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).

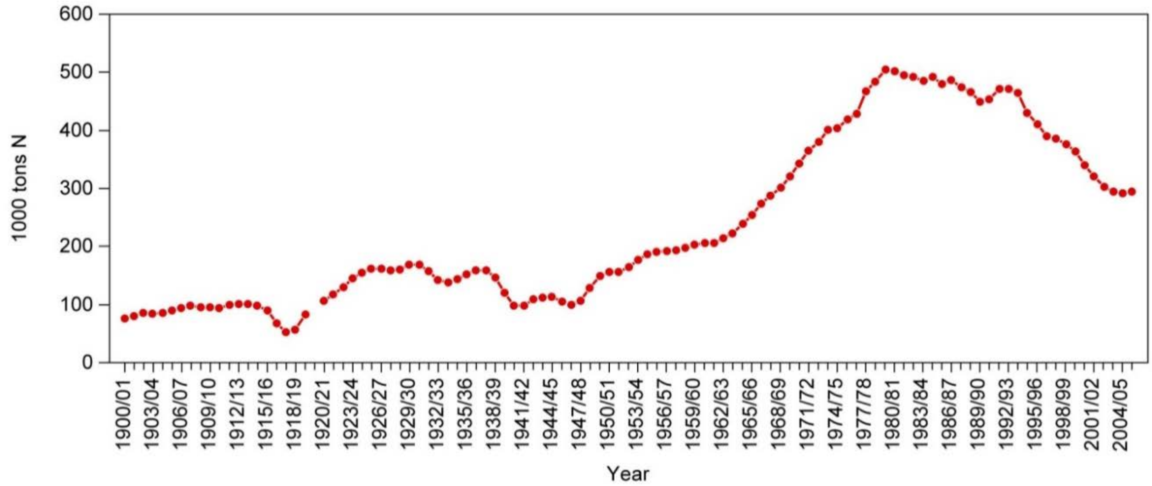


Figure 2.1. Surplus of nitrogen in Danish agriculture (three-year average) (from Kyllingsbæk 2008).

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

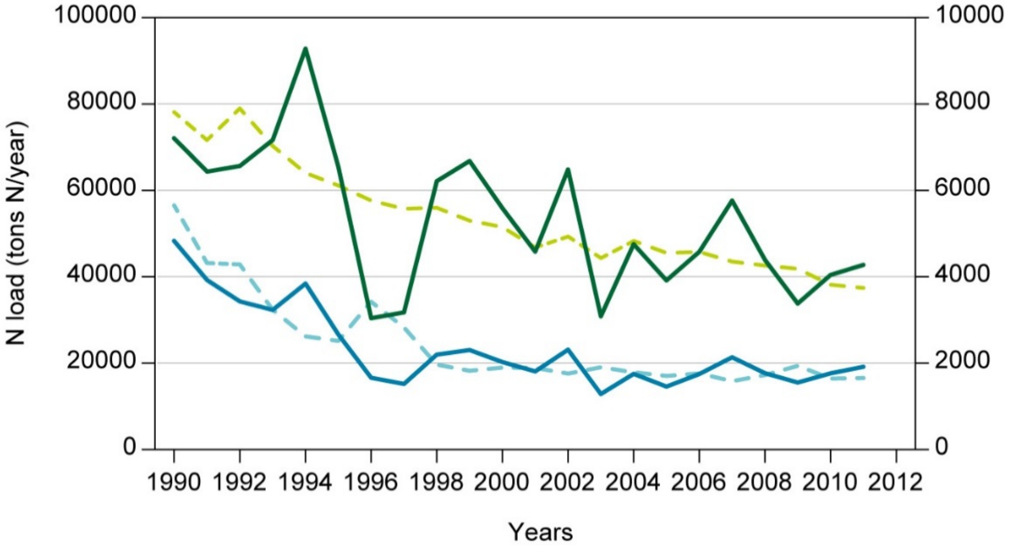


Figure 2.2. Time series of Danish land-based loading of nitrogen (green) and phosphorous (blue) from 1990 to 2011. Full line is actual loadings, dotted lines are flow-normalised loadings (Windolf et al. 2013).

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:

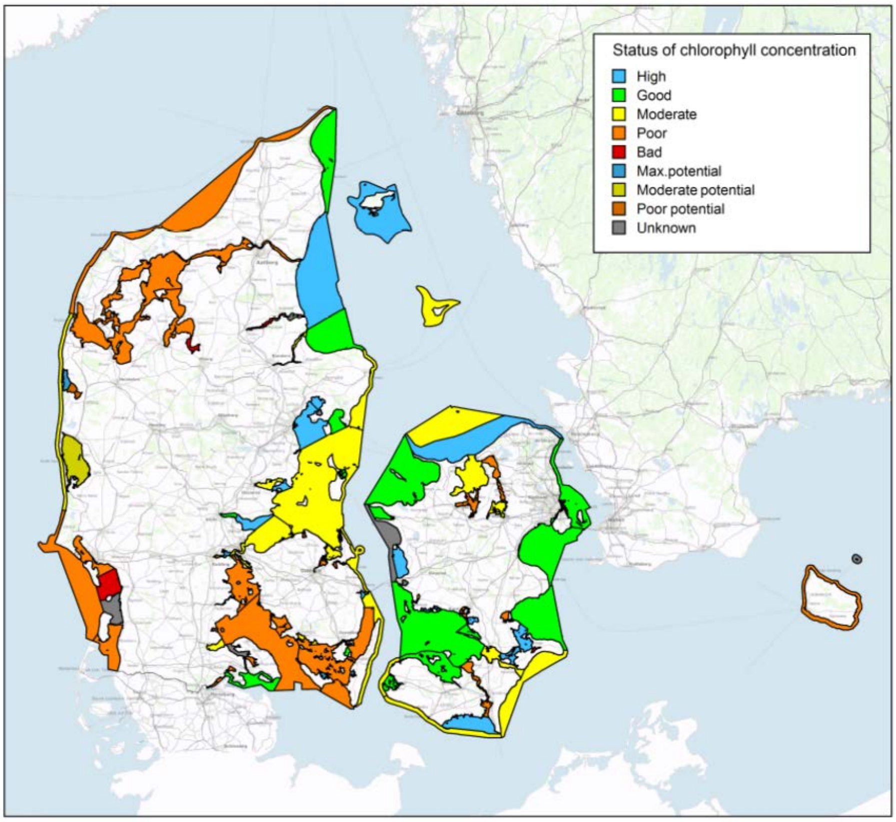
- Chlorophyll-*a* concentration is an indicator for phytoplankton biomass and is assessed as the average (May-September) chlorophyll-*a* concentration within the inner Danish waters and 90-percentile of the March to September chlorophyll-*a* concentrations for water bodies located in the North Sea and 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.

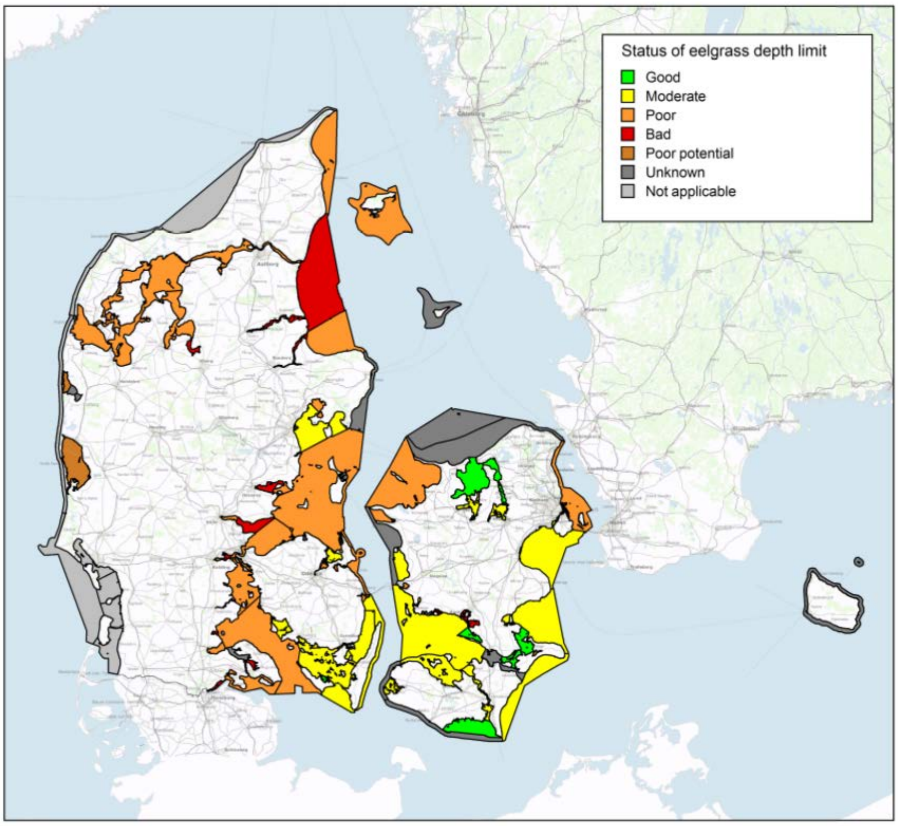
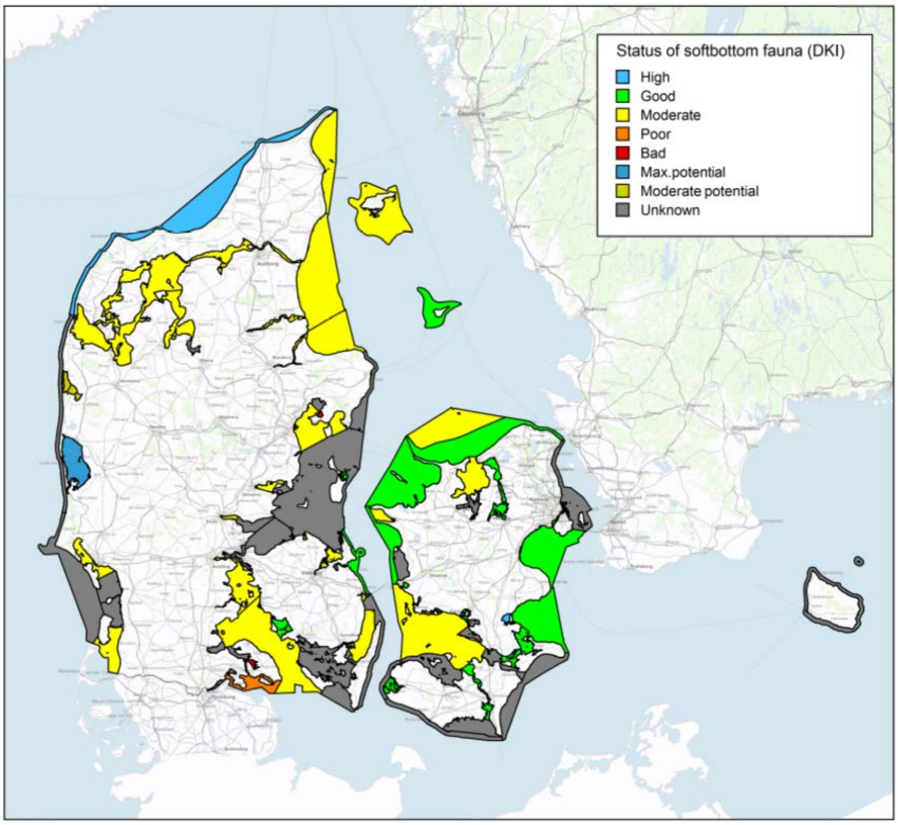


Figure 2.5. Assessment of DKI status of Danish marine water bodies.



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*.

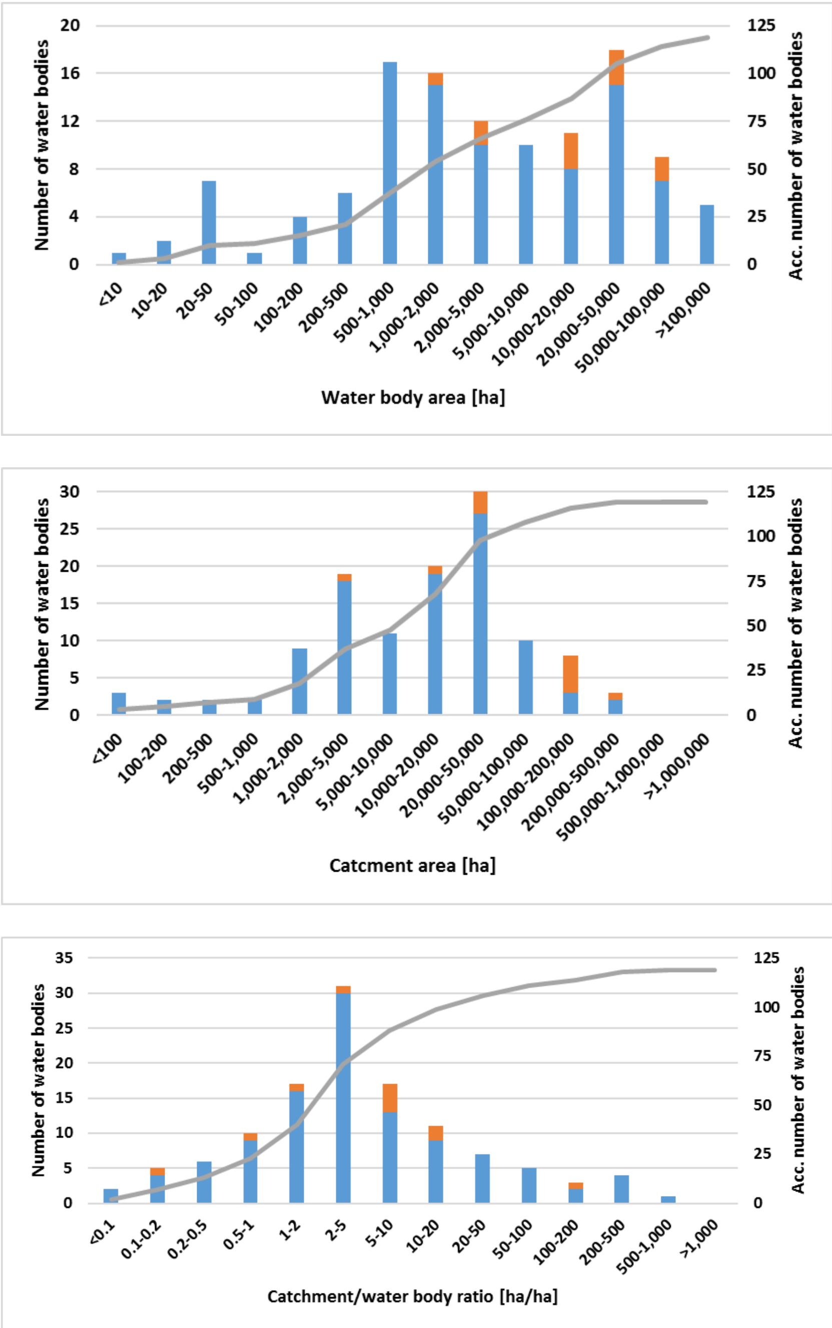


Figure 3.1. Frequency plot of water body area (top panel) and catchment area (middle panel) and the relation between catchment area and water body area (bottom panel). Blue bars represent water bodies of the inner Danish waters and orange bars water bodies in the Skagerrak, North Sea and Wadden Sea areas. 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.

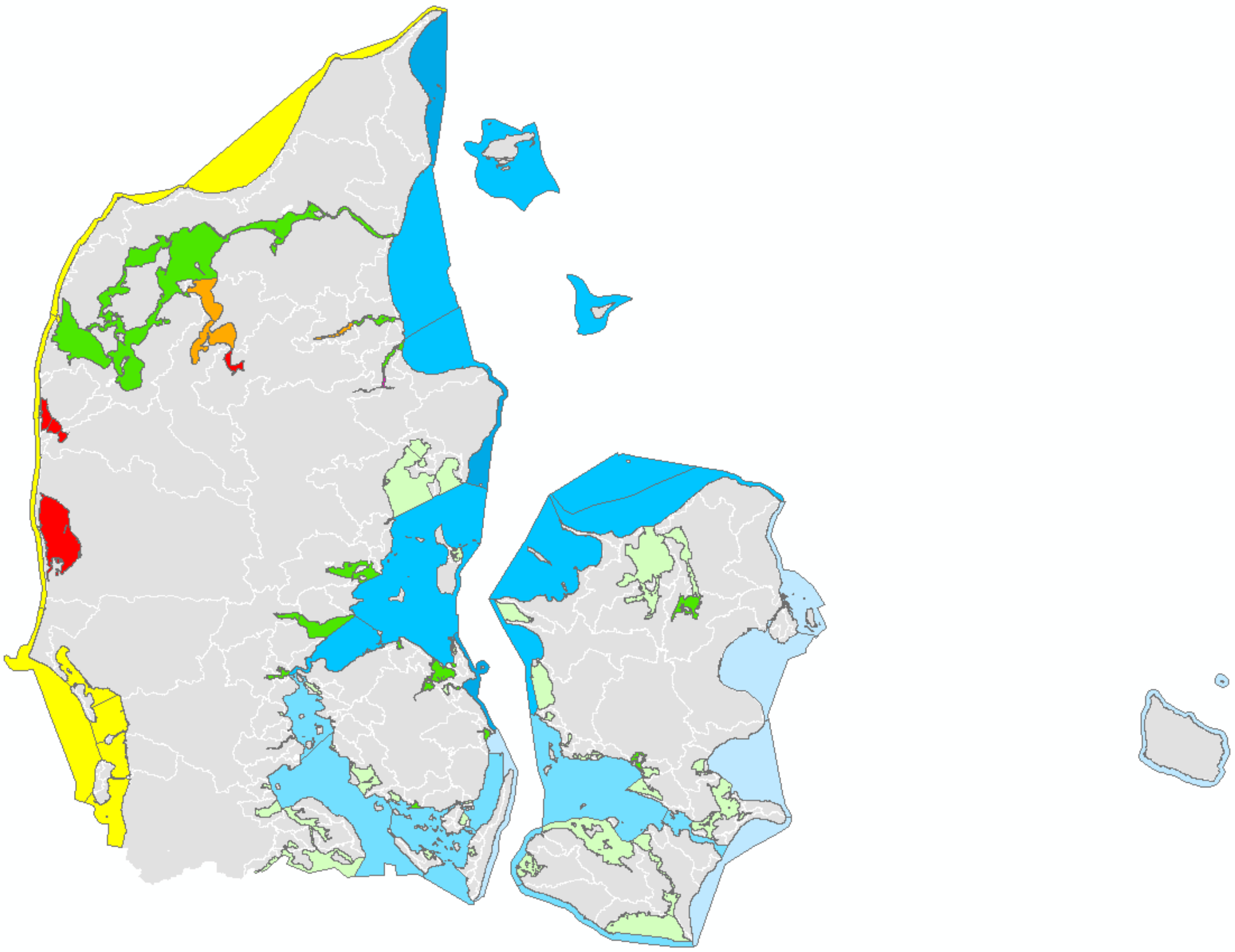


Figure 3.2. Applied typology for Danish WFD water bodies (modified from Dahl et al. 2005). Type 1 (blue – the four different blue colours indicate sub-types), Type 2 (light green), Type 3 (dark green), Type 4 (pink[4]), Type 5 (red), original North Sea types[5] (yellow) and areas not included in the 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.

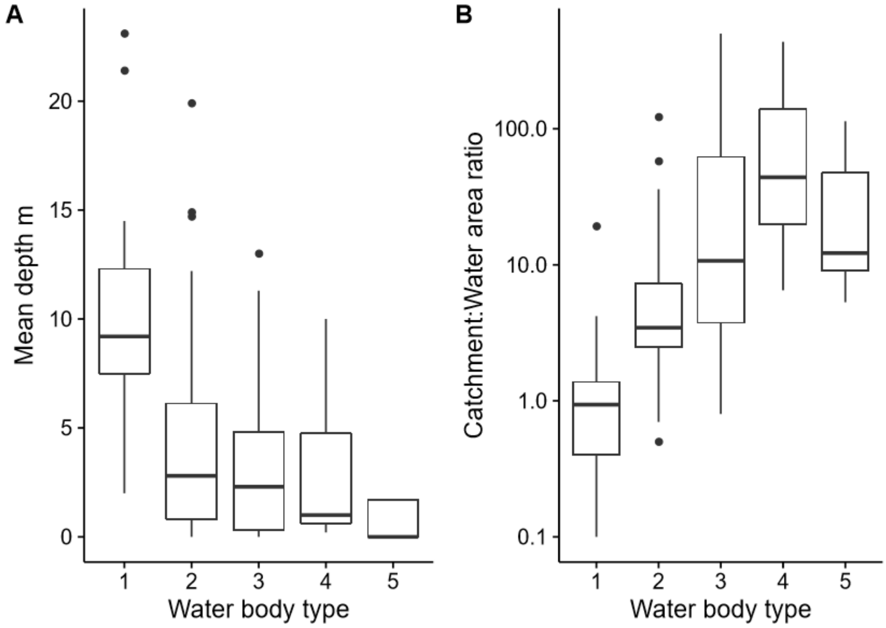
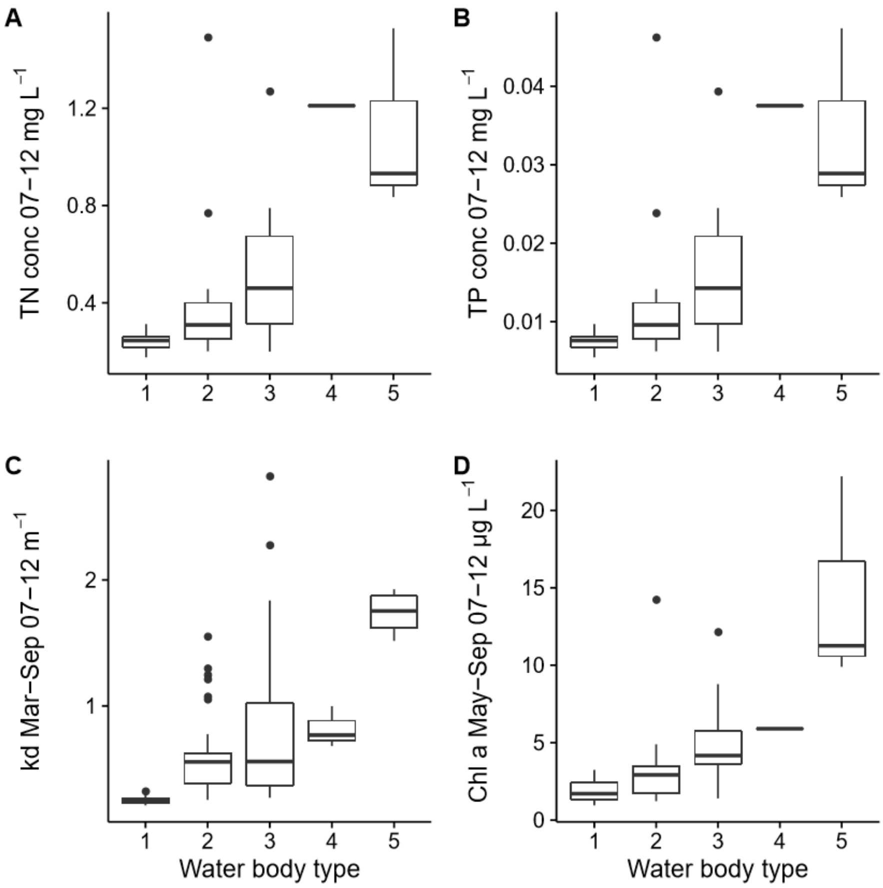


Figure 3.4. Statistics of the physical/chemical and biological state of the different water body types based on observations during 2007-2012 from available monitoring stations. A) Total nitrogen (TN) concentration in mg L^{-1}, B) total phosphorus (TP) concentration in mg L^{-1}, C) light attenuation coefficient K_d (m^{-1}) from March to September, D) chlorophyll-*a* concentration (µg L^{-1}) from May to September.



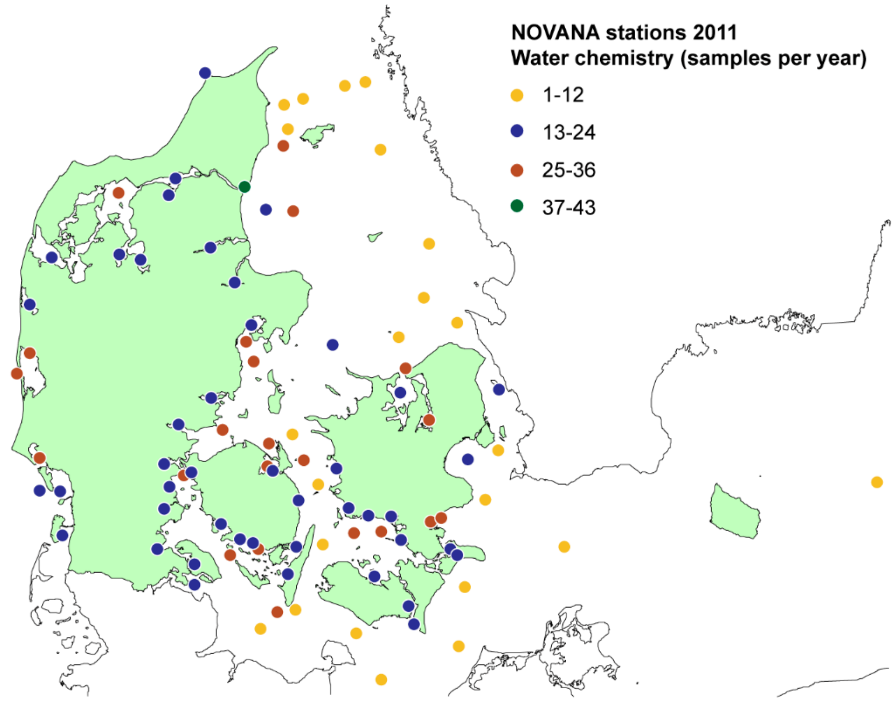
4 Danish monitoring data DNAMAP (NOVANA)

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%.

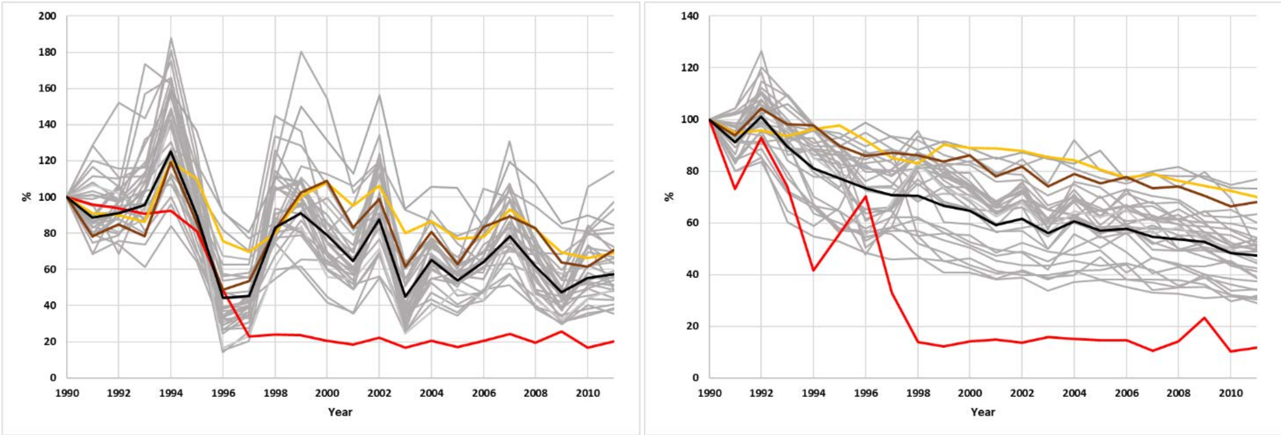


Figure 4.2. Changes in N loadings from the catchments accounting for 80% of total loadings to the inner Danish waters. Calculated as percentages of 1990 loadings based on actual loadings (left) and flow-normalised loadings (right). Black line = average of all catchments. Grey lines = individual catchments. 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.

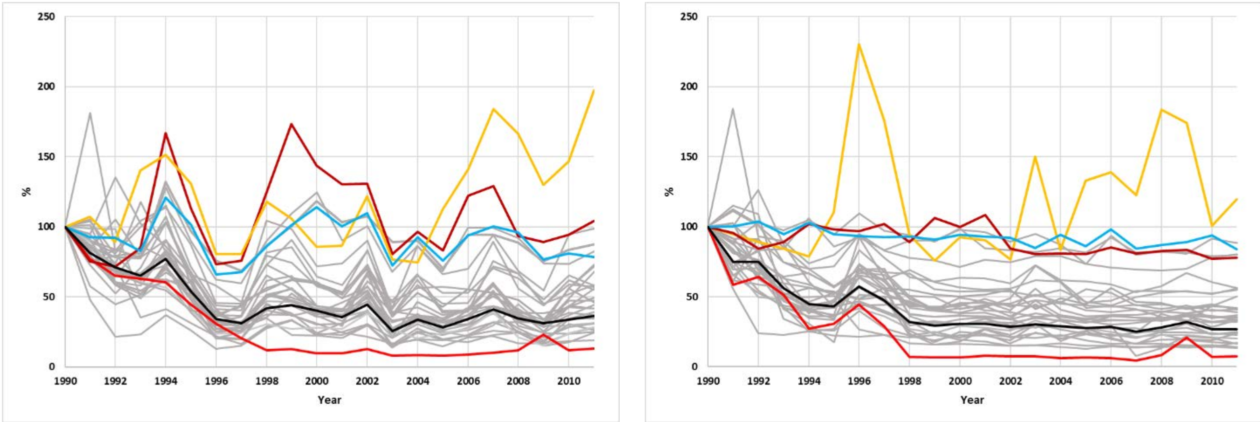


Figure 4.3. Changes in P loadings from the catchments accounting for 80% of the total loadings to the inner Danish waters. Calculated as percentages of the 1990 loadings based on actual loadings (left) and flow-normalised loadings (right). Black line = average of all catchments. Grey lines = individual 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.

