Nitrogen Working Group 2012) it was concluded that fishery, sand mining, sediment disposal and maintenance of navigation channels could potentially affect especially eelgrass recovery and abundance. Fishery has briefly been assessed above and no other assessment has been carried out.

The working group also discussed xenobiotics as a potential pressure on eelgrass and chlorophyll-*a*. The conclusion from the working group was that in general xenobiotics do not pose a risk to the marine environmental status although some locations might be impacted locally.

9.2 Indicators

9.2.1 Chlorophyll-*a* targets

Due to lack of historical data, chlorophyll-*a* reference values and corresponding chlorophyll-*a* targets were estimated using statistical and mechanistic models.

By forcing the models with ”reference nutrient loadings” the corresponding ”reference” summer chlorophyll-*a* values were modelled for water bodies south of Skagen. However, as reference N loadings on average correspond to less than 30% of the 2007-2011 N loadings, the models were used to extrapolate far from the range used during development and calibration. This invariably introduces uncertainties (Rossberg et al. 2017). To minimise these, type-specific reference values were preferred over site-specific values, and for the estuarine types only water body areas where both statistical and mechanistic models are available were used to estimate ensemble reference values. The ensemble approach was applicable for two estuarine water body types (Type 2 and Type 3) and was also used to calculate chlorophyll-*a* reference values for the southern part of the Limfjorden, an area assumed to be affected by hypoxia even under reference conditions.

The application of typologies to support the establishment of type-specific reference values is in accordance with the WFD guidelines, but although the typologies allow targets to be set for all water bodies and reduce uncertainties by averaging potential odd values (from models or observations), they might produce values that do not necessarily fit all the specified water bodies.

Here, we applied a typology with five main types; to differentiate further, additional model development is required.

With respect to Type 1 water bodies, only mechanistically modelled reference values were available, whereas site-specific mechanistic and statistical models were applied for water body Type 2, 3 and 5 as well as for the southern part of Limfjorden, which was treated outside the typology.

An alternative model-based approach to estimate reference conditions for areas with no or poor data coverage was applied by Schernewski et al. (2015). Briefly, as in the present study, they modelled present-day conditions (i.e. status) and reference conditions (based on reference nutrient loads), but used reference to status ratios calculated for every model grid cell to estimate targets for unmeasured areas.

As discussed in section 8.1, the reference chlorophyll-*a* values obtained by modelling in this study are in good agreement with values obtained using corresponding (modelling) methods and other approaches. However, inclusion of additional water bodies would have been desirable to increase the amount of available data for each type and maybe to differentiate even further between types.

9.2.2 Chlorophyll-*a* as indicator

Summer chlorophyll-*a* is one of the intercalibrated WFD indicators and is generally sensitive to land-based N loadings in the more enclosed water bodies, whereas the more open waters are less sensitive to Danish land-based N loadings.

The close relation between nutrient loadings and chlorophyll-*a* is well established for both marine and freshwater systems, making chlorophyll-*a* an obvious indicator for eutrophication. However, whereas elevated levels of chlorophyll-*a* are a clear sign of eutrophication, low chlorophyll-*a* concentrations can be related to both low nutrient loadings and/or a high grazing pressure. For example, in open waters primary production will most likely respond to changes in N loadings, also during summer where pelagic grazing on phytoplankton may prevent increases in biomass, and hence chlorophyll-*a* concentrations, despite an increase in nitrogen concentrations (Cloern et al. 2014; Duarte et al. 2000; Olsen et al. 2006).

Likewise, in shallow, well-mixed coastal water bodies benthic filter feeders may occur in high densities, allowing "control" of chlorophyll-*a*. In these water bodies, changes in nutrient loadings do not necessarily result in changes in chlorophyll-*a* concentrations but in modifications of the overall turnover of phytoplankton (primary production and respiration). In consequence, even nutrient-enriched estuaries do not necessarily show elevated summer chlorophyll-*a* concentrations.

9.2.3 Light attenuation (K_d) as proxy indicator

Eelgrass is an important biological element for a wide range of the Danish water bodies. The intercalibrated indicator for eelgrass is the depth limit, but it was not possible to develop the models to describe this indicator reliably. In the mechanistic models, eelgrass is described as a state variable and a number of processes affecting eelgrass abundance and biomass are included: light availability, hypoxia, sediment quality (H_2S, organic content), seabed shear stress etc. Some of these processes are affected by nutrients, and some have a feedback to the ecosystem, such as eelgrass beds reducing resuspension and further promoting growth of beds. However, even though eelgrass is included as a state variable in the models, we cannot extract the depth limit as a valid model indicator. Hence, we applied the proxy indicator light attenuation coefficient, K_d, and as both the statistical and the mechanistic models include K_d, both model types can be applied for the estimation of N reductions and corresponding MAI, which we regard as a strength.

Light availability is one of the main drivers determining the maximum depth distribution of eelgrass (Duarte 1991) and the K_d-indicator reflects the light habitat and thereby expresses the potential depth to which eelgrass can grow.

However, as demonstrated by Flint et al. (2016) and Canal-Vergés et al. (2016), sufficient light availability alone does not ensure eelgrass growth and proliferation. Several physico-chemical and biological factors prevent or restrict the re-establishment of eelgrass, and even if the indicator "summer K_d" achieves GES, this might not result in GES for the eelgrass depth limit indicator.

Even though K_d is a more "model-friendly" parameter than the eelgrass depth limit, several factors such as dissolved organic matter, detritus and scattering of light affect K_d, making the link to eutrophication complex. The light climate is, however, of fundamental importance for the structure and functioning of coastal ecosystems, and the K_d-indicator is therefore highly relevant also in a WFD context.

9.2.4 Additional indicators

Three indicators are developed and applied in the assessment by the Danish EPA of the ecological status: summer chlorophyll-*a*, eelgrass depth limit and the Danish Quality Index (DKI) for benthic communities. Unfortunately, direct links between nutrient loadings and the fauna index have not been established, why benthic fauna is not included in the RBMP 2015-2021.

It is likely, though, that links between DKI and secondary eutrophication effects related to increased ecosystem productivity (e.g. bottom water hypoxia and enrichment of sediment organic matter) can be established, which could be valuable for the preparation of RBMP 2021-2027. An obvious action would be to establish linkages between benthic fauna (biomass, diversity) and areas affected by hypoxia, thereby introducing oxygen (concentrations, duration of hypoxia) as a specific indicator. Although an oxygen depletion indicator is applied in the statistical model approach, the link between nutrient loadings and hypoxia is complicated by a considerable time lag and sensitivity to climate variables and has not been directly addressed in this study. However, bottom water oxygen is highly relevant for both nutrient cycling and structure as well as the functioning of benthic communities and should thus be included in the assessment of ecosystem status (Diaz & Rosenberg 1995; Carstensen et al. 2014).

As mentioned above, chlorophyll-*a* in open deep waters is not likely to respond to changes in nutrient concentrations, as the phytoplankton biomass is partly controlled by a high grazing pressure during summer. In contrast, pelagic process rates such as primary production are expected to respond to nutrient reductions, why primary production could be suggested as a future indicator (Cloern 2014). However, today's monitoring program does not support this, and introduction of primary production as an indicator would require substantial updates.

Only few of the developed models responded to reduced P loadings when analysing the two indicators: summer chlorophyll-*a* and the proxy indicator K_d. This does not imply that P loadings are not important, but for the indicators adopted by the Danish EPA the models did not demonstrate any significant response in most areas. However, phosphorous is often the limiting nutrient for primary production in spring, and we consequently suggest development and introduction of indicators sensitive to P loadings in order to be able to manage P loadings as well.

Hence, in the preparation of the RBMP 2021-2027 the development and adaptation of additional indicators would most likely increase the certainty of the effort to manage Danish marine ecosystems.

9.3 Modelling approaches

Two rather different modelling approaches (i.e. statistical and mechanistic) have been applied to estimate chlorophyll-*a* reference concentrations (section 8.1) and to calculate MAI for Danish WFD water bodies. The statistical models build solely on long-term monitoring data and describe observations using linear relations without including any process descriptions or mechanisms. Their simplicity, direct link to observations and high transparency are an advantage; however, the lack of mechanistic descriptions makes model predictions outside the validation range challenging, and the uncertainty will increase when moving away from the conditions where the models have been calibrated. In the statistical approach, a suite of ecological relevant indicators besides K_d and chlorophyll-*a* was introduced in order to obtain a more holistic approach to evaluate the status of the ecosystems and assess the load reductions needed to obtain GES.

The mechanistic models represent the other end of the continuum of model types. They build on complex process descriptions and interactions, comprising a (simplified) ecosystem. In addition, the mechanistic models applied include both sediment pools of nutrients and benthic primary producers and may account for nutrient loadings from other sources than Danish catchments. However, mechanistic models rely heavily on parameterisations and parameter estimations, making predictions sensitive to model assumptions. As for the statistical models, uncertainty in model predictions will increase when moving away from conditions where the models have been calibrated.

Despite the inherent differences, the ensemble results revealed an overall satisfactory agreement between the two model approaches, both with regard to estimates of MAI (*Table 8.10*) and reference concentrations of chlorophyll-*a* (*Table 8.3* and *8.4*), which gives confidence to the model predictions. Apparently, the model derived sensitivity of the indicators to changes in N loading is similar between both model approaches, except in open areas. This might be related to variations in loadings to the Baltic Sea, which cannot be separated from variations in the Danish loads in the statistical models and/or assumptions regarding remineralisation of organic matter in the mechanistic models.

Both the statistical and mechanistic models are site-specific and thus expected to reflect the local physical and ecological characteristics. The site-specific approach could not be applied to several water bodies due to lack of data. Here, the type-specific meta-model approach was used instead. This approach utilises the information from several areas (and models) and, at least in theory, this will provide more robust model predictions. An obvious disadvantage is that the type-specific models are not necessarily a good representation of the single individual water body and the meta-model approach could very likely be improved by refining the typology. A more differentiated typology would, however, require an increased amount of monitoring data and number of site-specific models for each type in order to obtain sufficient power to establish robust meta models.

MAI calculations performed by meta models are considered more uncertain than MAI calculated using site-specific models. This is partly related to the potential lack of representativity of the type-specific model, but mostly to the general lack of data from these areas, resulting in, for instance, uncertain estimates of status values.

9.4 Achievement of GES

The objective of the present project was to develop models and methods for RBMP 2015-2021 that will lead to fulfilment of Danish WFD obligations by ensuring that all marine waters do obtain GES no later than 2027. The models and methods developed target the Danish water bodies and Danish N loadings and provide a central estimate of the MAI that most probably will ensure fulfilment of Danish obligations. However, full implementation of the estimated MAI will not necessarily result in achievement of GES in all water bodies. To achieve GES according to the maps presented in section 2.2, additional factors play a role:

- Eelgrass might not recover even though light is sufficient to support growth. Worldwide, only few examples exist of eelgrass recovery following significant losses and research is ongoing to investigate how to promote recovery.

- Benthic fauna quality is not linked to the models and, assessment of measures to obtain GES for fauna was consequently not made. Even if eelgrass recovers and chlorophyll-*a* achieves GES, we do not have evidence that this will also be the case for benthic fauna especially since other pressures, like bottom trawling might influence DKI.

- A number of Danish water bodies rely on an N reduction effort in neighbouring countries to meet GES.

In addition, the methods presented here basically violate the one-out-all-out principle, which is defined when evaluating the ecological status and not when estimating measures to ensure GES. When reductions based on chlorophyll-*a* or K_d are averaged instead of choosing the maximum reductions, we do, in theory, not obtain GES for both indicators. The sensitivity analysis showed that the estimated reductions, and corresponding MAIs, were sensitive to the estimated status of both indicators. Some of the variability originating from observations can be minimised by averaging, but in theory, this will only ensure GES for one of the two indicators.

On the other hand, regime shifts might significantly alter the sensitivity to N loadings. From lake research, we know that regime shifts may have a strong impact on the entire ecosystem structure (Scheffer & Jeppesen 2007). Examples from estuaries include a change from turbid to clear water following invasion of benthic filter feeders (Petersen et al. 2008; Cloern & Jassby 2012), and some scientists suggest that similar structural changes will occur when, for example, eelgrass recovers. Besides a few local areas (Orth et al. 2012), however, no evidence exists of the magnitude of such large-scale structural changes. Thus, many factors other than direct anthropogenic pressure influence GES, but we regard the estimated MAI as central estimate to achieve GES in Danish waters and to fulfil WFD obligations.

10 Conclusion

We have developed and applied model tools and methods to calculate maximum allowable nutrient inputs (MAI) to all Danish marine WFD water bodies. The calculations are based on knowledge of the current (2007-2012) status values of indicators, target values representing the good-moderate boundary value and model-based relations between N loading and indicator values. For approximately half of the water bodies, site-specific mechanistic and/or statistical models are applied. For a large part of the remaining water bodies, type-specific meta-models are used to calculate MAI. In total, the models cover approximately 90% of the Danish water body area. Summarising MAI for all 119 Danish WFD water bodies results in a N load from Danish catchment of 42 ktons N year^{-1}, which is equivalent to a reduction of approx. 30% compared with the current (2007-2012) N load. However, this national scale reduction requirement exhibits huge variation between individual catchments, reflecting variations in the current eutrophication level, as well as in the sensitivity of each water body.

Calculations of MAI are associated with uncertainties, especially related to target setting and model predictions. The tool development process has been focused on reducing uncertainties, for instance by averaging indicators and applying a type-specific approach whenever site-specific values and estimates were considered too uncertain. The ensemble model results reveal good agreement between the two very different model approaches, thus indicating that the estimated MAIs are reliable.

To obtain more certain MAI estimates, it is important to continuously monitor the ecosystems as they approach GES and to evaluate, update and improve the models and methods accordingly based on new knowledge. Thus, the model tools and methods developed in this project should be regarded as part of an ongoing process towards better understanding and improved predictability of the behaviour of marine ecosystems in a changing world.

11 Epilogue

The development of models and methods to support the establishment of the Danish River Basin Management Plan 2015-2021 (RBMP) commenced 2013 and was terminated at the end of 2014. Besides defining target chlorophyll-*a* values for all Danish Water Framework Directive water bodies from Skagen and southwards, the main project result was the estimation of a maximum allowable nitrogen input (MAI) from Danish land-based run-off at 42 kton N year^{-1}. The estimated MAI would account for Denmark to meet the requirements of the Water Framework Directive (WFD) according to present day (2007-2012) loadings, indicator status values, meteorology and exchanges with neighbouring waters.

The main results of this project (MAI to each Danish water body) were used by the Danish EPA to establish the first version of the Danish RMBP 2015-2021. The RMBP was presented to the public by the Minister of Environment in December 2014, and went into a 6-month public consultation.

Following, the public consultation, some adjustments to the work presented in this report were acquired. These adjustments were addressed during the second half of 2015 end resulted in an adjusted RBMP 2015-2021 and a second version of MAI from Danish land-based run-off of 44.5 kton N year^{-1}, corresponding to an increase of 2.5 kton N year^{-1}, adopted by the Danish Parliament in 2016.

The changes in MAI were based on three adjustments: i) Optimization of reductions in a few fjords by accounting for upstream N-load reductions, ii) implementation of suggested reductions in emissions and corresponding N depositions for 2027 according to 2030 WPE 2014[19], and iii) a historic load scenario combined with the implementation of 2030 WPE 2014.

[19] Suggestions by the EU Commission in 2013 to a new NEC Directive developed by GAINS/IIASA. Since 2015, the assumptions for this scenario have changed.

12 References

Abdi H (2010) Partial least squares regression and projection on latent structure regression (PLS Regression). Wiley Interdisciplinary Reviews: Computational Stratistics 2:97-106. doi:10.1002/wics.051.

Acevedo-Trejos E, Brandt G, Steinacher M & A Merico (2014) A glimpse into the future composition of marine phytoplankton communities. Front. Mar. Sci. 1:15. doi: 10.3389/fmars.2014.00015.

Alexander M, Scott J & Deser C (2000) Processes that influence sea surface temperature and ocean mixed layer depth variability in a coupled model. Journal of Geophysical Research: Oceans 105:16823-16842 http://onlinelibrary.wiley.com/doi/10.1029/2000JC900074/full.

Almroth E & Skogen MD (2010) A North Sea and Baltic Sea Model Ensemble Eutrophication Assessment. Ambio 39(1):59-69.

Andersen JH, Carstensen J, Conley DJ, Dromph, K., Fleming-Lehtinen V, Gustafsson BG, Josefson JB, Norkko A, Villnäs A, Murray C (2015) Long-term temporal and spatial trends in eutrophication status of the Baltic Sea. Biological Reviews 92(1):135-149.

Andersen JH, Axe P, Backer H, Carstensen J, Claussen U, Fleming-Lehtinen V, Jarvinen M, Kaartokallio H, Knuuttila S, Korpinen S, Kubiliute A, Laamanen M, Lysiak-Pastuszak E, Martin G, Murray C, Mohlenberg F, Nausch G, Norkko A, Villnas A 2011: Getting the measure of eutrophication in the Baltic Sea: towards improved assessment principles and methods. Biogeochemistry 106:137-156.

Armstrong RA (1994) Grazing limitation and nutrient limitation in marine ecosystems: Steady state solutions of an ecosystem model with multiple food chains. Limonology and Oceanography 39:597-608. doi:10.4319/lo.1994.39.3.0597.

Barber RT, Hilting AK (2002) History of the study of plankton productivity. In: Williams PJ LeB, Thomas DN, Reynolds CS (Eds.) Phytoplankton productivity in marine and aquatic environments. Blackwell Science, Oxford, pp 16-43.

Baretta JW, Ebernhöh W, Ruardij P (1995) The European regional seas ecosystem model, a complex marine ecosystem model. Netherlands J Sea Res 33 (3/4): 233-246.

Basset A, Barbone E, Borja A, Elliott M, Jona-Lasinio G, Marques JC, Mazik K, Muxika I, Neto JM, Reizopoulou S, Rosati I, Teixeira H (2013) Natural variability and reference conditions: setting type-specific classification boundaries for lagoon macroinvertebrates in the Mediterranean and Black Seas. Hydrobiologia 704:325-345.

Behrenfeld MJ, O'Malley RT, Siegel DA, McClain CR, Sarmiento JL, Feldman GC, Milligan AJ, Falkowski PG, Letelier RM, Boss ES (2006) Climate-driven trends in contemporary ocean productivity. Nature 444:752-755. doi:10.1038/nature05317

Bek. nr. 837 (2016) Bekendtgørelse om basisanalyser, nr. 837 af 27/6/16.

Bek. nr. 1001 (2016) Bekendtgørelse om overvågning af overfladevandets, grundvandets og beskyttede områders tilstand og om naturovervågning af internationale naturbeskyttelsesområder, nr. 1001 from 29/06/2016. https://www.retsinformation.dk/pdfPrint.aspx?id=181970 (in Danish).

Bennion H, Fluin J, Simpson GL (2004) Assessing eutrophication and reference conditions for Scottish freshwater lochs using subfossil diatoms. Journal of Applied Ecology 41:124-138.

Bendtsen J, Hansen JLS (2013) Effects of global warming on hypoxia in the Baltic Sea-North Sea transition zone. Ecological Modelling 264:17-26.

Berge T, Chakraborty S, Hansen PJ, Andersen KH (2017) Modeling succession of key resource-harvesting traits of mixotrophic plankton. The ISME Journal 11:212-223.

Bianchi TS, Engelhaupt E, Westman P, Andren T, Rolff C, Elmgren R. (2000) Cyanobacterial blooms in the Baltic Sea: natural or human-induced? Limnology and Oceanography 45:716-726.

Boesch DF (2002) Challenges and Opportunities for Science in Reducing Nutrient Over-enrichment of Coastal Ecosystems. Estuaries 25(4b):886-900.

Boesch DF, Brinsfield RB, Magnien RE (2001) Chesapeake Bay Eutrophication: Scientific Understanding, Ecosystem Restoration, and Challenges for Agriculture. Journal of Environmental Quality 30(2)303-320.

Bopp L, Monfray P, Aumont O, Dufresne JL, Le Treut H, Madec G, Terray L, Orr JC (2001) Potential impact of climate change on marine export production. Global Biogeochem. Cycles 15:81-99. doi:10.1029/1999GB001256.

Borja A, Dauer DM, Grémare A (2012) The importance of setting targets and reference conditions in assessing marine ecosystem quality. Ecological Indicators 12:1-7.

Boyd PW, Doney SC (2002) Modelling regional responses by marine pelagic ecosystems to global climate change. Geophys. Res. Lett. 29:53-1-53–4 . doi:10.1029/2001GL014130.

Bundesministerium (2014) Harmonisierte Hintergrund- und Orientierungswerte für Nährstoffe und Chlorophyll-a in den deutschen Küstengewässern der Ostsee sowie Zielfrachten und Zielkonzentrationen für die Einträge über die Gewässer. Bund/Länder-Ausschuss Nord- und Ostsee, Bundesministerium für Umwelt, Naturschutz, Bau und Reaktorsicherheit.

Butenschön M, Clark J, Aldridge JN, Allen JI, Artioli Y, Blackford J, Bruggeman J, Cazenave P, Ciavatta S, Kay S, Lessin G, van Leeuwen S, van der Molen J, de Mora L, Polimene L, Sailley S, Stephens N & Rtorres (2016) ERSEM 15.06: a generic model for marine biogeochemistry and the ecosystem dynamics of the lower trophic levels. Geosci Model Dev 9:1293-1339.

Bøgestrand J, Windolf J, Kronvang B (2014a) Næringsstofbelastningen til vandområder omkring år 1900. 9 s. Aarhus Universitet. Notat fra DCE - Nationalt Center for Miljø og Energi.

Bøgestrand J, Windolf J, Kronvang B, Kjeldgaard A (2014b) Baggrunds-belastning med total N og nitrat-N. 11 s. Aarhus Universitet. Notat fra DCE-Dansk Center for Miljø og Energi.

Canal-Vergés P, Petersen JK, Rasmussen EK, Erichsen AC, Flindt MR (2016) Validating GIS tool to assess eelgrass potential recovery in the Limfjorden (Denmark). Ecological Modelling 338:135-148 http://dx.doi.org/10.1016/j.ecolmodel.2016.04.023.

