GUIDELINES FOR SELECTION OF APPROPRIATE MANAGEMENT MODEL

Technical Report from DCE – Danish Centre for Environment and Energy No. 198 2021

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Abstract: Ecosystem models are used to support the management and implementation of European directives (e.g. the Water Framework Directive (WFD)). Some of these models are based on sophisticated 3-D mechanistic marine ecosystem models, while others have a more simplistic setup. However, few attempts have been made to understand limitations, commonalities and differences between these models. In this report the main strengths and weaknesses of the models will be outlined and an evaluation given of how this influences the use of these model types.

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Contents

1 Preface

This report is the outcome of work package (WP) 6: Modelling synthesis as part of the project SeaStatus. In this report, a short comparison of the marine model setup used in Northern European countries (Denmark, Netherlands, Sweden) as part of the European Union WFD monitoring and reporting is presented. The general focus is on evaluating strengths and weaknesses of different mechanistic modelling setups. The statistical modelling approach used in Denmark is also shortly presented.

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2 Summary

Ecosystem models are used to support the management and implementation of European directives (e.g. the Water Framework Directive (WFD)). Some of these models are based on sophisticated 3–D mechanistic marine ecosystem models, while others have a more simplistic setup. However, few attempts have been made to understand limitations, commonalities and differences between these models. In this report the main strengths and weaknesses of the models will be outlined and an evaluation given of how this influences the use of these model types.

Choosing the right model for managing marine ecosystems is challenging and depends strongly on the nature of the issues to be addressed and on the available dataset. However, generally, the user should select a model that has the least amount of complexity necessary to address the problem and where data are available to support the model setup and verification. However, choosing a too simple model also implies a risk for management as they might miss important details, thereby limiting its use.

3 Introduction

The use of ecological models for environmental management began to increase rapidly in the early 1970's. At that time, almost all of the models in use were population or biogeochemical dynamic models (Jørgensen 2008). The models in the 1970's lacked key components as a description of spatial distributions, accounting for adaptable species and stochastic ecological processes etc. It was the need for including such parameters and processes that led to the development of the ecological models as we know them today (Jørgensen 2008). It is now widely recognised that ecological models are essential for conservation and management policies (Rombouts et al. 2013) and the use of ecological models is currently among the best approaches for understanding changes in the natural environment (Piroddi et al. 2015).

Ecosystem models with different complexity and setups have previously been developed and applied to manage European marine waters. Some of these models are based on sophisticated 3–D biogeochemical hydrodynamical marine ecosystem models, while others have a more simplistic setup. These models are used widely to support the management and implementation of European directives (e.g. the Water Framework Directive (WFD)). However, few attempts have been made to understand the conceptual limitations, commonalities and differences between these models. Such differences could make some models the preferred tool in certain circumstances and hamper their use in others.

In this synthesis, we present a short comparison of the marine model setup used in Northern European countries (Denmark, Netherlands, Sweden) as part of the European Union WFD monitoring and reporting. Here, the main conceptual strengths and weaknesses of the approaches will be outlined and a short evaluation given of how this influences the use of these model types.

4 Overview of different modelling approaches

In our comparison, we only consider 3D marine biogeochemical hydrodynamical and statistical models that have been used for describing the biological (e.g. flora and fauna) and physical-chemical quality (e.g. temperature, oxygenation and nutrient conditions) of coastal waters, while those used to describe the flows of, for example, specific pollutants (e.g. heavy metals) relevant for the WFD have not been included.

The Netherlands and Sweden have used models as a supplementary tool to identify problems but not as a direct decision support tool for river basin management planning (RBMP). This is contrary to the Danish modelling setup that combines mechanistic and statistical models to inform the RBMP for most water bodies. The reliance on models in the Danish RBMP is therefore more prevalent than in the other countries.

4.1 Denmark

Two methods are used in the Danish WFD monitoring and reporting, a mechanistic and statistical modelling approach.