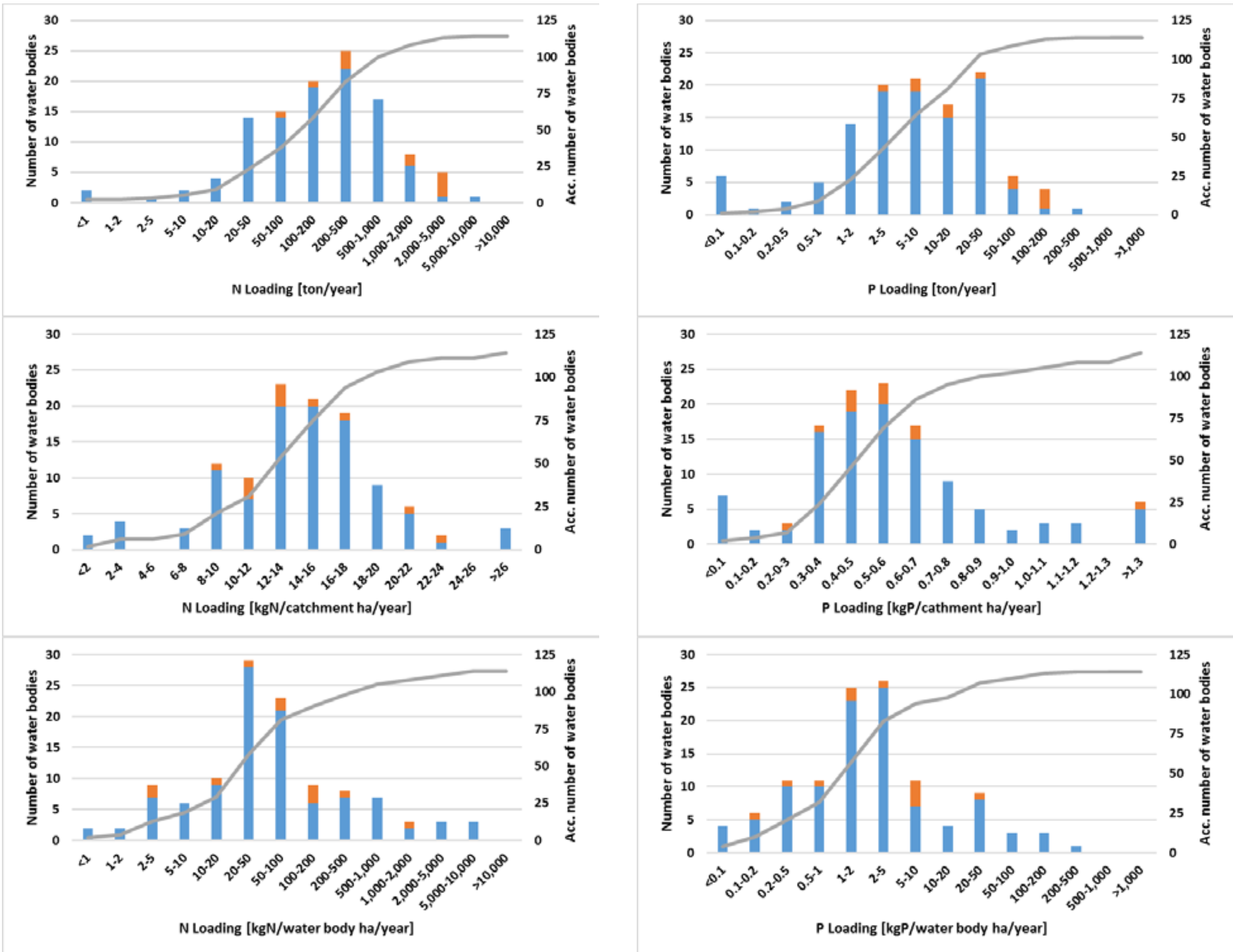


Figure 4.4. Average nutrient loadings for the period 2007-2011 divided into water bodies. Left panels show N loadings and right panels P loadings. Top panels are the yearly loadings per water body, middle panels include the areal loadings per catchment, and the bottom panels represent the areal loadings 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.

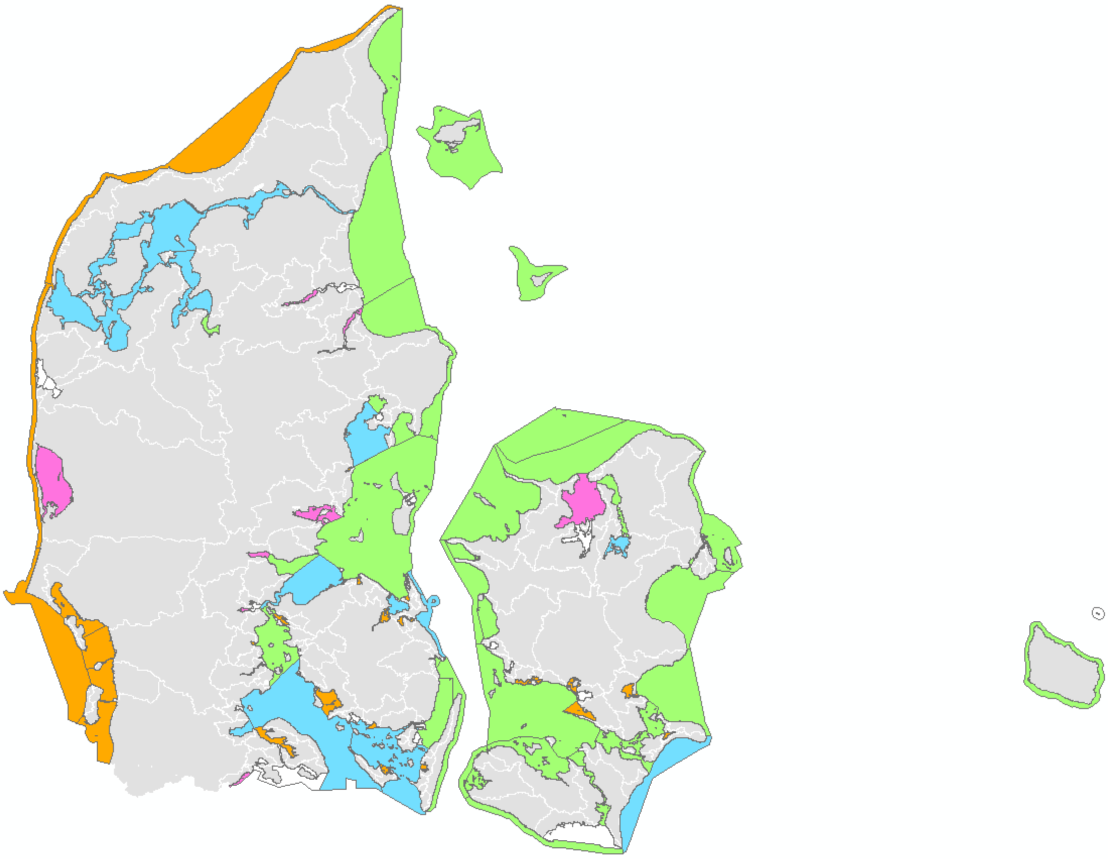


Figure 5.1. Water bodies covered by the different models developed for calculating nutrient reduction requirement and corresponding MAI to obtain GES. Pink colour indicates water bodies for which only statistical models have been developed, green colour indicates water bodies for which only mechanistic 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)

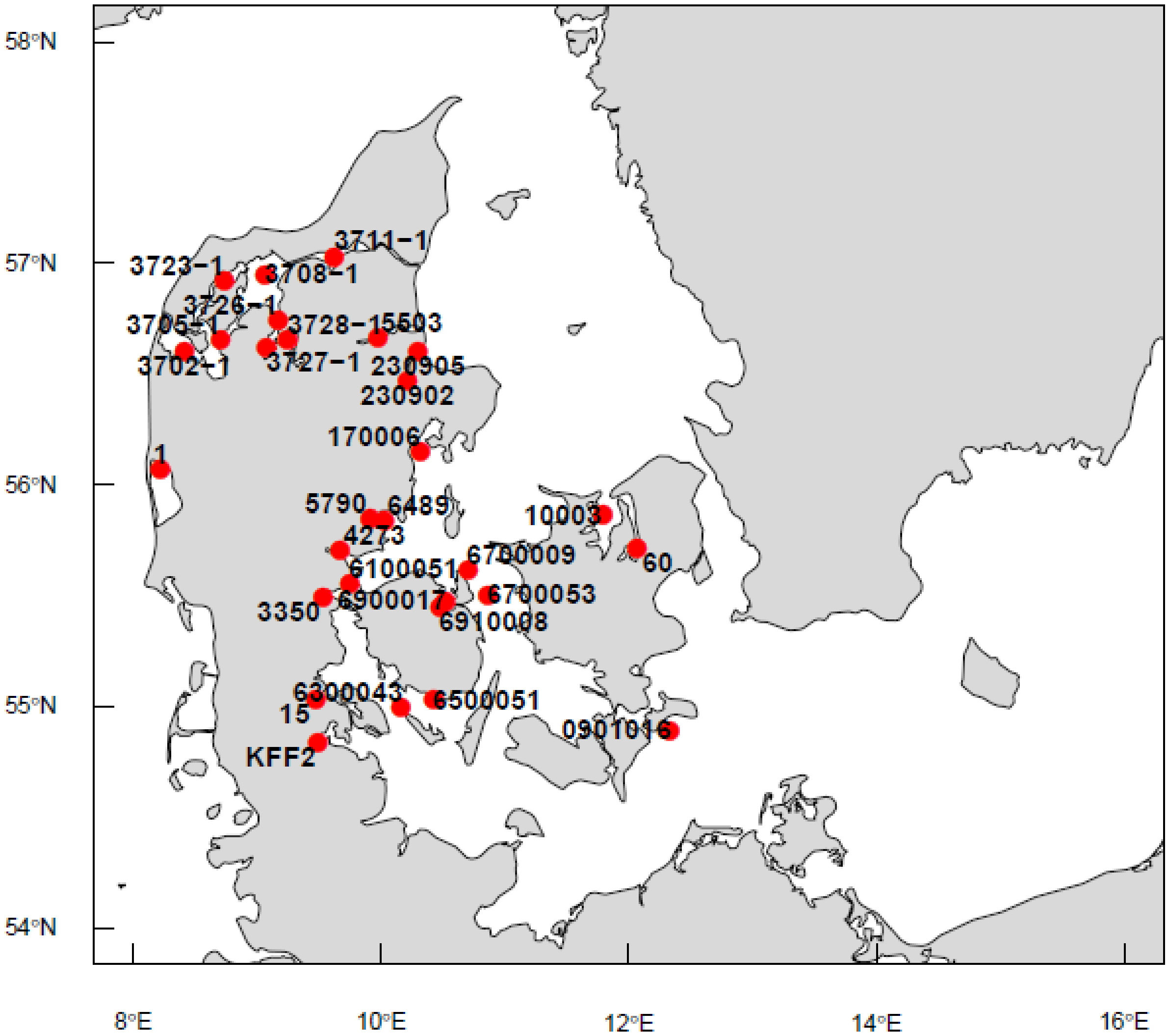


Figure 6.1. Marine monitoring stations used in the development of statistical models. The numbers refer to official station identification numbers.

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.

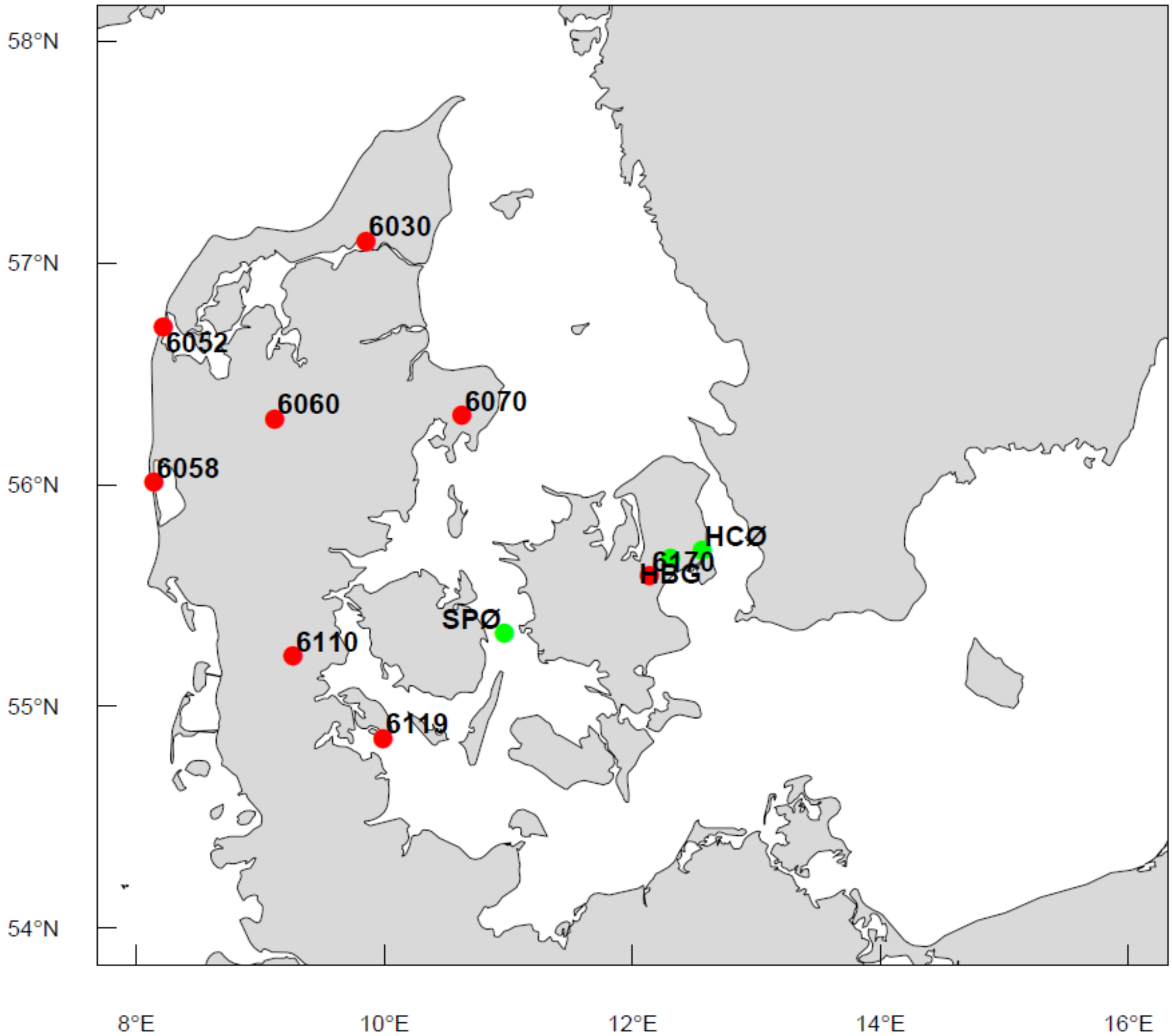


Figure 6.2. Location of meteorological stations providing wind data (red dots) or irradiance data (green dots).

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 sensor 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:

$$Xvar\_detr_i = Xvar_i - (\alpha + \beta \cdot year_i) \quad \text{Eq. 6.2}$$

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:

$$-\ln\left(\frac{L_z}{L_0}\right) = Kd \cdot Zs \quad \text{Eq. 6.3}$$

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*)

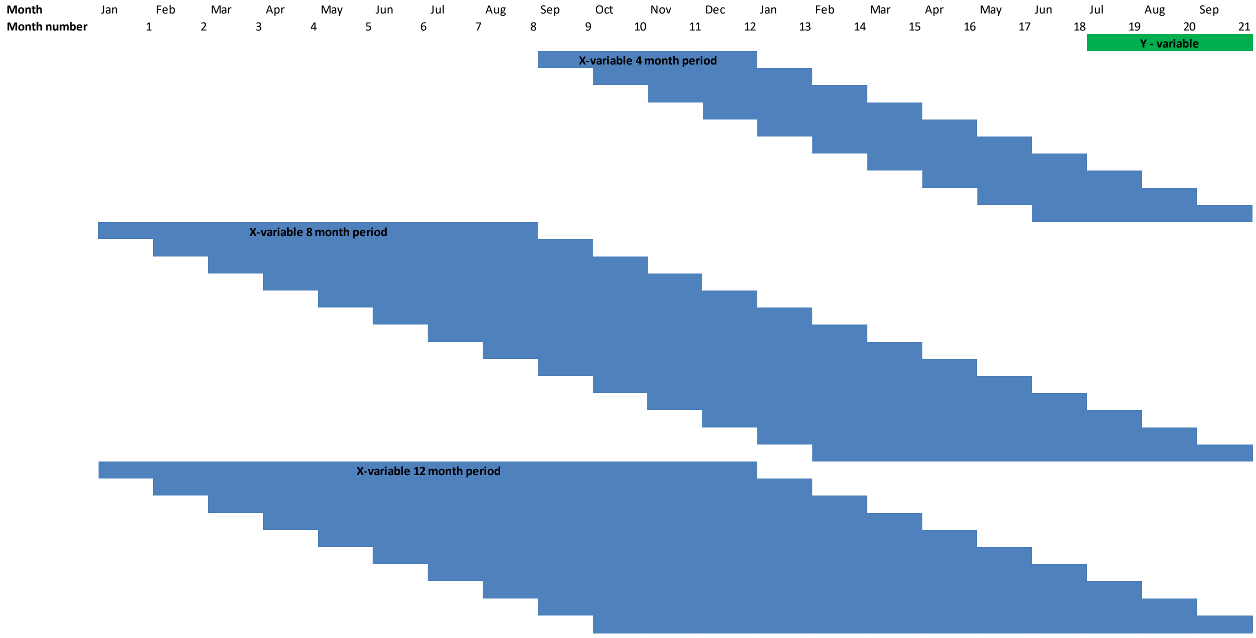


Figure 6.3. Example of predictor variable time periods based on a 3-month response variable period.

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*).

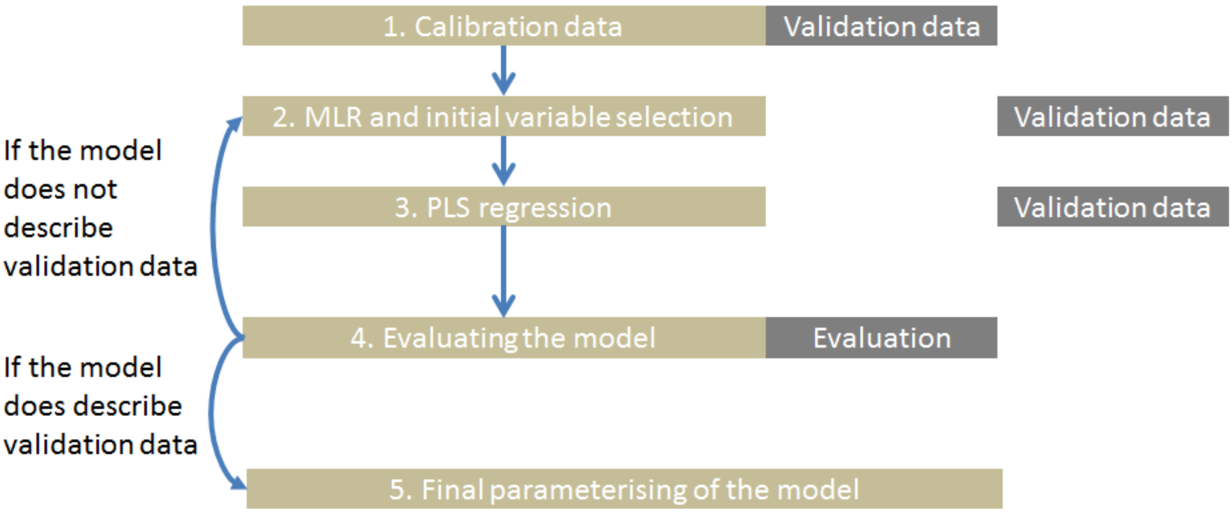


Figure 6.4. Schematic diagram of model development used in this work. For the development of each model we went through a calibration and a validation stage to do so one must: 1) divide the data set in a calibration set and a validation set (25% of the data set); 2) select a suite of possible explanatory 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.

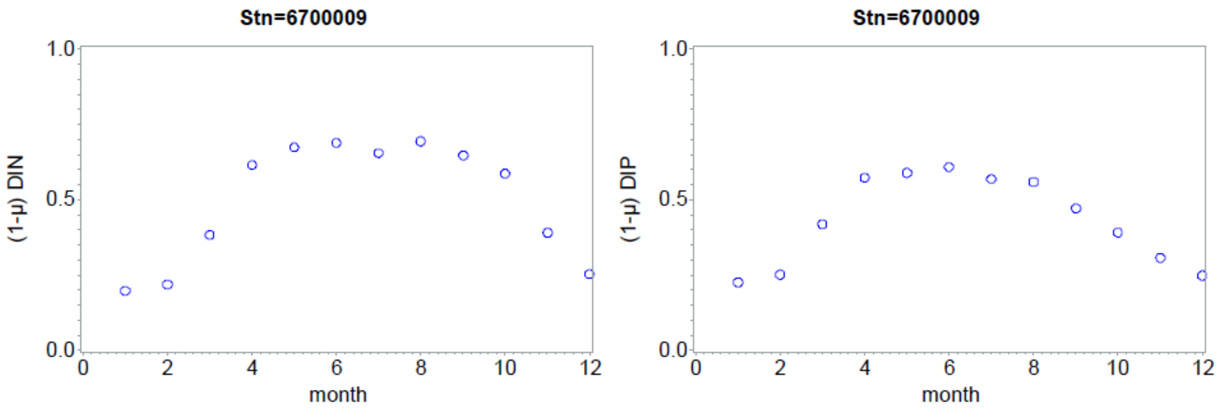


Figure 6.5. Potential nutrient limitation for each month at station 6700009 in the northern part of the Great Belt (Fynshoved). Left = potential limitation of nitrogen, right = potential limitation of phosphorus. 0 is no nutrient limitation and 1 is maximum nutrient limitation (no dissolved inorganic 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.

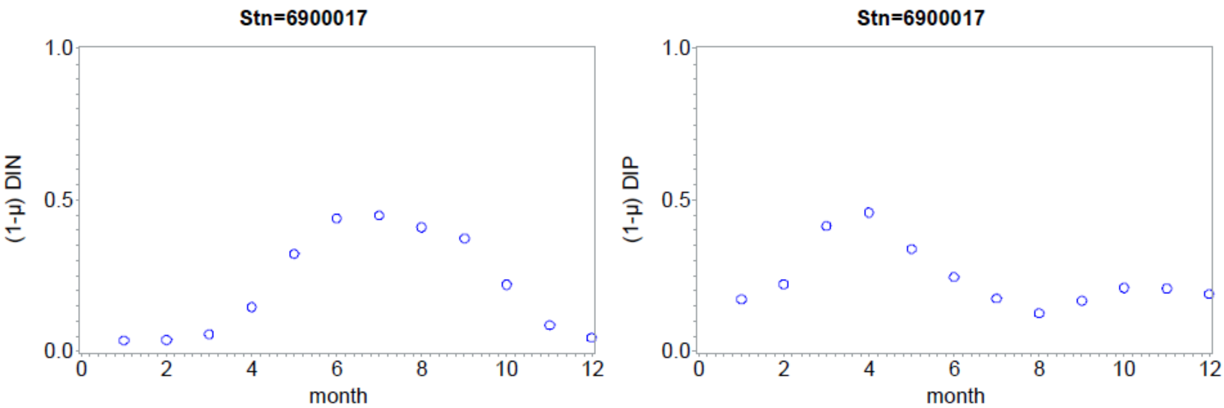


Figure 6.6. Potential nutrient limitation for each month at station 6900017 in Odense Fjord. Left = potential limitation of nitrogen, right = potential limitation of phosphorus. 0 is no nutrient limitation and 1 is maximum nutrient limitation (no dissolved inorganic nutrient) This station is an example 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).

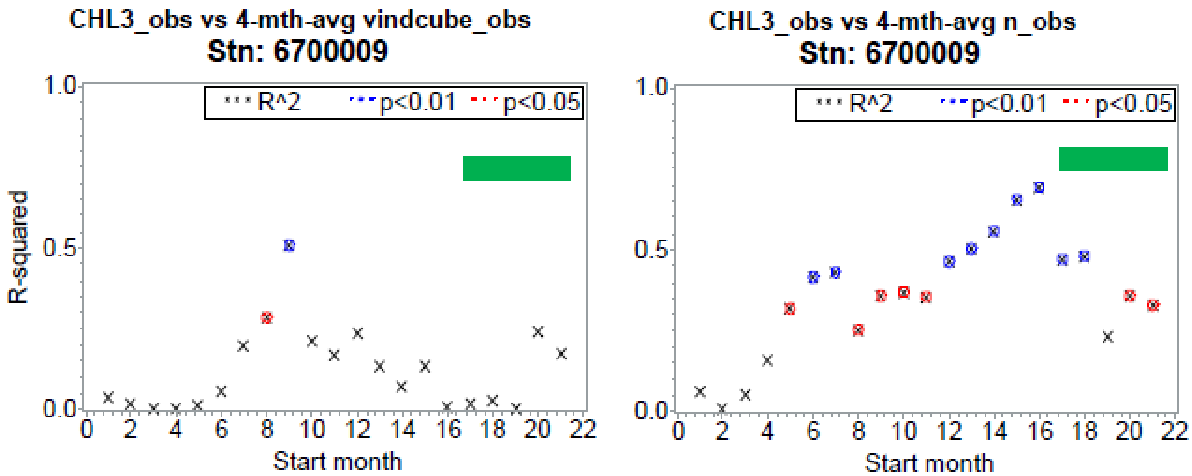


Figure 6.7. R-squared plotted over the starting month of the predictor variables for the responding variable Chl3 (average chl. a month 5-9). The predictor variables are four-month average of vindcube (cubed wind speed) to the left and n-obs (N loading) to the right. Month 1 is January a year before 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.

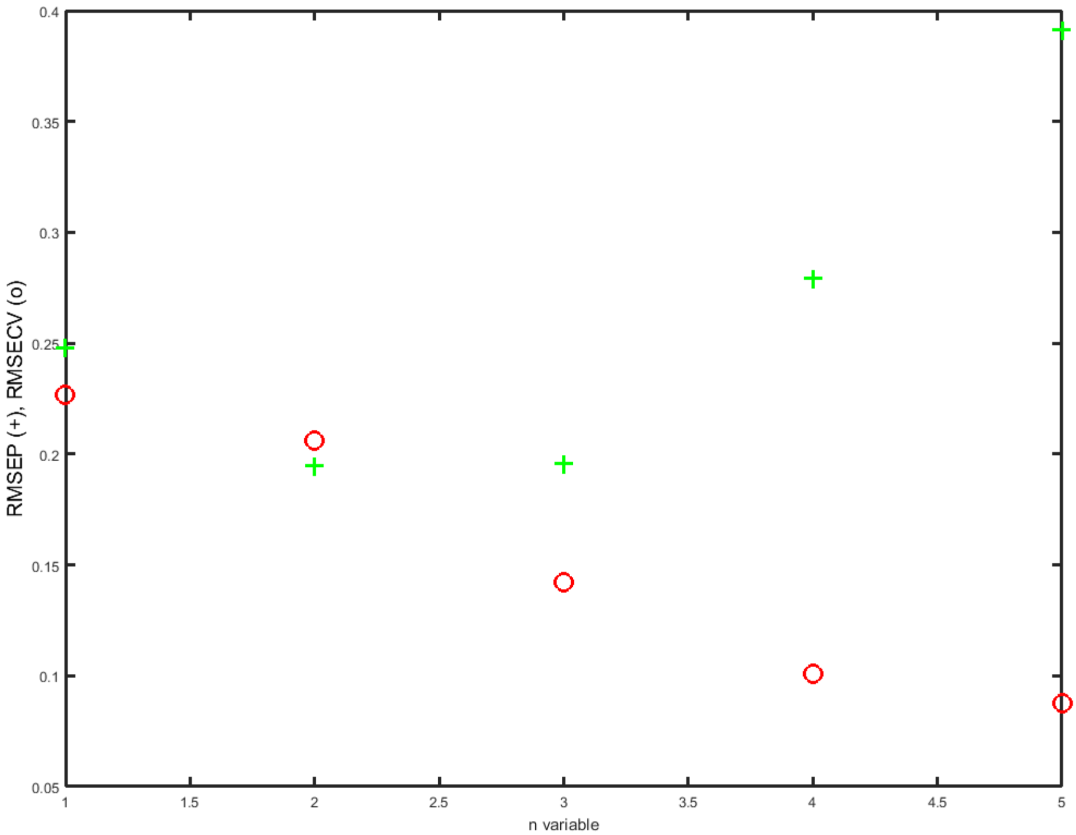


Figure 6.8. An example (from Isefjord) of RMSECV and RMSEP plotted against the number of variables in the PLS regression. In this case, the number of variables would be three since this is the point where the independent validation data start to increase in error RMSEP (+), and after three variables 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.

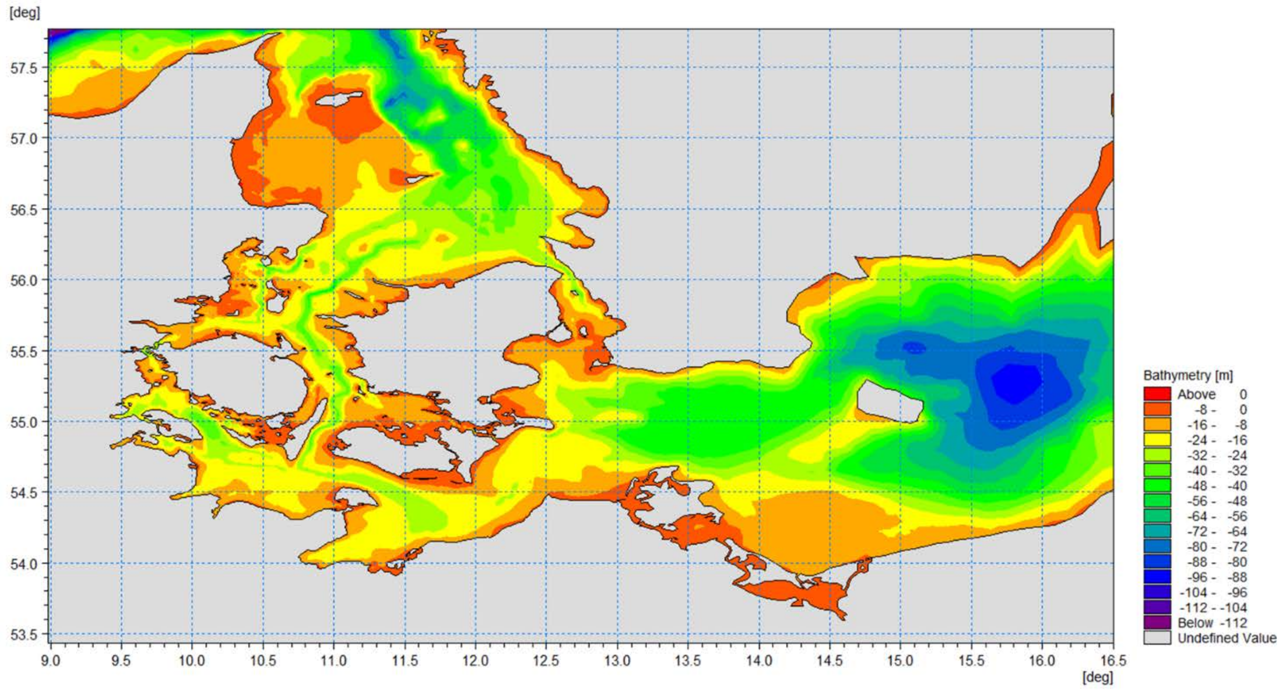


Figure 7.1. Model zoom of the inner Danish waters from the regional IDW model. The model covers the waters from the Skagerrak in the north-west to the Bothnian Bay in the east. Colours indicate water depths relative to mean sea level.