Carman R, Cederwall H (2001) Sediments and Macrofauna in the Baltic Sea - Characteristics, Nutrient Contents and Distribution. In: Wulff F, Rahm RA., Larsson RA (eds) A Systems Analysis of the Baltic Sea. Springer-Verlag Berlin Heidelberg, pp 289-328.

Caron DA (2016) Mixotrophy stirs up our understanding of marine food webs. PNAS 113:2806-2808.

Carrascal L M, Galvan I, and Gordo O (2009) Partial least squares regression as an alternative to current regression methods used in ecology. Oikos 118: 81-690.

Carstensen J, Henriksen P (2009) Phytoplankton biomass response to nitrogen inputs: a method for WFD boundary setting applied to Danish coastal waters. Hydrobiologia 633:137-149. doi:10.1007/s10750-009-9867-9.

Carstensen J, Sanchez-Camacho M, Duarte CM, Krause-Jensen Dsalmon farms & N Marba (2011). Connecting the dots: responses of coastal ecosystems to changing nutrient concentrations. Environmental Science and Technology 45:9122-9132.

Carstensen J, Krause-Jensen D, Markager S, Timmermann K, Windolf J (2013a) Water clarity and eelgrass responses to nitrogen reductions in the eutrophic Skive Fjord, Denmark. Hydrobiologia 704:293-309.

Carstensen J, Andersen JH, Dromph K, Fleming-Lehtinen V, Simis S, Gustafsson B.G, Norkko A, Radtke H, Petersen DLJ, Uhrenholdt T (2013b) Approaches and methods for eutrophication target setting in the Baltic Sea region. Helsinki Commission. Baltic Sea Enviroment Proceedings 133:1-134.

Carstensen J, Andersen JH, Gustafsson BG, Conley DJ (2014) Deoxygenation of the Baltic Sea during the last century. Proc. Natl. Acad. Sci. U. S. A. 111:5628-33. doi:10.1073/pnas.1323156111.

Carstensen J, Conley DJ, Bonsdorff E, Gustafsson BG, Hietanen S, Janas U et al. (2014) Hypoxia in the Baltic Sea: Biogeochemical cycles, benthic fauna, and management. AMBIO 43:26-36. doi:10.1007/s13280-013-0474-7.

Carstensen J (2016) Intercalibration of chlorophyll *a* between Denmark, Norway and Sweden. Western Baltic (BC6), Kattegat (NEA8b) and Skagerrak (NEA8a, NEA9 and NEA10). Aarhus University, DCE – Danish Centre for Environment and Energy, 38 pp. Technical Report from DCE – Danish Centre for Environment and Energy No. 76.
http://dce2.au.dk/pub/TR76.pdf

Christensen PB, Hansen OS, Ærtebjerg G (red.) (2004) Iltsvind. Miljøbiblioteket nr 4, Danmarks Miljøundersøgelser og Forlaget Hovedland. 128 pp.

Christianen MJA, van Belzen J, Herman PMJ, van Katwijk MM, Lamers LPM et al. (2013) Low-canopy seagrass beds still provide important coastal protection services. PLoS ONE 8(5): e62413. doi:10.1371/journal.pone.0062413

Clarke A, Juggins S, Conley D (2003) A 150-year reconstruction of the history of coastal eutrophication in Roskilde Fjord, Denmark. Marine Pollution Bulletin 46:1615-1618. doi:10.1016/S0025-326X(03)00375-8.

Clarke AL, Weckström K, Conley DJ, Anderson NJ, Adser F, Andrén E, de Jonge VN, Ellegaard M, Juggins S, Kauppila P (2006) Long-term trends in eutrophication and nutrients in the coastal zone. Limnology and Oceanography 51:385-397. doi:10.4319/lo.2006.51.1_part_2.0385.

Cloern JE (1984) Temporal dynamics and ecological significance of salinity stratification in an estuary (South San Francisco Bay, USA). Oceanol. Acta 7: 137-141.

Cloern JE (1999) The relative importance of light and nutrient limitation of phytoplankton growth: A simple index of coastal ecosystem sensitivity to nutrient enrichment. Aquatic Ecology 33:3-16. doi:10.1023/A:1009952125558.

Cloern JE (2001) Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210:223-253.
doi:10.3354/meps210223.

Cloern JE, Jassby AD (2012) Drivers of change in estuarine-coastal ecosystems: Discoveries from four decades of study in San Francisco Bay. Reviews of Geophysics 50, RG4001. doi:10.1029/2012RG000397.

Cloern JE, Foster SQ, Kleckner AE (2014) Phytoplankton primary production in the world's estuarine-coastal ecosystems. Biogeosciences 11: 2477-2501.

Conley DJ, Kaas H, Mohlenberg F, Rasmussen B, Windolf J (2000) Characteristics of Danish estuaries. Estuaries 23:820-837.

Conley DJ, Markager S, Andersen J, Ellermann T, Svendsen LM (2002) Coastal eutrophication and the Danish national aquatic monitoring and assessment program. Estuaries 25:848-861. doi:10.1007/BF02804910.

Conley DJ, Carstensen J, Ærtebjerg G, Christensen PB, Dalsgaard T, Hansen JLS, Josefson AB (2007) Long-term changes and impacts of hypoxia in Danish coastal waters, in: Ecological Applications. p. 472. doi:10.1890/05-0766.1

Conley DJ, Paerl HW, Howarth RW, Boesch DF, Seitzinger SP, Havens KE, Lancelot C, Likens GE (2009a). Ecology - Controlling eutrophication: Nitrogen and phosphorus. Science 323(5917):1014-1015.

Conley DJ, Carstensen J, Vaquer-Sunyer R, Duarte CM (2009b). Ecosystem thresholds with hypoxia. Hydrobiologia 629:21-29. doi: 10.1007/s10750-009-9764-2.

Dahl K (red.), Andersen JH (red.), Riemann B (red.), Carstensen J, Christiansen T, Krause-Jensen D, Josefson AB, Larsen MM, Petersen JK, Rasmussen MB, Stand J (2005) Redskaber til vurdering af miljø- og naturkvalitet i de danske farvande. Typeinddeling, udvalgte indikatorer og eksempler på klassifikation. Danmarks Miljøundersøgelser. 158 s. - Faglig rapport fra DMU nr. 535.

Dennison, W., Orth, R., Moore, K., 1993. Assessing water quality with submersed aquatic vegetation. Bioscience 43, 86–94.

DHI (2013a). MIKE 21 & MIKE 3 Flow Model FM. Hydrodynamic and Transport Module. Scientific Documentation. DHI Water Environment Health, Hørsholm, Denmark, 54 pp.

DHI (2013b) MIKE 21/3 Ecological Modelling. MIKE 21/3 ECO Lab FM module. Short description. DHI Water Environment Health, Hørsholm, Denmark, 14 pp. Retrieved January 6, 2013 from
http://www.dhisoftware.com/Download/DocumentsAndTools/~/media/Microsite_MIKEbyDHI/Publications/PDF/Short%20descriptions/MIKE213_FM_EL_Short_Description.ashx.

DHI (2014) DHI 3 Algae and Sediment Model. ECO Lab Template. Scientific Description. DHI Water Environment Health, Hørsholm, Denmark, 134 pp.

Di Toro DM, O'Connor DJ & RV Thormann (1971) A dynamic model of phytoplankton population in the Sacramento-San Joaquin Delta. In "Non-equilibrium Systems in Natural Water Chemistry", Adv. Chem. Ser. 106. American Chemical Society, Washington DC, pp. 131-180.

Diaz RJ, Rosenberg R (1995) Marine benthic hypoxia: a review of its ecological effects and the behavioral responses of benthic macrofauna. Oceanography and Marine Biology 33: 245-303.

Directive 2000/60/EC of the European Parliament and ofthe Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Official Journal L 327/1.

Droop MR (1968) Vitamin B12 and marine ecology IV: the kinetics of uptake, growth and inhibition in Monochrysis lutheri. J Mar Biol Ass UK 48:689-733

Duarte CM (1991) Seagrass depth limits. Aquatic Botany 40(4):363-377. doi:10.1016/0304-3770(91)90081-F.

Duarte CM (1995) Submerged aquatic vegetation in relation to different nutrient regimes. Ophelia 41:87-112 http://www.tandfonline.com/doi/pdf/10.1080/00785236.1995.10422039?needAccess=true.

Duarte CM, Augustí S, Agawin NSR (2000) Response of a Mediterranean phytoplankton community to increased nutrient inputs: A mesocosm experiment. Marine Ecology Progress Series 195:61-70.

Duarte CM, Conley DJ, Carstensen J, Sanchez-Camacho M. (2009) Return to Neverland: Shifting Baselines Affect Eutrophication Restoration Targets. Estuaries and Coasts 32: 29-36.

Edelvang K, Kaas H, Erichsen AC, Alvarez-Berastegui1 D, Bundgaard B, Jørgensen PV (2005) Numerical modelling of phytoplankton biomass in coastal waters. Journal of Marine Systems 57:13-29.

Edwards AM, Bees MA (2001) Generic dynamics of a simple plankton population model with a non-integer exponent of closure. Chaos, Solitons & Fractals 12:289-300.

Edwards KF, Thomas MK, Klausmeier CA, E Litchman E (2012) Allometric scaling and taxonomic variation in nutrient utilization traits and maximum growth rate of phytoplankton. Limnology and Oceanography 57(2):554-566.

Eilola K, Gustafsson B G, Kuznetsov I, Meier HEM, Neumann T, Savchuk O. P (2011) Evaluation of biogeochemical cycles in an ensemble of three state-of-the-art numerical models of the Baltic Sea. Journal of Marine Systems 88: 267-284.

Ellegaard M, Clarke AL, Reuss N, Drew S, Weckström K, Juggins S, Anderson NJ, Conley DJ (2006) Multi-proxy evidence of long-term changes in ecosystem structure in a Danish marine estuary, linked to increased nutrient loading. Estuarine, Coastal and Shelf Science 68:567-578. doi:10.1016/j.ecss.2006.03.013.

Ellermann T, Andersen HV, Bossi R, Christensen J, Løfstrøm P, Monies C, Grundahl L, Geels C (2013) Atmosfærisk deposition 2012. NOVANA. Aarhus Universitet, DCE – Nationalt Center for Miljø og Energi. 85 s. – Videnskabelig rapport fra DCE – Nationalt Center for Miljø og Energi nr. 73. http://dce2.au.dk/pub/SR73.pdf.

Elser JJ, Bracken MES, Cleland EE, Gruner DS, Harpole WS, Hillebrand H, Ngai JT, Seabloom EW, Shurin JB, Smith JE (2007) Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecol. Lett. 10:1135-42. doi:10.1111/j.1461-0248.2007.01113.x.

Eppley, R.W., 1972. Temperature and phytoplankton growth in the sea. Fish. Bull. 70, 1063–1085.

Eppley RW, Rogers JN, Mccarthy JJ (1969) Half-saturation constants for uptake of nitrate and ammonium by marine photoplankton. Limnology and Oceanography 14(6)912-920. doi: 10.4319/lo.1969.14.6.0912.

Erichsen AC, Rasch PS (2001) Two- and Three-dimensional Model System Predicting the Water Quality of Tomorrow. Proc. of the Seventh International Conference on Estuarine and Coastal Modelling. Spaulding ML (Ed), American Society of Civil Engineers 2001.

EU (2016) Commission decision (EU) laying down criteria and methodological standards on good environmental status of marine waters and specifications and standardised methods for monitoring and assessment, and repealing Decision 2010/477/EU.
http://ec.europa.eu/environment/marine/good-environmental-status/descriptor-10/pdf/good_environmental_status.pdf

Executive Order no.1001 from 29/6-2016 (Bekendtgørelse). Bekendtgørelse om overvågning af overfladevandets, grundvandets og beskyttede områders tilstand og om naturovervågning af internationale naturbeskyttelsesområder1. https://www.retsinformation.dk/pdfPrint.aspx?id=181970 (in Danish).

Fath BD, Jørgensen SE & Scharler UM (2011) Ecological Modeling in Environmental Management: History and Applications. In: Wolanski E and McLusky DS (eds.) Treatise on Estuarine and Coastal Science 9:23-33. Waltham: Academic Press.

FEMA (2013) Fehmarnbelt Fixed Link. Environmental Impact Analyses. Ecological Modelling Report. No. E2TR0031. 100 pp.

FEHY (2012) Fehmarnbelt Fixed Link. Marine Water Baseline. Hydrography of the Fehmarnbelt Area. Report. No E1TR0057, Volume II.

Fitzpatrick J, Brady D, DePinto J, DiToro D, Kemp M & D Scavia (2016) Scenario-Based Forecasts in Support of Regional Coastal Management: Concepts of Operation. Prepared for NOAA/NOS National Centers for Coastal Ocean Science.http://graham.umich.edu/scavia/wp-content/uploads/2016/01/Scenario-based-Forecasts.pdf

Flindt MR, Rasmussen EK, Valdemarsen T, Erichsen AC, Kaas H (2016) Using a GIS-tool to evaluate potential eelgrass reestablishment in estuaries. Ecological Modelling 338:122-134.

Forrest DR, Hetland RD, DiMarco SF (2012) Corrigendum: Multivariable statistical regression models of the areal extent of hypoxia over the Texas–Louisiana continental shelf. Environ. Res. Lett. 7:19501. doi:10.1088/1748-9326/7/1/019501.

Gallegos CL, Kenworthy WJ (1996) Seagrass Depth Limits in the Indian River Lagoon (Florida, U.S.A.): Application of an Optical Water Quality Model. Estuar. Coast. Shelf Sci. 42:267-288. doi:10.1006/ecss.1996.0020

Ganju NK, Brush MJ, Rashleigh B, Aretxabaleta AL, del Barrio P, Grear JS, Harris LA, Lake SJ, McCardell G, O'Donnell J, Ralston DK, Signell RP, Testa JM, Vaudrey JMP (2016) Progress and Challenges in Coupled Hydrodynamic-Ecological Estuarine Modeling. Estuaries and Coasts 39:311-332.

Geels C, Hansen KM, Christensen JH, Ambelas Skjøth C, Ellermann T, Hedegaard GB, Hertel O, Frohn LM, Gross A, Brandt J. (2012) Projected change in atmospheric nitrogen deposition to the Baltic Sea towards 2020. Atmospheric Chemistry and Physics 12:2615-2629.

Gothenburg Protocol (2012) Revision of the annexes to the 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone: draft revised annex II. Economic and Social Council, Executive Body for the Convention on Long-range Transboundary Air Pollution.

Griffiths JR, Kadin M, Nascimento FJA, Tamelander T, Törnroos A, Bonaglia S, Bonsdorff E, Brüchert V, Gårdmark A, Järnström M, Kotta J, Lindegren M, Nordström MC, Norkko A, Olsson J, Weigel B, Žydelis R, Blenckner T, Niiranen S & M Winder (2017) The importance of benthic-pelagic coupling for marine ecosystem functioning in a changing world. Global Change Biology 23:2179-2196. doi:10.1111/gcb.13642.

Guidance Document No. 5 (2003) Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance Document No. 5. Transitional and Coastal Waters – Typology, Reference Conditions and Classification Systems. Working Group 2.4 - COAST. European Communities.

Guidance Document No. 14 (2005) Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance Document No. 14. Guidance on the Intercalibration Process 2004-2006. European Communities.

Gustafsson BG, Schenk F, Blenckner T, Eilola K, Meier HEM, Muller-Karulis B, Neumann T, Ruoho-Airola T, Savchuk OP, Zorita E (2012) Reconstructing the Development of Baltic Sea Eutrophication 1850-2006. Ambio 41:534-548.

Haney JD, Jackson GA (1996). Modeling phytoplankton growth rates. Journal of Plankton Research 18:63-85.

Hansen JLS, Dinesen GR, Bastardie F, Eigaard OR (2015) Natoat om effekter af fiskeri med bundslæbende redskaber på bundfaunaen i de indre danske farvande. Notat fra DCE-nationalt Center for Miljø og Energi.

HELCOM (2007) HELCOM Baltic Sea Action Plan. HELCOM Ministerial Meeting, Krakow, Poland, 15 November 2007.
(http://helcom.fi/Documents/Baltic%20sea%20action%20plan/BSAP_Final.pdf).

HELCOM (2011) The Fifth Baltic Sea Pollution Load Compilation (PLC-5). Baltic Sea Environment Proceedings No. 128. Helsinki Commission, 217 pp. http://www.helcom.fi/Lists/Publications/BSEP128.pdf.

HELCOM (2013) Approaches and methods for eutrophication target setting in the Baltic Sea region. Baltic Sea Environment Proceedings No. 133. Helsinki Commission, 134 pp.

HELCOM (2013) Climate change in the Baltic Sea Area: HELCOM thematic assessment in 2013. Balttic Sea Environmental Proceedings No. 137. Helsinki Commission, 66 pp.

HELCOM (2015) Updated Fifth Baltic Sea pollution load compilation (PLC-5.5). Baltic Sea Environment Proceedings No. 145. Helsinki Commission, 142 pp http://www.helcom.fi/Lists/Publications/BSEP145_Lowres.pdf.

Henriksen P (2009) Reference conditions for phytoplankton at Danish Water Framework Directive intercalibration sites. Hydrobiologia 629:255-262.

Hernes PJ, 2003. Photochemical and microbial degradation of dissolved lignin phenols: Implications for the fate of terrigenous dissolved organic matter in marine environments. J. Geophys. Res. 108:3291. doi:10.1029/2002JC001421.

Hinsby K, Markager S, Kronvang B, Windolf J, Sonnenborg TO, and Thorling L. (2012) Threshold values and management options for nutrients in a catchment of a temperate estuary with poor ecological status. Hydrology and Earth System Sciences 16:2663-2683.

Ioannidis JPA (2014) How to make more published research true. PLoS Medicine 11(10)e1001747.

Jakobsen F, Ottavi J (1997) Transport through the concentration area in the Little Belt. Estuarine, Coastal and Shelf Science 45:6:759-767.

Janse JH, De Senerpont Domis LN, Scheffer M, Lijklema L, Van Liere L, Klinge M, Mooij WM (2008) Critical phosphorus loading of different types of shallow lakes and the consequences for management estimated with the ecosystem model PCLake. Limnologica 38:203-219. doi:10.1016/j.limno.2008.06.001.

Janssen ABG, Arhonditsis GB, Beusen A, Bolding K, Bruce L, Bruggeman J, Mooij WM (2015) Exploring, exploiting and evolving diversity of aquatic ecosystem models: a community perspective. Aquatic Ecology 49(4):513-548.

Josefson AB, Rasmussen B (2000) Nutrient retention by benthic macrofaunal biomass of Danish estuaries: Importance of nutrient load and residence time. Estuarine Coastal and Shelf Science 50:205-216.

Josefson AB, Blomqvist M, Hansen JLS, Rosenberg R, Rygg B (2009) Assessment of marine benthic quality change in gradients of disturbance: Comparison of different Scandinavian multi-metric indices. Marine Pollution Bulletin 58:1263-1277.

Jørgensen BB (1996). Case Study – Aarhus Bay. In: Eutrophication in Coastal Marine Ecosystems (Eds.: Jørgensen BB and Richardson). Coastal and Estuarine Studies 53:137-154.

Jørgensen BB (ed.) (1996) Stoftransport og stofomsætning i Århus Bugt. Havforskning fra Miljøstyrelsen nr. 59, p. 284.

Jørgensen L, Markager S, Maar M (2014) On the importance of quantifying bioavailable nitrogen instead of total nitrogen. Biogeochemistry 117: 455-472. doi:10.1007/s10533-013-9890-9.

Karle I, Hall POJ, Dahllöf I (2007) Biogeochemical response of an intact coastal sediment to organic matter input : a multivariate approach Marine Ecology Progress Series 342:15-25.

Kemp WM, Boynton WR, Adolf J.E, Boesch D.F, Boicourt WC, Brush G, Cornwell JC, Fisher TR, Glibert PM, Hagy JD, Harding LW, Houde E., Kimmel DG, Miller WD, Newell RIE, Roman MR, Smith EM, Stevenson JC (2005) Eutrophication of Chesapeake Bay: historical trends and ecological interactions. Marine Ecology Progress Series 303:1-29.

Kiørboe T, Møhlenberg F & K Hamburger (1985) Bioenergetics of the planktonic copepod Acartia tonsa: relation between feeding, egg production and respiration, and composition of specific dynamic action. Mar. Ecol. Prog. Ser. 26:85-97.

Kiørboe T, Nielsen TG (1994) Regulation of zooplankton biomass and production in a temperate, coastal ecosystem. I. Copepods. Limnol Oceanogr 39: 493-507.

Klais R, Norros V, SLehtinen S, Tamminen T & K Olli (2017) Community assembly and drivers of phytoplankton functional structure. Functional Ecology 31:760–767.

Klausmeier CA, Litchman E, Daufresne T, Levin SA (2004) Optimal nitrogen-to-phosphorus stoichiometry of phytoplankton. Nature 429:171-174. doi:1.1029/2001GL014649.

Knudsen-Leerbeck H, Mantikci M, Bentzon-Tilia M, Jo-Traving S, Riemann L, Hansen JLS, Markager S (submitted to Biogeochemistry) Seasonal dynamics and bioavailability of dissolved organic matter in two contrasting temperate estuaries.

Krause-Jensen D, Middelboe A (2000) Eelgrass, *Zostera marina*, growth along depth gradients: upper boundaries of the variation as a powerful predictive tool. Oikos 91:233-244. doi:10.1034/j.1600-0706.2001.910204.x

Krause-Jensen D, Greve TM, Nielsen K (2005) Eelgrass as a bioindicator under the European Water Framework Directive. Water Resources Management 19:63-75.

Krause-Jensen D, Rasmussen MB (2009) Historisk udbredelse af ålegræs i danske kystområder. Danmarks Miljøundersøgelser, Aarhus Universitet. 38 s. Faglig rapport fra DMU nr. 755. .

Krause-Jensen D, Carstensen J, Nielsen SL, Dalsgaard T, Christensen PB, Fossing H, Rasmussen MB (2011) Sea bottom characteristics affect depth limits of eelgrass Zostera marina. Mar. Ecol. Prog. Ser. 425:91-102. doi:10.3354/meps09026

Krause-Jensen D, Markager S, Dalsgaard T (2012) Benthic and Pelagic Primary Production in Different Nutrient Regimes. Estuaries and Coasts 35: 527-545.

Kuusemäe K, Rasmussen EK, Vergés PC, Flindt MR (2016) Modelling stressors on the eelgrass recovery process in two Danish estuaries. Ecological Modelling 333:11-42.

Kyllingsbæk A (2008) Landbrugets husholdning med næringsstoffer 1900-2005: Kvælstof, Fosfor og Kalium. Intern rapport. Aarhus Universitet, Det Jorbrugsvidenskabelige Fakultet. 79s.

Kaas H, Møhlenberg F (1990) Kvælstof og fosfor i havet. NPO forskning fra Miljøstyrelsen, Nr C8, p. 64.

Kaas H, Møhlenberg F, Josefson A, Rasmussen B, Krause-Jensen D, Jensen HS, Svendsen LM, Windolf J, Middelboe AL, Sand-Jensen K, Foldager Pedersen M (1996). Marine Områder. Danske fjorde – status over miljøtilstand, årsagssammenhænge og udvikling. Vandmiljøplanernes overvågningsprogram 1995. Danmarks Miljøundersøgelser. 205 s. – Faglig rapport fra DMU nr. 171.

Kaas H, Markager S, Eds. (1998) Technical guidelines for marine monitoring. National Environmental Research Institute, Denmark.

Latimer JS, Rego SA (2010) Empirical relationship between eelgrass extent and predicted watershed-derived nitrogen loading for shallow New England estuaries. Estuar. Coast. Shelf Sci. 90:231-240. doi:10.1016/j.ecss.2010.09.004

Lee KS, Park SR, Kim YK (2007) Effects of irradiance, temperature, and nutrients on growth dynamics of seagrasses: A review. J. Exp. Mar. Bio. Ecol. 350:144-175. doi:10.1016/j.jembe.2007.06.016

Lefèvre N, Taylor A, Gilbert F, Geider R (2003) Modeling carbon to nitrogen and carbon to chlorophyll *a* ratios in the ocean at low latitudes: Evaluation of the role of physiological plasticity. Limnology and Oceanography 48:1796-1807.

Legates DR & GJ Jr McCabe (1999) Evaluating the use of "goodness-of-fit" measures in hydrological and hydroclimatic model validation. Water Resour Res 35:233-241.

Lenhart H-J, Mills DK, Baretta-Bekker H, Van Leeuwen SM, Van Der Molen J, Baretta JW, Blaas M, Desmit X, Kühn W, Lacroix G (2010) Predicting the consequences of nutrient reduction on the eutrophication status of the North Sea. Journal of Marine Systems 81:148-170.

Lessin G, Raudsepp U (2006) Water quality assessment using integrated modeling and monitoring in Narva Bay, Gulf of Finland. Environmental Modelling and Assessment 11:315-33.

Litchman E and CA Klausmeier (2008) Trait-based community ecology of phytoplankton. Annual Review of Ecology, Evolution & Systematics 39:615-639.

Litchman E, Edwards KF & CA Klausmeier (2015) Microbial resource utilization traits and trade-offs: implications for community structure, functioning and biogeochemical impacts at present and in the future. Frontiers in Microbiology 6:254. doi: 10.3389/fmicb.2015.00254.

Lomstein, BAa, Blackburn TH (1992) Havbundens kvælstofomsætning i Århus Bugt. Havforskning for Miljøstyrelsen, Nr. 16. (in Danish).

Lyngsgaard MM, Markager S, Richardson K (2014) Changes in the vertical distribution of primary production in response to land-based nitrogen loading. Limnology and Oceanography 59: 1679-1690. doi:10.4319/lo.2014.59.5.1679.

Lyngsgaard M, Richardson K, Markager S, Nielsen M, Olesen M, Christensen J (2014) Deep primary production in coastal pelagic systems: importance for ecosystem functioning. Mar. Ecol. Prog. Ser. 517:15–33. doi:10.3354/meps11015

Lyngsgaard MM, Markager S, Richardson K (2014) Changes in the vertical distribution of primary production in response to land-based nitrogen loading. Limnol. Oceanogr. 59:1679-1690. doi:10.4319/lo.2014.59.5.1679

MacIsaac JJ, Dugdale RC (1969) The kinetics of nitrate and ammonia uptake by natural populations of marine phytoplankton. Deep-Sea Research and Oceanographic Abstracts 16(1):45-57. doi:10.1016/0011-7471(69)90049-7.

Malve O, Qian SS (2006) Estimating nutrients and chlorophyll *a* relationships in Finnish lakes. Environ. Sci. Technol.40: 7848-7853.

Mariani P, Andersen KH, Visser AW, Barton AD, Kiørboe T (2013) Control of plankton seasonal succession by adaptive grazing. Limnology and. Oceanography 58:173-184.

Markager S, Storm L (2003a) Miljøeffektvurdering for Havmiljøet Del 1: Empirisk modellering af miljøtilstanden i de åbne indre farvande. Institut for Miljøvurdering, 60 pp.

Markager S, Storm L (2003b) Empirisk modellering af sammenhængen mellem vandets klarhed og tilførsler af kvælstof og fosfor for Mariager Fjord. Fagligt notat, 11. juli 2003.

Markager S, Storm L, Stedmon C (2006) Limfjordens miljøtilstand 1985 til 2003. Sammenhæng mellem næringsstoftilførsler, klima og hydrografi belyst ved hjælp af emperiske modeller. Danmarks Miljøundersøgelser. 219 s. - Faglig rapport fra DMU, nr. 577.

Markager S, Bassompierre M, Petersen DLJ (2008) Analyse af miljøtilstanden i Mariager Fjord. Empirisk modellering af miljøtilstanden. Danmarks Miljøundersøgelser, Aarhus Universitet. 50 s. - Faglig rapport fra DMU nr. 685. http://www.dmu.dk/Pub/FR685.pdf

Markager SS, Bassompierre M, Petersen DLJ (2010) Analyse af miljøtilstanden i Horsens Fjord fra 1985 til 2006: Empirisk modellering. Danmarks Miljøundersøgelser, Aarhus Universitet. 84 s. - Faglig rapport fra DMU nr. 733.

Marra J (2009) Review - Net and gross productivity: weighing in with ^{14}C. Aquatic Microbial Ecology 56:123-131. doi:10.3354/ame01306.

Martens HA, Dardenne P (1998) Validation and verification of regression in small data sets. Chemom. Intell. Lab. Syst. 44:99-121. doi:10.1016/S0169-7439(98)00167-1

McCauley E, Downing JA, Watson S (1989) Sigmoid relationships between nutrients and chlorophyll among lakes. Canadian Journal of Fisheries and Aquatic Sciences 46: 1171-1175.

McGlathery KJ, Reidenbach MA, D'Odorico P, Fagherazzi S, Pace ML, Porter JH (2013) Nonlinear Dynamics and Alternative Stable States in Shallow Coastal Systems. Oceanography 26: 220-231.

Meier HEM, Kari Eilola K, Almroth E (2011) Climate-related changes in marine ecosystems simulated with a 3-dimensional coupled physical-biogeochemical model of the Baltic Sea. Climate Research 48: 31-55.

Meier HEM, Andersson HC, Eilola K, Gustafsson BG (2011) Hypoxia in future climates: A model ensemble study for the Baltic Sea. Geophysical Research Letters 38:L24608.

Meier HEM, Muller-Karulis B, Andersson HC, Dieterich C, Eilola K, Gustafsson BG, Hoglund A, Hordoir R, Kuznetsov I, Neumann T, Ranjbar Z, Savchuk OP, Schimanke S (2012) Impact of Climate Change on Ecological Quality Indicators and Biogeochemical Fluxes in the Baltic Sea: A Multi-Model Ensemble Study. AMBIO 41: 558-573.

Meier HEM, Andersson HC, Arheimer B, Donnelly C, Eilola K, Gustafsson B G, Kotwicki L, Neset TS, Niiranen S, Piwowarczyk J, Savchuk OP, Schenk F, Weslawski JM, Zorita E (2014) Ensemble Modeling of the Baltic Sea Ecosystem to Provide Scenarios for Management. AMBIO 43:37-48.

Moll A, Radach G (2003) Review of three-dimensional ecological modelling related to the North Sea shelf system Part 1: models and their results. Progress in Oceanography 57:175-217.

Monod J (1949) The growth of Bacterial cultures. Annual Review of Microbiology 3:371-394.

Moran MA, Pomeroy LR, Sheppard ES, Atkinson LP, Hodson RE (1991) Distribution of terrestrially derived dissolved organic matter on the southeastern U.S. continental shelf. Limnol. Oceanogr. 36:1134–1149. doi:10.4319/lo.1991.36.6.1134

Morán XAG, Lopez-Urrutia A, Calvo-Diaz A & Li WKW (2010) Increasing importance of small phytoplankton in a warmer ocean. Global Change Biol 16:1137–1144.

Morel FMM (1987) Kinetics of nutrient uptake and growth in phytoplankton. J. Phycol. 23:137-150Murray C (2015) Light attenuation in natural waters. Ph.D. thesis, 158 pp.

Murray C (2015) Light attenuation in natural waters. Ph.D. thesis, 158 pp.

Muxika I, Borja A, Bald J (2007) Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. Marine Pollution Bulletin 55:16-29.

Møhlenberg F (1995) Regulating mechanisms of phytoplankton growth and biomass in a shallow estuary. Ophelia 42: 239-256.

Møhlenberg F, Petersen S, Petersen AH, Gameiro C (2007) Long-term trends and short-term variability of water quality in Skive Fjord, Denmark - Nutrient load and mussels are the primary pressures and drivers that influence water quality. Environ. Monit. Assess. 127, 503–521 . doi:10.1007/s10661-006-9298-2.

Maar M (2014) Notat om betydningen af fiskeri for zooplankton og fytoplankton samt en vurdering af det relative græsningstryk på fytoplankton fra zooplankton og bunddyr. Notat fra DCE- Nationalt Center for Miljø og Energi.

Naturstyrelsen (2012) Danmarks Havstrategi. Miljømålsrapport. Danish EPA, 32 pp.

Naturstyrelsens arbejdsgruppe (2012) Ålegræs og marine kvalitetselementer. Naturstyrelsen – Arbejdspapir, december 2012.

Newton A, Icely J, Cristina S, Brito A, Cardoso AC, Colijn F, Riva SD, Gertz F, Hansen JW, Holmer M, Ivanova K, Leppakoski E, Canu DM, Mocenni C, Mudge S, Murray N, Pejrup M, Razinkovas A, Reizopoulou S, Perez-Ruzafa A, Schernewski G, Schubert H, Carr L, Solidoro C, PierluigiViaroli, Zaldivar JM (2014) An overview of ecological status, vulnerability and future perspectives of European large shallow, semi-enclosed coastal systems, lagoons and transitional waters. Estuarine Coastal and Shelf Science 140: 95-122.

Neumann T, Schernewski G (2005) An ecological model evaluation of two nutrient abatement strategies for the Baltic Sea. Journal of Marine Systems 56:195-206.

Neumann T, Schernewski G (2008) Eutrophication in the Baltic Sea and shifts in nitrogen fixation analyzed with a 3D ecosystem model. Journal of Marine Systems 74:592-602.

Neumann T (2010) Climate-change effects on the Baltic Sea ecosystem: A model study. Journal of Marine Systems 81:213-224.

Nielsen TG & T Kiørboe (1994) Regulation of zooplankton biomass and production in a temperate, coastal ecosystem. II. Ciliates. Limnol Oceanogr, 39: 508-519.

Nielsen K, Nielsen LP, Rasmussen P (1995) Estuarine nitrogen retention independently estimated by the denitrification rate and mass balance methods: a study of Norsminde Fjord, Denmark. Oceanographic Literature Review 42(11):958.

Nielsen K, Risgaard-Petersen N, Sømod B, Rysgaard S, Bergø T (2001) Nitrogen and phosphorus retention estimated independently by flux measurements and dynamic modelling in the estuary, Randers Fjord, Denmark. Marine Ecology Progress Series 219:25-40.

Nielsen SL, Sand-Jensen K, Borum J, Geertz-Hansen O (2002) Depth colonization of eelgrass (*Zostera marina*) and macroalgae as determined by water transparency in Danish coastal waters. Estuaries 25:1025-1032

Nitrogen Working Group (2012) http://www.naturoglandbrug.dk/kv%c3%a6lstof.aspx?ID=50720 (in Danish).

Nixon SW (1995) Coastal marine eutrophication: A definition, social causes, and future concerns. Ophelia. doi:10.1080/00785236.1995.10422044.

Nixon SW, Ammerman SW, Atkinson LP, Berounsky VM, Billen G, Boicourt WC, Boynton WR, Church TM, Ditoro DM, Elmgren R, Garber JH, Giblin AE, Jahnke RA, Owens NJP, Pilson MEQ. Seitzinger SP (1996) The fate of nitrogen and phosphorus at the land-sea margin of the North Atlantic Ocean. Biogeochemistry 35(1):141-180.

Nobre AM, Ferreira JG, Newton A, Simas T, Icely JD, Neves R (2005) Management of coastal eutrophication: Integration of field data, ecosystem-scale simulations and screening models. Journal of Marine Systems 56: 375-390.

Nowicki BL, 1994. The Effect of Temperature, Oxygen, Salinity, and Nutrient Enrichment on Estuarine Denitrification Rates Measured with a Modified Nitrogen Gas Flux Technique. Estuar. Coast. Shelf Sci. doi:10.1006/ecss.1994.1009

Nowicki B, Kelly J, Requintina E, Keuren D van (1997) Nitrogen losses through sediment denitrification in Boston Harbor and Massachusetts Bay. Estuaries 20, 626–639.

Nyholm N (1977) Kinetics of phosphate limited algal growth. Biotechnology and Bioengineering 19(4): 467–492.

Nyholm N (1978) A simulation model for phytoplankton growth and nutrient cycling in eutrophic, shallow lakes. Ecological Modelling 4(2): 279-310.

Olesen B (1996) Regulation of light attenuation and eelgrass *Zostera marina* depth distribution in a Danish embayment. Marine Ecology Progress Series 134:187-194. doi:10.3354/meps134187.

Olesen B, Sand-Jensen K (1993) Seasonal acclimatization of eelgrass Zostera marina growth to light. Mar. Ecol. Prog. Ser. 94:91-99. doi:10.3354/meps094091

Olsen Y, Agustí S, Andersen T, Duarte CM, Gasol JM, Gismervik I, Heiskanen AS, Hoell E, Kuuppo P, Lignell R, Reinertsen H, Sommer U, Stibor H, Tamminen T, Vadstein O, Vaqué D, Vidal M (2006) A comparative study of responses in planktonic food web structure and function in contrasting European coastal waters exposed to experimental nutrient addition. Limnology and Oceanography 51:488-503.

Orth RJ, Moore KA, Marion SR, Wilcox DJ, Parrish DB (2012) Seed addition facilitates eelgrass recovery in a coastal bay system. Marine Ecology Progress Series 448:177-195.

OSPAR (1988) PARCOM Recommendation 88/2 of 17 June 1988 on the reduction in inputs of nutrients to the Paris Convention area. OSPAR Commission.

OSPAR (2005) Common Procedure for the identification of the eutrophication status of the OSPAR maritime area. OSPAR Commission. OSPAR agreement 2005-3.

OSPAR (2006) Interim report on the use of eutrophication modelling for predicting expected eutrophication status of the OSPAR maritime area following the implementation of agreed measures. OSPAR Commission. OSPAR publication 286/2006.

OSPAR (2008a) Overview of the state-of-the-art of models and their use in predictive eutrophication assessments. OSPAR Commission. OSPAR publication 373/2008.

OSPAR (2008b) Nutrient reduction scenarios for the North Sea. Environmental consequences for problems areas with regard to eutrophication following nutrient reductions in models scenarios. OSPAR Commission. Publication Number: 374/2008.

Pedersen MF, Borum J (1996) Nutrient control of algal growth in estuarine waters: nutrient limitation and the importance of nitrogen requirements and nitrogen storage among phytoplankton and species of macroalgae. Mar. Ecol. Prog. Ser. 142: 261–272.

Petersen JK, Hansen JW, Laursen MB, Clausen P, Carstensen J, Conley DJ (2008) Regime shift in a coastal marine ecosystem. Ecological Applications 18:497-510.

Pingree RD, Holligan PM, Mardell GT, Head RN (1976) The influence of physical stability on spring, summer and autumn phytoplankton blooms in the Celtic Sea. J. Mar. Biol. Assoc. United Kingdom 56: 845–873.
doi: 10.1017/S0025315400020919.

Pingree RD, Holligan PM, Mardell GT (1978) The effects of vertical stability on phytoplankton distributions in the summer on the northwest European Shelf. Deep. Res. 25. doi:10.1016/0146-6291(78)90584-2.

Plattner GK, Joos F, Stocker TF, Marchal O (2001) Feedback mechanisms and sensitivities of ocean carbon uptake under global warming. Tellus B 53:564–592. doi:10.1034/j.1600-0889.2001.530504.x.

Radach G, Patsch J (1997) Climatological annual cycles of nutrients and chlorophyll in the North Sea. Journal of Sea Research 38:231-248.

Radach G, Moll A (2006) Review of three-dimensional ecological modelling related to the North Sea shelf system - Part II: Model validation and data needs. Oceanography and Marine Biology 44:1-60.

Rasheed MA, Unsworth RKF (2011) Long-term climate-associated dynamics of a tropical seagrass meadow : implications for the future 422:93–103.
doi:10.3354/meps08925.

Rasmussen B, Josefson AB (2002) Consistent estimates for the residence time of micro-tidal estuaries. Estuarine Coastal and Shelf Science 54:65-73.

Rasmussen EK, Petersen OS, Thompson JR, Flower RJ, Ayache F, Kraiem M, Chouba L (2009) Model Analyses of the Future Water Quality of the Eutrophicated Ghar El Melh Lagoon (Northern Tunisia). Hydrobiologia 622:173-193. doi:10.1007/s10750-008-9681-9.

Riemann B, Carstensen J, Dahl K, Fossing H, Hansen JW, Jakobsen H, Josefson AB, Krause-Jensen D, Markager S, Stæhr PA, Timmermann K, Windolf J, Andersen JH (2016) Recovery of Danish coastal ecosystems after reductions in nutrient loading: A holistic ecosystem approach. Estuaries and Coasts 39(1):82-97. doi:10.1007/s12237-015-9980-0.

Rossberg AG, Uusitalo L, Berg T, Zaiko A, Chenuil A, Uyarra AC, Borja A & CP Lynam (2017) Quantitative criteria for choosing targets and indicators for sustainable use of ecosystems. Ecological Indicators 72:215–224.

Scavia D, Rabalais NN, Turner RE, Justić D, Wiseman WJ (2003) Predicting the response of Gulf of Mexico hypoxia to variations in Mississippi River nitrogen load. Limnologi and Oceanography 48:951-956.
doi:10.4319/lo.2003.48.3.0951

Sand-Jensen K, Nielsen SL, Borum J, Geertz-Hansen O (1994). Fytoplankton- og makrofytudvikling i danske kystområder. Miljøstyrelsen.

Savchuk OP, Wulff ., Hille S, Humborg C, Pollehne F (2008) The Baltic Sea a century ago - a reconstruction from model simulations, verified by observations. Journal of Marine Systems 74:485-494.

Savchuk OP, Eremina TR, Isaev AV, Neelov IA (2009) Response of eutrophication in the eastern Gulf of Finland to nutrient load reduction scenarios. Hydrobiologia 629:225-237.

Savchuk OP, Eilola K, Gustafsson BG, Medina MR, Ruoho-Airol T (2012a). Long-term reconstruction of nutrient loads to the Baltic Sea, 1850-2006. Baltic Nest Institute. Technical Report Series Nr 6.

Savchuk OP, Gustafsson BG, Medina MR, Sokolov AV, Wulf FV. (2012b). External nutrient loads to the Baltic Sea, 1970-2006. Baltic Nest Institute. Technical Report no. 5.

Scheffer M, Jeppesen E (2007) Regime Shifts in Shallow Lakes. Ecosystems 10:1-3.

Schernewski G, Neumann T (2005) The trophic state of the Baltic Sea a century ago: a model simulation study. Journal of Marine Systems 53:109-124.

Schernewski G, Friedland R, Carstens M, Hirt U, Leujak W, Nausch G, Neumann T, Petenati T, Sagert S, Wasmund N, von Weber M. (2015) Implementation of European marine policy: New water quality targets for German Baltic waters. Marine Policy 51:305-321.

Seitzinger S, Harrison JA, Böhlke JK, Bouwman AF, Lowrance R, Peterson, B, Tobias C, and Van Drecht G (2006) Denitrification Across Landscapes and Waterscapes: A Synthesis. Ecological Applications 16(6):2064–2090.

Short FT, Burdick DM, Kaldy JEI (1995) Mesocosm experiments quantify the effects of eutrophication on eelgrass, *Zostera marina*. Limnology and Oceanography 40:740-749. doi:10.4319/lo.1995.40.4.0740.

Skogen D, Svendsen E, Ostrowski M (1998) Quantifying volume transports during SKAGEX with the Norwegian ecological model system. Continental Shelf Research 17: 15: 1817-1837.

Smith SL, Merico A, Wirtz KW, Pahlow M (2014) Leaving misleading legacies behind in plankton ecosystem modelling. Journal of Plankton Research 36:613-620.

Smith SL, Pahlow M, Merico A, Acevedo-Tréjos E, Sasai Y, Yoshikawa C, Sasaoka K, Fujiki T, Matsumoto K, Honda MC (2015) Flexible phytoplankton functional type (FlexPFT) model: size-scaling of traits and optimal growth. Journal of Plankton Research 38(4):977-992. doi:10.1093/plankt/fbv038.