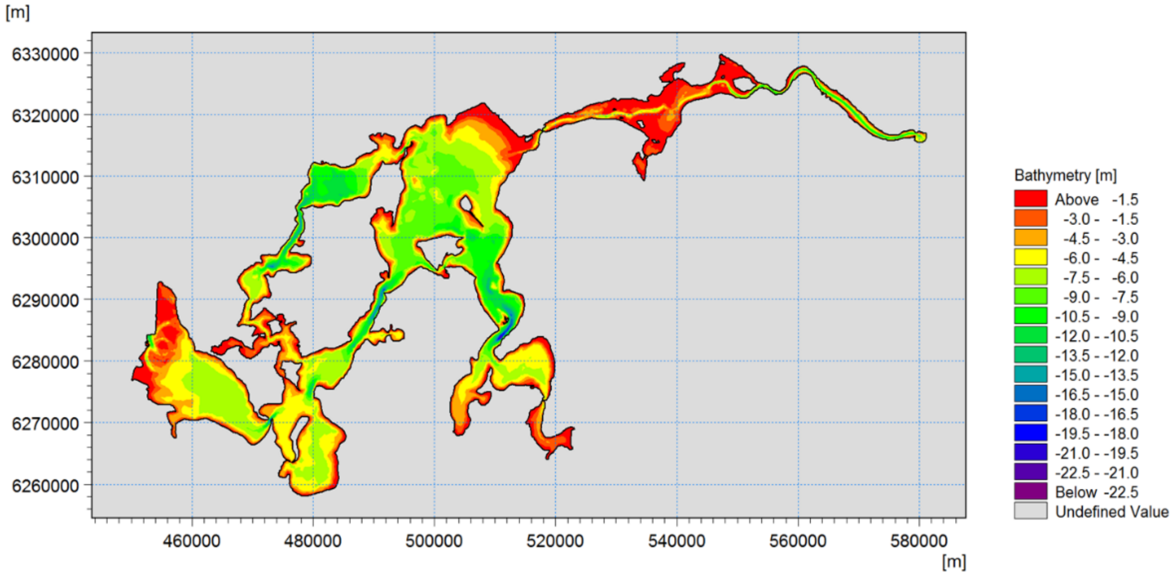


Figure 7.2. The model covering the Limfjorden extending from Thyborøn Canal in the west to Hals in the east. Colours indicate water depths relative to mean sea level.

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).

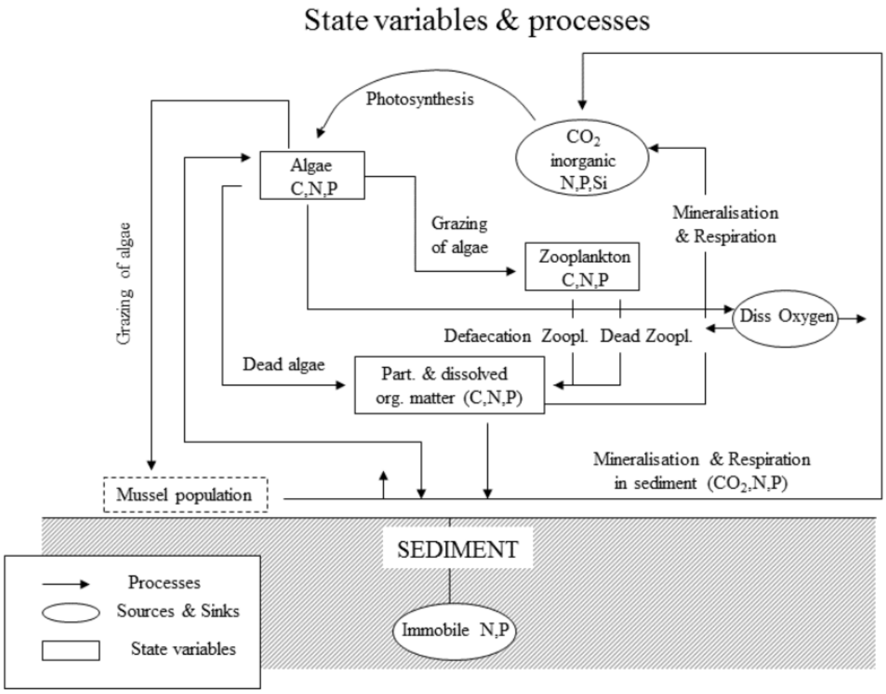
In the IDW model, nitrogen is also made available through nitrogen fixation by cyanobacteria. N_2 fixation is an important source of N to the Baltic Sea, and some estimates suggest that N fixation is comparable to the total land-based N load to the Baltic Sea (Neumann & Schernewski 2008). Since cyanobacteria 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.

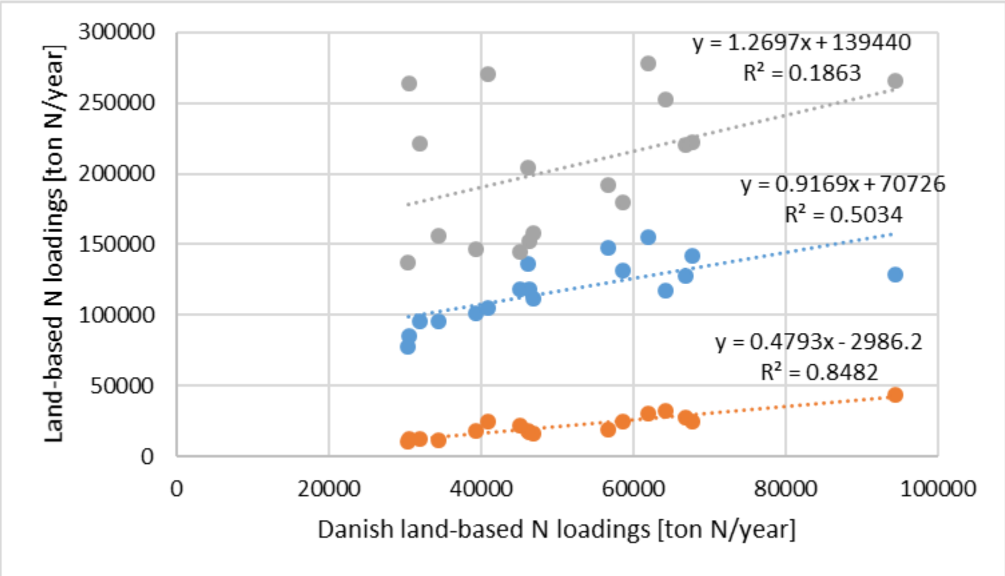


Figure 7.4. The relation between Danish annual loadings (tons) in the period 1994-2010 and the corresponding loadings from Germany (orange line), Sweden (blue line) and Poland (grey line).

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).

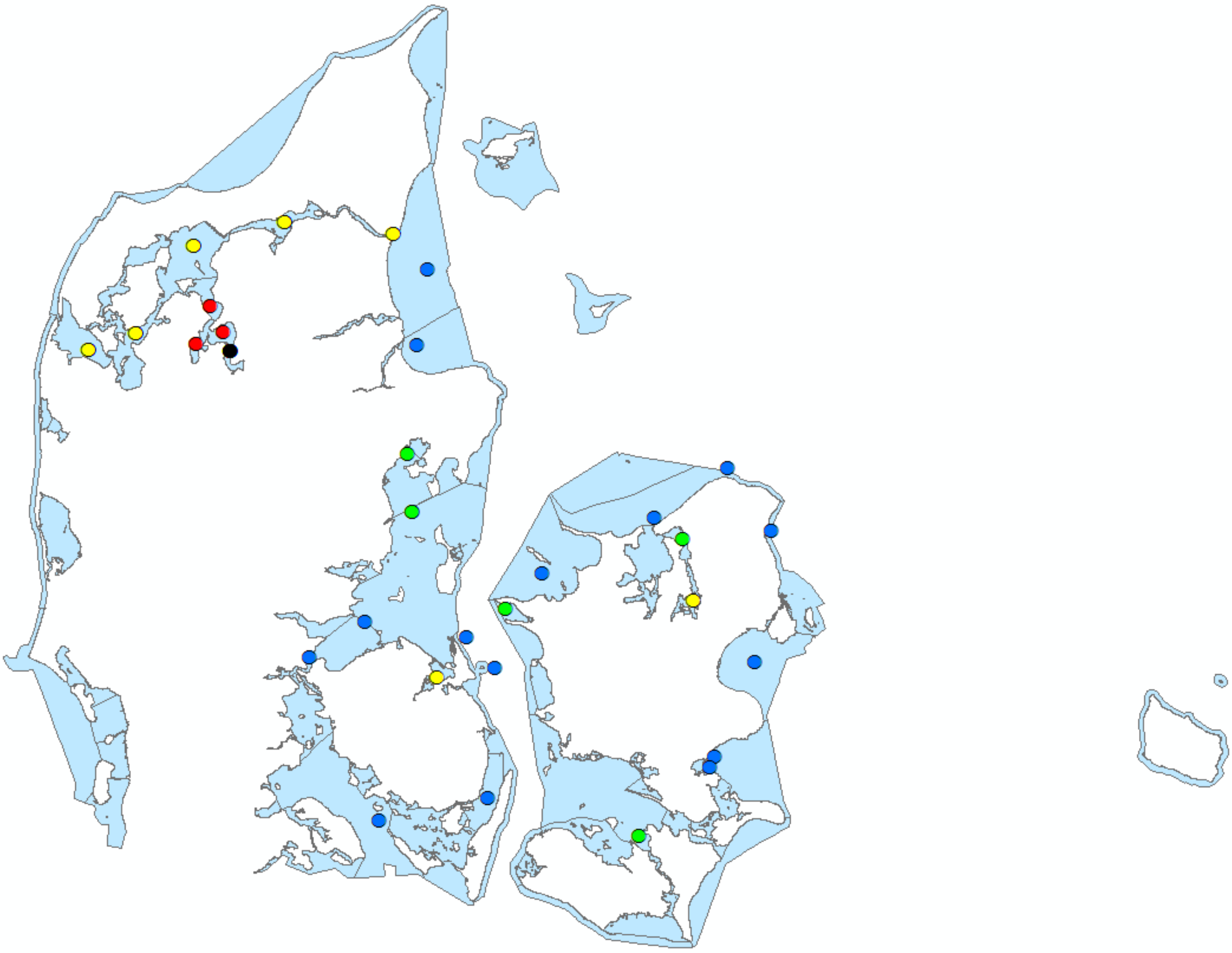


Figure 7.5. Water bodies and monitoring network applied for assessment of the model skills. The different colours of the dots indicate monitoring stations representing different water body types. Black dot in the most inner part of the Limfjorden represent a sluice-fjord (Type 5), yellow dots in the 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.

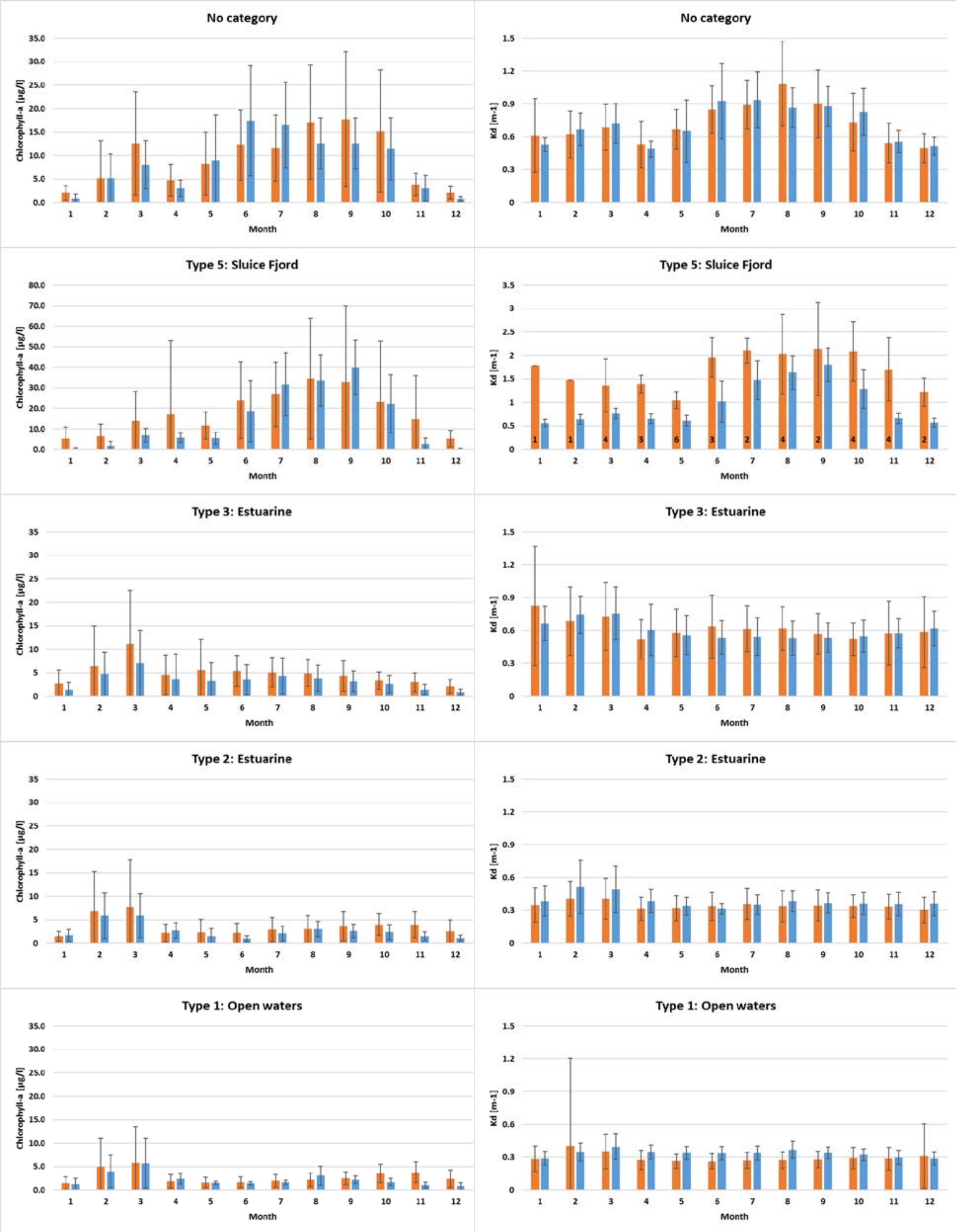


Figure 7.6. Average monthly measured (orange bars) and modelled (blue bars) concentrations of surface chlorophyll-*a* (left column) and K_d (right column). Error bars represent one standard deviation. From top panel to bottom panel the different water body categories are included: No category, Type 5, 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

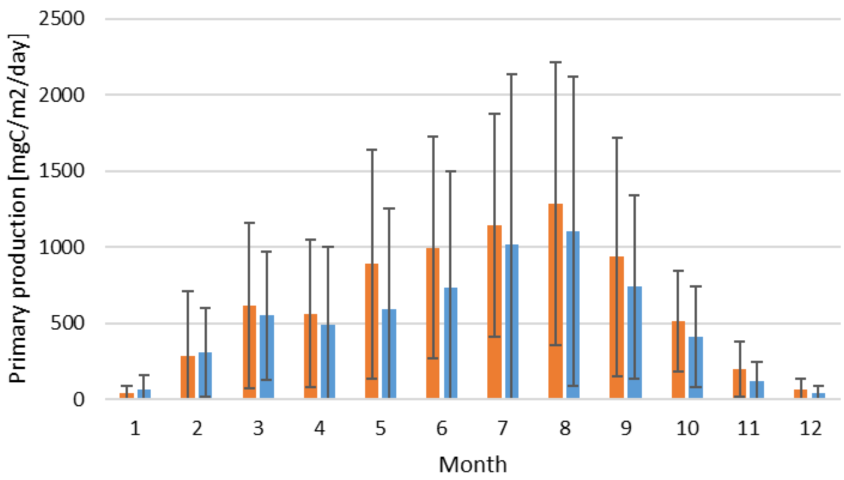


Figure 7.7. Average monthly measured (orange bars) and modelled (blue bars) primary production. Error bars included represent one standard deviation. The evaluation covers the years 2002-2011.

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.

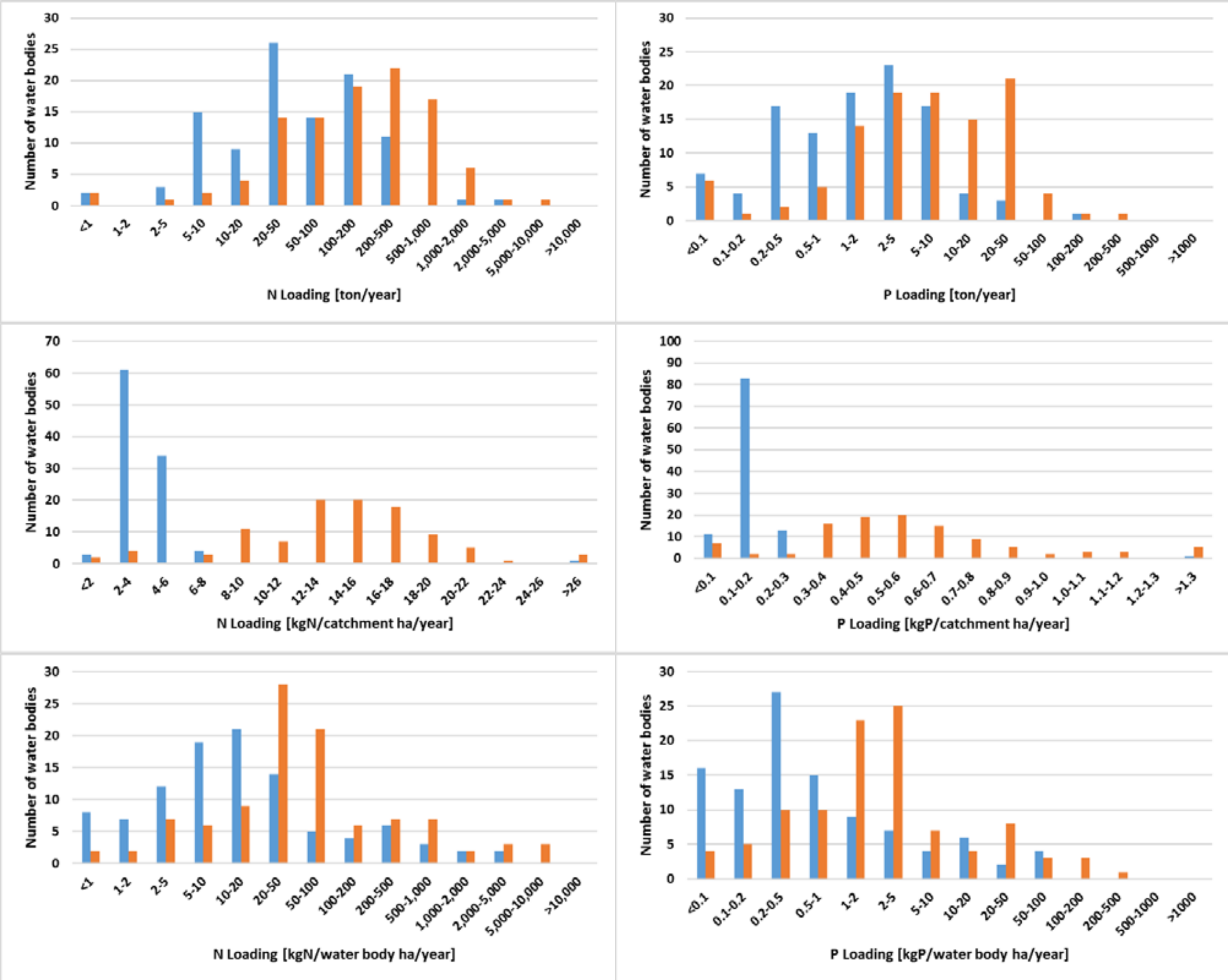


Figure 8.1. Statistics based on average yearly nutrient loadings to each of the 119 Danish WFD water bodies under reference (year 1900; blue columns) and present (2007-2011; orange columns) conditions. The water bodies are divided according to the grouping indicated by the x-axes. Left vertical panel 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.

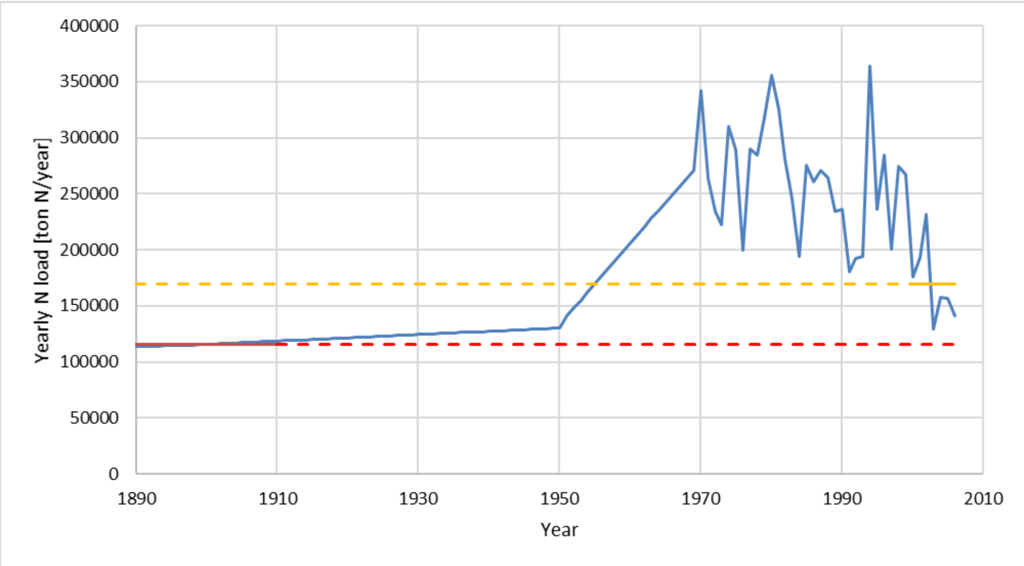


Figure 8.2. Yearly riverine N loadings to the Gotland Basin (basin no. 9), modified from Savchuk et al. (2012a). Red line indicates average 1890-1910 N loadings whereas orange line indicates average 2000-2006 N loadings.

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*:

$$EQR_{chl}^{i,j} = \frac{Reference\ value\ for\ chlorofyll{-}a}{Target\ i,j\ chlorophyll{-}a\ concentration} \qquad \text{Eq. 8.1}$$

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:

$$Target\ value_{chlorophyll-a\ indicator} = \frac{\text{Reference value for chlorofyll-}a}{0.6} \qquad \text{Eq. 8.2}$$

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).

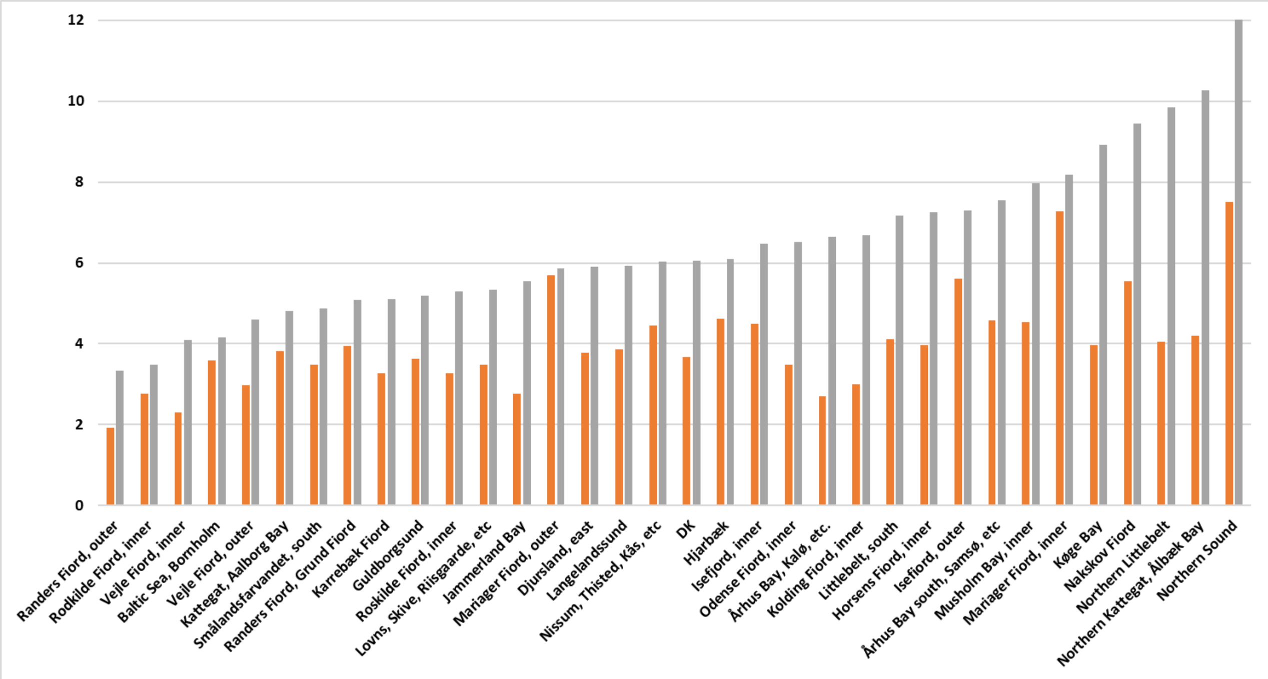


Figure 8.3. Relative increase in nitrogen loadings (catchments accounting for 80% of present day land-based N loading) to Danish water bodies from reference load (approx. year 1900) to present day (2007-2012) loadings (orange) and the 1990s (grey).The y-axis has been cut at a factor of 12, but for the 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:

$$PLR = (PLR_1 * w_1 + PLR_2 * w_2 + PLR_3 * w_3 + PLR_4 * w_4 + PLR_5 * w_5 + PLR_6 * w_6) / \sum w_1\text{-}w_6 \qquad \text{Eq. 8.5}$$

where PLR_n is PLR for each indicator and w_n is the associated weight

and

$$L_t = L_p - (PLR/100 * L_p) \qquad \text{Eq. 8.6}$$

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*).

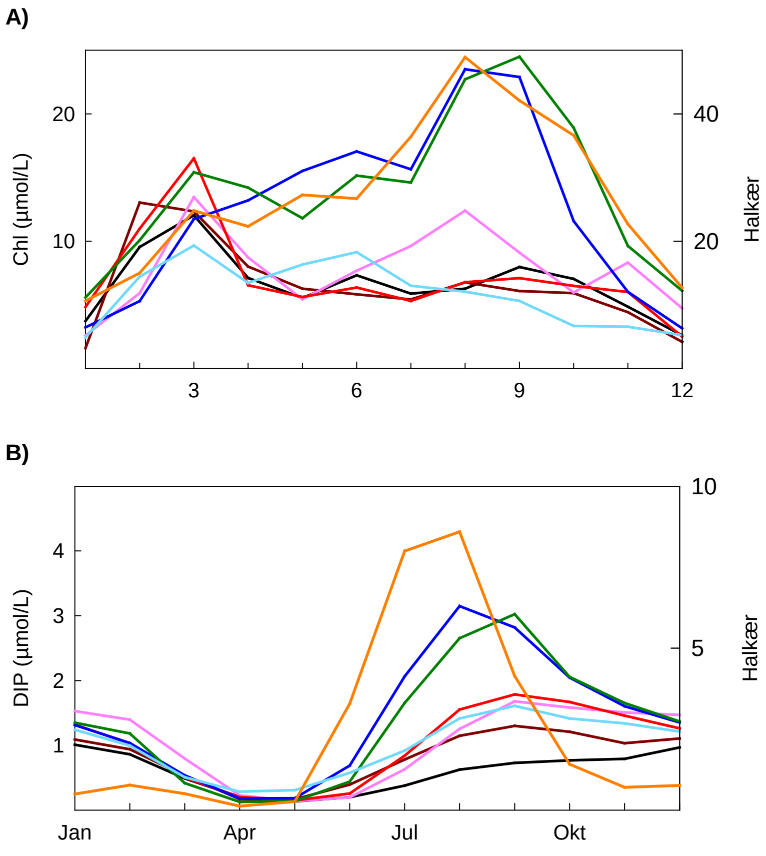


Figure 8.5. A) Seasonal patterns in eight basins of the eutrophic estuary the Limfjorden. The corresponding distribution of oxygen depletion is shown in **B)**. The blue and green lines are from the south-eastern basins Skive Fjord and Lovns Broad where anoxia is widespread, frequent shifts occur between 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.