Soetaert K, Middelburg JJ, Herman PMJ, Buis K (2000) On the coupling of benthic and pelagic biogeochemical models. Earth-Science Reviews 51:173-201.

Sommer U, Peter KH, Genitsaris S, Moustaka-Gouni M (2016) Do marine phytoplankton follow Bergmann's rule *sensu lato*? Biological Reviews 92(2): 1011-1026. doi:10.1111/brv.12266.

Stedmon CA, Markager S, Søndergaard M, Vang T, Laubel A, Borch NH, Windelin A, (2006) Dissolved organic matter (DOM) export to a temperate estuary: seasonal variations and implications of land use. Estuaries and Coasts 29:388-400. doi:10.1007/BF02784988.

Steemann Nielsen E (1955) The interaction of photosynthesis and respiration and its importance for the determination of ^{14}C-discrimination in photosynthesis. Physiologia Plantarum 8:945-953.

Stepanauskas R, Jørgensen NOG, Eigaard OR, Zvikas A, Tranvik LJ, Leonardson L (2002) Summer inputs of riverine nutrients to the Baltic Sea: bioavailability and eutrophication relevance. Ecological Monographs 72:579-597.

Tabachnick BG, Fidell LS, Osterlind SJ (2001) Using multivariate statistics.

Taucher J, Oschlies A (2011) Can we predict the direction of marine primary production change under global warming? Geophys. Res. Lett. 38:1–6. doi:10.1029/2010GL045934

Taucher J, Schulz KG, Dittmar T, Sommer U, Oschlies A, Riebesell U (2012) Enhanced carbon overconsumption in response to increasing temperatures during a mesocosm experiment. Biogeosciences 9: 3531–3545. doi:10.5194/bg-9-3531-2012

Thieu V, Garnier J, Billen G (2010) Assessing the effect of nutrient mitigation measures in the watersheds of the Southern Bight of the North Sea. Science of the Total Environment 408:1245-1255.

Thomann RV, DM Di Toro & DJ O'Connor (1974) Preliminary Model of Potomac Estuary Phytoplankton. J. Environ. Engr. 100:699-715.

Tian RC (2006) Toward standard parameterizations in marine biological modeling. Ecological Modelling 193:363-386.

Timmermann K, Markager S, and Gustafsson KE (2010) Streams or open sea? Tracing sources and effects of nutrient loadings in a shallow estuary with a 3D hydrodynamic-ecological model. Journal of Marine Systems 82: 111-121.

Timmermann K, Dinesen GE, Markager S, Ravn-Jonsen L, Bassompierre M, Støttrup JG (2013) Development and use of a bioeconomic model for management of mussel fisheries under different nutrient regimes in the temperate estuary of the Limfjord, Denmark. Ecology and Society 19(1):14. doi:10.5751/ES-06041-190114.

Timmermann K, Stæhr P, Göke C, Egekvist J (2015) Dansk erhvervsfiskeri's påvirkning af ålegræs. Notat fra DCE- Nationalt Center for Miljø og Energi.

Topcu D, Behrendt H, Brockmann U, Claussen U (2011) Natural background concentrations of nutrients in the German Bight area (North Sea). Environmental Monitoring and Assessment 174:361-388.

UNESCO (1981) The practical salinity scale 1978 and the international equation of state of seawater 1980. UNESCO technical papers on marine science.

Van Denderen PD, Hintzen NT, Rijnsdorf AD, Ruardij P, van Kooten T (2014) Habitat-specific effects of fishing disturbance on benthic species richness in marine soft sediments. Ecosystems 17:1216-1226.

van der Heide T, van Nes E., van Katwijk MM, Olff H, Smolders AJ (2011) Positive Feedbacks in Seagrass Ecosystems - Evidence from Large-Scale Empirical Data. PLoS ONE 6(1): e16504. https://doi.org/10.1371/journal.pone.0016504

Vichi M, Pinardi N, Masina S (2007) A generalized model of pelagic biogeochemistry for the global ocean ecosystem. Part I: Theory. J Marine Syst 64:89–109.

Villars M, de Vries I (1998) Report of the ASMO Modelling Workshop on Eutrophication Issues, 5-8 November 1996, The Hague, The Netherlands. M. Villars M. & de Vries I. (Eds). Assessment and Monitoring, OSPAR Commission.

Vodacek A, Blough NV, DeGrandpre MD, DeGrandpre MD, Nelson RK, (1997) Seasonal variation of CDOM and DOC in the Middle Atlantic Bight: Terrestrial inputs and photooxidation. Limnol. Oceanogr. 42:674–686 . doi:10.4319/lo.1997.42.4.0674

Våge S, Castellani M, Giske J, Thingstad TF (2013) Successful strategies in size structured mixotrophic food webs. Aquatic Ecology 47:329-347.

Ward BA, Schartau M, Oschlies A, Martin AP, Follows MJ, Anderson TR (2013) When is a biogeochemicalmodel too complex? Objective model reduction and selection for North Atlantic time-series sites. Prog. Oceanogr. 116,: 49-65, doi:10.1016/j.pocean.2013.06.002.

Ward BA, Follows MJ (2016) Marine mixotrophy increases trophic transfer efficiency, mean organism size, and vertical carbon flux. Proceeding of the National Academy of Sciences of the USA 113:2958-2963.

Windolf J, Timmermann A, Kjeldgaard A, Bøgestrand J, Larsen SL, Thodsen H (2013) Landbaseret tilførsel af kvælstof og fosfor til danske fjorde og kystafsnit, 1990-2011. Aarhus Universitet, DCE – Nationalt Center for Miljø og Energi, 110 s. – Teknisk rapport fra DCE - Nationalt Center for Miljø og Energi nr. 31. http://dce2.au.dk/pub/TR31.pdf.

Wold S, Sjöström M, Eriksson L (2001) PLS-regression: A basic tool of chemometrics. Chemometrics and Intelligent Laboratory Systems 58:109-130.

Wulff F, Stigebrandt A, and Rahm L (1990) Nutrient Dynamics of the Baltic Sea. Ambio 19:126-133.

Zaldívar J-M, Cardoso AC, Viaroli P, Newton A, de Wit R, Ibañez C, Reizopoulou S, Somma F, Razinkovas A, Basset A, Holmer M, Murray N (2008) Eutrophication in transitional waters: an overview. Transitional Waters Monographs 1:1-78.

Ærtebjerg G (ed), Andersen JH (ed), Bendtsen J, Carstensen J, Christiansen T, Dahl K, Dahllöf I, Ellermann T Fossing H, Greve TM, Gustafsson K, Hansen JLS, Henriksen P, Josefson AB, Krause-Jensen D, Larsen MM, Markager S, Nielsen TG, Ovesen NB, Petersen JK, Riemann B, Risgaard-Petersen N, Skjøth CA, Stedmon C, Strand J, Nielsen S, Jensen JB, Madsen H (2004) Marine områder 2003 – Miljøtilstand og udvikling. 97 s. – Faglig rapport fra DMU nr. 513.

Appendix A – Danish water bodies

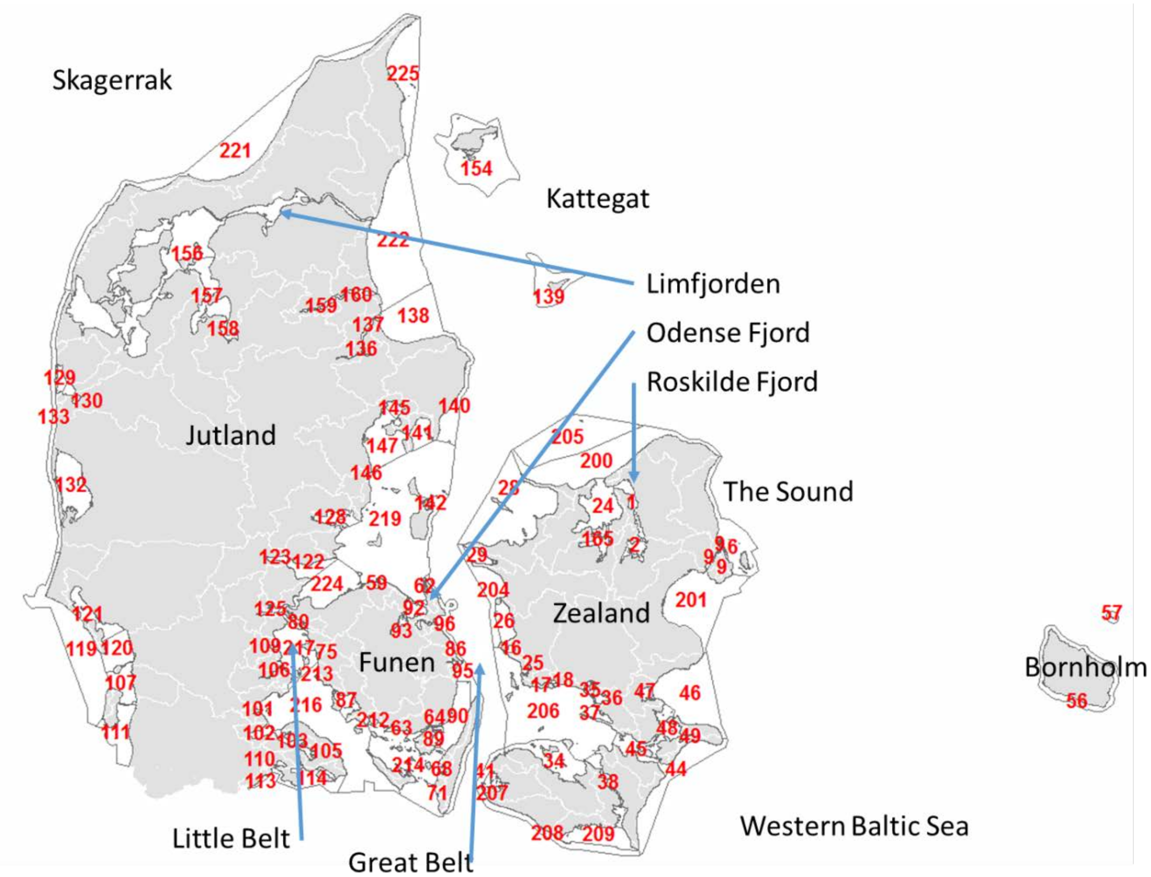


Figure A1. Danish water bodies. Numbers indicate the specific water body number and the corresponding name is included in *Table A1*.

Table A1. Water body numbers and names.

Water body no.	Water body name	Danish typology based on Dahl et al. 2005	Water body type this study
1	Roskilde Fjord, ydre	M2	2
2	Roskilde Fjord, indre	M2	3
6	Nordlige Øresund	OW2	1.1
9	København Havn	OW3a	1.1
16	Korsør Nor	M2	2
17	Basnæs Nor	M2	2
18	Holsteinsborg Nor	M2	2
24	Isefjord, ydre	P2	2
25	Skælskør Fjord og Nor	M2	2
26	Musholm Bugt, indre	M2	2
28	Sejerøbugt	OW2	1.3
29	Kalundborg Fjord	P3	2
34	Smålandsfarvandet, syd	M1	2
35	Karrebæk Fjord	M3	3
36	Dybsø Fjord	M2	2
37	Avnø Fjord	M2	2

Water body no.	Water body name	Danish typology based on Dahl et al. 2005	Water body type this study
38	Guldborgssund	M2	2
41	Langelandsbælt, øst	OW3a	1.2
44	Hjelm Bugt	OW3a	1.1
45	Grønsund	OW3a	1.2
46	Fakse Bugt	OW3a	1.1
47	Præstø Fjord	M2	2
48	Stege Bugt	M2	2
49	Stege Nor	M2	2
56	Østersøen, Bornholm	OW3b	1.1
57	Østersøen, Christiansø	OW3b	1.1
59	Nærå Strand	M4	3
61	Dalby Bugt	P4	3
62	Lillestrand	P2	2
63	Nakkebølle Fjord	M3	3
64	Skårupøre Sund	M2	2
65	Thurøbund	M2	2
68	Lindelse Nor	M2	2
69	Vejlen	O4	4
70	Salme Nor	O4	4
71	Tryggelev Nor	O4	4
72	Kløven	M2	2
74	Bredningen	M3	3
75	Emtekær Nor	M4	3
76	Orestrand	M2	2
78	Gamborg Nor	O4	2
80	Gamborg Fjord	P1	2
81	Bågø Nor	M2	2
82	Aborgminde Nor	M3	3
83	Holckenhavn Fjord	M3	3
84	Kerteminde Fjord	P3	3
85	Kertinge Nor	P2	2
86	Nyborg Fjord	P3	3
87	Helnæs Bugt	M1	2
89	Lunkebugten	M2	2
90	Langelandssund	OW3a	1.2
92	Odense Fjord, ydre	P3	3
93	Odense Fjord, indre	M4	3
95	Storebælt, SV	OW3a	1.1
96	Storebælt, NV	OW2	1.4
101	Genner Bugt	P1	2
102	Åbenrå Fjord	P1	2
103	Als Fjord	P1	2
104	Als Sund	M1	2
105	Augustenborg Fjord	M2	2
106	Haderslev Fjord	M1	2
107	Juvre Dyb, tidevandsområde	OW5	NS
108	Avnø Vig	M2	2
109	Hejlsminde Nor	M2	2
110	Nybøl Nor	P1	2
111	Lister Dyb	OW5	NS
113	Flensborg Fjord, indre	P1	2

Water body no.	Water body name	Danish typology based on Dahl et al. 2005	Water body type this study
114	Flensborg Fjord, ydre	P1	2
119	Vesterhavet, syd	OW4	NS
120	Knudedyb tidevandsområde	OW5	NS
121	Grådyb tidevandsområde	OW5	NS
122	Vejle Fjord, ydre	P3	3
123	Vejle Fjord, indre	P3	3
124	Kolding Fjord, indre	P3	3
125	Kolding Fjord, ydre	P3	3
127	Horsens Fjord, ydre	P3	3
128	Horsens Fjord, indre	P3	3
129	Nissum Fjord, ydre	Slusefjord	5
130	Nissum Fjord, mellem	Slusefjord	5
131	Nissum Fjord, Felsted Kog	Slusefjord	5
132	Ringkøbing Fjord	Slusefjord	5
133	Vesterhavet, nord	OW4	NS
135	Randers Fjord, Grund Fjord	O4	4
136	Randers Fjord, Randers-Mellerup	O3	4
137	Randers Fjord, ydre	M3	3
138	Hevring Bugt	OW2	1.3
139	Anholt	OW2	1.3
140	Djursland Øst	OW2	1.4
141	Ebeltoft Vig	P1	2
142	Stavns Fjord	P2	2
144	Knebel Vig	P1	2
145	Kalø Vig, indre	P1	2
146	Norsminde Fjord	M4	3
147	Århus Bugt, Kalø og Begtrup Vig	P3	2
154	Kattegat, Læsø	OW2	1.3
156	Nissum Bredning, Thisted Bredning, Kås Bredning, Løgstør Bredning, Nibe Bredning og Langerak	P4	3
157	Bjørnholms Bugt, Riisgårde Bredning, Skive Fjord og Lovns Bredning	P3	UK
158	Hjarbæk Fjord	Slusefjord	5
159	Mariager Fjord, indre	M1	3
160	Mariager Fjord, ydre	P1	3
165	Isefjord, indre	P2	2
200	Kattegat, Nordsjælland	OW2	1.3
201	Køge Bugt	OW3a	1.1
204	Jammerland Bugt	OW2	1.3
205	Kattegat, Nordsjælland >20 m	OW1	1.3
206	Smålandsfarvandet, åbne del	OW3a	1.2
207	Nakskov Fjord	M1	2
208	Femerbælt	OW3a	1.2
209	Rødsand	M2	2
212	Faaborg Fjord	M2	2
213	Torø Vig og Torø Nor	M2	2
214	Det Sydfynske Øhav	OW3a	1.2
216	Lillebælt, syd	OW3a	1.2
217	Lillebælt, Bredningen	OW3a	1.2
219	Århus Bugt syd, Samsø og Nordlige Bælthav	OW2	1.3

Water body no.	Water body name	Danish typology based on Dahl et al. 2005	Water body type this study
221	Skagerrak	OW4	NS
222	Kattegat, Aalborg Bugt	OW2	1.3
224	Nordlige Lillebælt	OW2	1.3
225	Nordlige Kattegat, Ålbæk Bugt	OW1	1.4

Appendix B – Statistical model evaluation

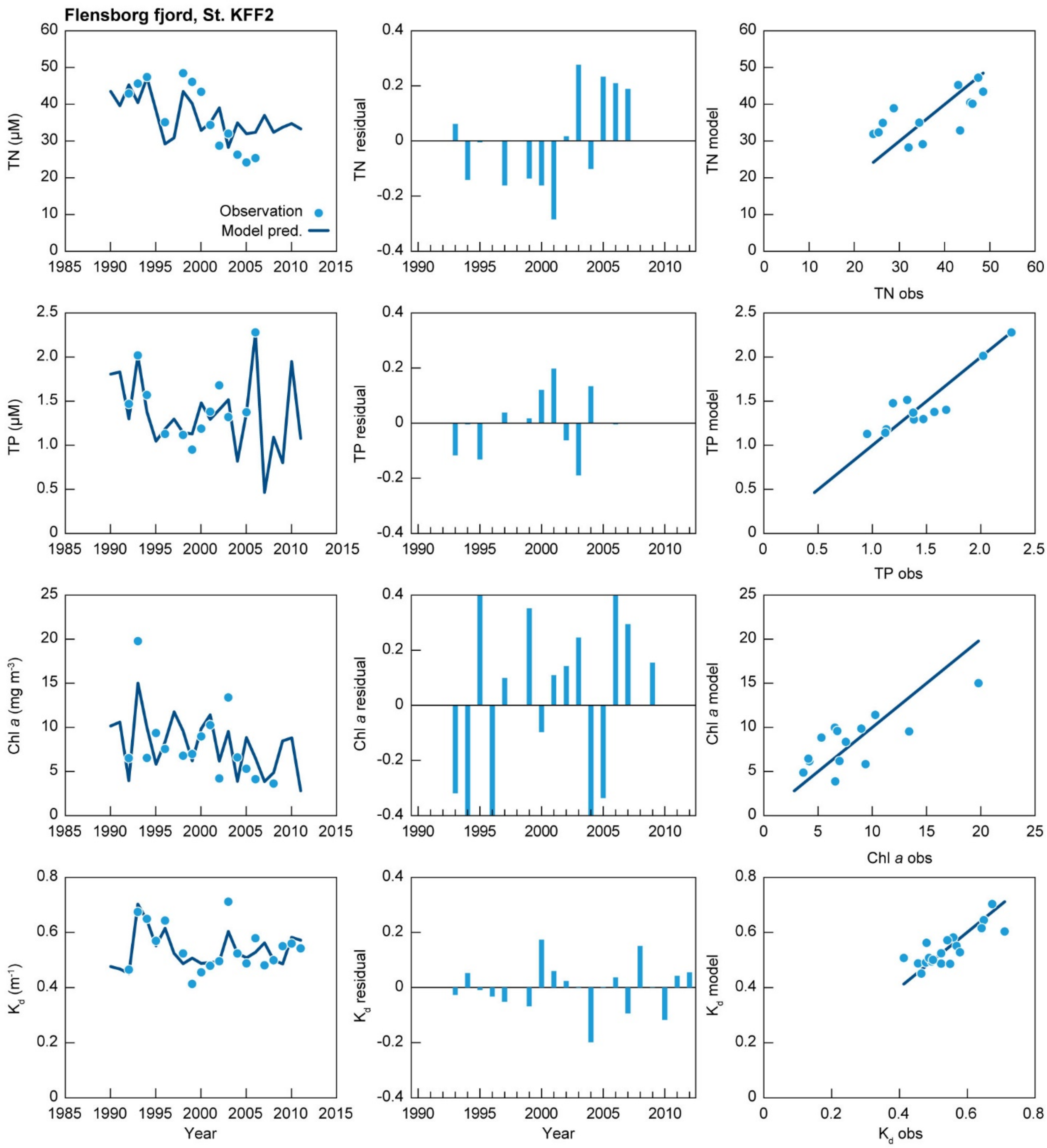
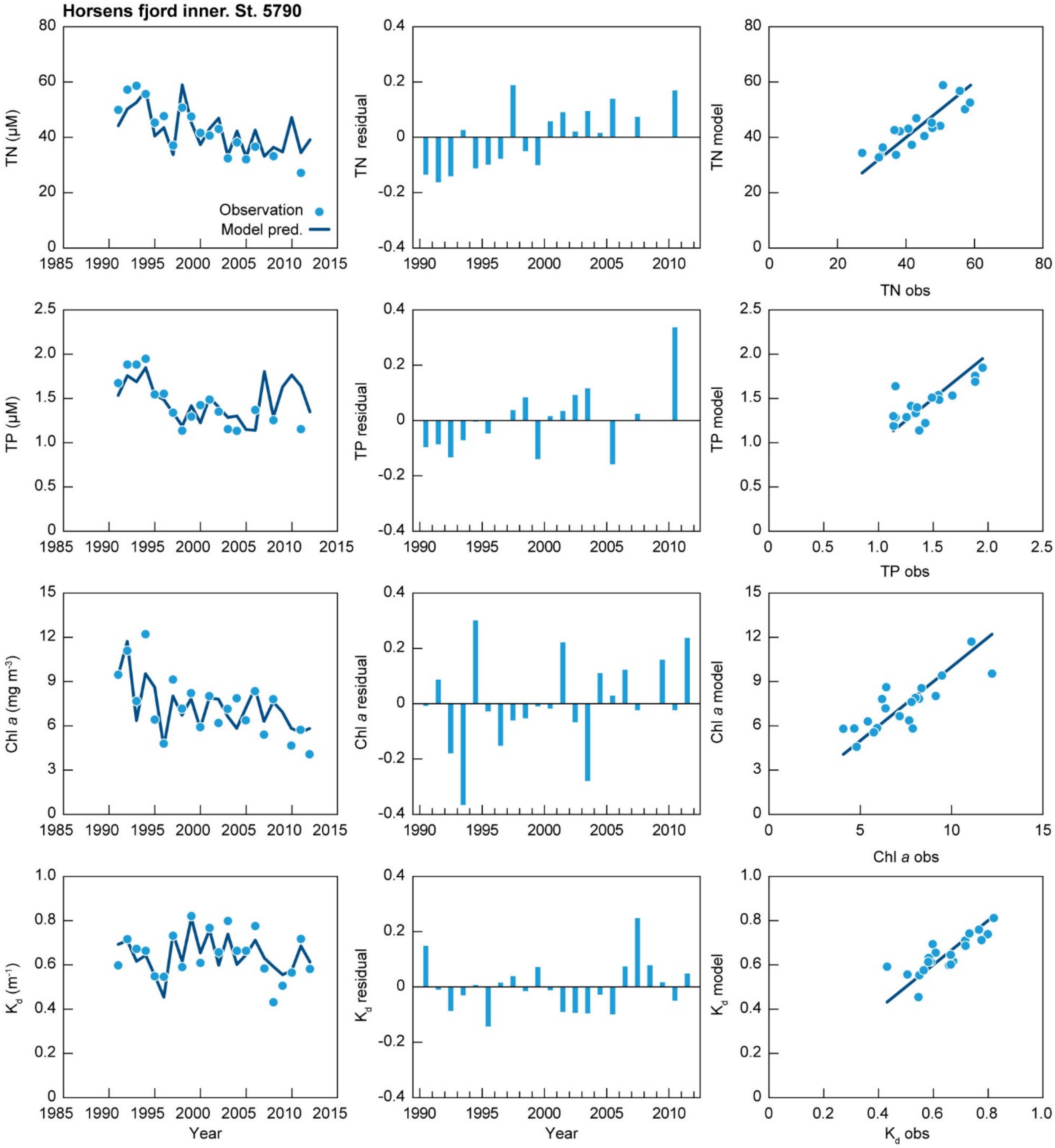
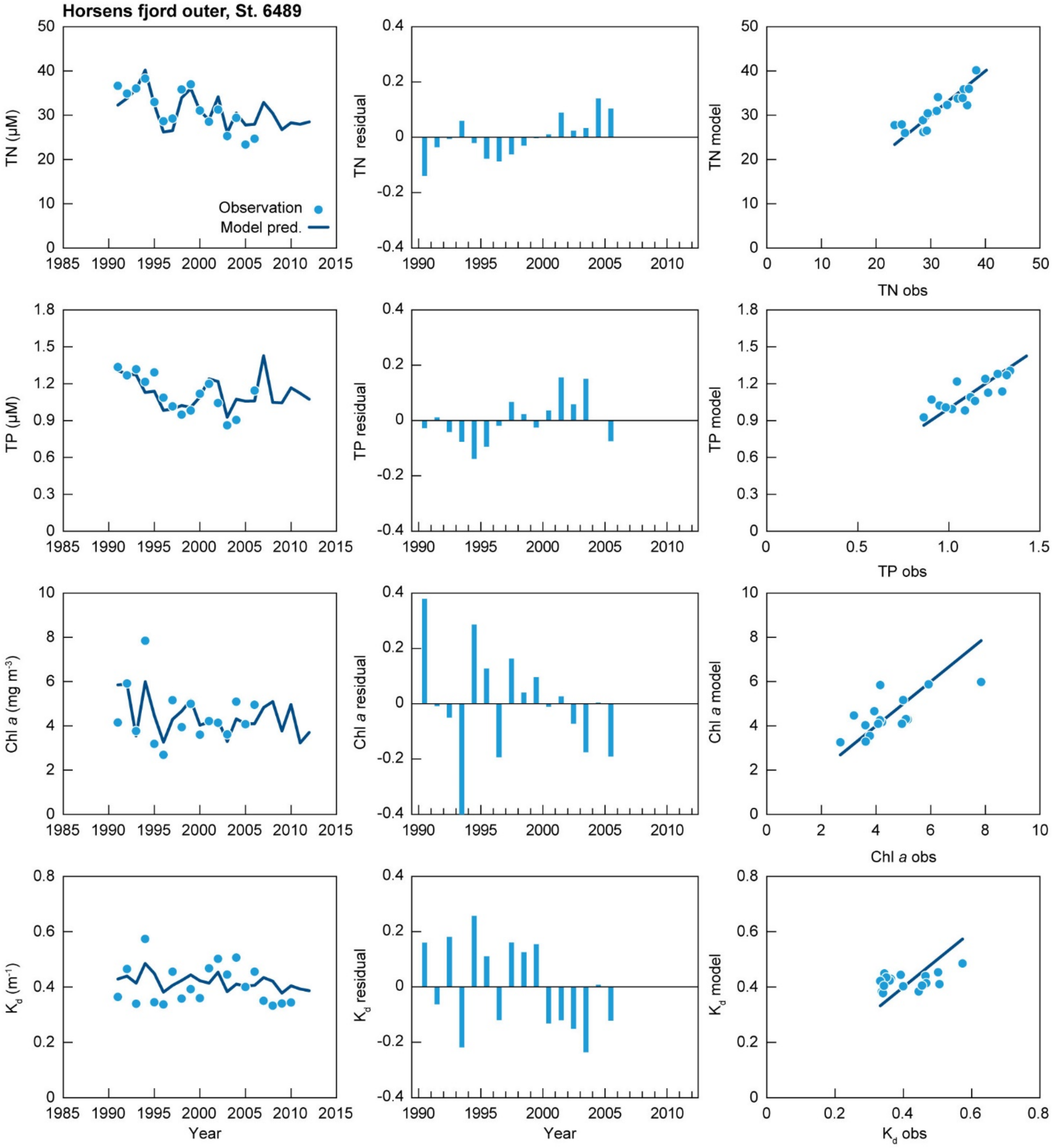