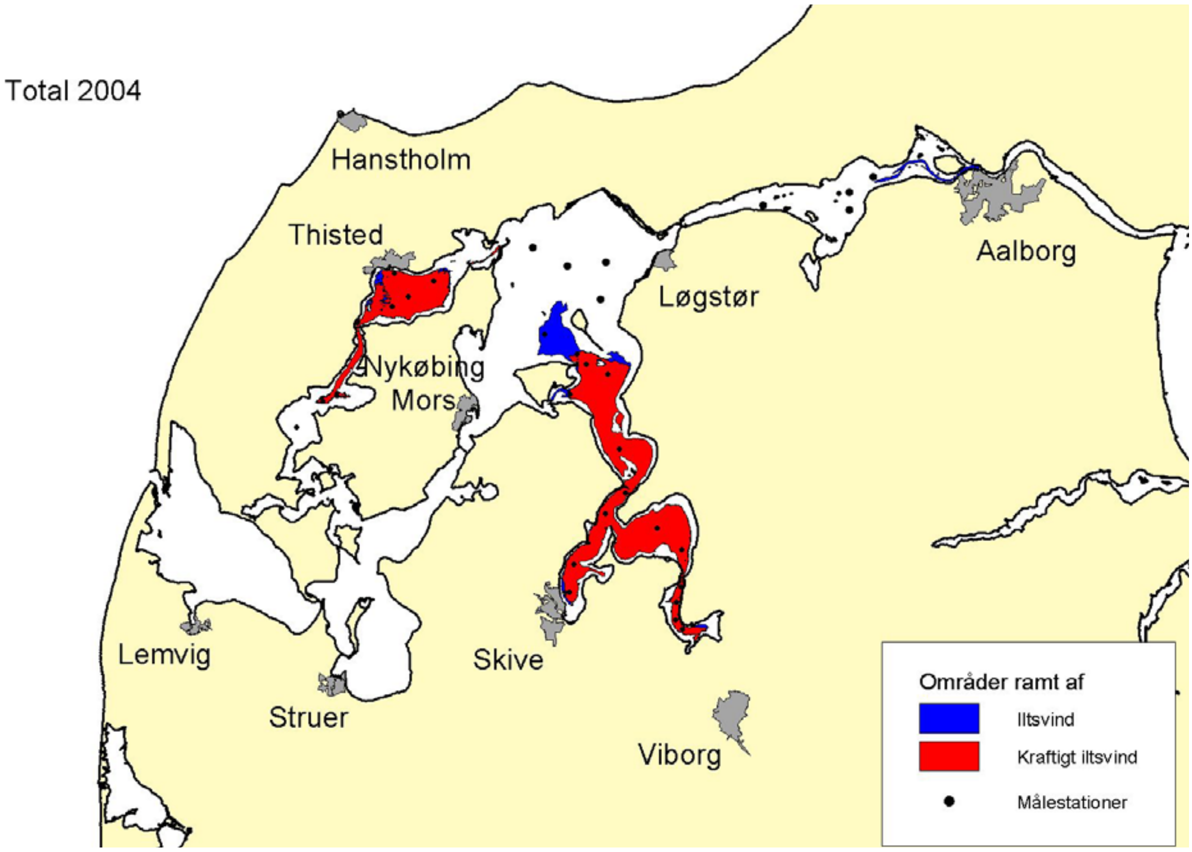


Figure 8.6. The spatial distribution of oxygen depletion in the Limfjorden in 2004. Blue and red zone correspond to concentrations below 4 and 2 mg oxygen l^{-1}, respectively.

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*).

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[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.

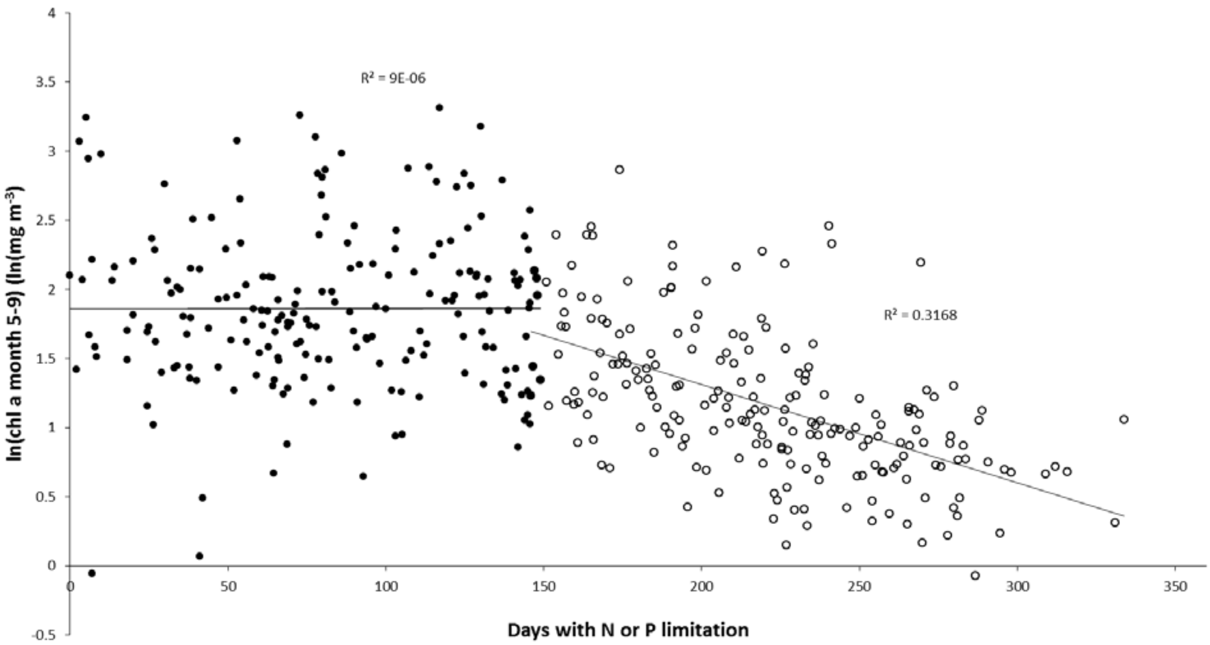


Figure 8.7. The relationship between ln(chl-*a* month 5-9) and the number of days with either N or P limitation. Black dots are observations (station and year) with fewer than 150 days, while the white dots are observations of chlorophyll-*a* concentration in years or at stations with more than 150 days of N or P limitation.

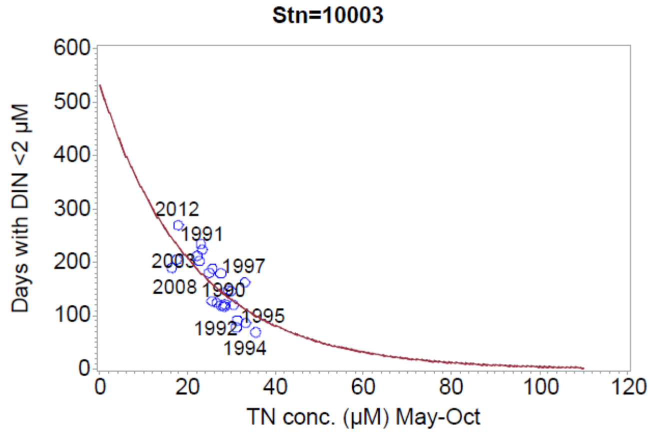
Similar analyses for coastal and open waters revealed threshold values of 200 days and 220 days, respectively.

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 $TN_{May-Oct}$ 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.

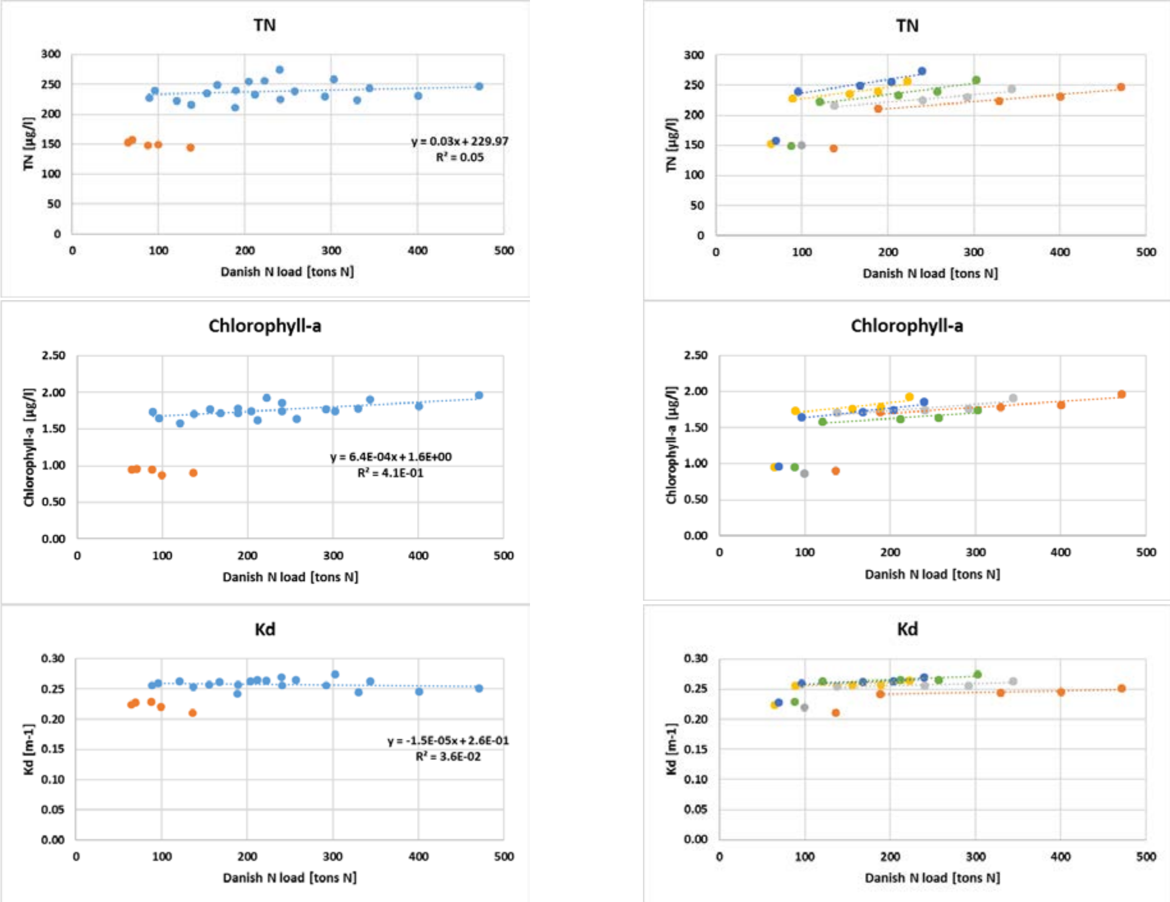


Figure 8.10. Indicator sensitivity to Danish land-based N loadings for water body No. 217. Top panel show TN concentrations, middle panel show chlorophyll-*a* concentrations and bottom panel show K_d. Blue dotes in left panels originate from different scenarios and different years, whereas orange dots 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.

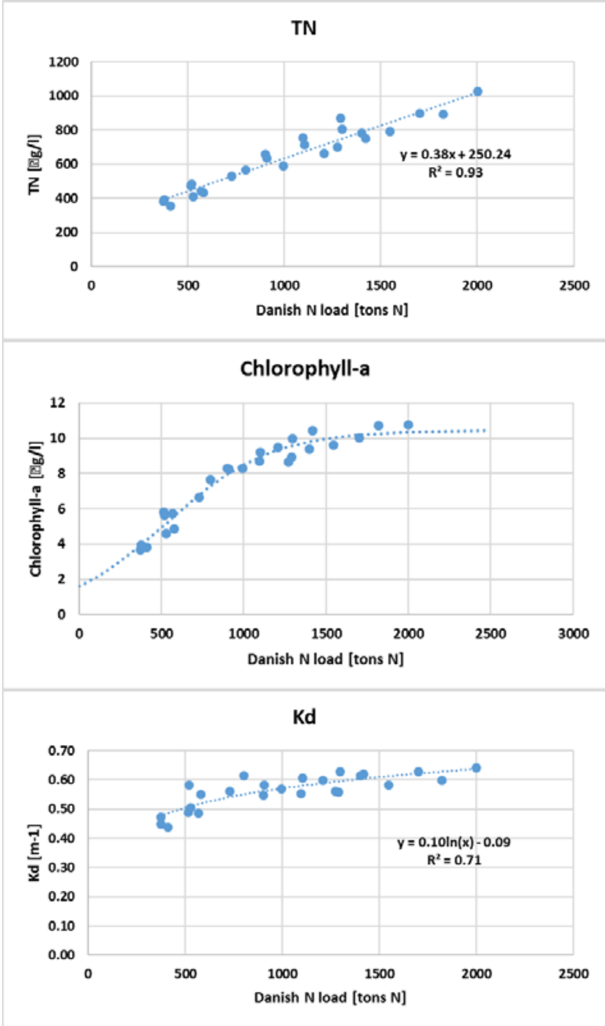


Figure 8.11. Indicator sensitivity to Danish land-based N loadings for water body No. 157. Top panel show TN concentrations, middle panel show chlorophyll-*a* concentrations and bottom panel show K_d. Blue dots originate from different scenarios and different years.

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.

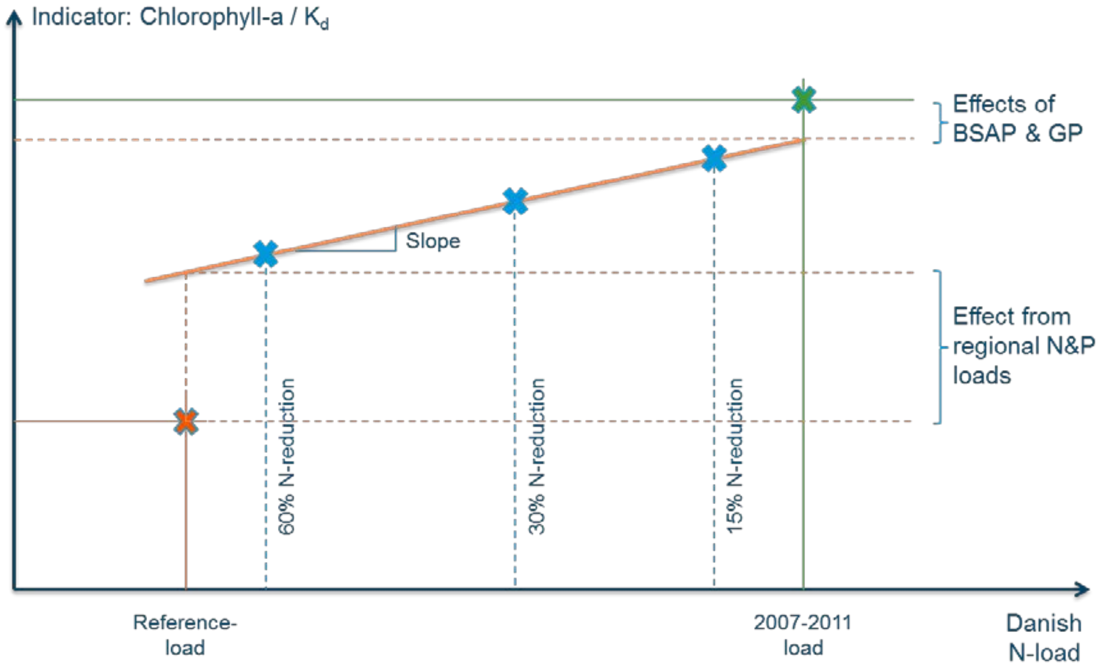


Figure 8.12. Schematic illustration of the surrogate models developed for each single water body covered by a mechanistic model. The y-axis show the value of the average indicator value (spatio-temporal average) covering the period 2007-2011. The x-axis show the Danish land-based N load. Loadings from 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*.

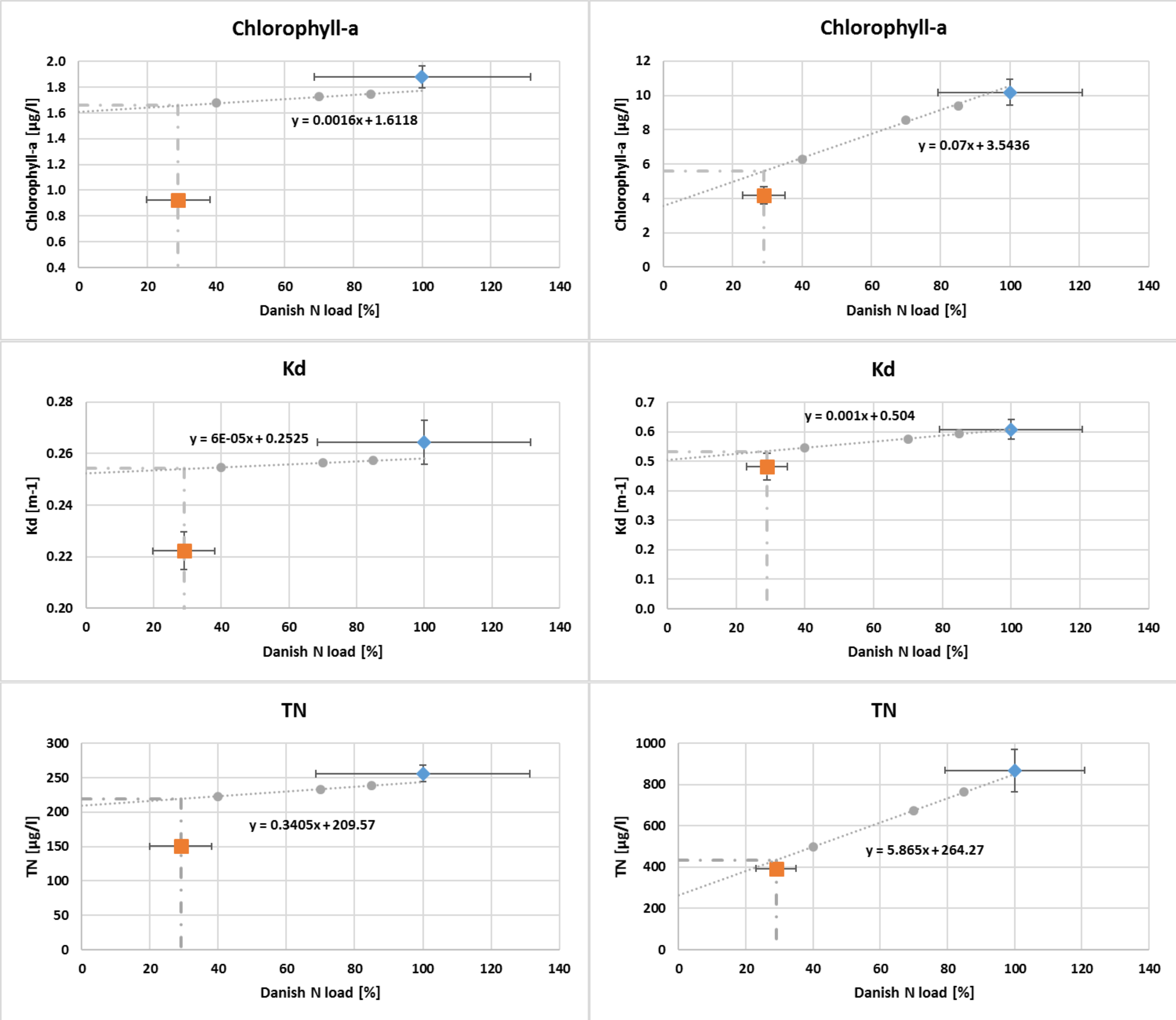


Figure 8.13. Impact correlations between Danish N loadings (StDev show the variability in loadings for the specific scenarios) and the two indicators chlorophyll-*a* (top panel) and K_d (middle panels) and the supporting parameter TN (bottom panels). Left figures are based on model results from water 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.

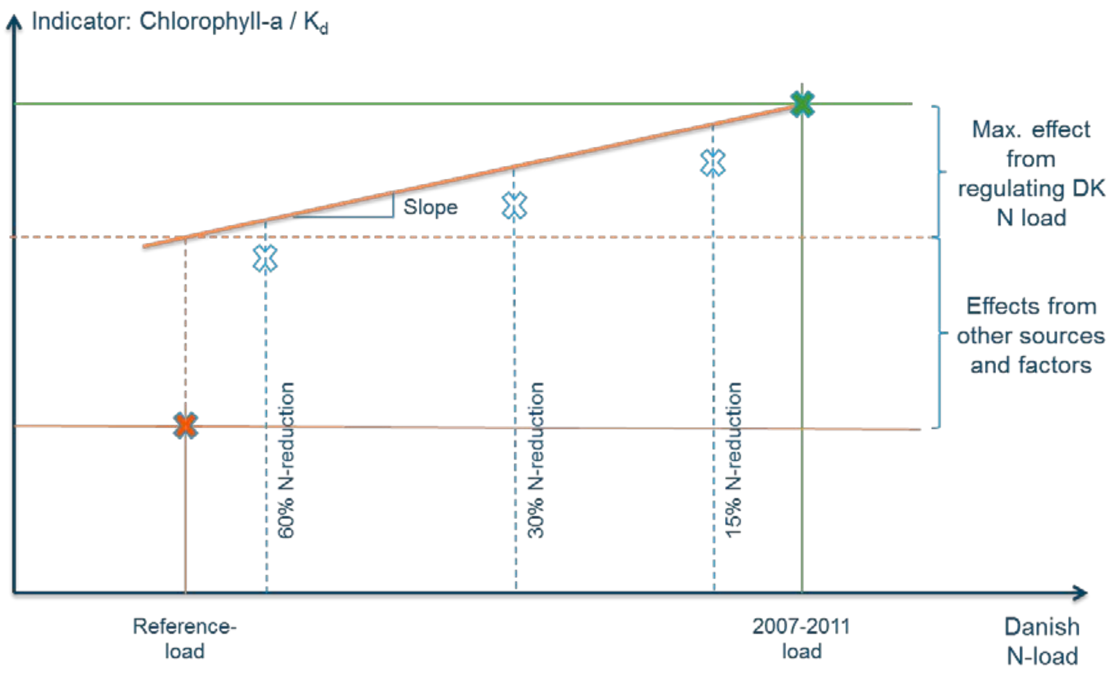


Figure 8.14. Schematic illustration of the surrogate models developed for each single water body describing the maximum effect achievable on the individual indicators from regulating Danish land-based N loadings alone. Orange line show the local sensitivity to Danish land-based N loads and is similar 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.

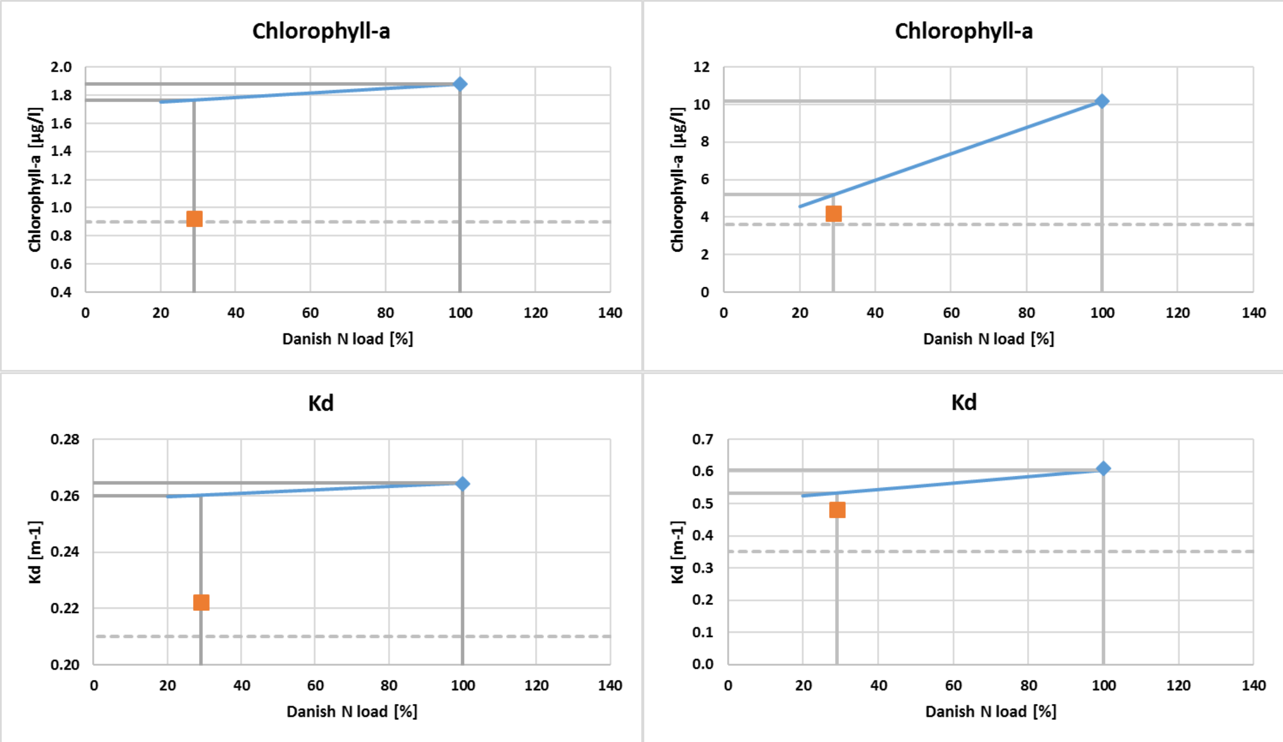


Figure 8.15. Impact correlations between Danish N loads and the two indicators chlorophyll-*a* (top panel) and K_d (bottom panels). Left figures are based on model results from water body No. 217 and right figures are based on results from water body No. 157. Blue dot corresponds to the status indicator 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.

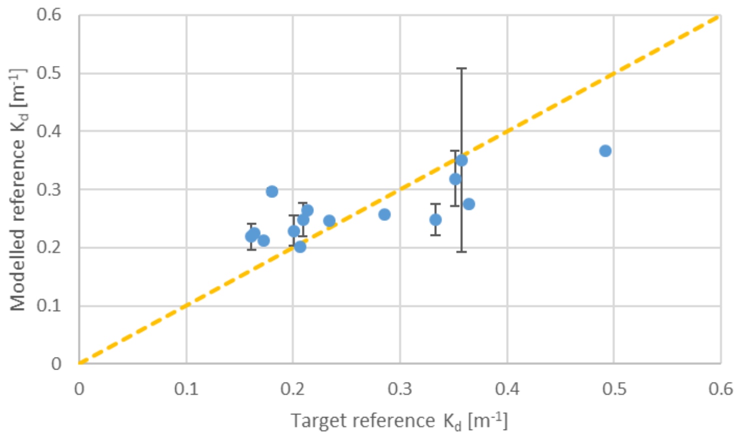


Figure 8.16. Reference K_d values derived from historical observations of eelgrass depth limits and corresponding modelled reference K_d values. Where more model results exist these values are averaged and include error bars of ± 1 StDev. Orange line show the 1:1 line.

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.

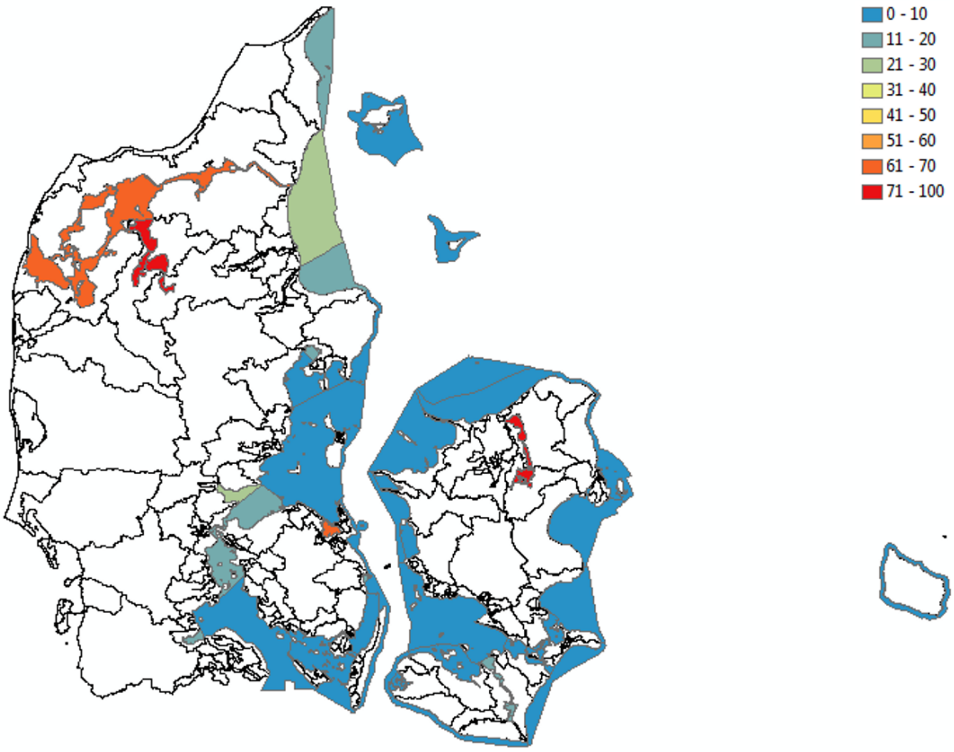


Figure 8.17. Proportion (in %) of chlorophyll-a, that can be regulated by Danish land-based N loads alone.

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*).

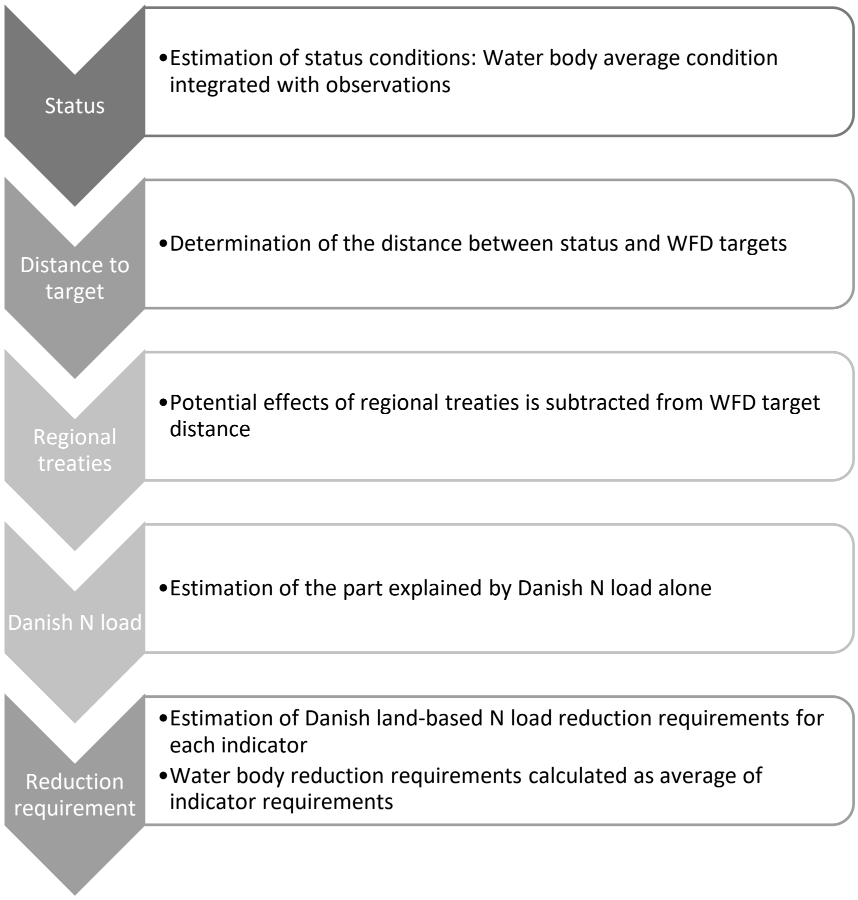


Figure 8.18. Step-by-step method used for calculating reduction targets for each specific water body covered by the mechanistic models.

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*.

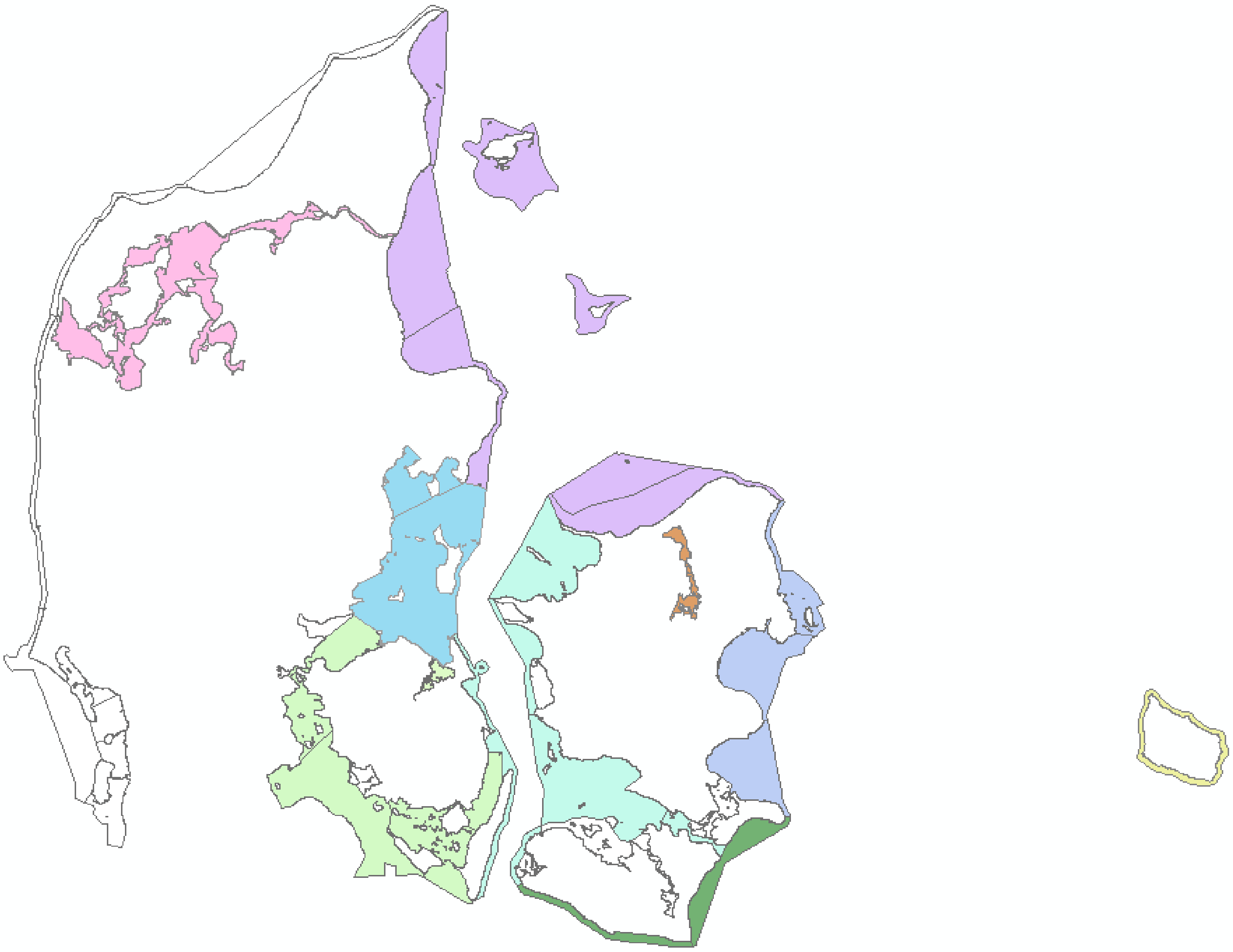


Figure 8.19. Map indicating local areas were individual estimated needs for reductions have been averaged.

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.

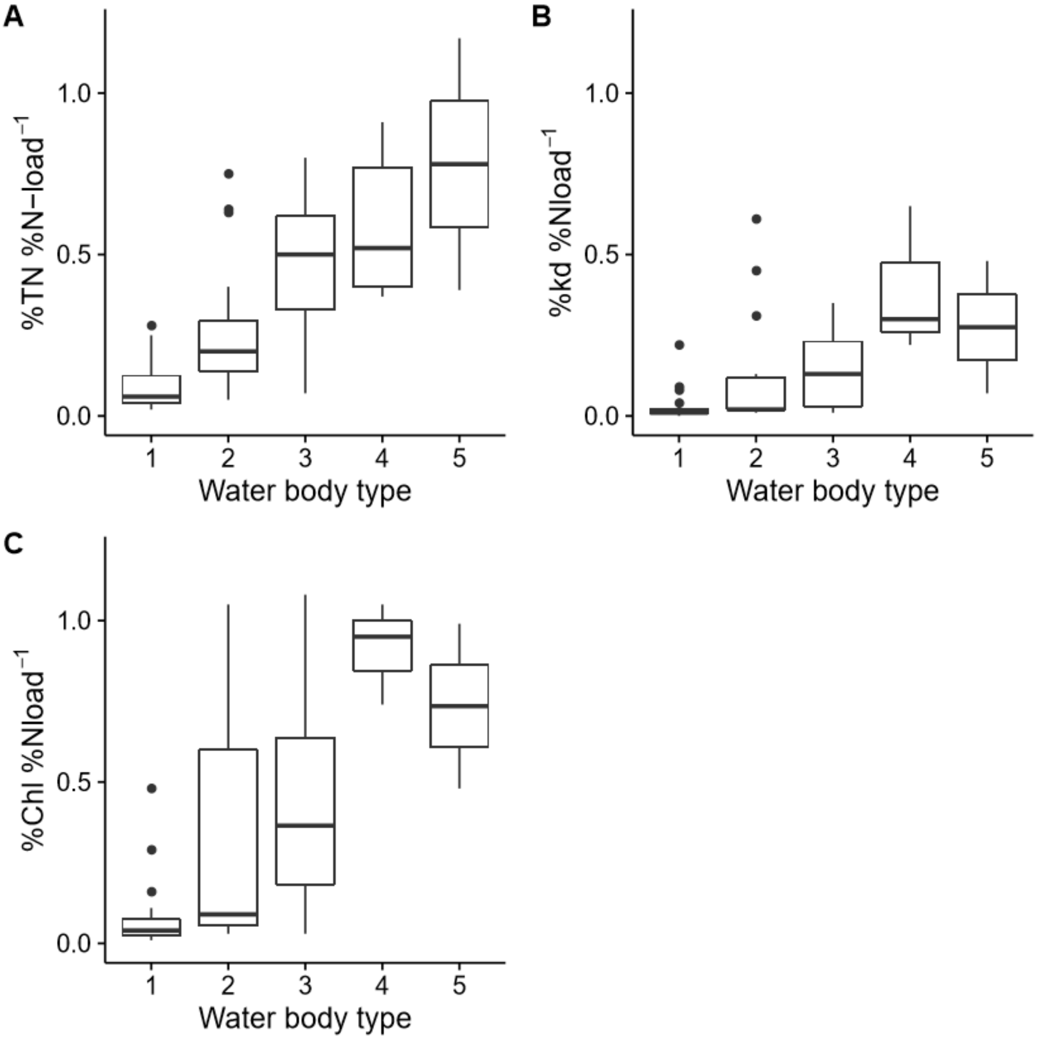


Figure 8.20. Slopes derived from statistical and the mechanistic models for each of the five water body types. A: relative change in TN concentration as a function of the relative change in nitrogen load from Danish catchments; B: relative change in light attenuation as a function of the relative change 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*.

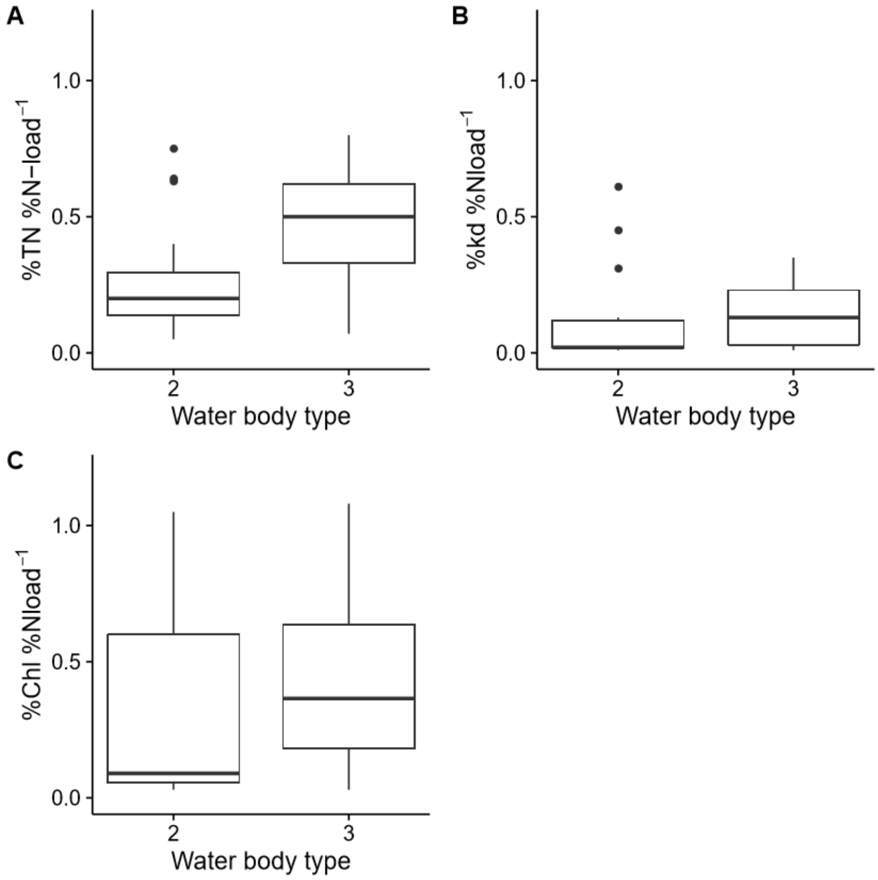


Figure 8.21. Statistics on the type-specific slopes for Type 2 and Type 3 water bodies between relative change in N loading and relative change in A) the yearly TN concentration, B) the K_d indicator and C) the chlorophyll-*a* indicator used in the meta models.

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*.

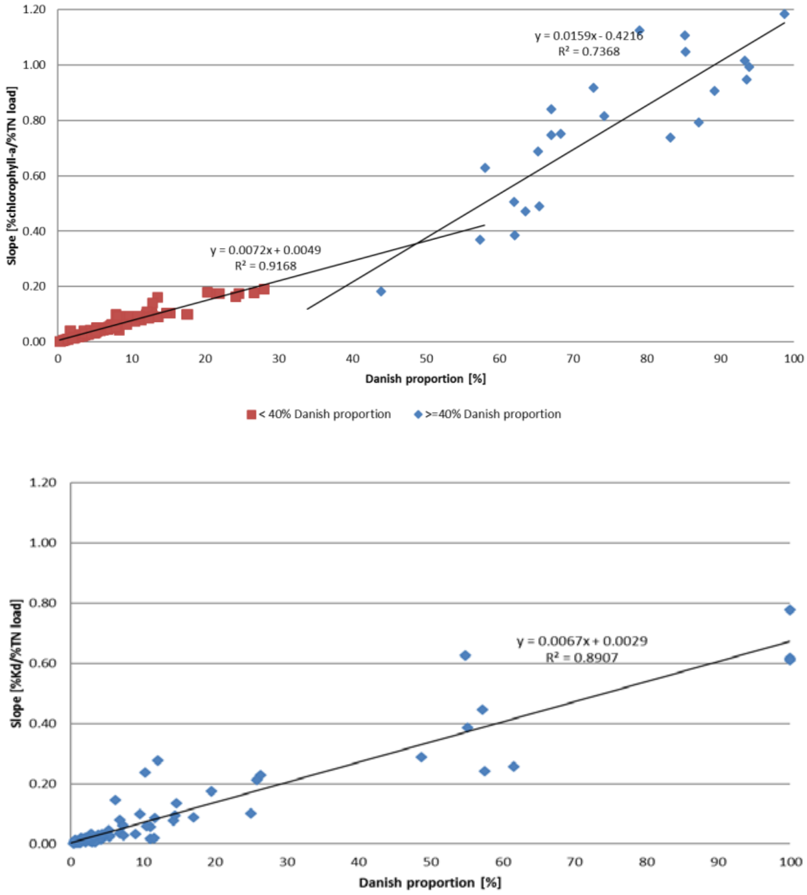
By extracting the trend line slopes in *Figure 8.22*, we get three uniform meta-slopes, two for chlorophyll-*a* and one for K_d:

- 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.

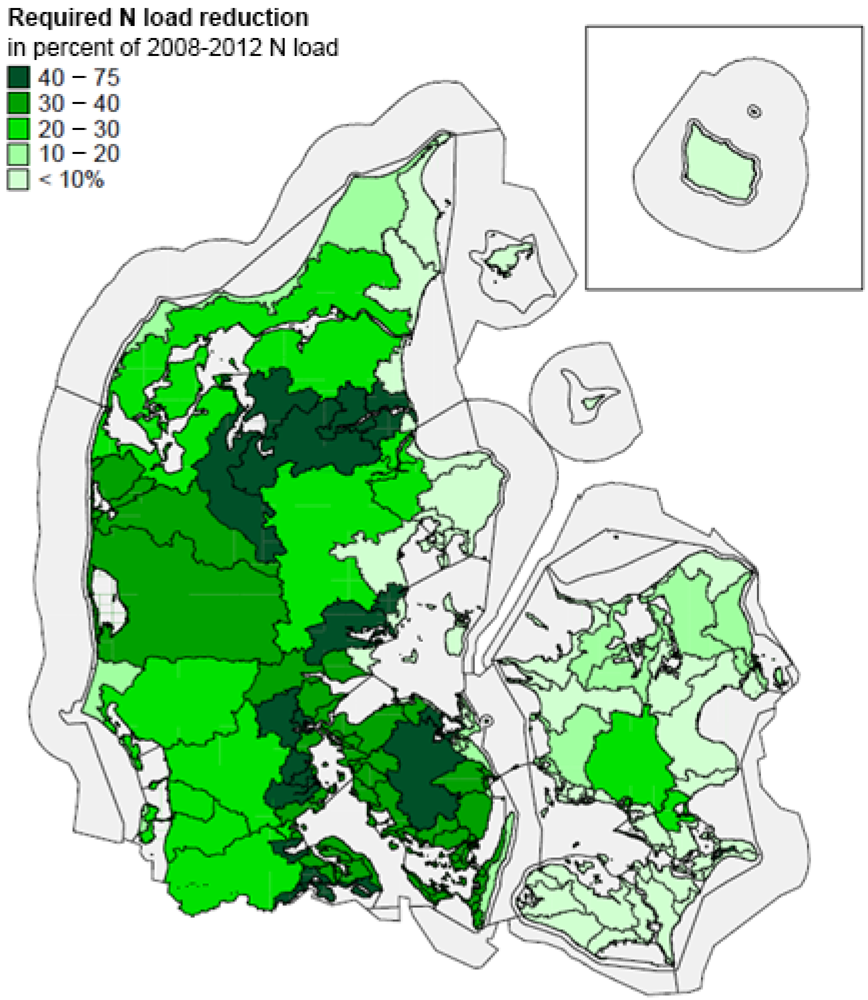
Table 8.9. Examples of application of the developed rules for integrating the results.

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.

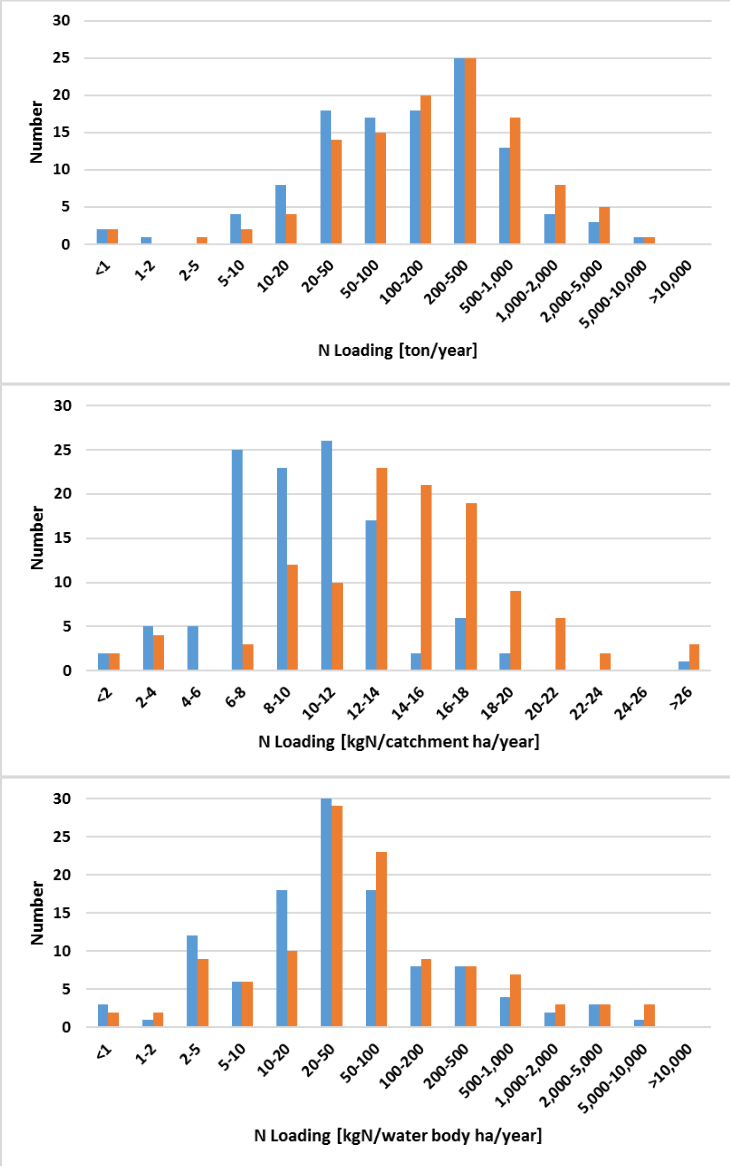


Figure 8.24. Average MAI (blue bars) and status (2007-2012) loadings (orange bars) divided into water bodies. Top panel is the yearly loadings per water body, middle panel is the areal loadings per catchment and bottom panel the areal loadings per water body.

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.

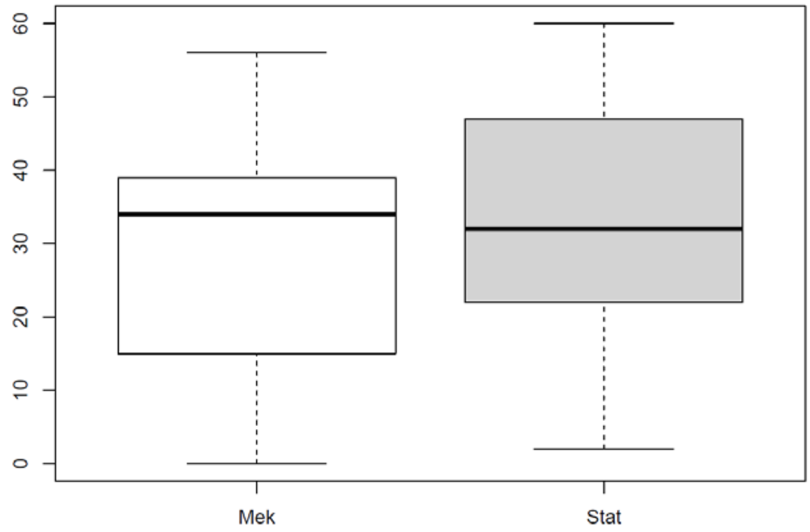


Figure 8.25. Boxplot of the load reduction results for the ensemble modelled water bodies estimated using the mechanistic model approach (Mek) and the statistical model approach (Stat).

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.

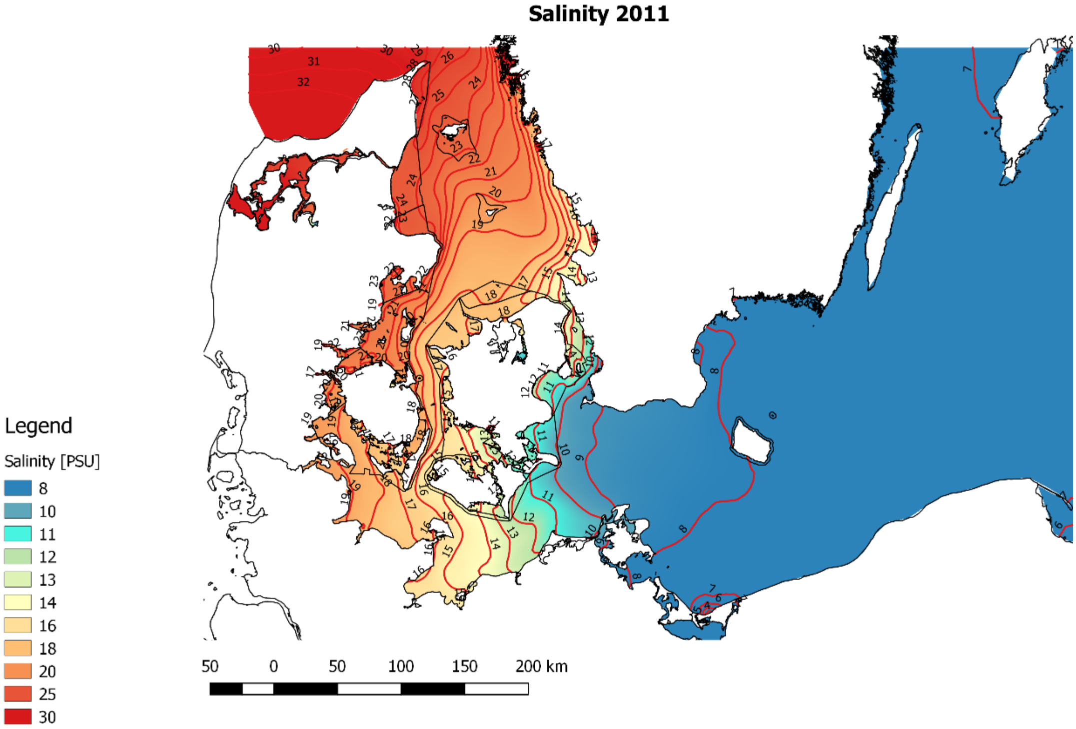


Figure 9.1. Modelled (IDW model) average yearly surface salinity, year 2011.

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).

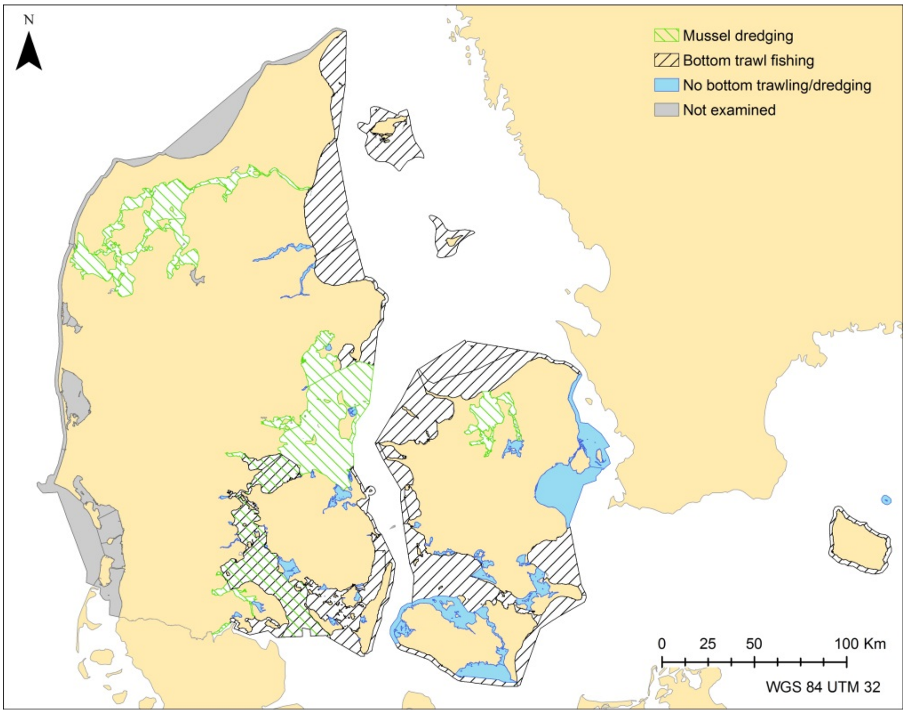


Figure 9.2. Danish WFD water bodies affected by mussel dredging and bottom trawl fishing.

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.

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Appendix A – Danish water bodies

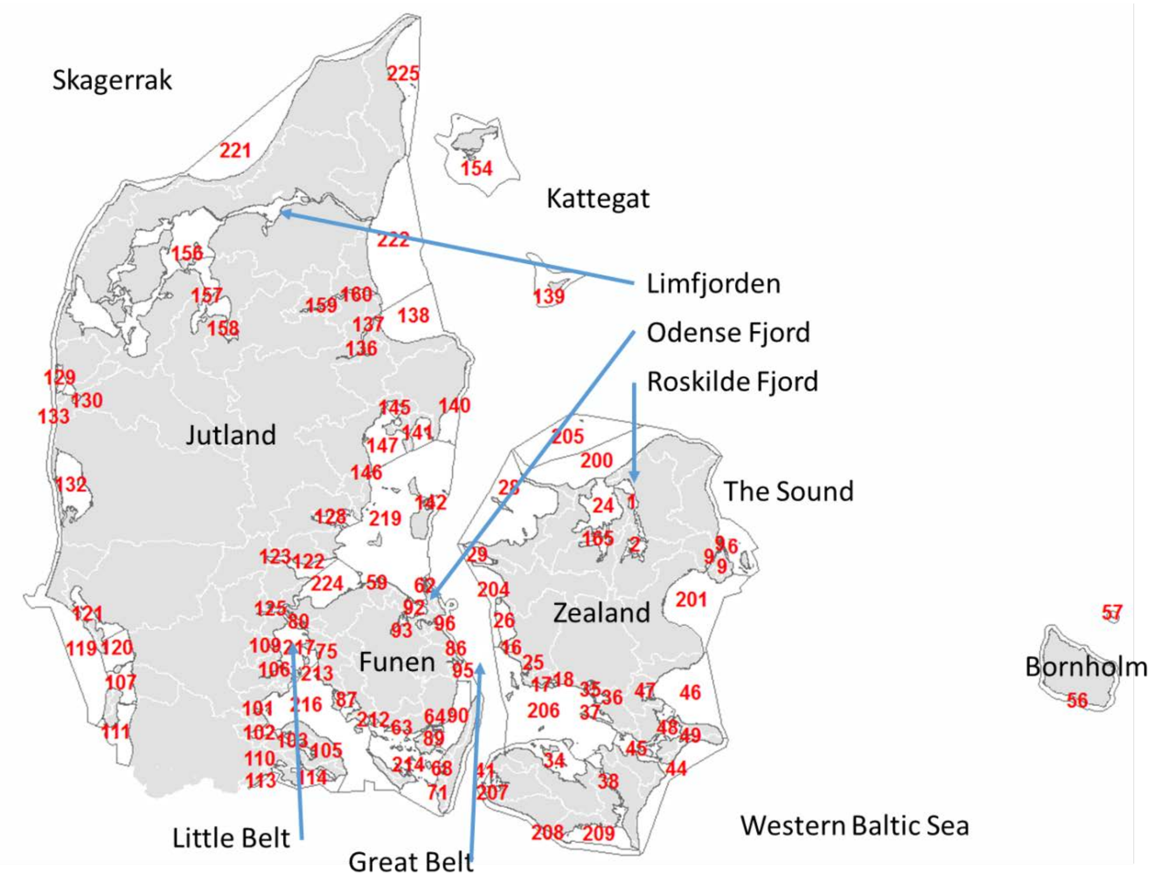


Figure A1. Danish water bodies. Numbers indicate the specific water body number and the corresponding name is included in *Table A1*.