Figure B1.



Horsens fjord inner. St. 5790
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
Kd (m⁻¹)
Kd residual
Kd model
Kd obs
Year

Figure B2.



Horsens fjord outer, St. 6489
TN (µM)
TN residual
TN model
TN obs
Observation
Model pred.
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year
Year

Figure B3.

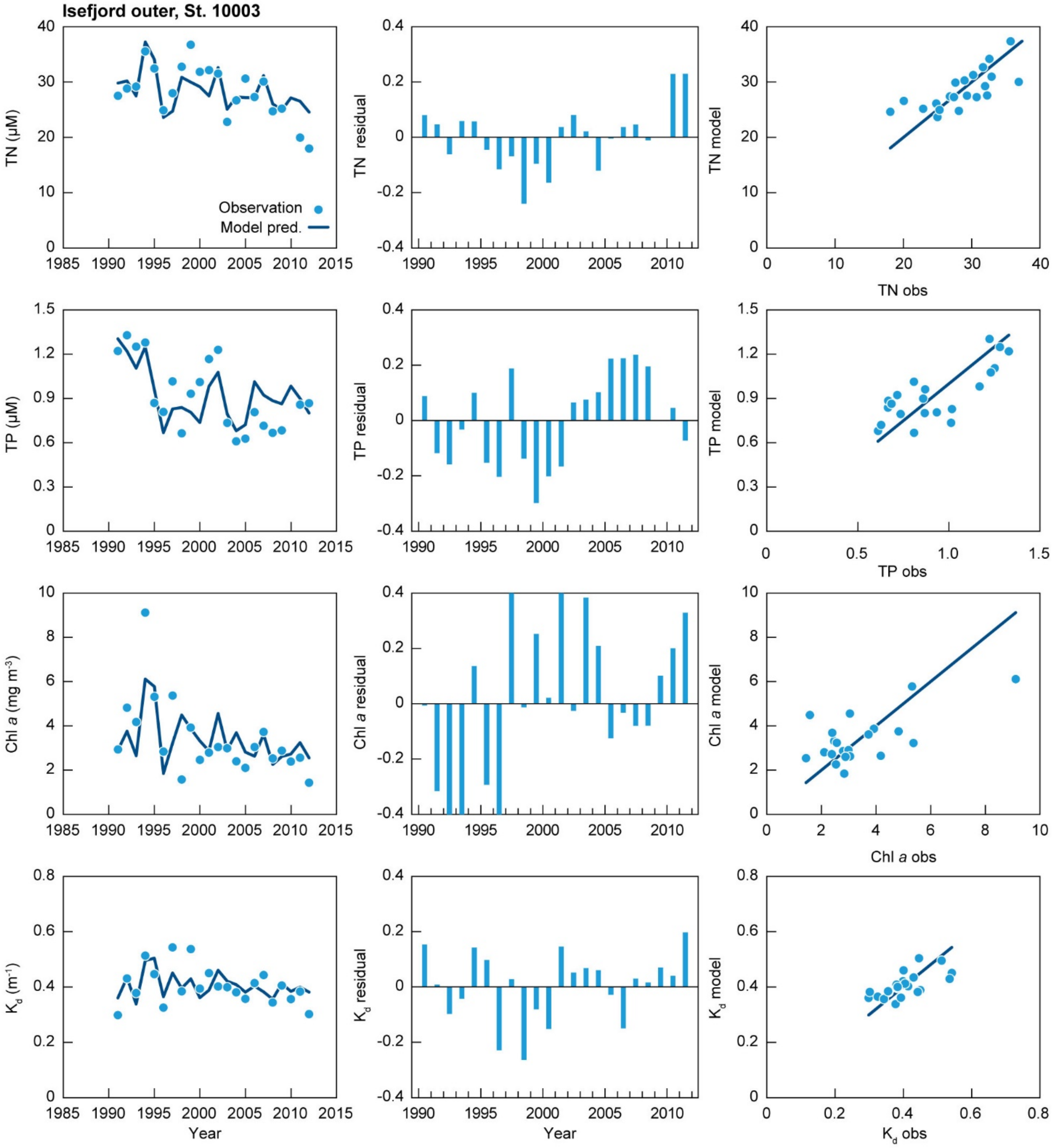

Isefjord outer, St. 10003
Observation
Model pred.
TN (µM)
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl *a* (mg m⁻³)
Chl *a* residual
Chl *a* model
Chl *a* obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year

Figure B4.

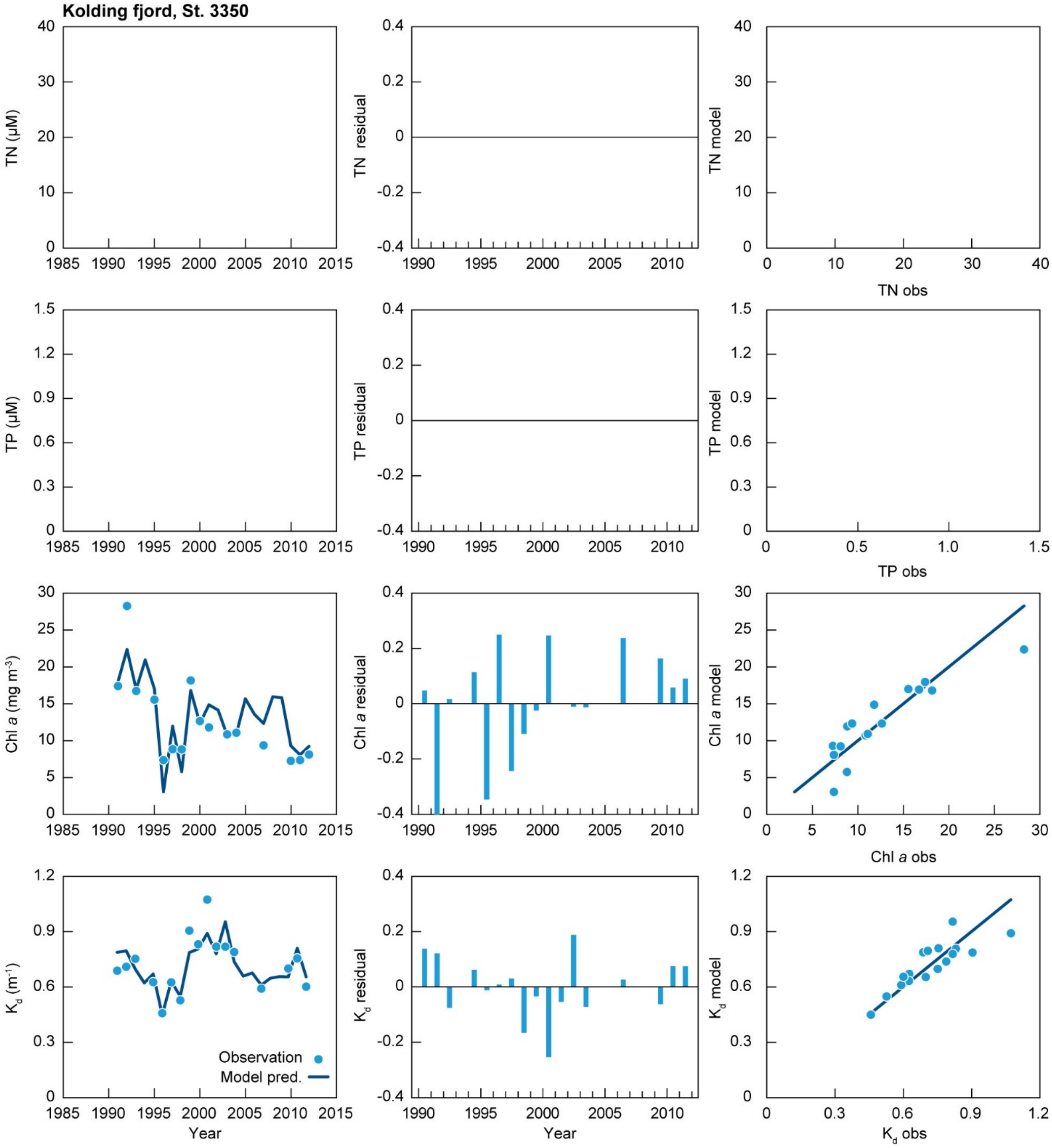
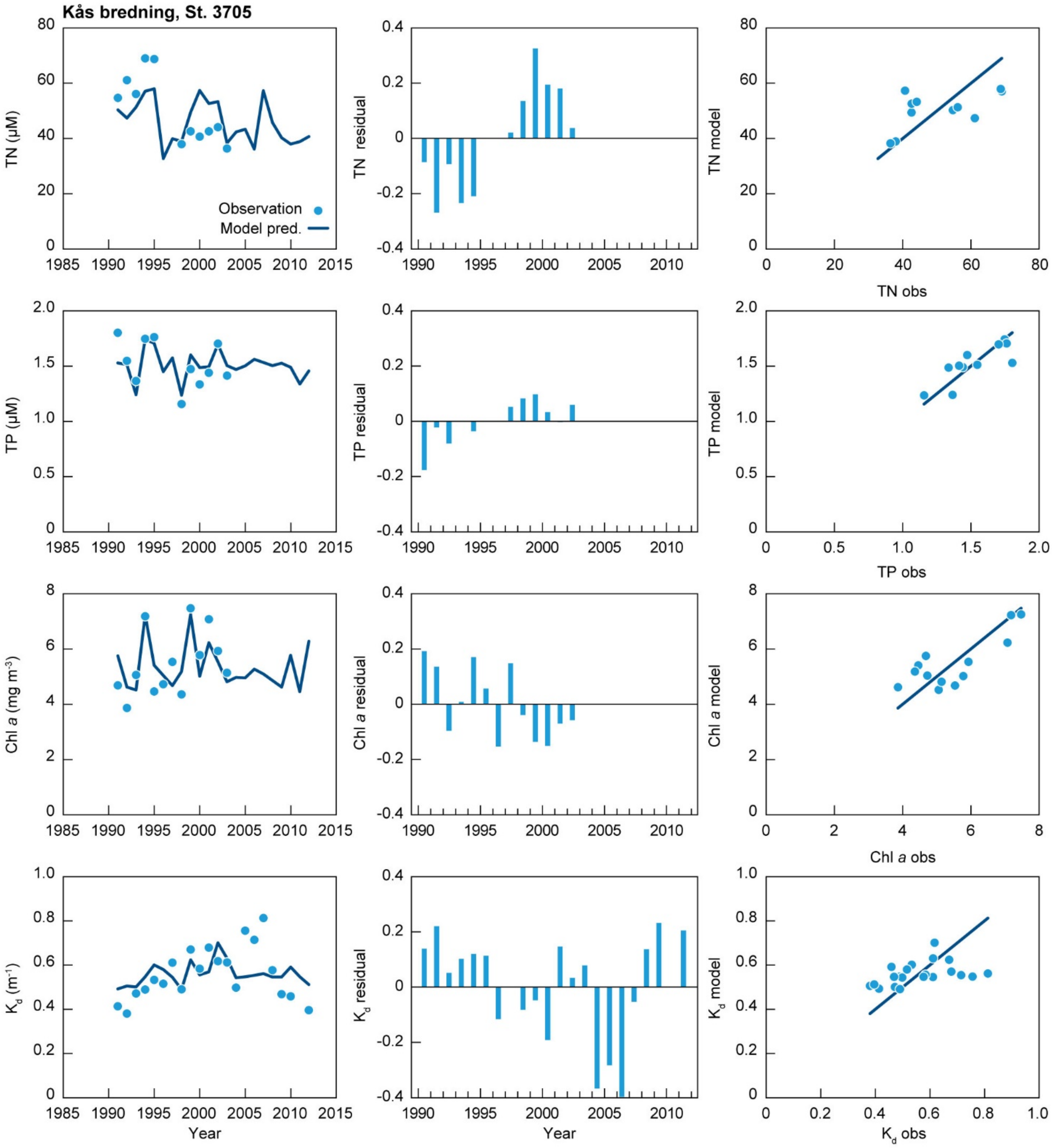
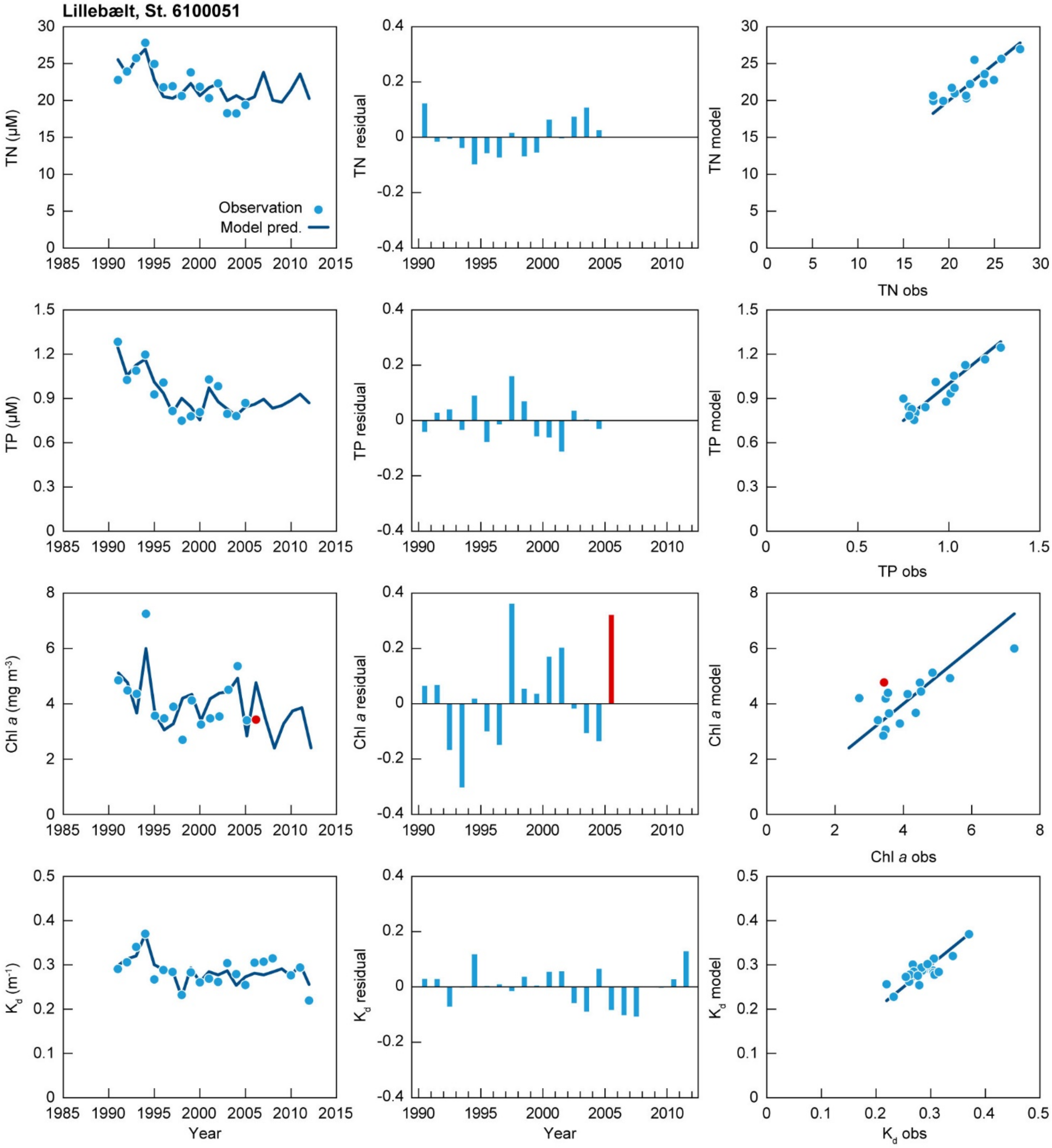

Kolding fjord, St. 3350
TN (µM)
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Observation
Model pred.
Year

Figure B5.



Kås bredning, St. 3705
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year
Year

Figure B6.



Lillebælt, St. 6100051
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year

Figure B7.