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

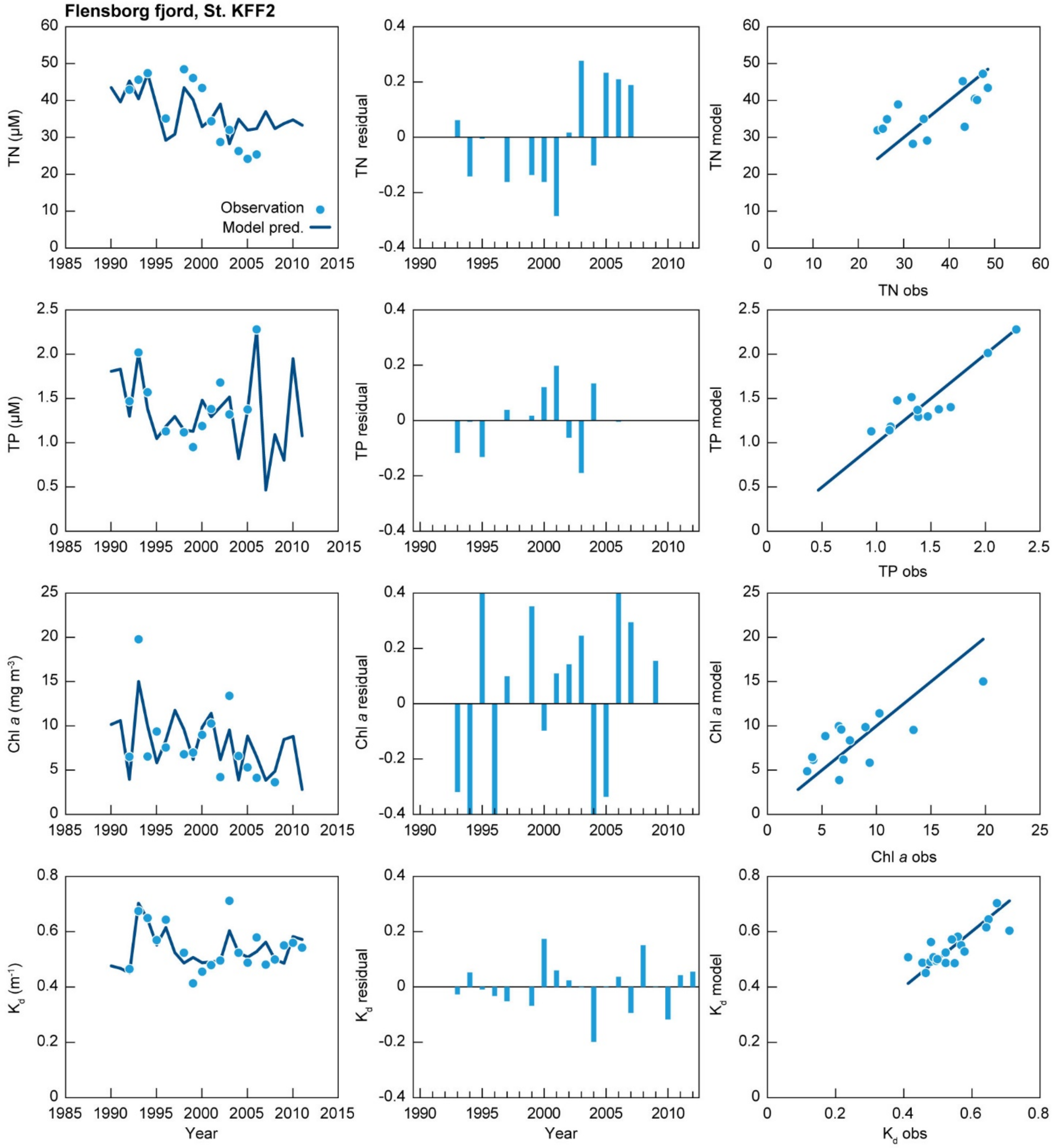


Figure B1.

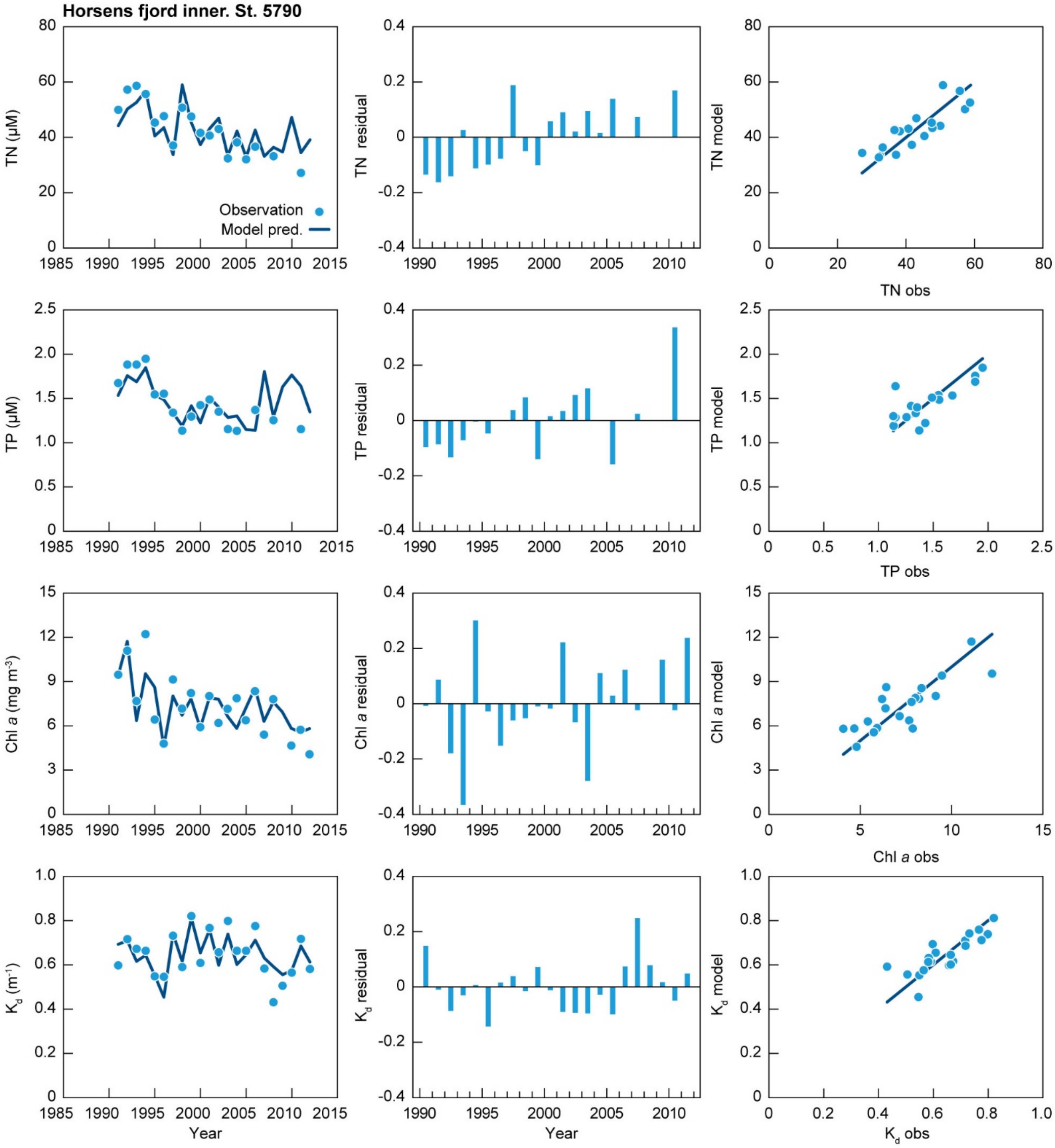
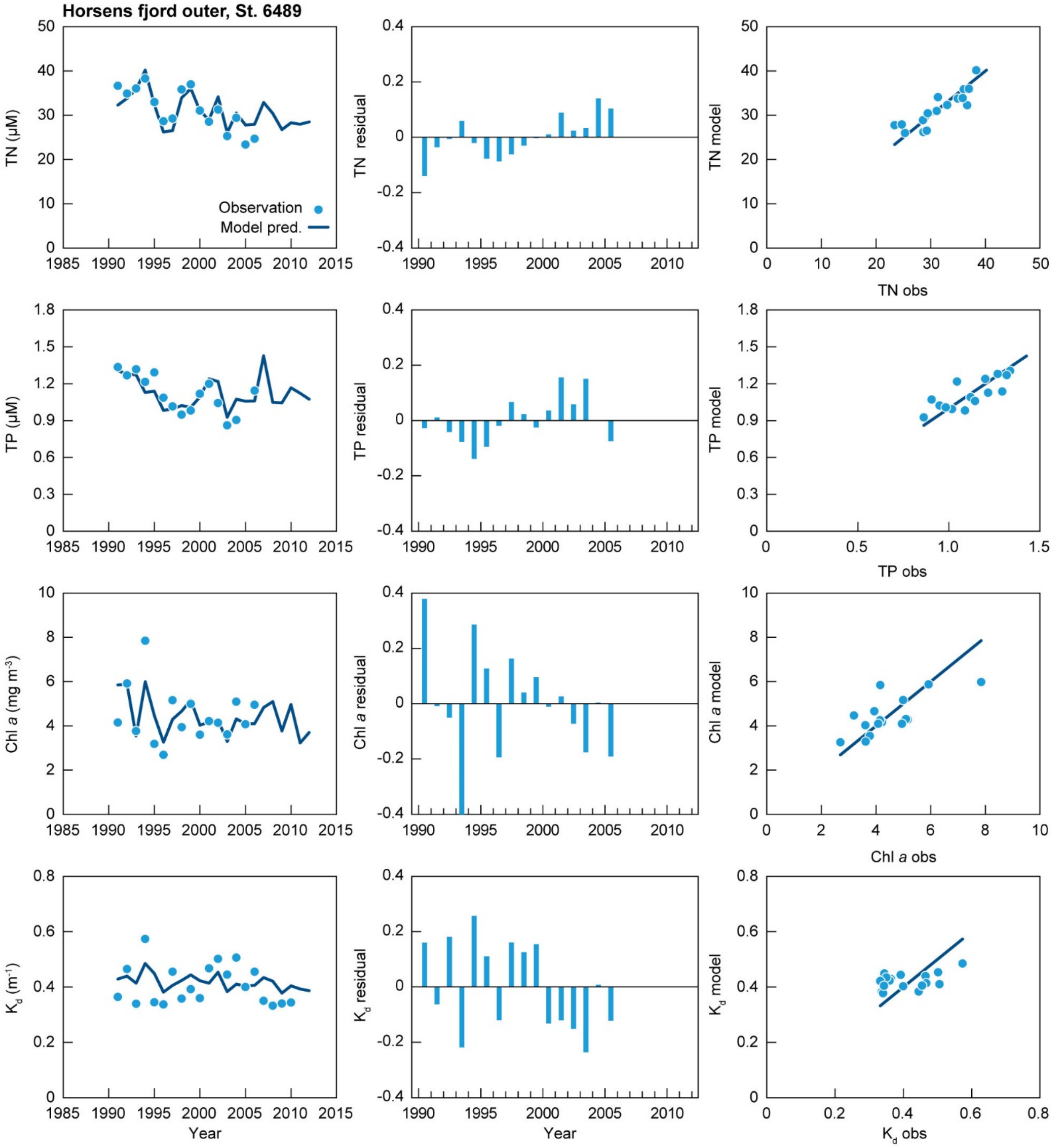

Horsens fjord inner. St. 5790
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year
Year

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

Figure B3.

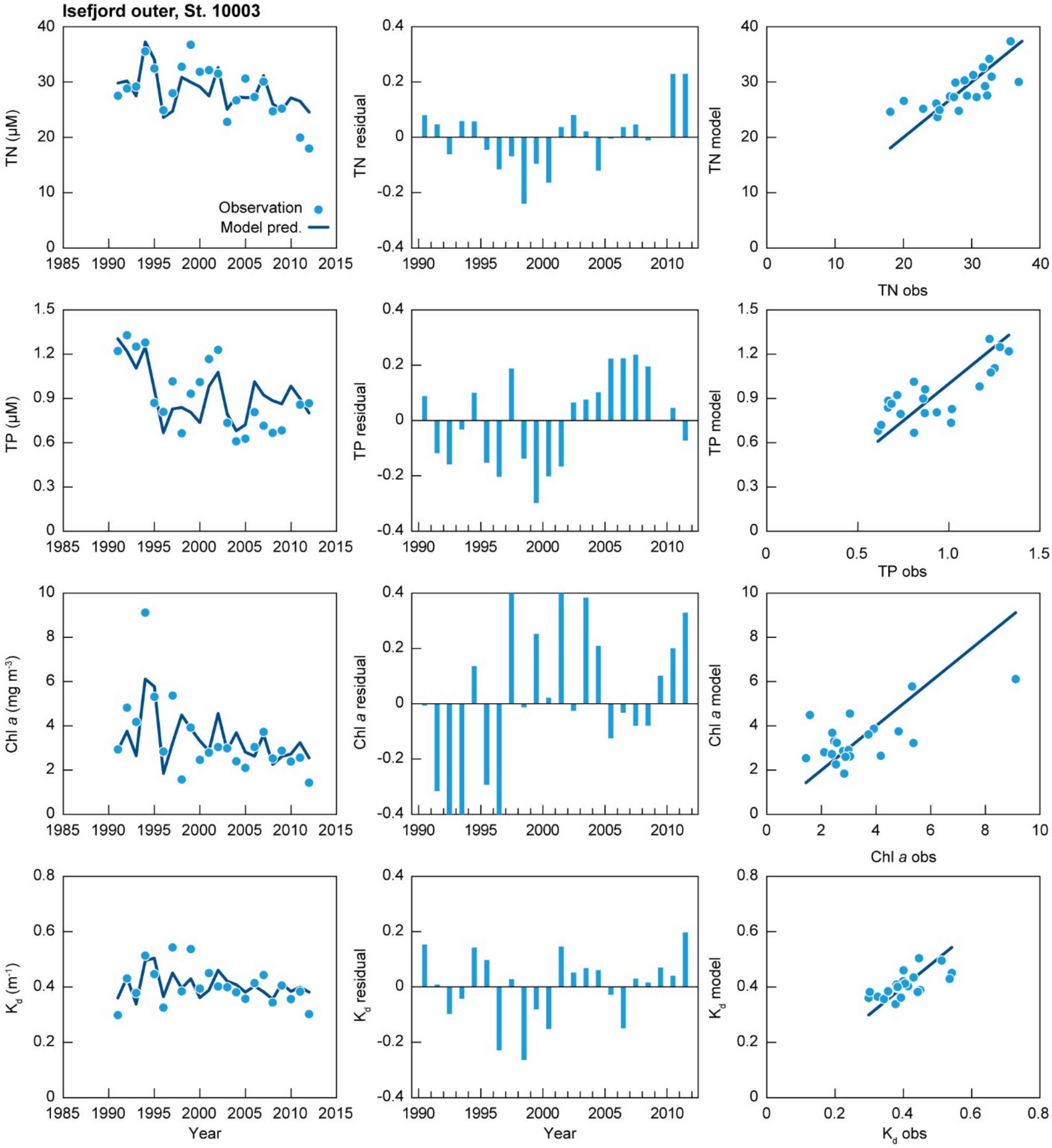

Isefjord outer, St. 10003
Observation
Model pred.
TN (µM)
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year
Year

Figure B4.

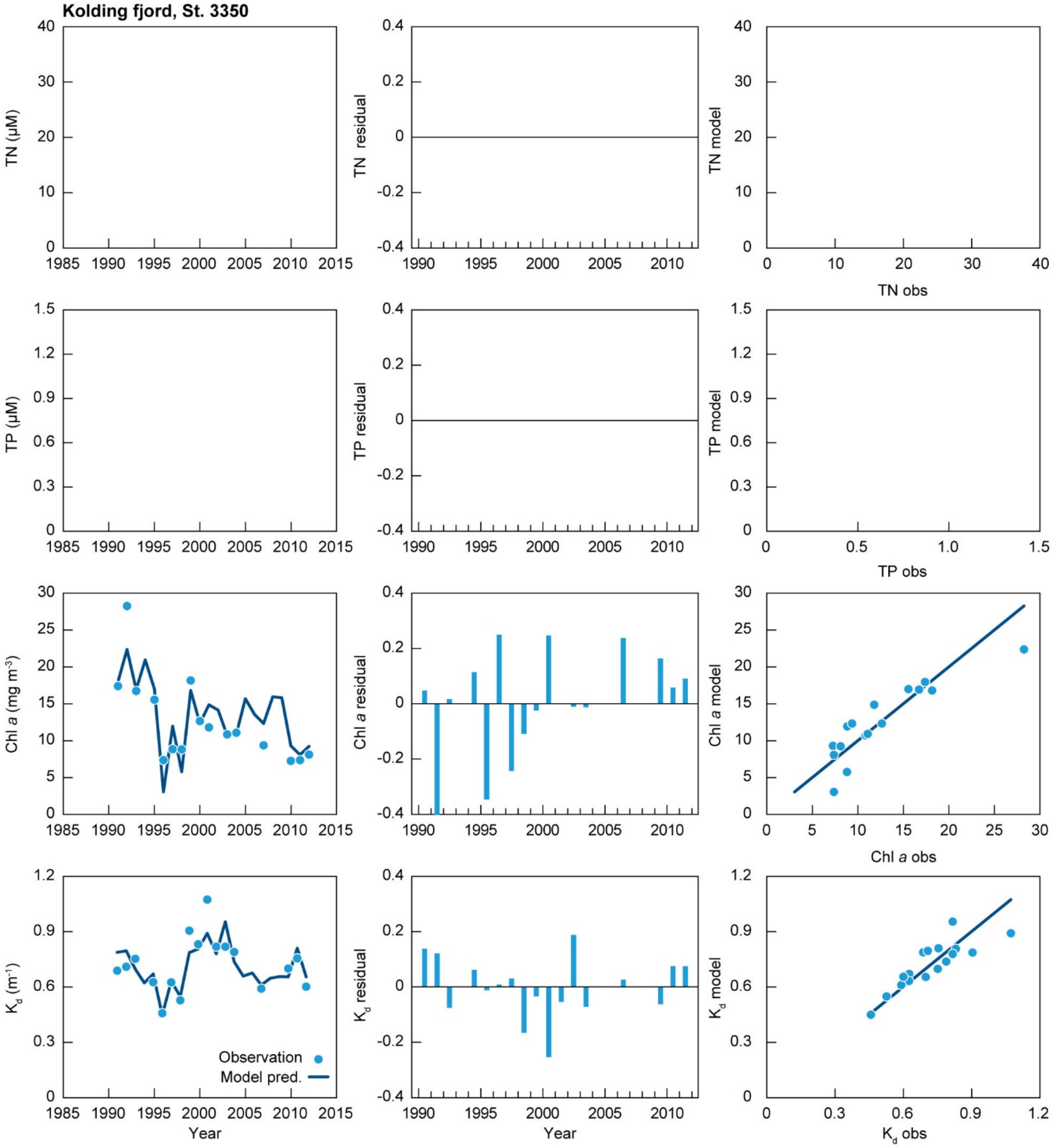
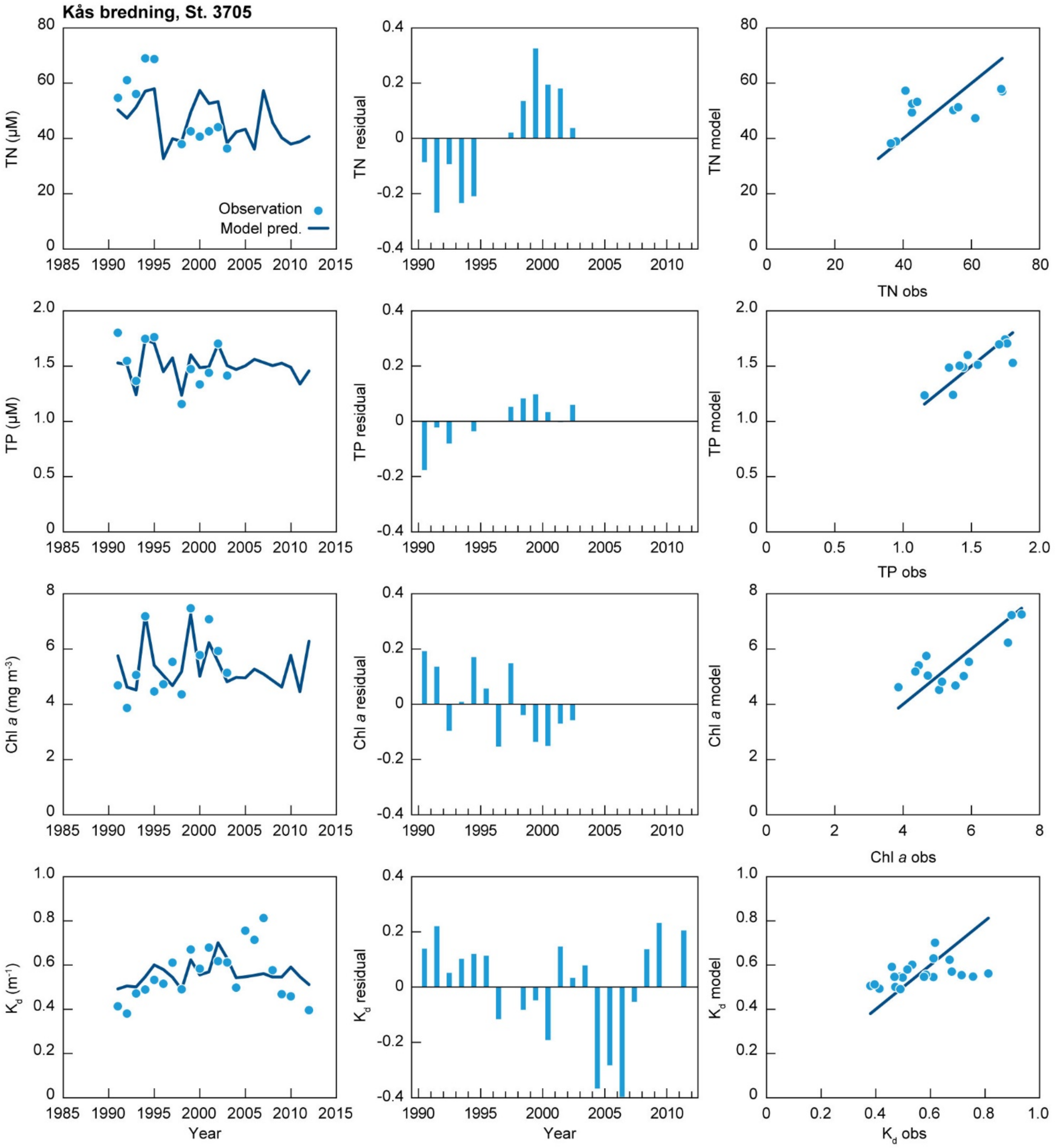
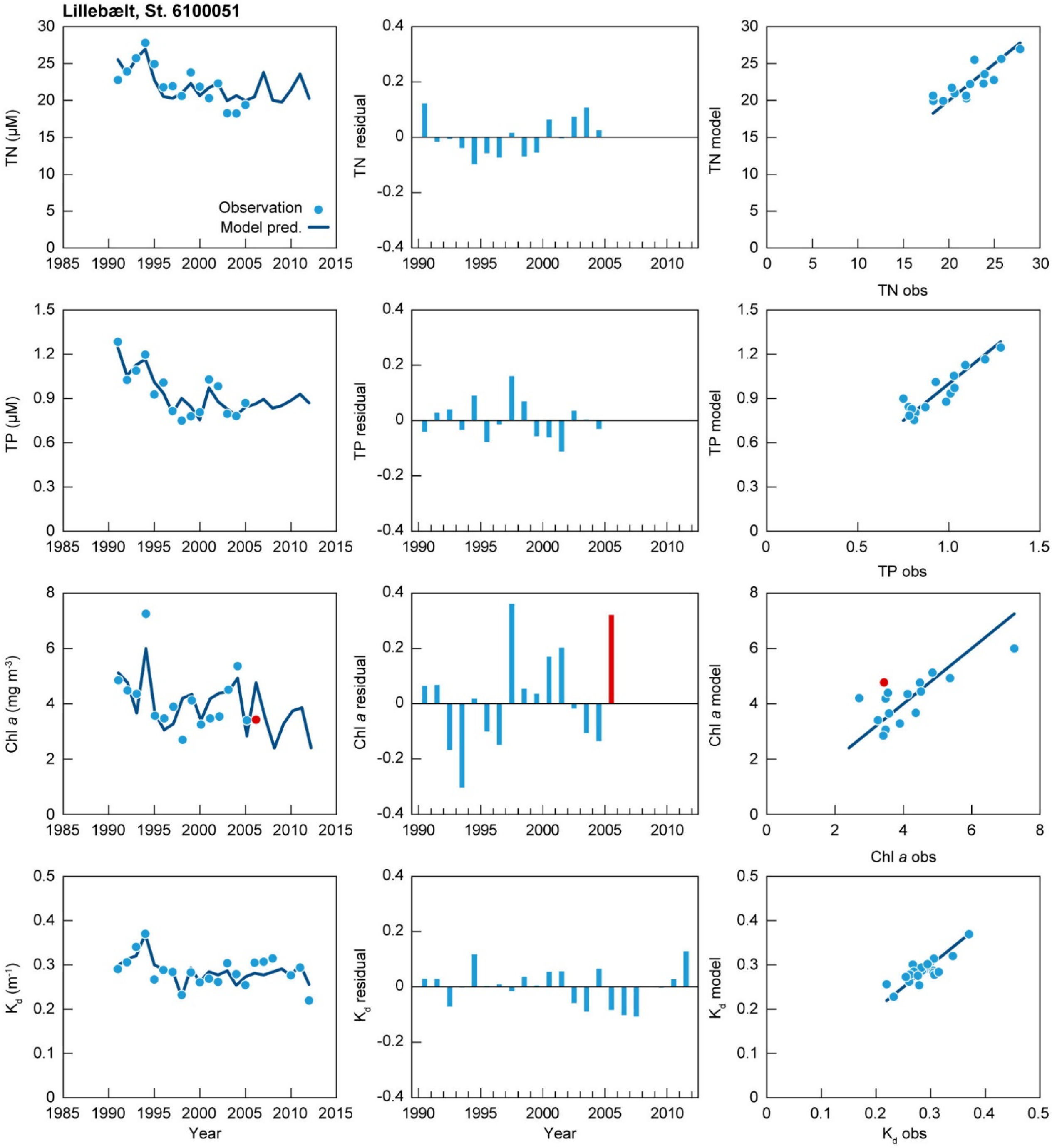

Kolding fjord, St. 3350
TN (µM)
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Observation
Model pred.
Year
Year

Figure B5.



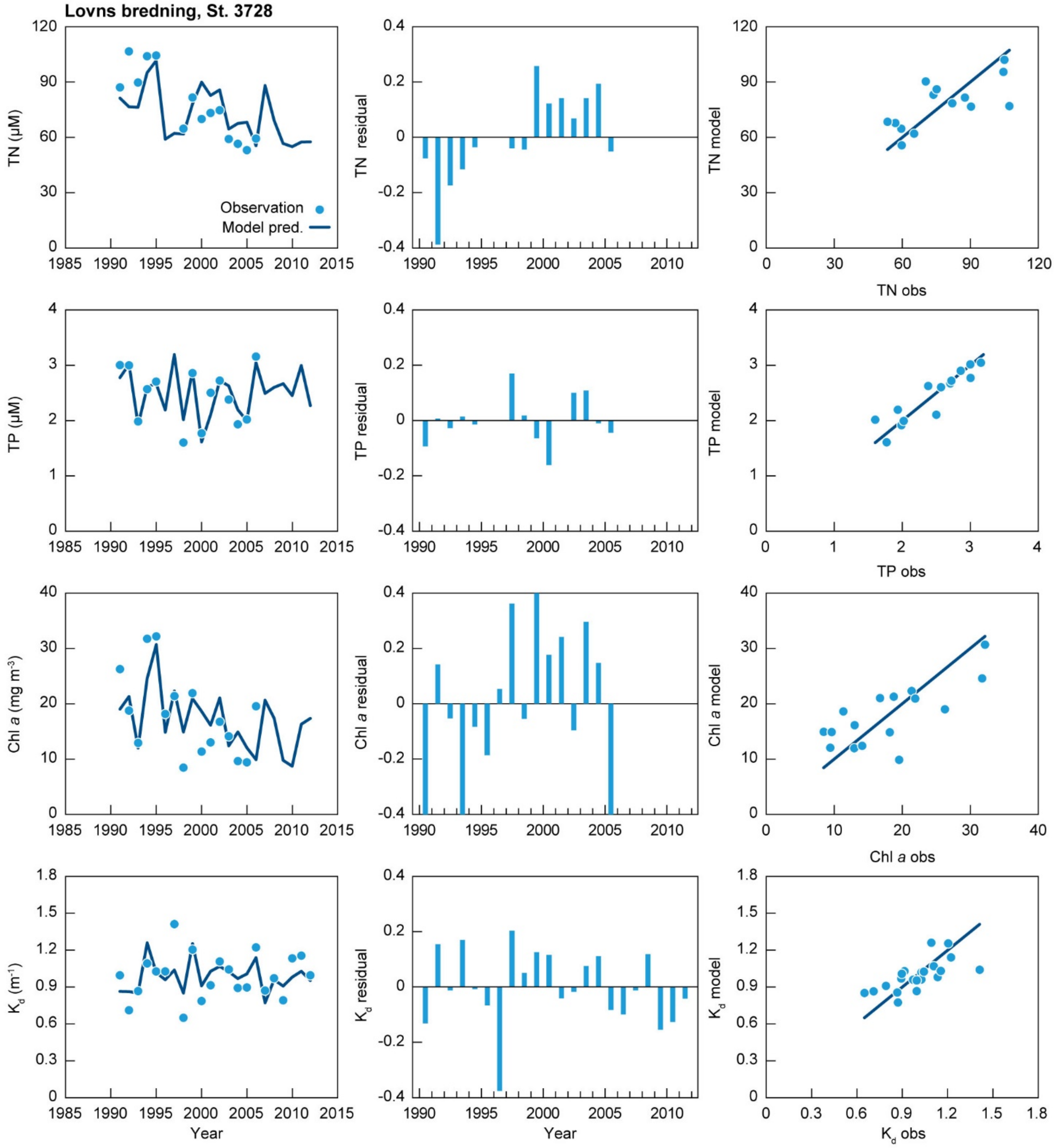
Kås bredning, St. 3705
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year
Year

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
Kd (m⁻¹)
Kd residual
Kd model
Kd 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.