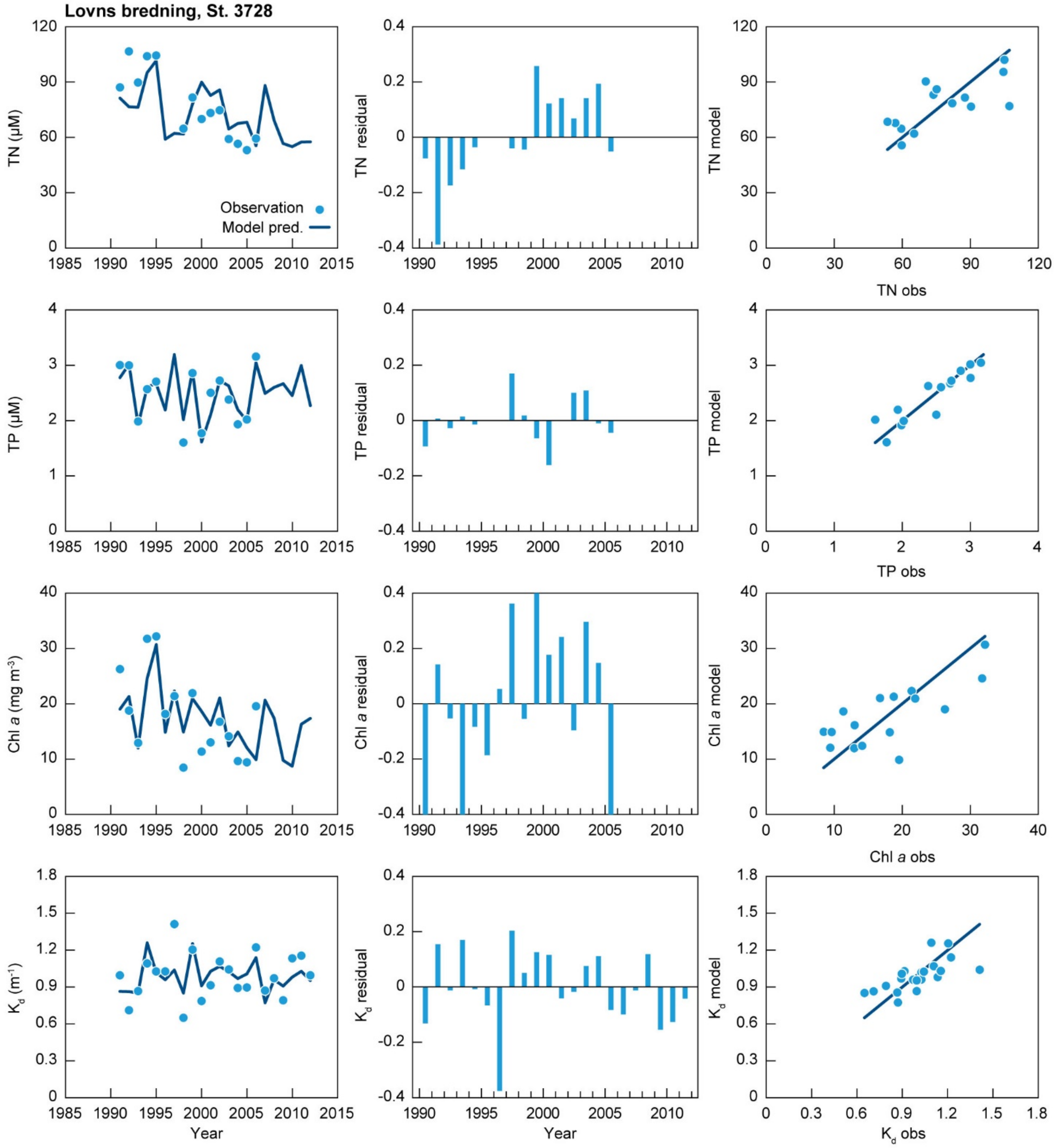
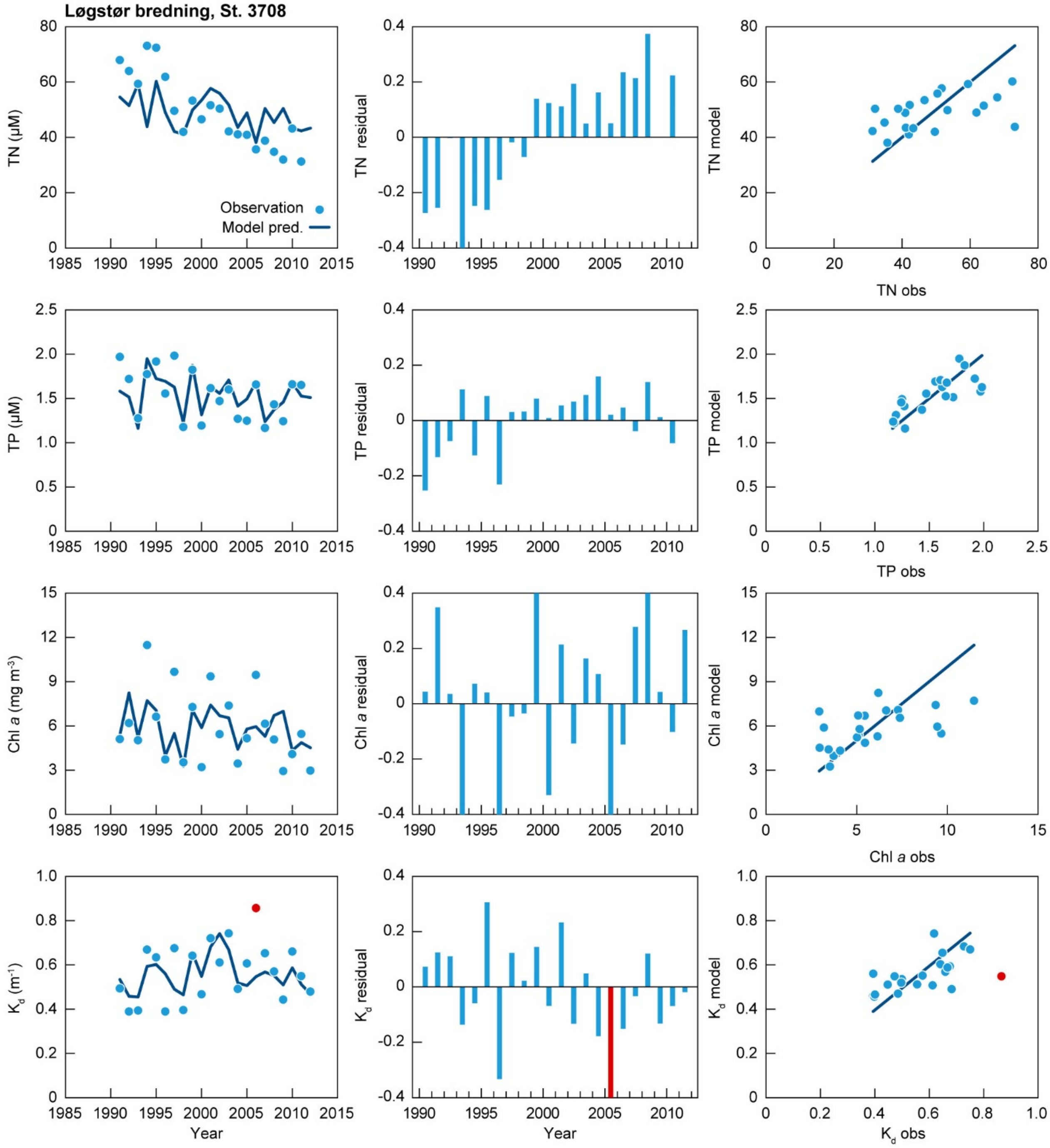

Lovns bredning, St. 3728
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year
Year

Figure B8.



Løgstør bredning, St. 3708
TN (µM)
TN residual
TN model
TN obs
Observation
Model pred.
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year

Figure B9

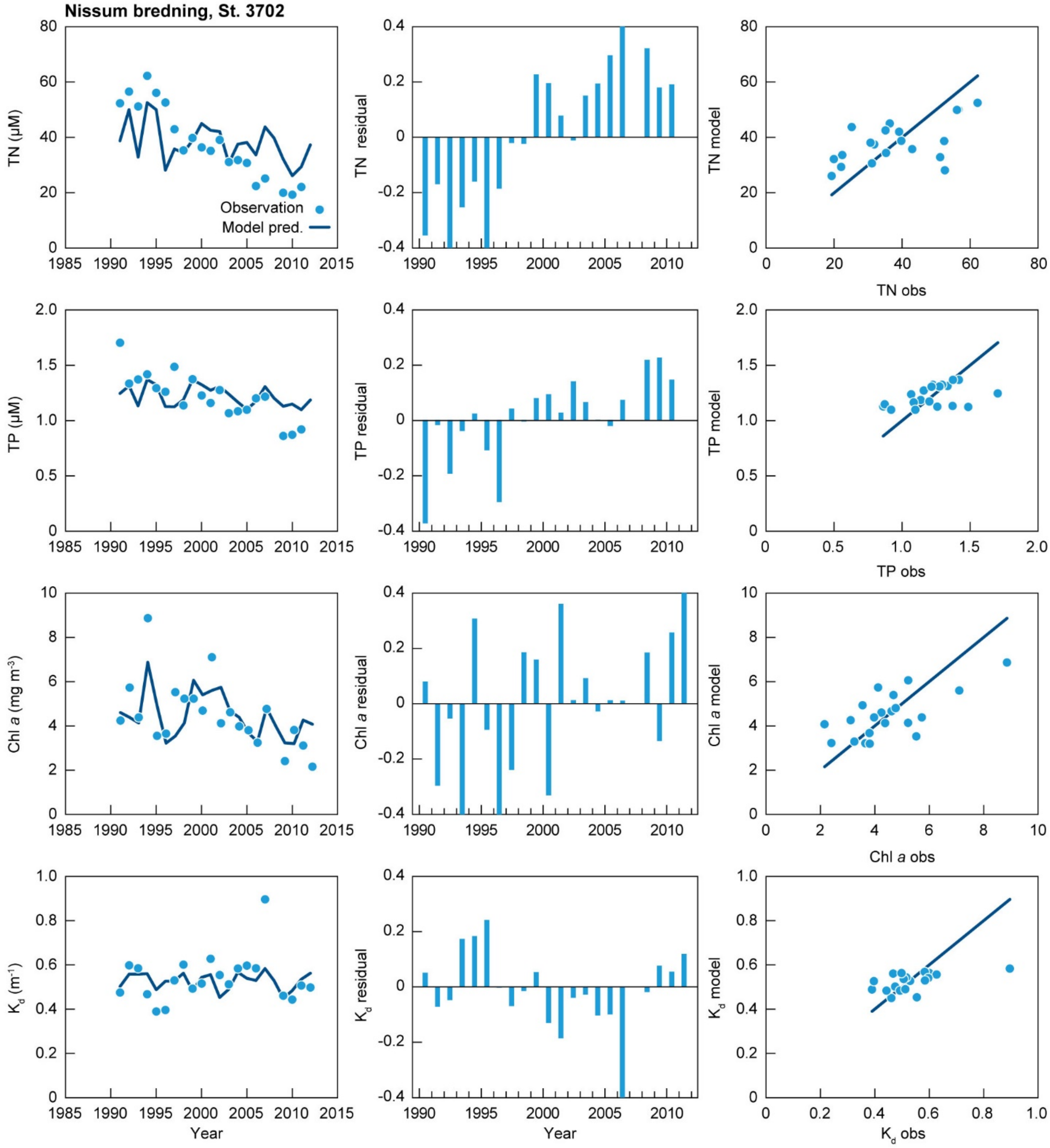
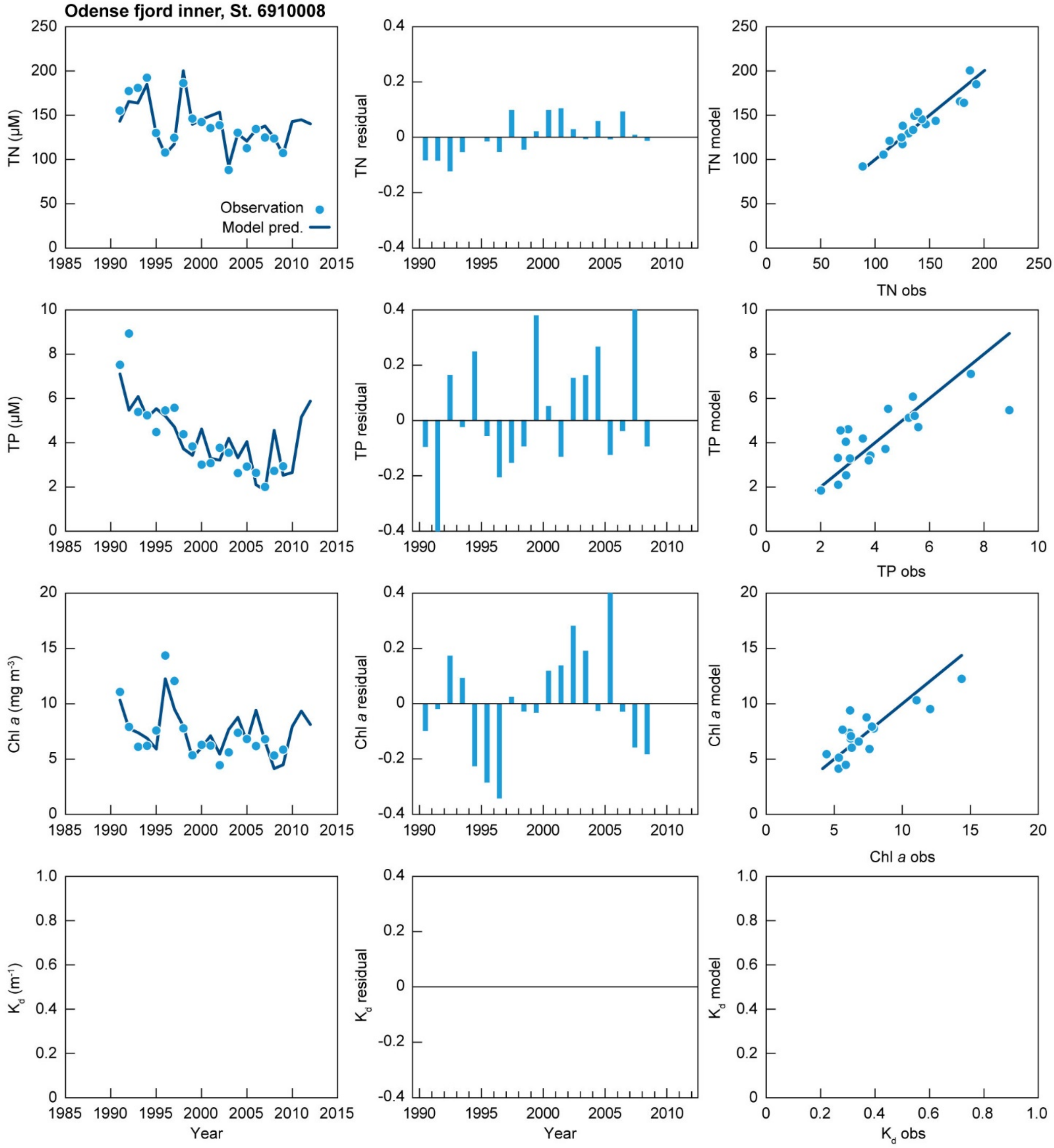
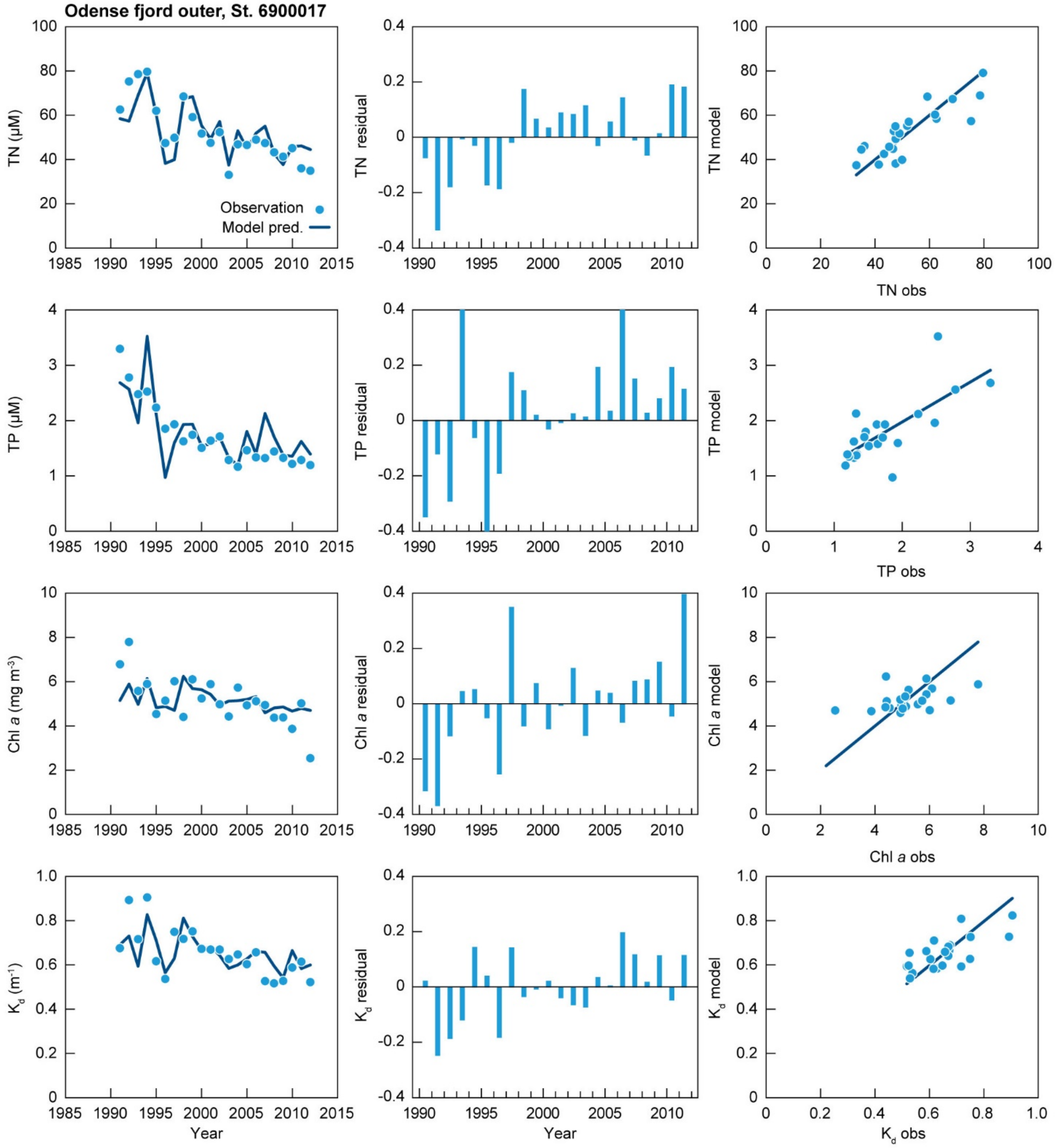

Nissum bredning, St. 3702
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year

Figure B10.



Odense fjord inner, St. 6910008
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year

Figure B11.



Odense fjord outer, St. 6900017
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year

Figure B12.

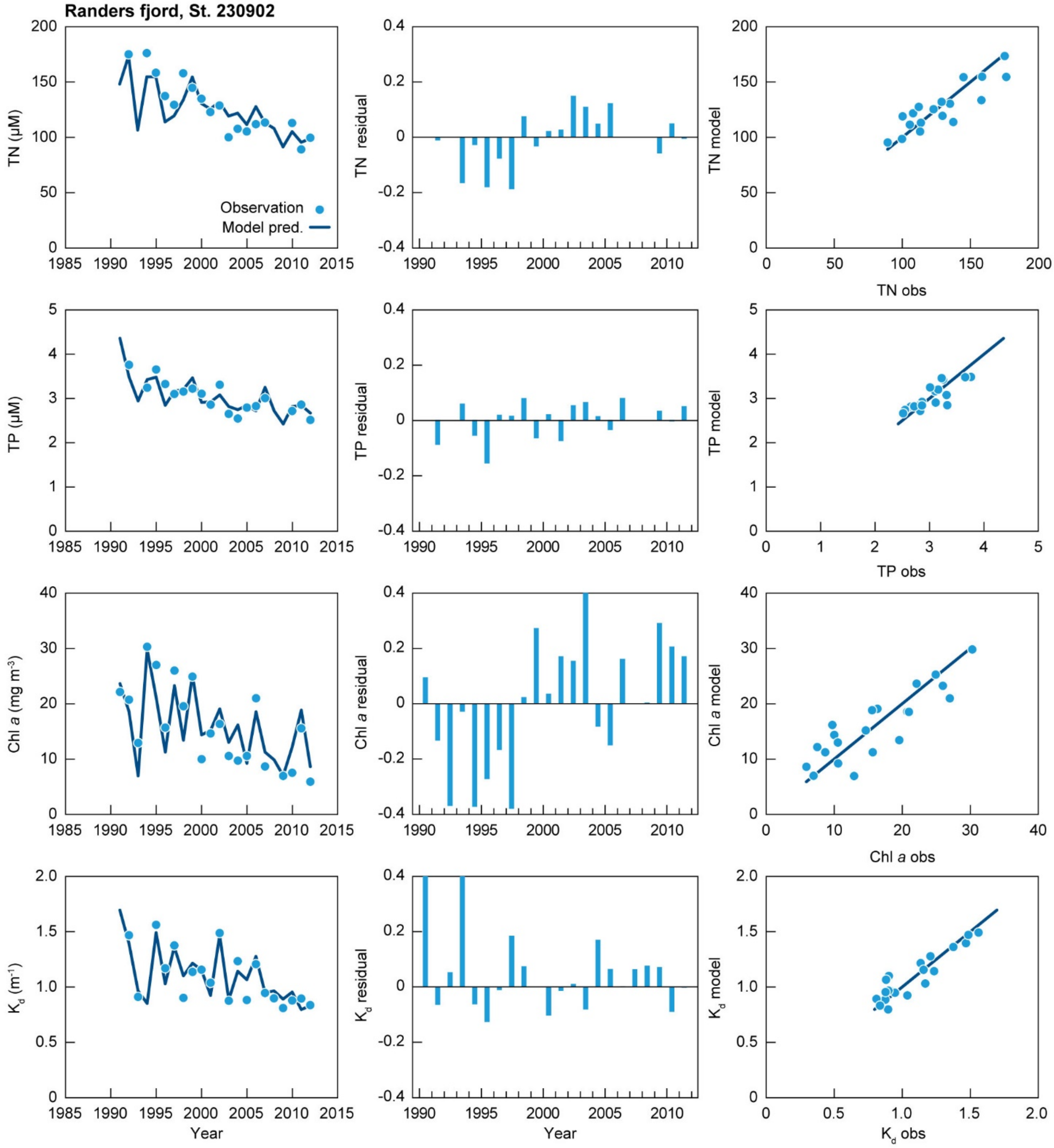
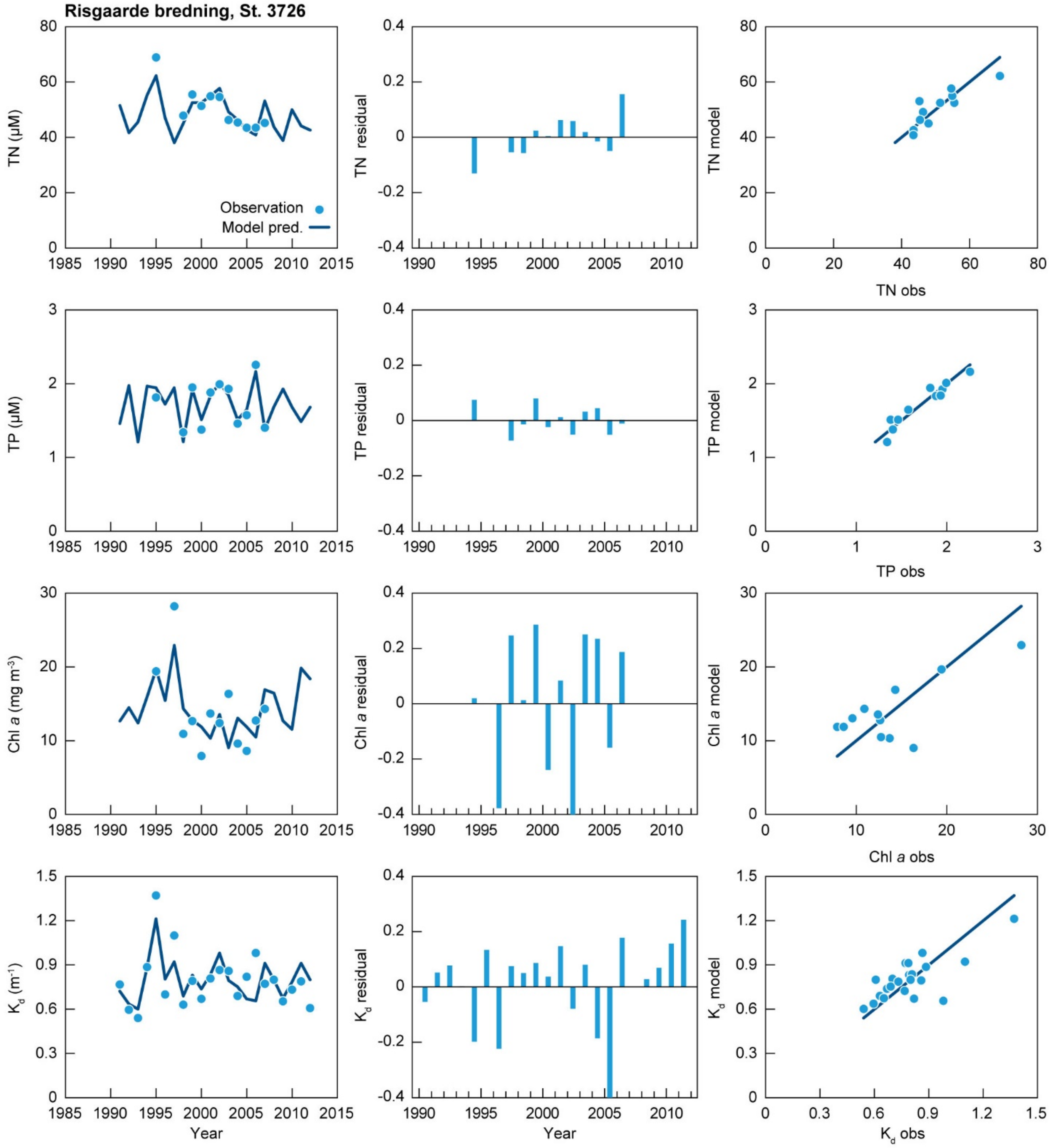

Randers fjord, St. 230902
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year

Figure B13.



Risgaarde bredning, St. 3726
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
Kd (m⁻¹)
Kd residual
Kd model
Kd obs
Year

Figure B14.

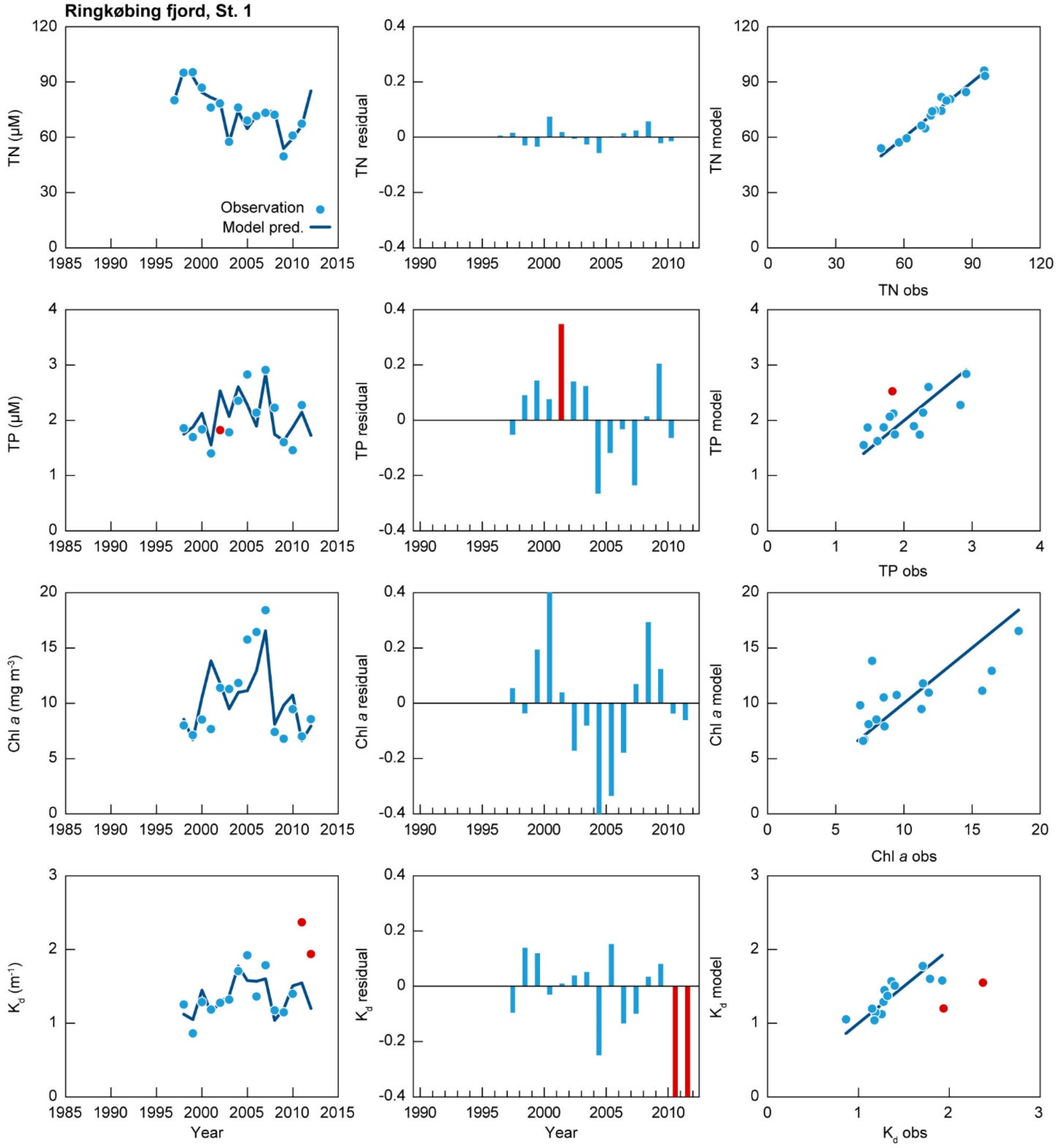
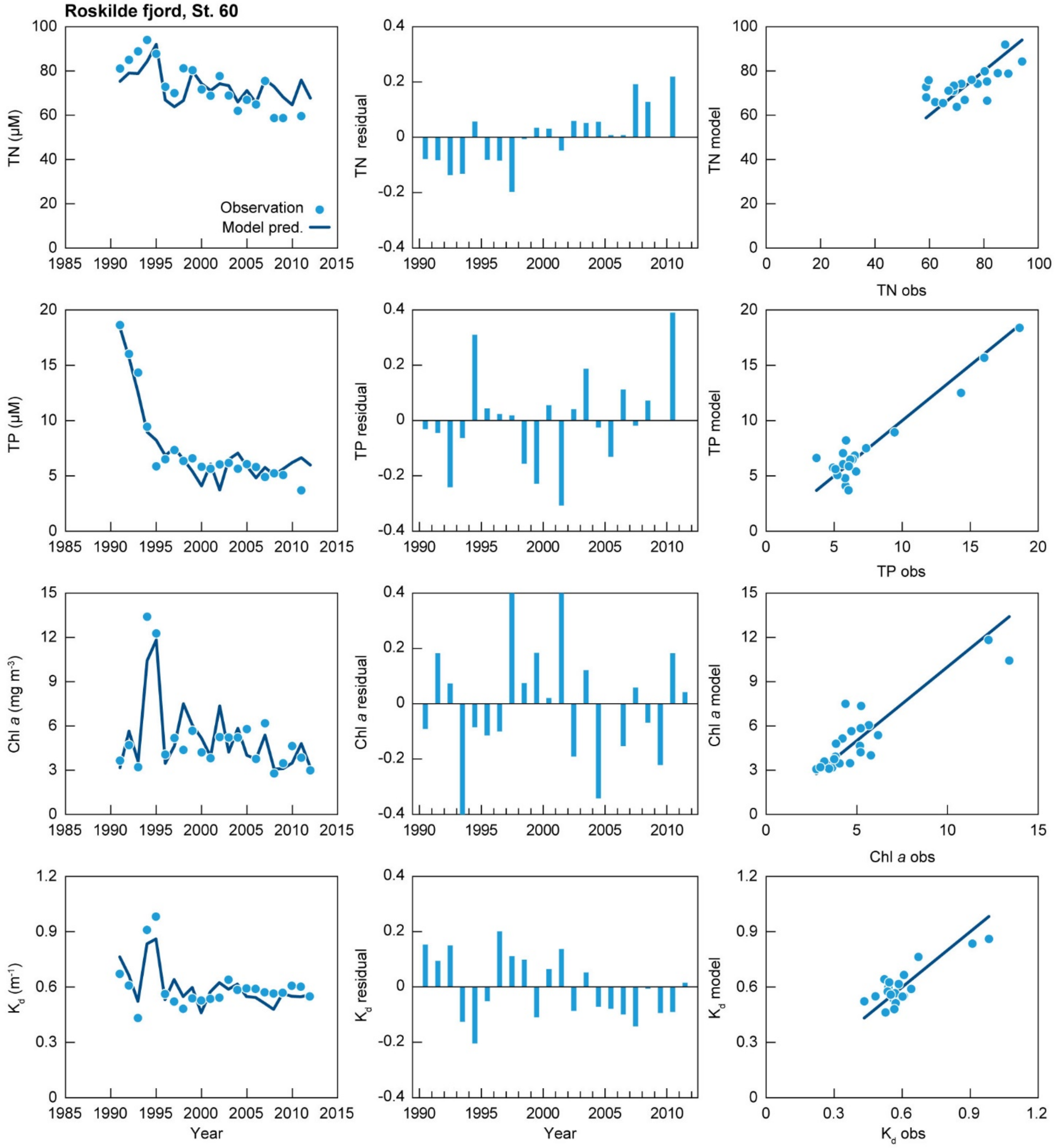

Ringkøbing fjord, St. 1
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year
Year

Figure B15.



Roskilde fjord, St. 60
TN (µM)
TN residual
TN model
TN obs
Observation
Model pred.
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year

Figure B16.

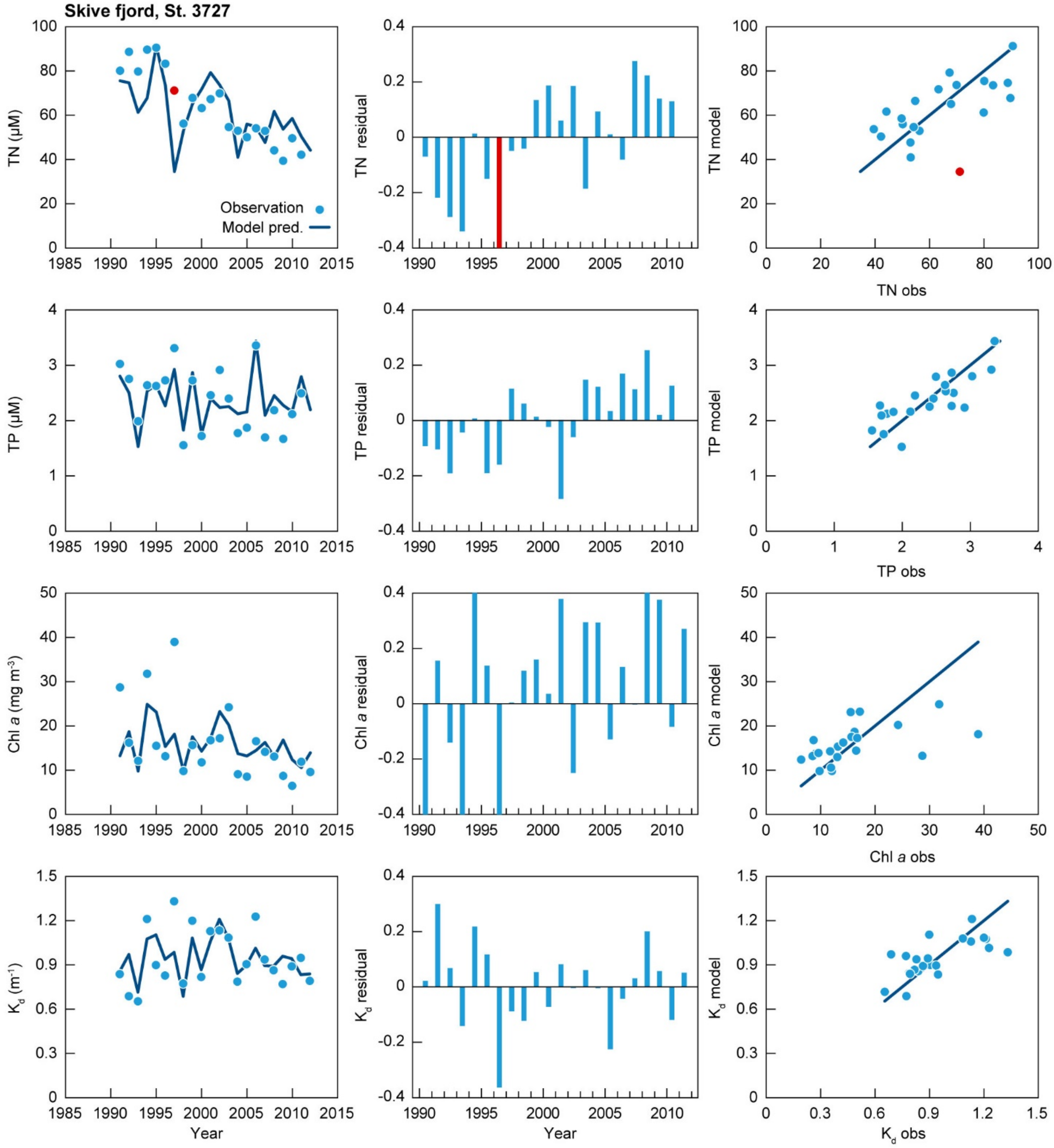

Skive fjord, St. 3727
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
Kd (m⁻¹)
Kd residual
Kd model
Kd obs
Year

Figure B17.

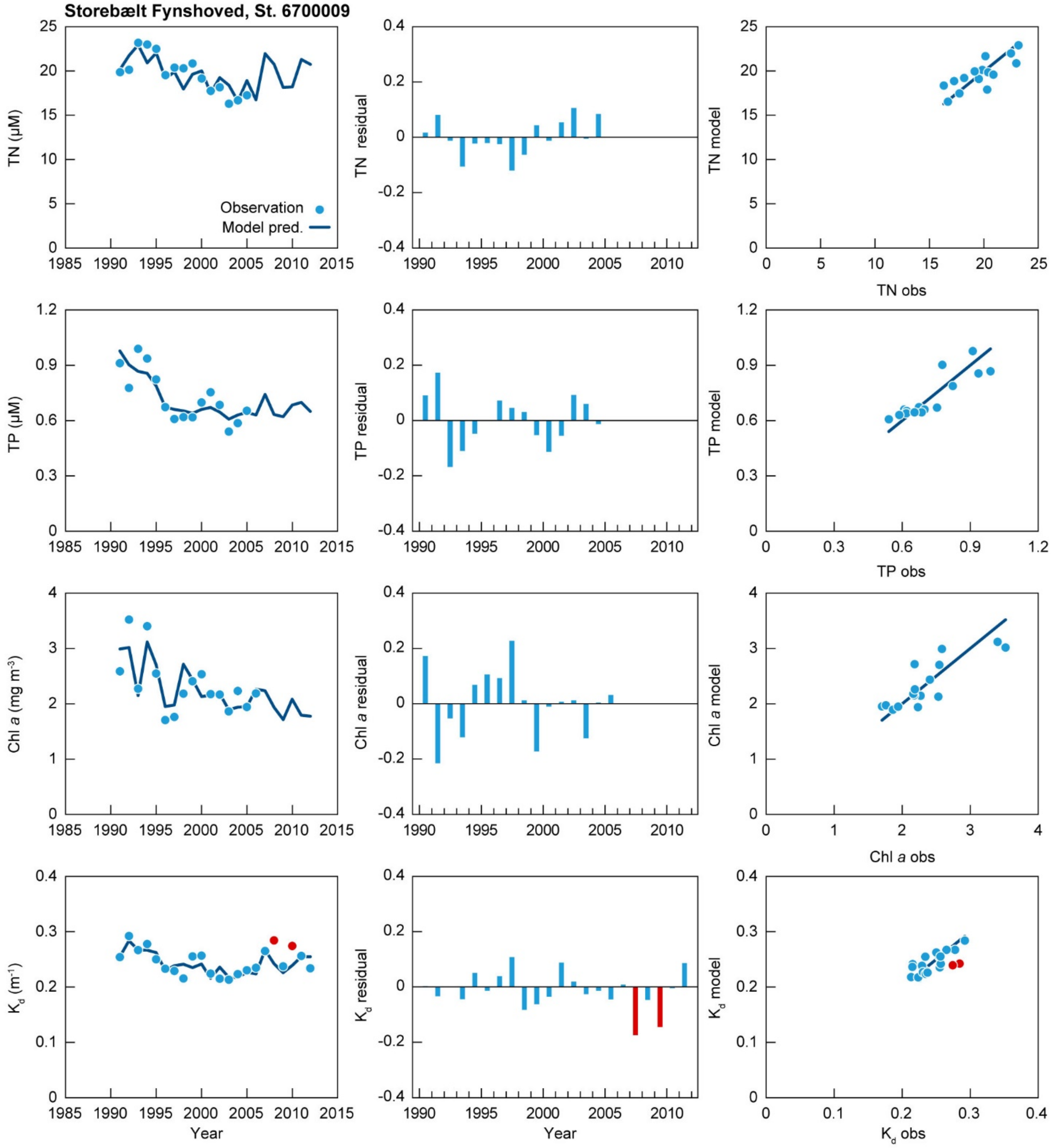

Storebælt Fynshoved, St. 6700009
Observation
Model pred.
TN (µM)
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year
Year

Figure B18.