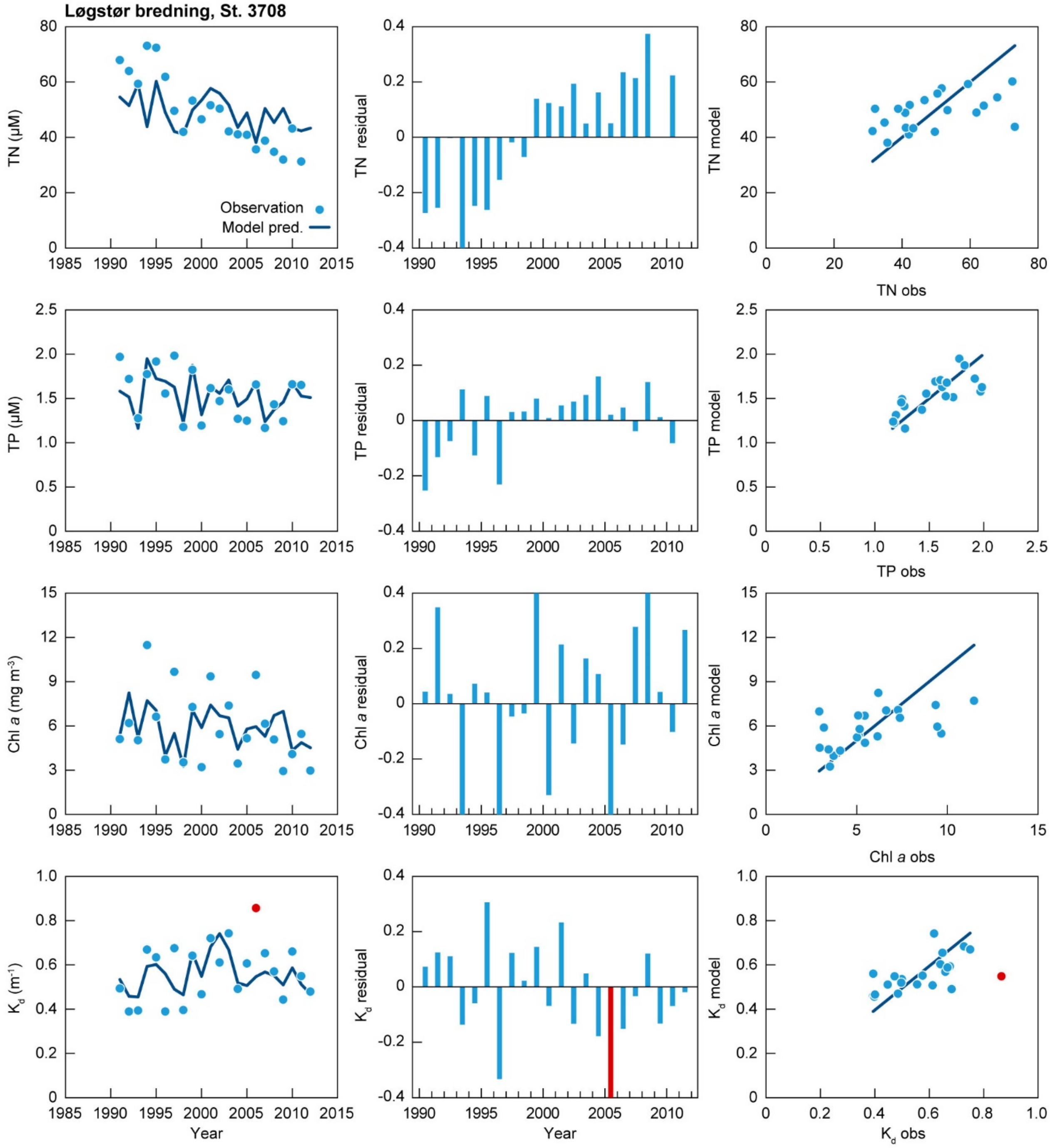
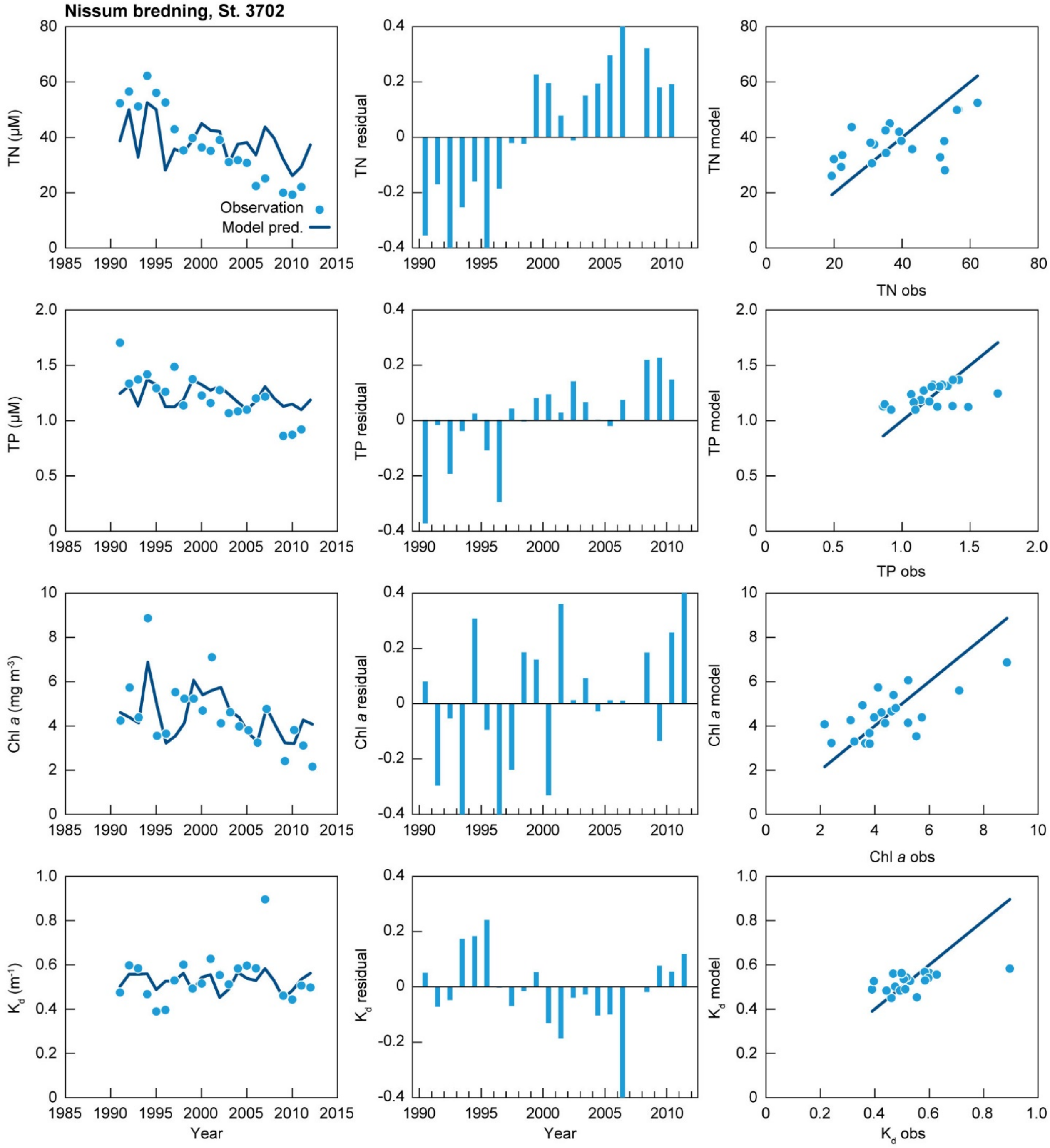

Løgstør bredning, St. 3708
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year

Figure B9



Nissum bredning, St. 3702
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

Figure B10.

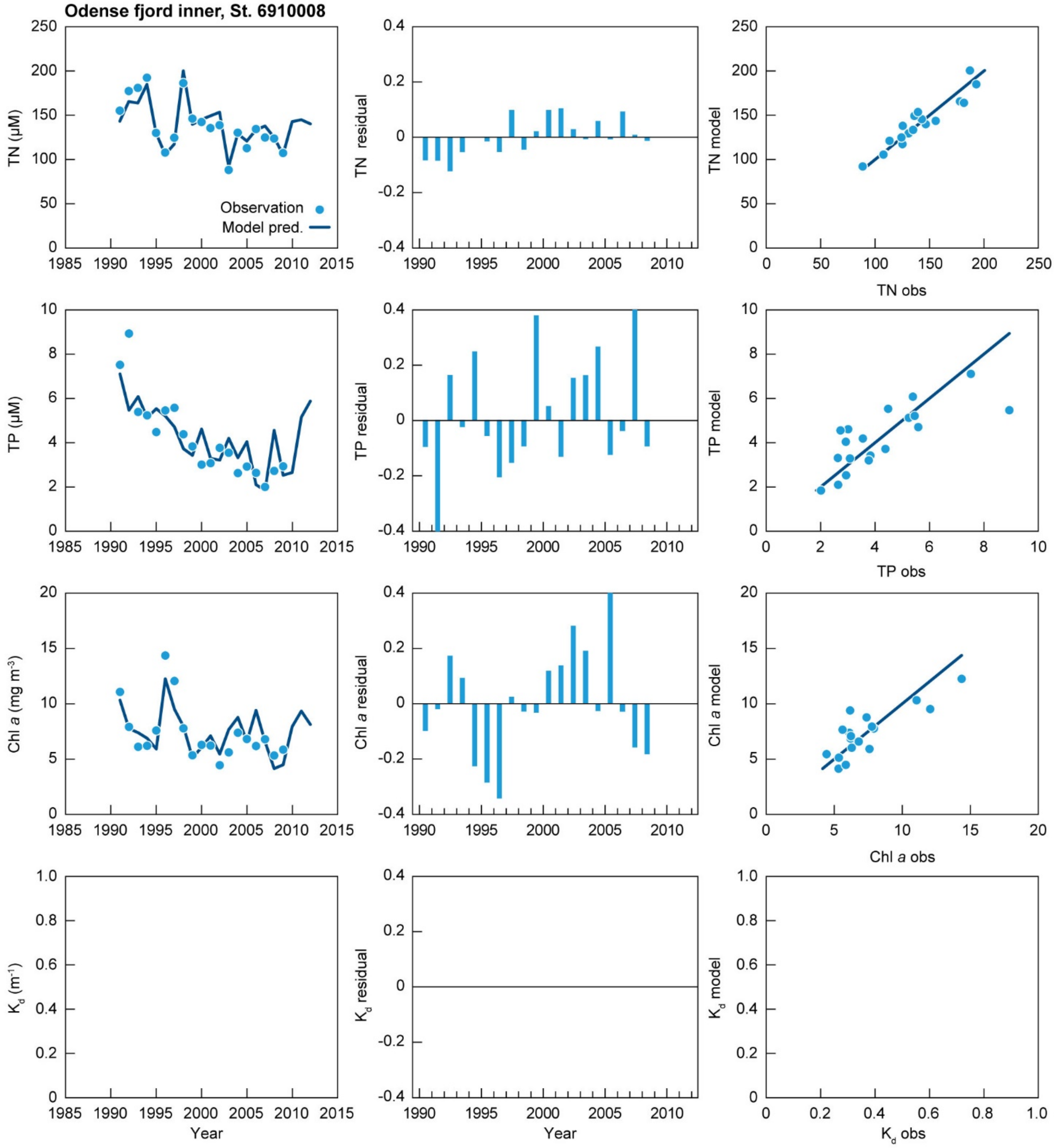

Odense fjord inner, St. 6910008
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year
Year

Figure B11.

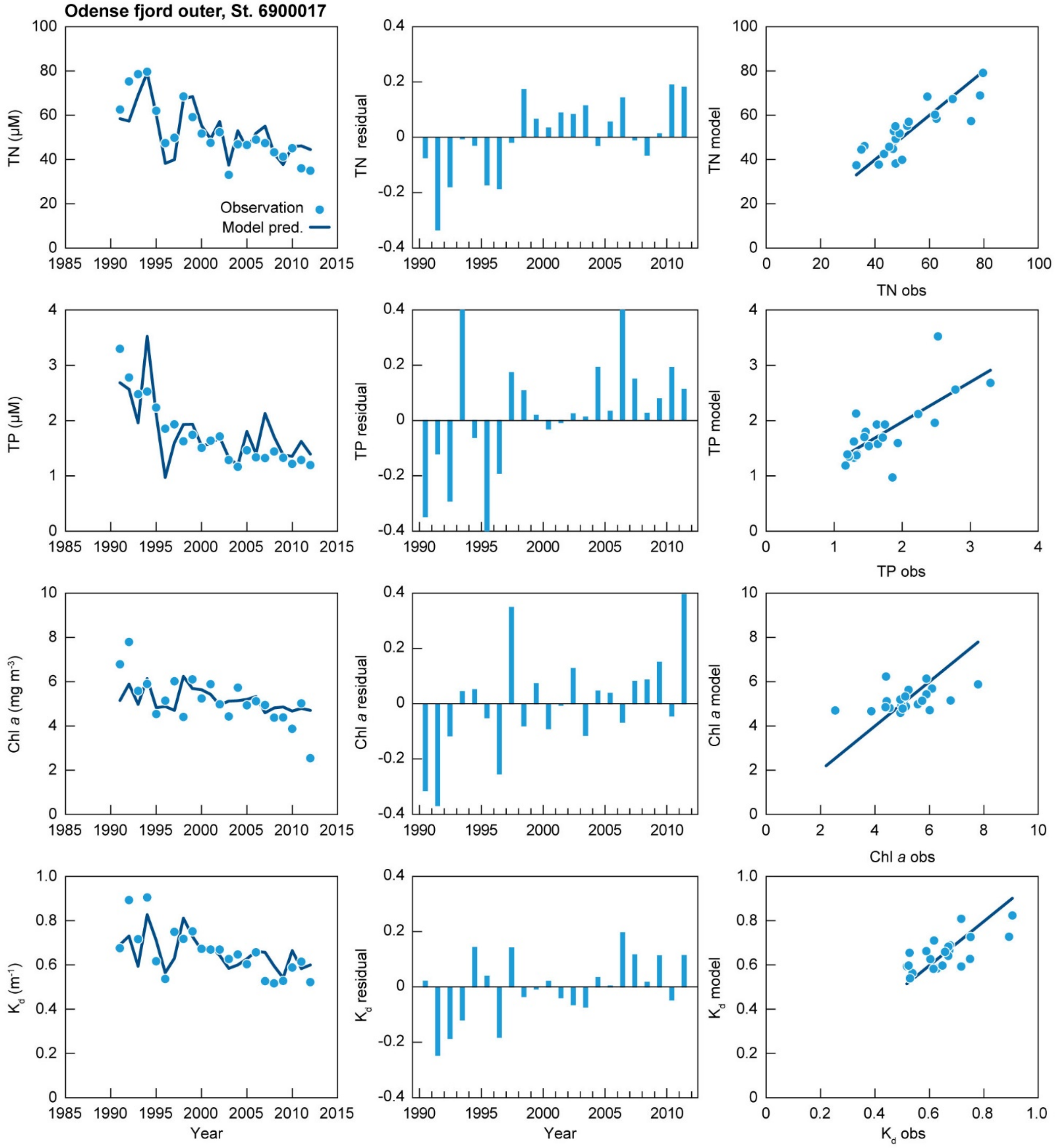
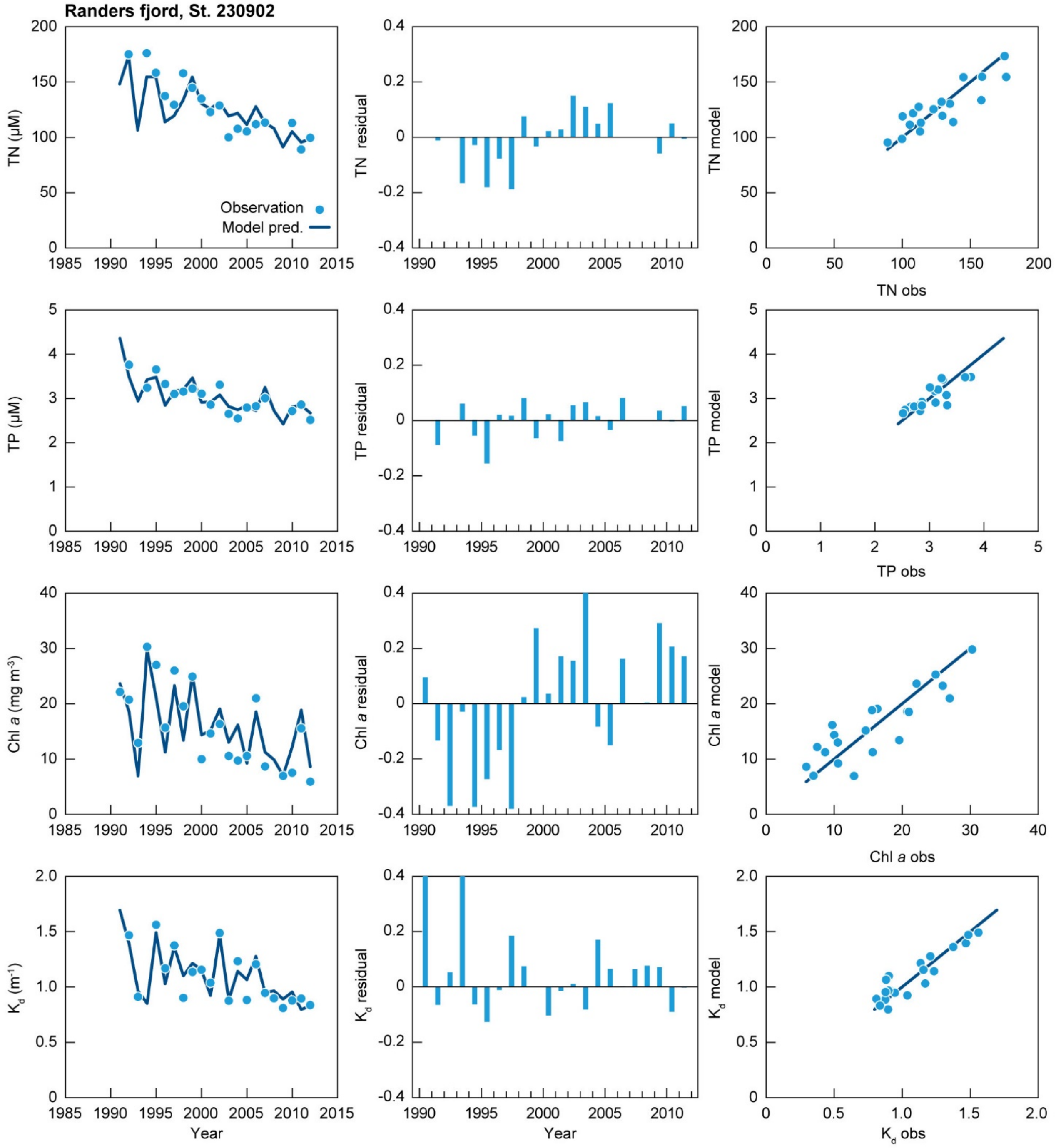

Odense fjord outer, St. 6900017
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year

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
Kd (m-1)
Kd residual
Kd model
Kd obs
Year
Year

Figure B13.

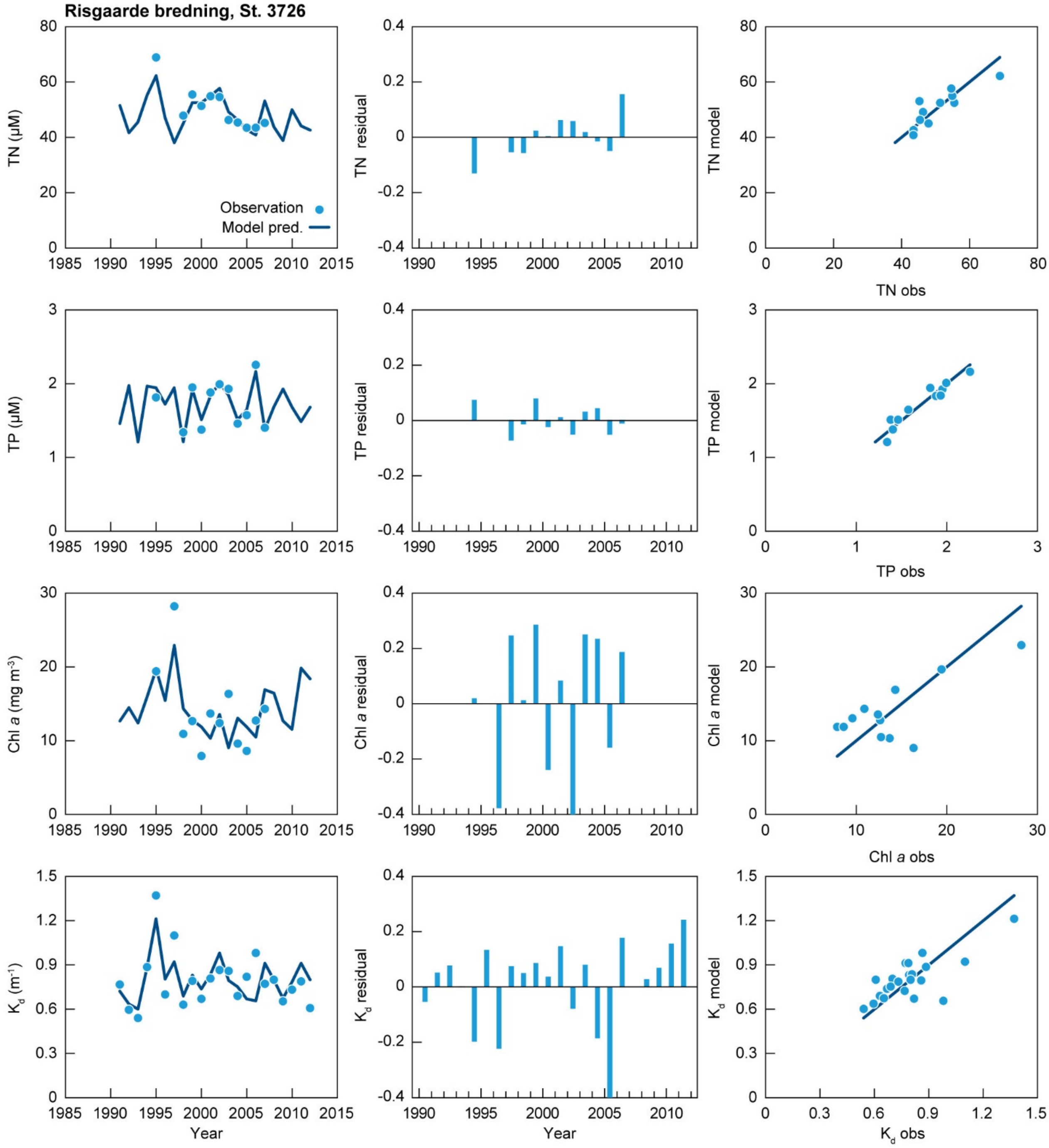
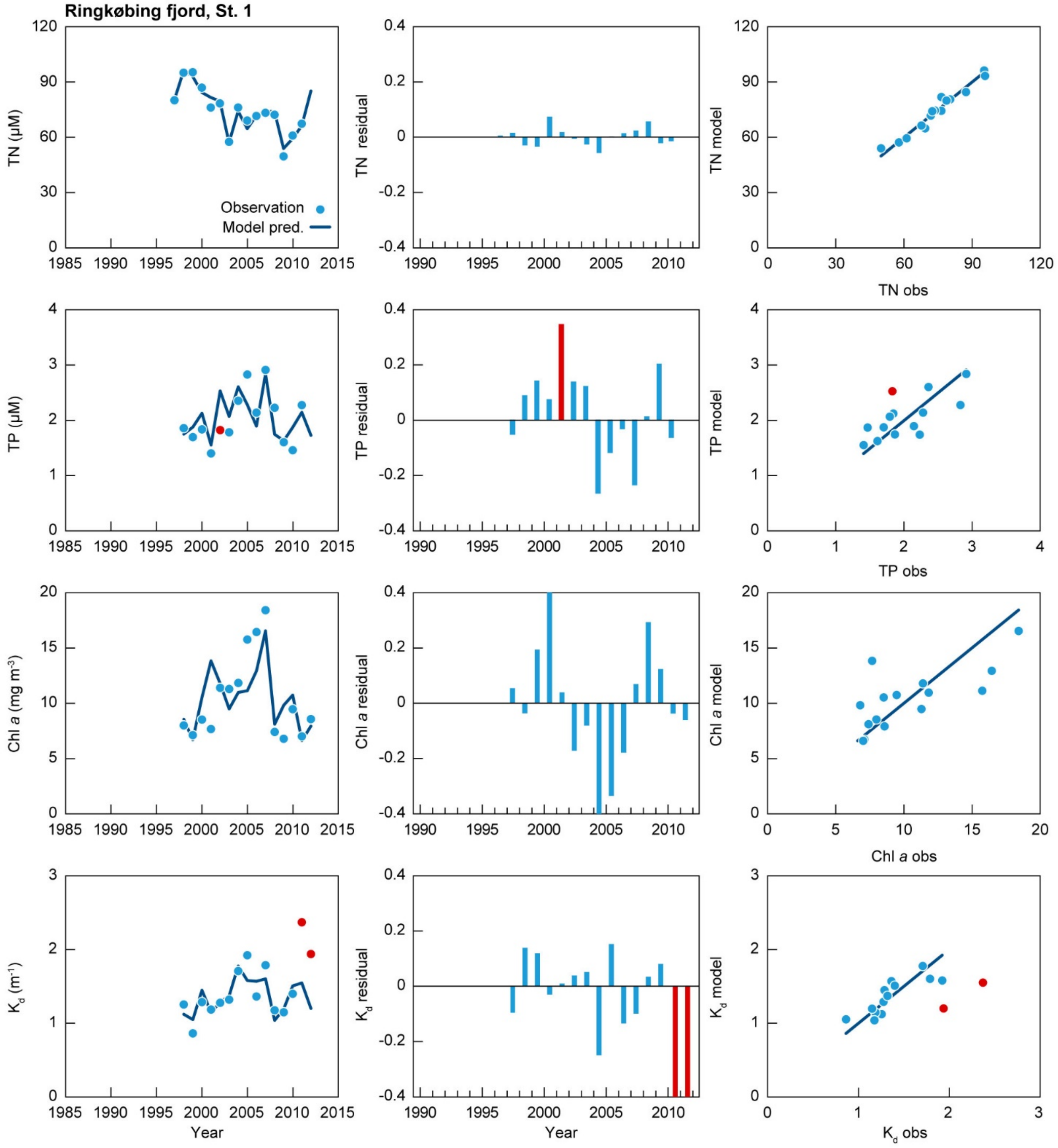

Risgaarde bredning, St. 3726
Observation
Model pred.
TN (µM)
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year

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.

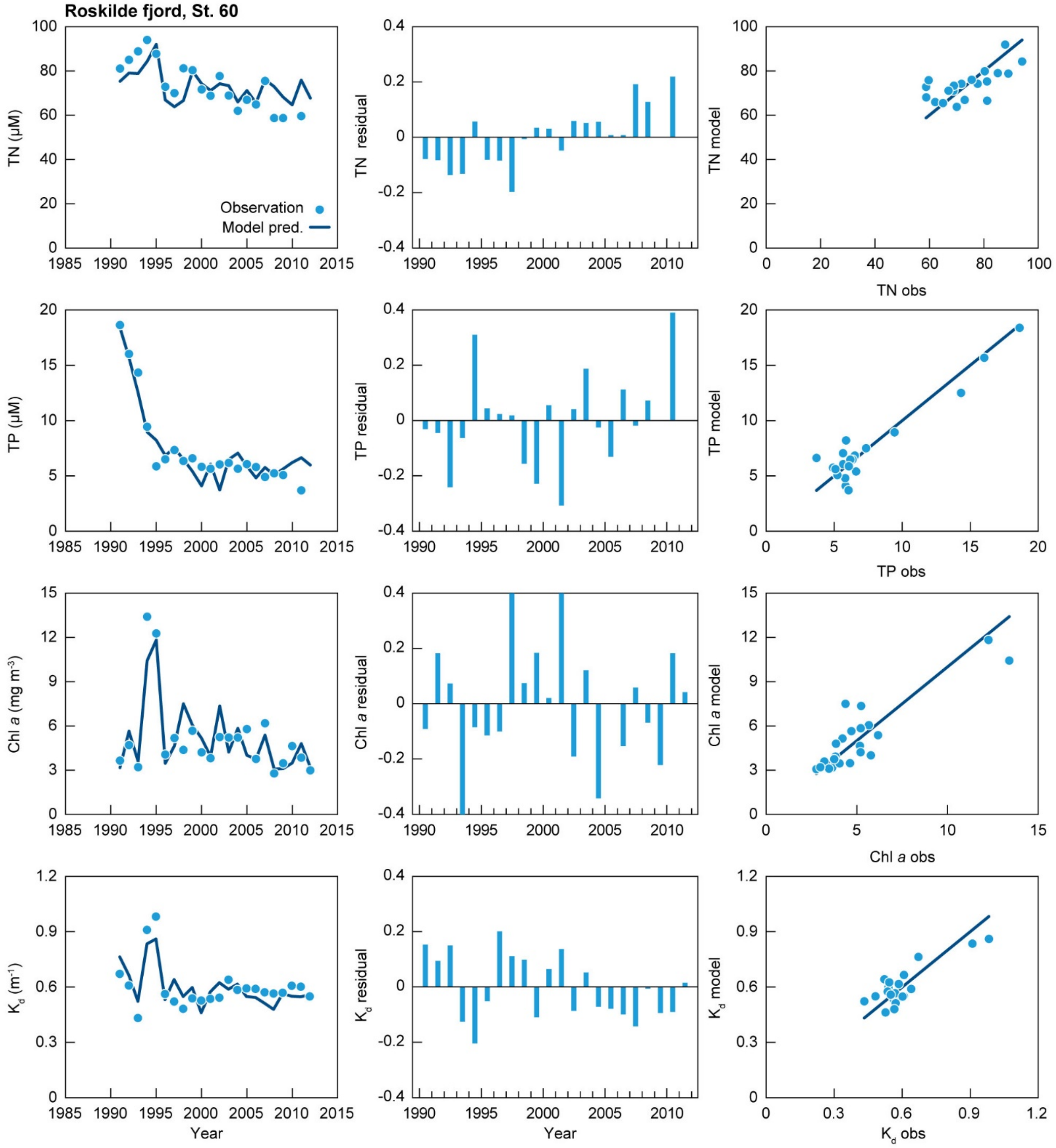

Roskilde fjord, St. 60
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
K_d (m⁻¹)
K_d residual
K_d model
K_d obs
Year

Figure B16.