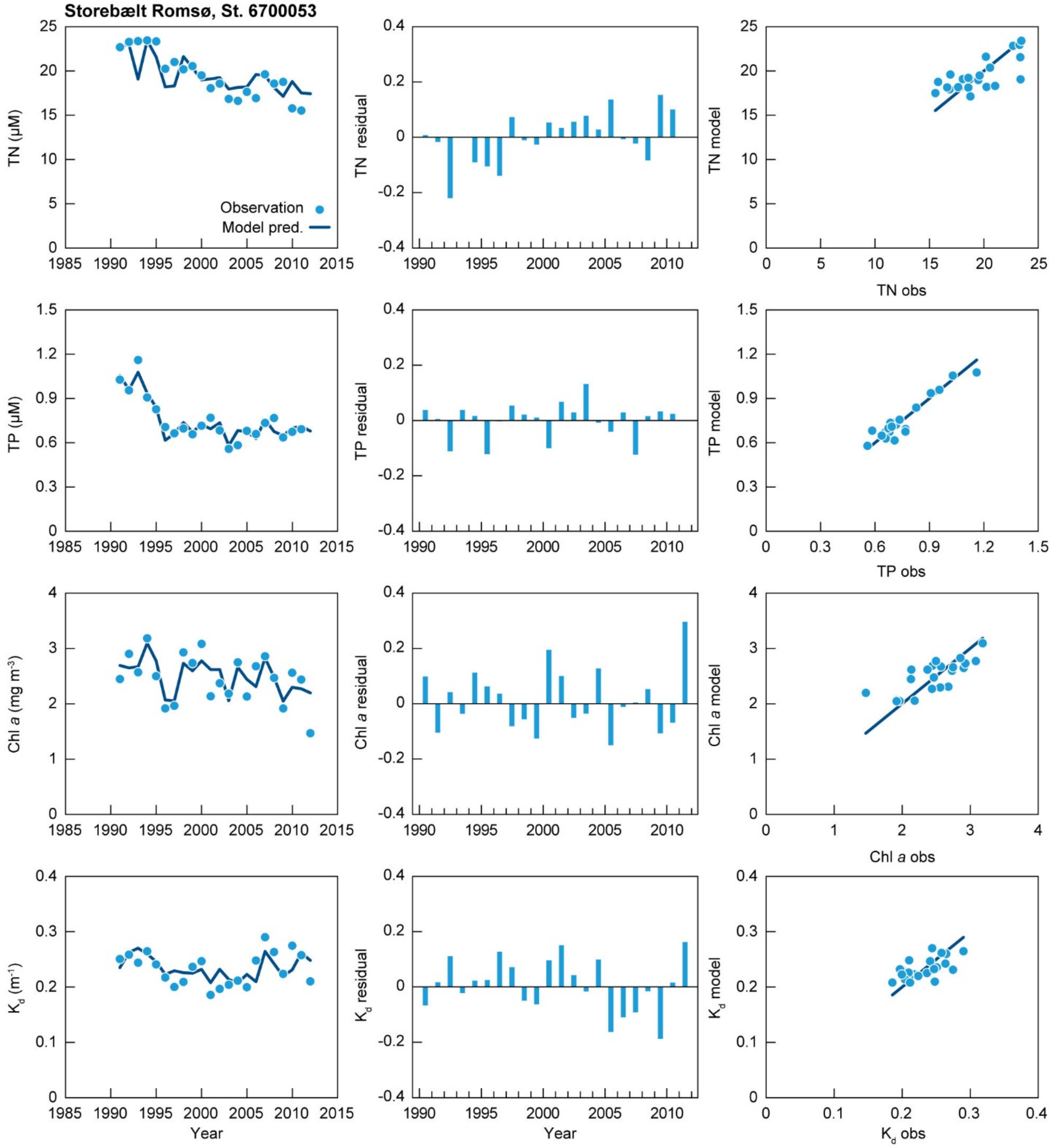
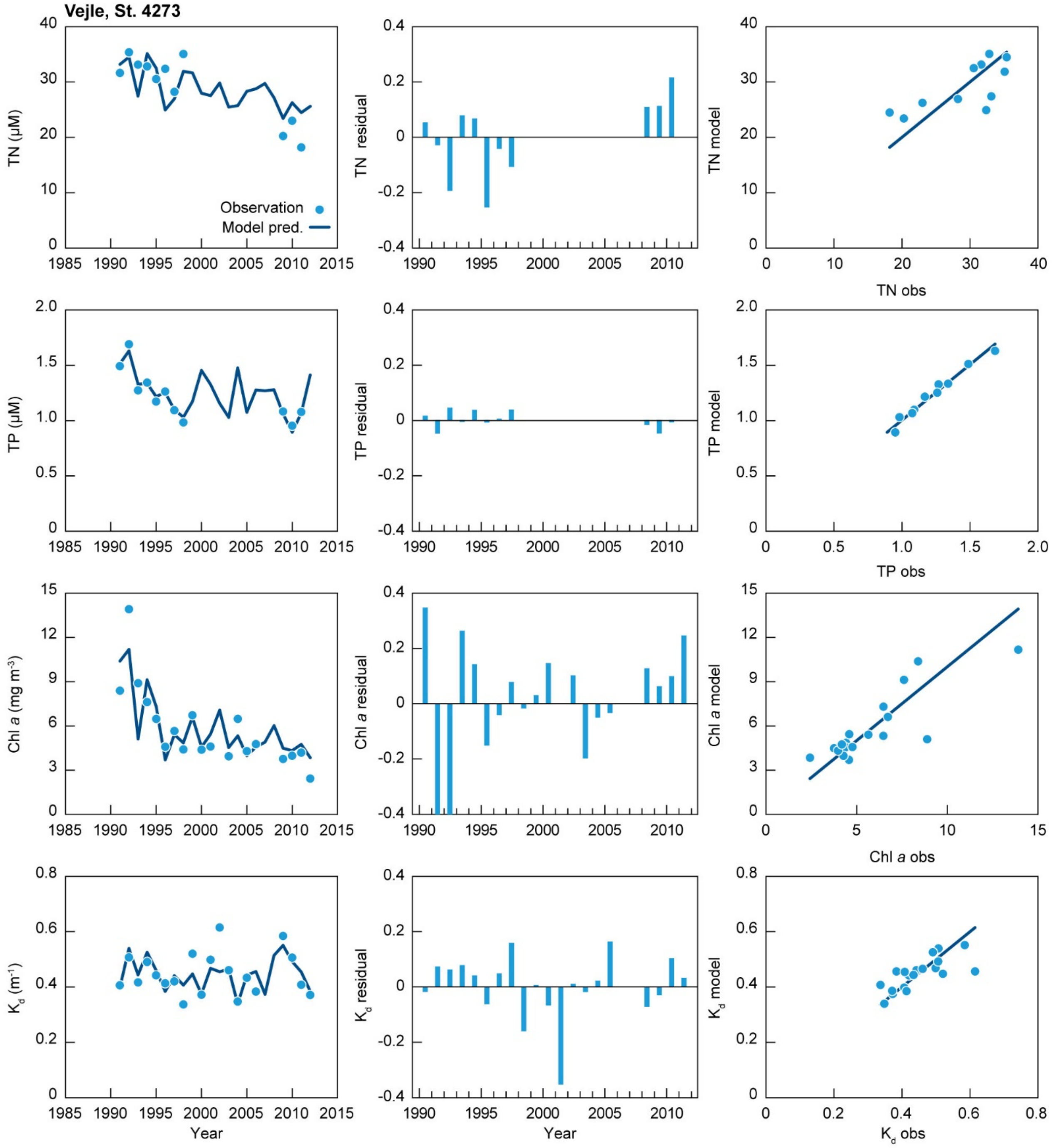
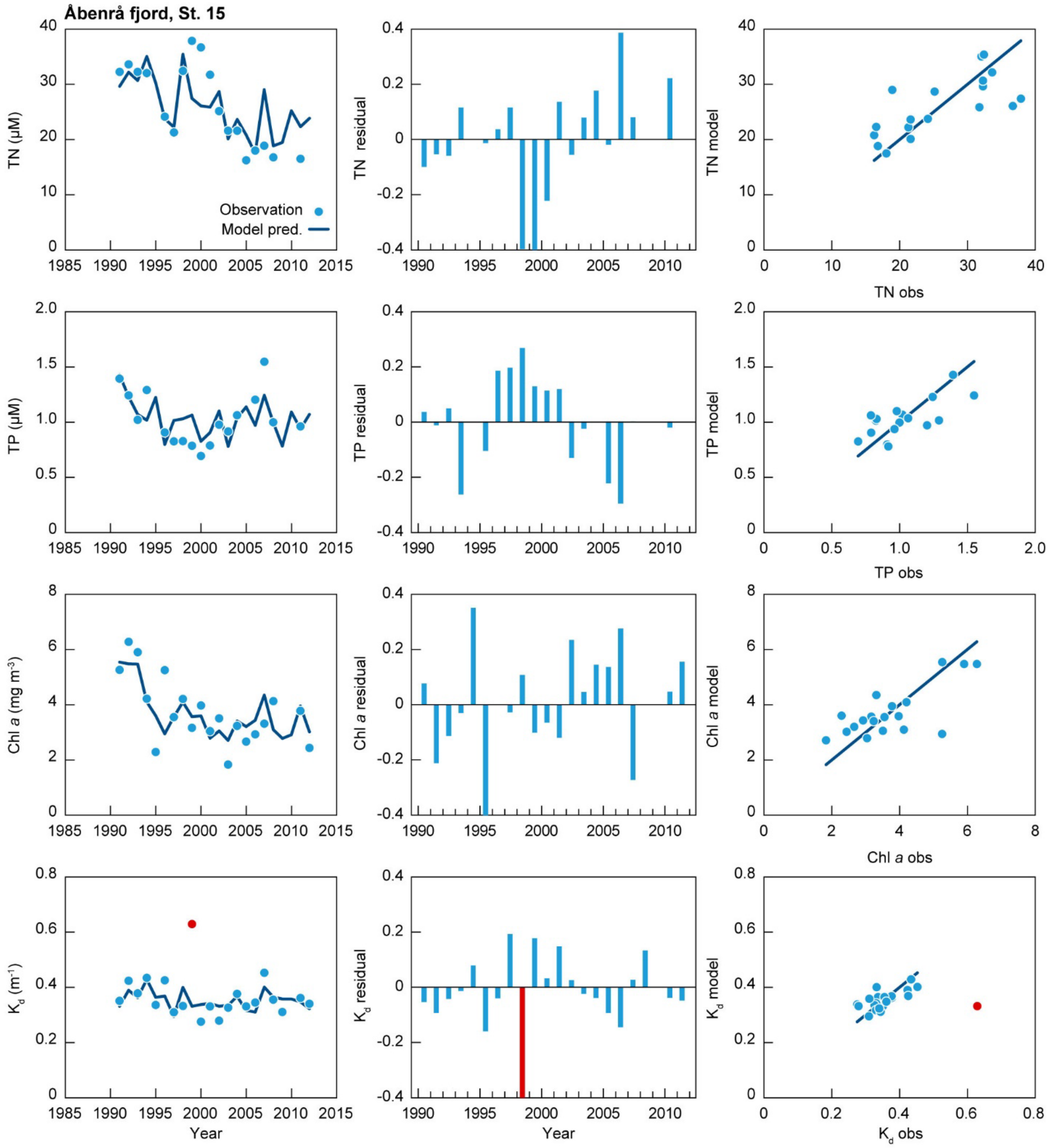

Storebælt Romsø, St. 6700053
Observation
Model pred.
TN (µM)
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
Kd (m⁻¹)
Kd residual
Kd model
Kd obs
Year
Year

Figure B19.



Vejle, St. 4273
TN (μM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (μM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
Kd (m⁻¹)
Kd residual
Kd model
Kd obs
Year
Year

Figure B20.



Åbenrå fjord, St. 15
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
Kd (m⁻¹)
Kd residual
Kd model
Kd obs
Year
Year

Figure B21.

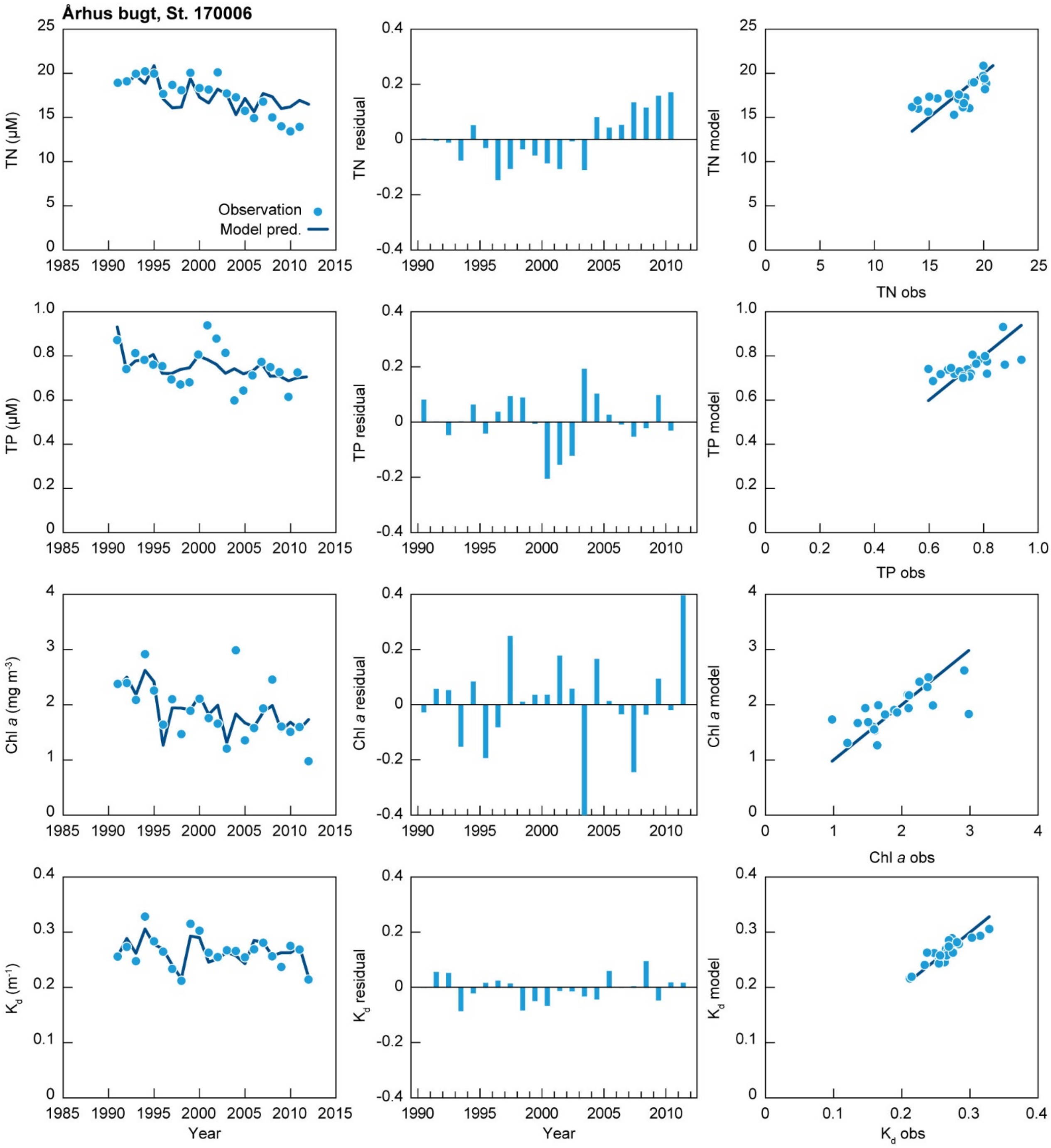

Århus bugt, St. 170006
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year

Figure B22.

Appendix C – Mechanistic model evaluation

Evaluation of modelled nitrogen

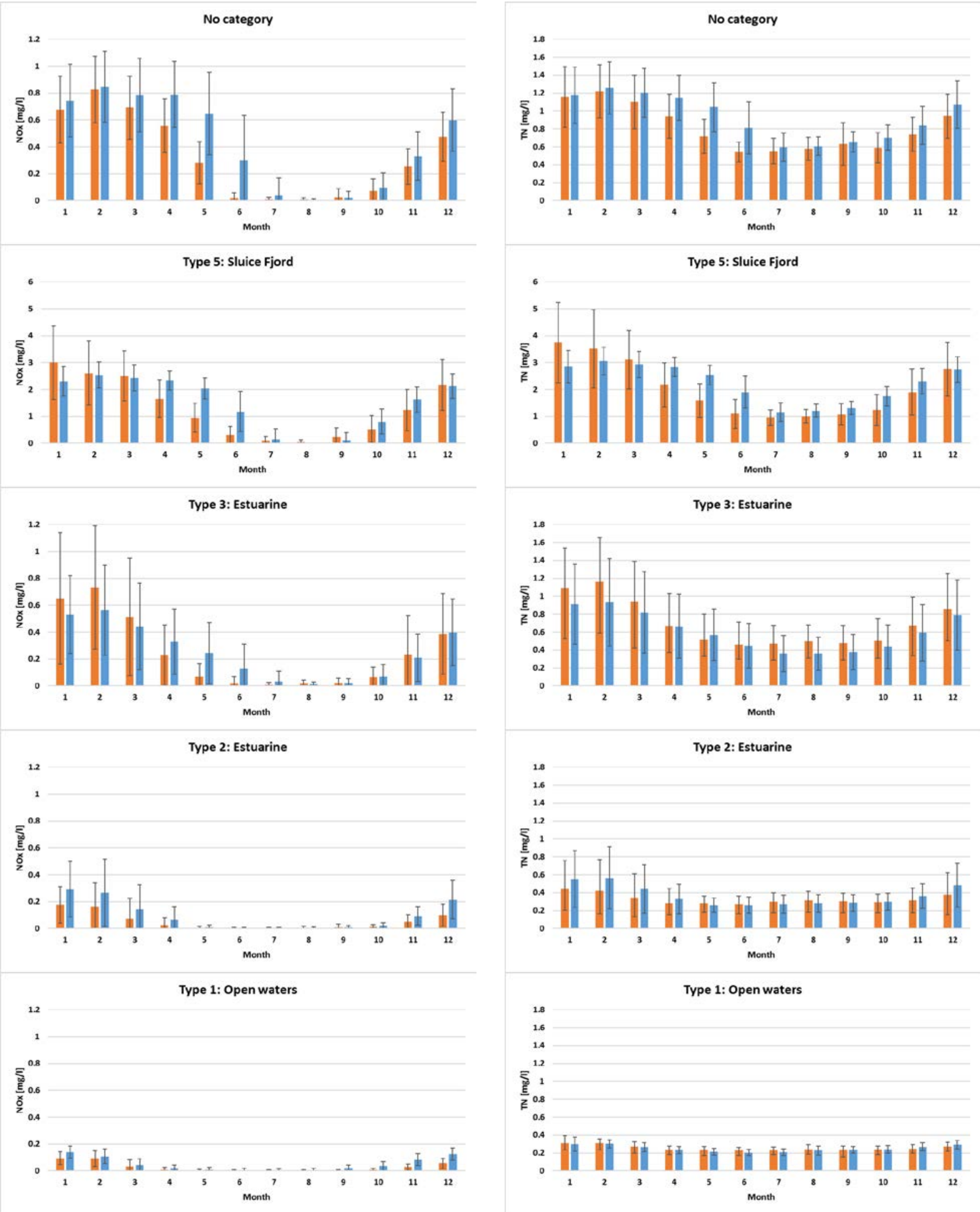


Figure C1. Average monthly measured (orange bars) and modelled (blue bars) concentrations of NO_x (left column) and TN (right column). Error bars represent one standard deviation. From top panel to bottom panel the different water body categories are included: No category, Type 5, Type 3, Type 2 and Type 1. Notice that Type 5 has different y-axis than the other panels.

Evaluation of modelled phosphorous

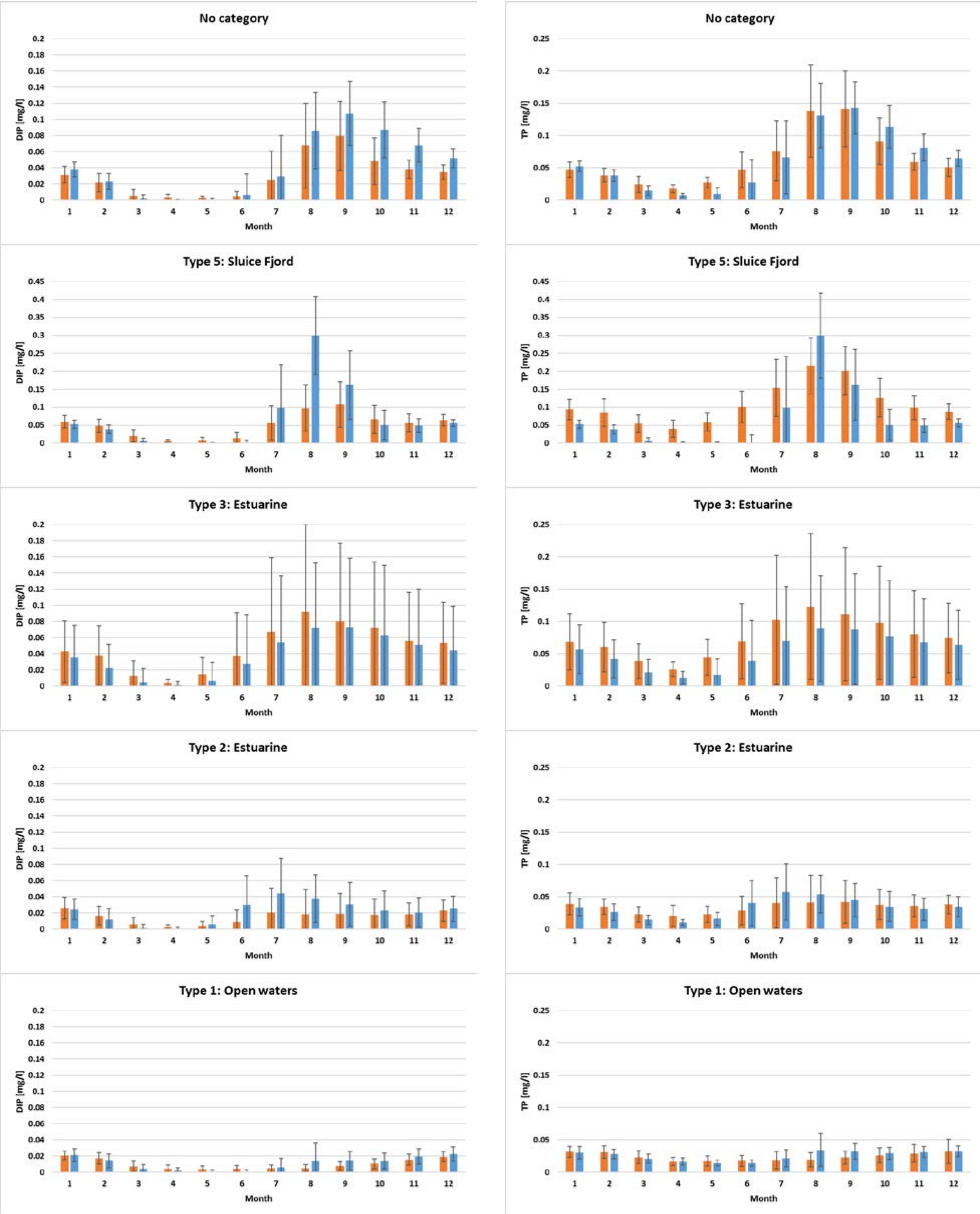


Figure C2. Average monthly measured (orange bars) and modelled (blue bars) concentrations of DIP (left column) and TP (right column). Error bars represent one standard deviation. From top panel to bottom panel the different water body categories are included: No category, Type 5, Type 3, Type 2 and Type 1. Notice that Type 5 has different y-axis than the other panels.

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