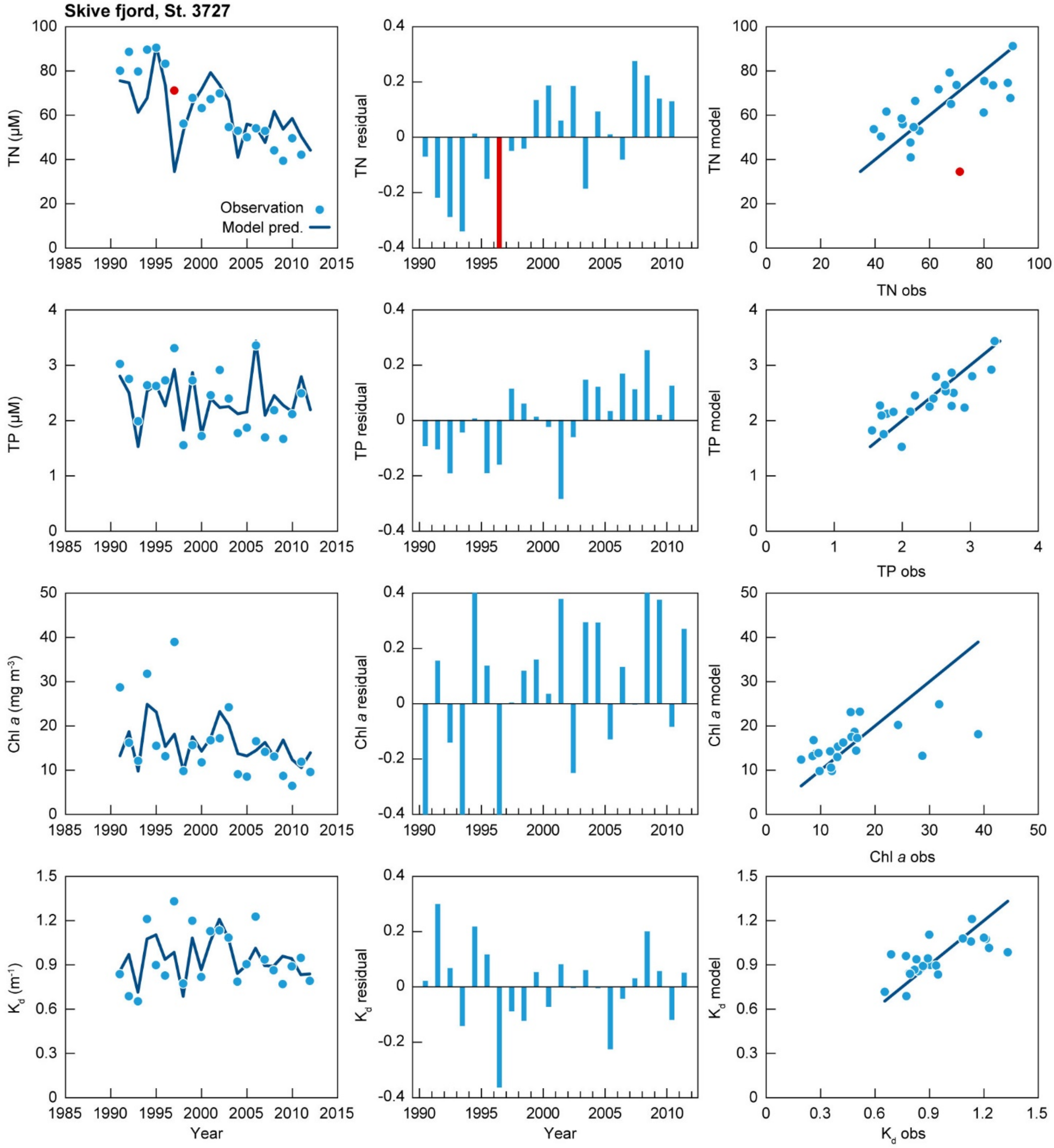
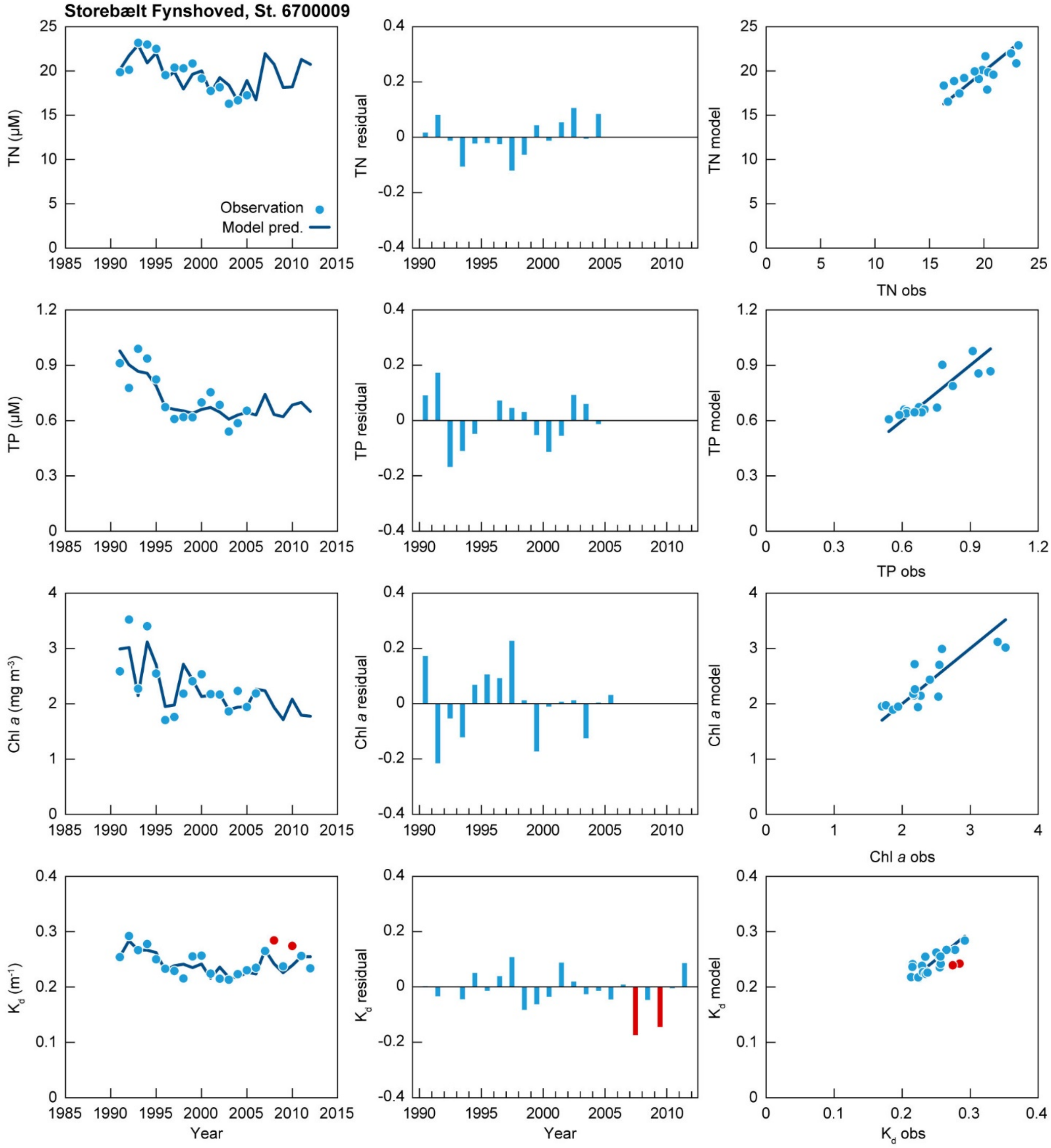

Skive fjord, St. 3727
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
Kd (m⁻¹)
Kd residual
Kd model
Kd obs
Year
Year

Figure B17.



Storebælt Fynshoved, St. 6700009
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

Figure B18.

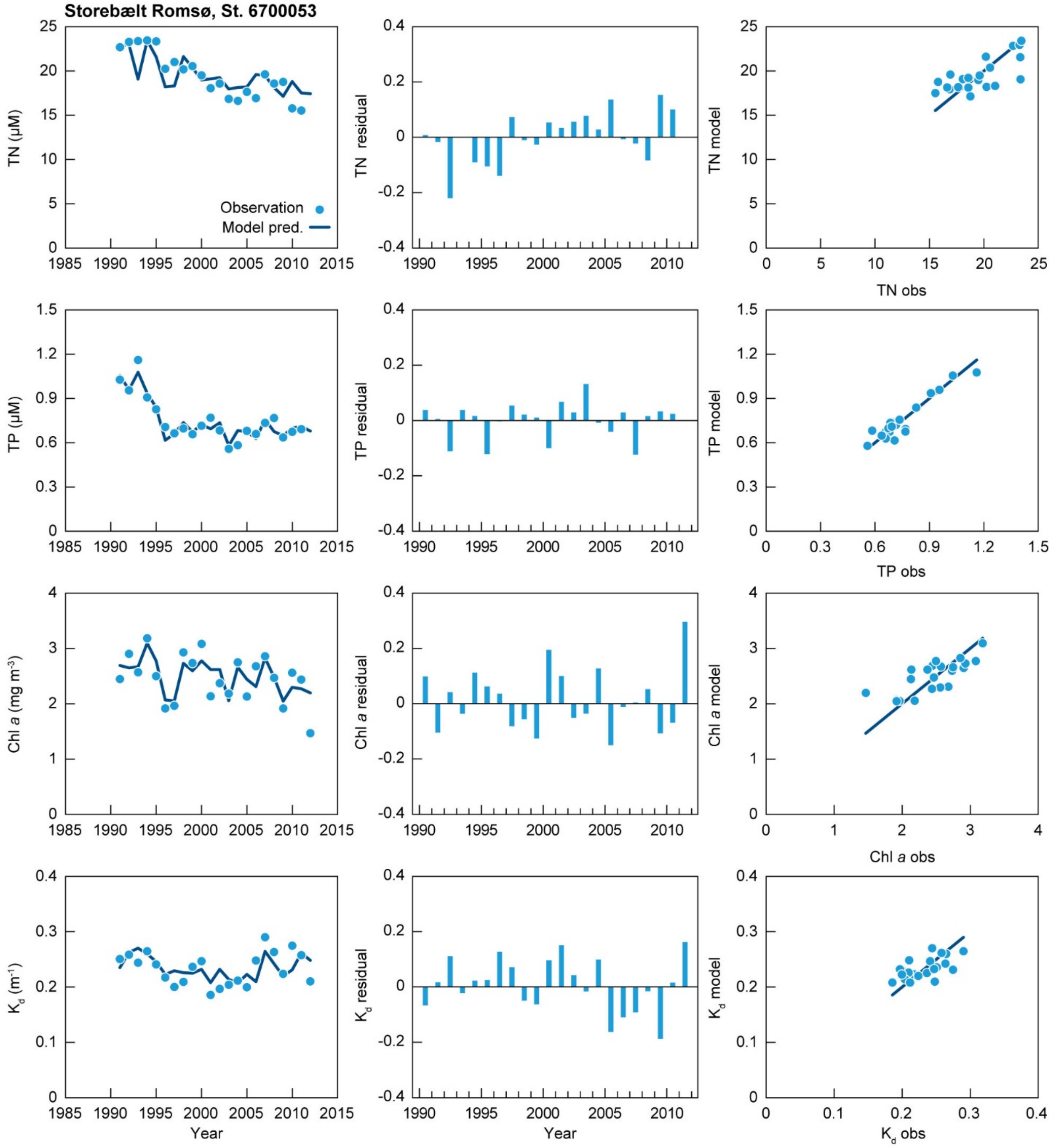
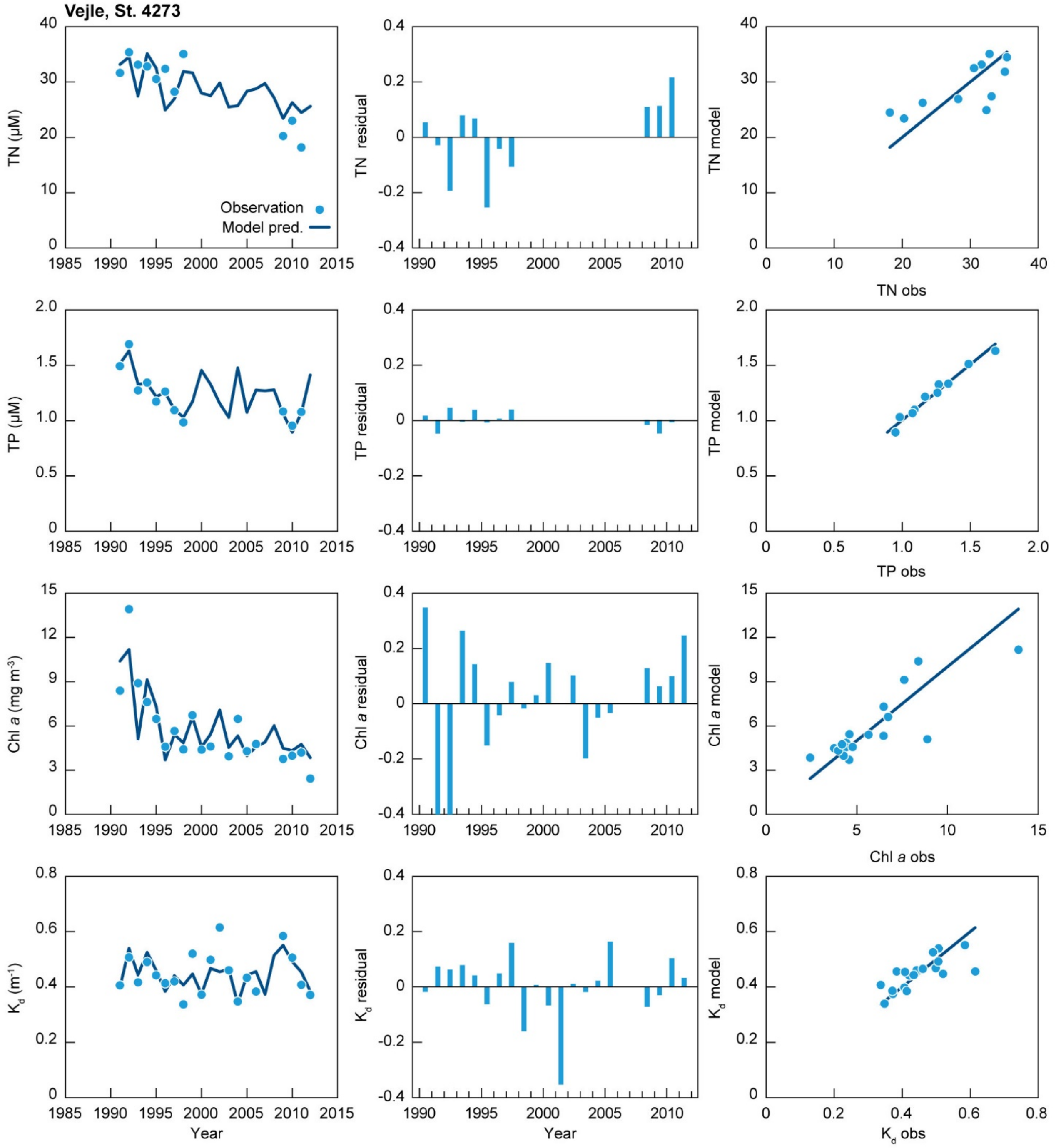
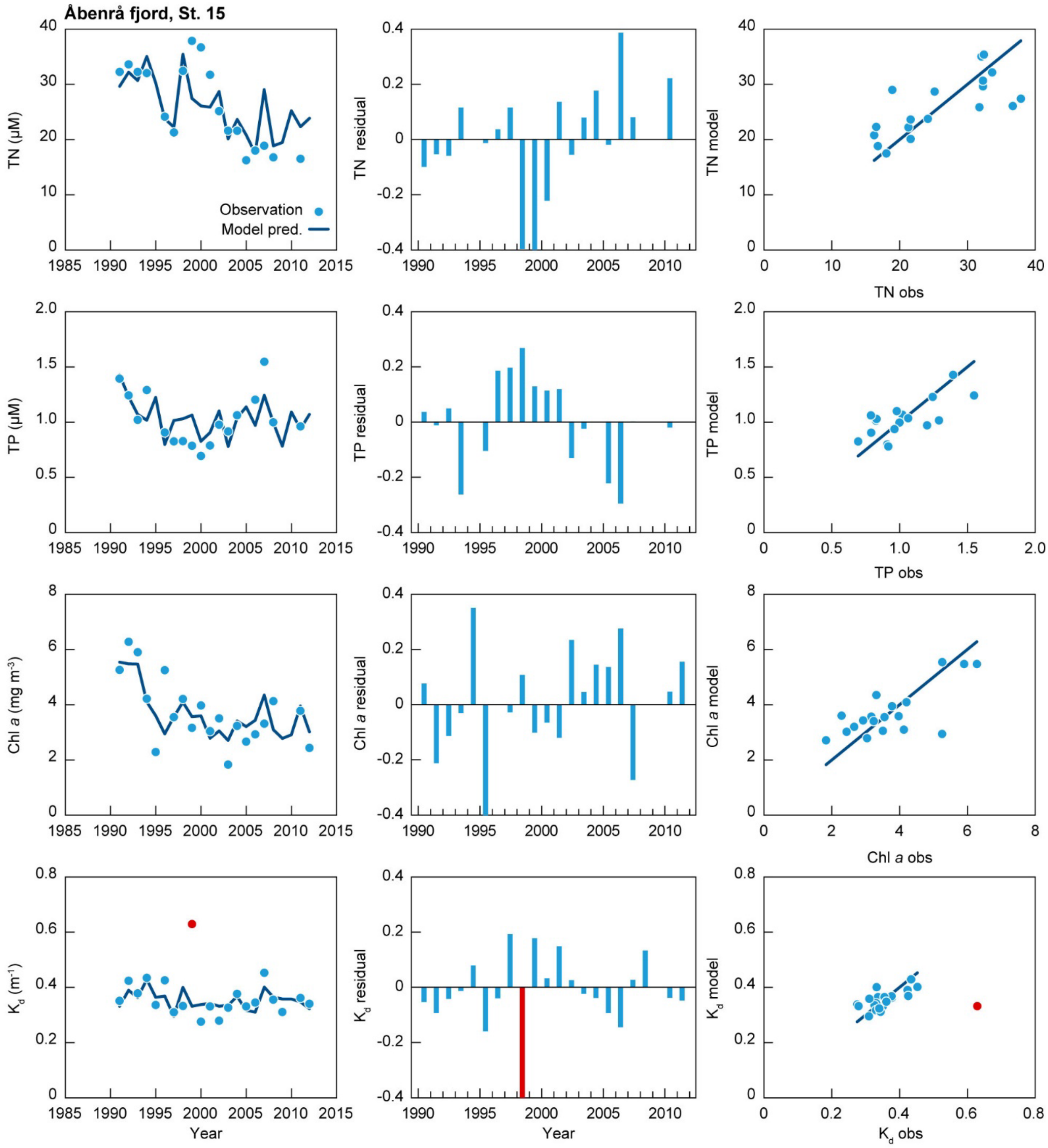

Storebælt Romsø, St. 6700053
Observation
Model pred.
TN (µM)
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m⁻³)
Chl a residual
Chl a model
Chl a obs
Kd (m⁻¹)
Kd residual
Kd model
Kd obs
Year
Year

Figure B19.



Vejle, St. 4273
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 B20.



Åbenrå fjord, St. 15
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
Year

Figure B21.

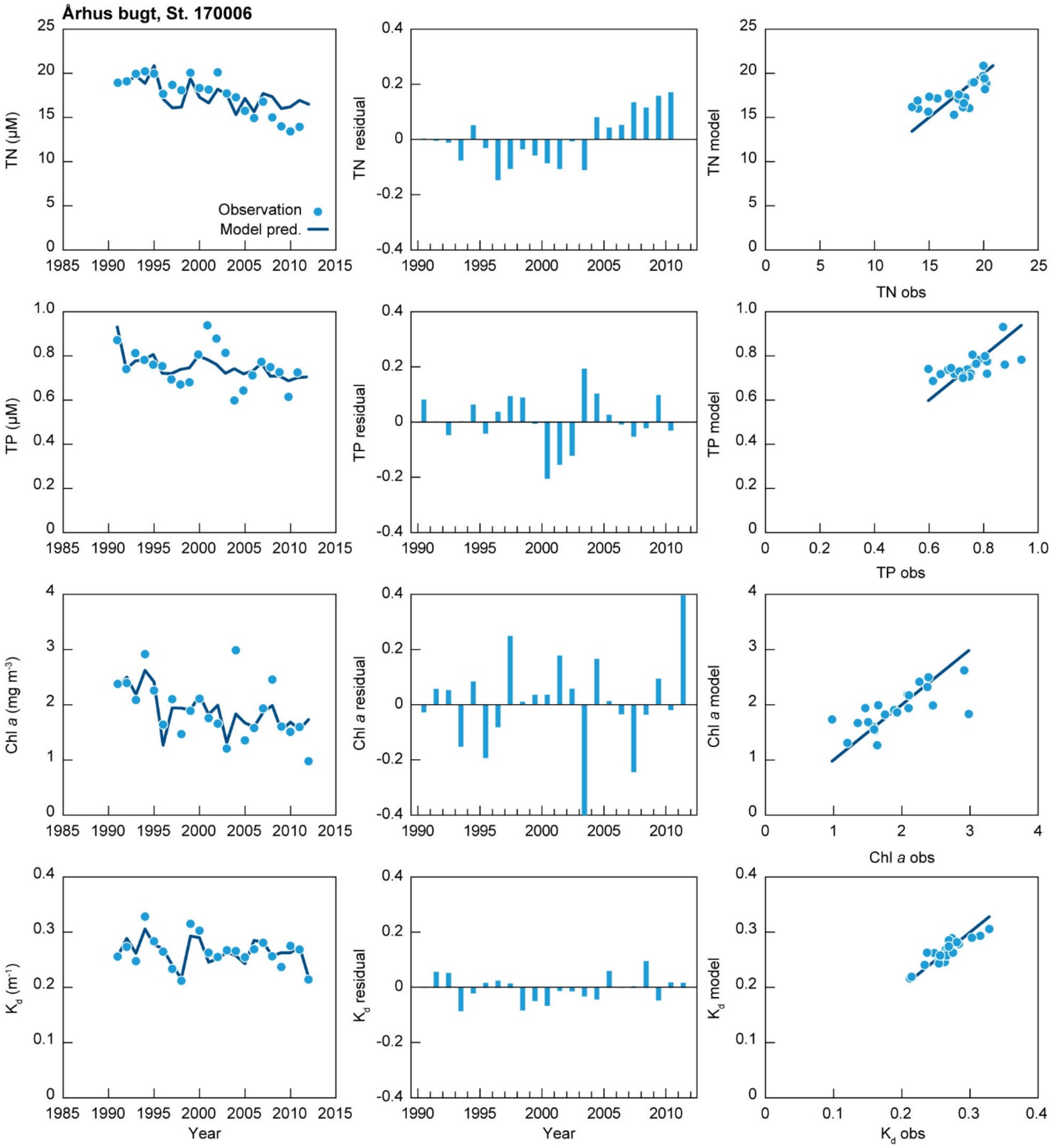

Århus bugt, St. 170006
TN (µM)
Observation
Model pred.
TN residual
TN model
TN obs
TP (µM)
TP residual
TP model
TP obs
Chl a (mg m-3)
Chl a residual
Chl a model
Chl a obs
Kd (m-1)
Kd residual
Kd model
Kd obs
Year
Year

Figure B22.

Appendix C – Mechanistic model evaluation

Evaluation of modelled nitrogen

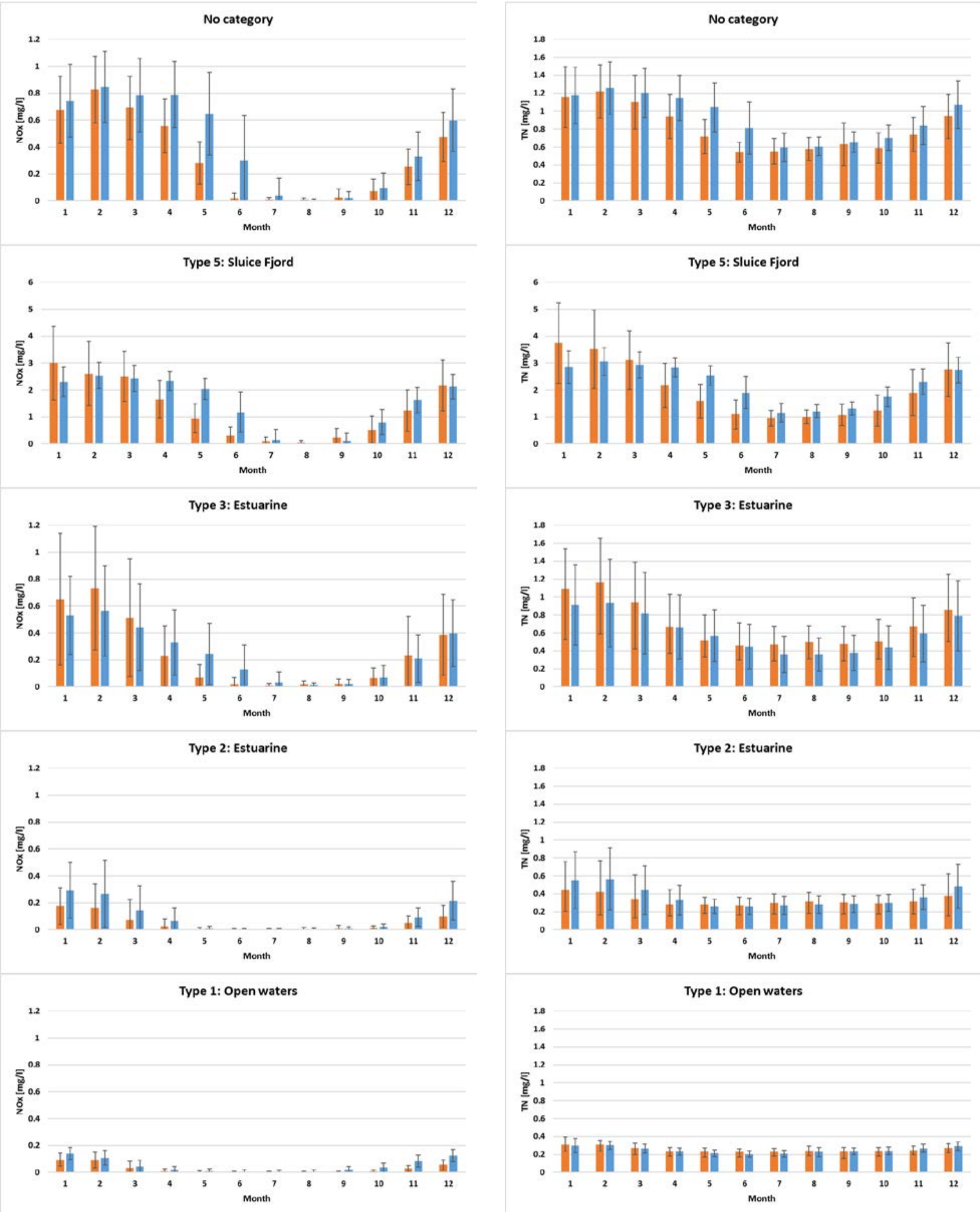


Figure C1. Average monthly measured (orange bars) and modelled (blue bars) concentrations of NO_x (left column) and TN (right column). Error bars represent one standard deviation. From top panel to bottom panel the different water body categories are included: No category, Type 5, Type 3, Type 2 and Type 1. Notice that Type 5 has different y-axis than the other panels.

Evaluation of modelled phosphorous

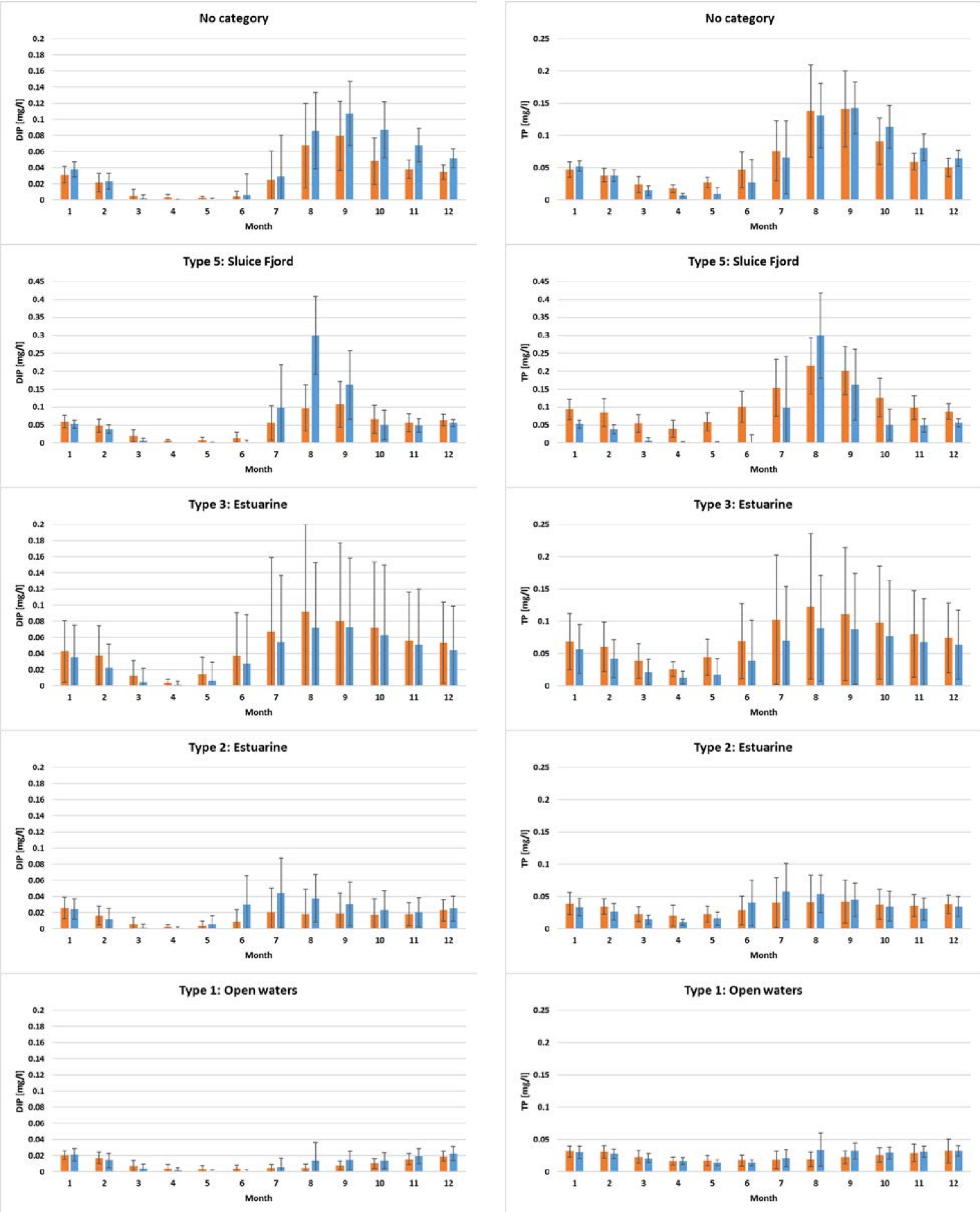


Figure C2. Average monthly measured (orange bars) and modelled (blue bars) concentrations of DIP (left column) and TP (right column). Error bars represent one standard deviation. From top panel to bottom panel the different water body categories are included: No category, Type 5, Type 3, Type 2 and Type 1. Notice that Type 5 has different y-axis than the other panels.

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