The mechanistic model (MIKE 3 FM – ECOLAB) currently employed for Danish waters was developed by DHI A/S and includes models for Odense and Roskilde Fjord, the Limfjorden as well as the inner Danish waters (the Danish Straits and the Kattegat) and the North Sea. In the current setup, all models, except the North Sea model, contain detailed representations of the physical, chemical and biological pelagic and benthic components (Erichsen et al. 2017). The model is a 3-D model with variable depth resolution (at least 1 m vertical resolution but finer resolution in waters shallower than 10 m and in local estuary models), and it contains a hydrodynamic ("physical") model, which is forced by meteorological data (e.g. precipitation, temperature, wind) as well as input data such as freshwater discharges and saltwater exchanges across model boundaries (Table 1). Combined, this setup can simulate salinity, temperature, water currents and transport. In addition, wave models covering the Odense and Roskilde Fjord, the Limfjorden as well as the inner Danish waters (the Danish Straits and the Kattegat) are included. Results from the wave models are used in the biogeochemical models as part of overall seabed stress calculations influencing, for example, resuspension.

The chemical and biological pelagic components of the models include up to three phytoplankton groups (flagellates, diatoms and cyanobacteria), two zooplankton groups (micro and meso), inorganic nutrients (nitrogen, phosphorus and silicate), particulate organic matter, two fractions of dissolved organic matter, inorganic materials and dissolved oxygen. The benthic compartment includes two-layer sediment pools of organic matter (carbon, nitrogen and phosphorus) and inorganic matter in the sediment (carbon, nitrogen and phosphorus) and, where relevant, benthic vegetation (perennial macroalgae, opportunistic macroalgae, eelgrass biomass, eelgrass shoot density and benthic microalgae) (Table 2).

In the model, the following processes are described: phytoplankton assimilation; phytoplankton mortality; nitrogen fixation; zooplankton and zoobenthos grazing; zooplankton excretion of detritus, dissolved inorganic nitrogen and phosphorus; oxygen- and temperature-dependent mineralisation of detritus and dissolved organic matter, sediment mineralisation and uptake/release from benthic vegetation and sediment fluxes; and oxygen- and temperature-dependent nitrification and denitrification. The sediment module also contains descriptions of permanent burial of organic matter and includes hydrogen sulphide in sediment and water phase (Table 2).

The statistical model developed by Danish Centre for Environment and Energy (DCE, Aarhus University) is used to simulate four indicators: total nitrogen (TN), total phosphorous (TP), chlorophyll a (Chla) and light attenuation coefficient (Kd), which vary depending on eight other variables (freshwater, nitrogen and phosphorus discharges from Danish catchments, wind, sunlight, salinity, level of stratification defined as the mean of the Brunt-Väisälä buoyancy frequency, temperature) (Erichsen et al. 2017). These relationships are based on decadal data from 29 sites in Danish waters. The main principle of the statistical model is to select the suite of variables that best describes a given indicator. The variables are then tested against the indicators by introducing them to the model back in time. All data sets are divided such that one part is used for calibration and one part for validation.

Table 1. Showing the type and key features (resolution and model components) of the mechanistic models used in Denmark, the Netherlands and Sweden.

Country:	**Denmark**	**The Netherlands**	**Sweden**
Model type:	**Mechanistic**	**Mechanistic**	**Mechanistic**
Model name: **Feature:**	**MIKE 3 FM - ECOLAB**	**Delft3D-GEM**	**Swedish Coastal zone Model (SCM)**
User interface	Yes	Yes	Yes
Dimension	3-D	3-D	1-D
Vertical resolution	< 1 metre	meters	meters
Horizontal resolution	Kilometres	Kilometres	Kilometres
Temporal resolution	Hours	Hours	Hours
Pelagic model component	Yes	Yes	Yes
Benthic model component	Yes	Yes	Yes

4.2 The Netherlands

The modelling approach for coastal waters used by the Dutch Government is mainly based on the Delft3D-GEM model for the North Sea (Blauw et al. 2008; Los et al. 2008; Troost et al. 2014), originally developed at DELTARES (Table 1 and 2). The model is a hydrodynamic transport model that is forced by climatological data (precipitation, temperature, wind) as well as by factors such as tides, external freshwater and saltwater inputs.

The components included in the model are: nitrogen, phosphorus and oxygen dynamics as well as a simple representation of plankton dynamics typical for coastal waters. In the water column, the following state variables are used: up to five groups of zooplankton and zoobenthos, functional phytoplankton groups (diatoms, flagellates and others), detritus, nitrate, ammonium, phosphate, oxygen, benthic nitrogen and benthic phosphorus (Table 2). In the model, the following processes are described: phytoplankton assimilation; phytoplankton mortality; nitrogen fixation; zooplankton and zoobenthos grazing; zooplankton excretion of detritus, dissolved inorganic nitrogen and phosphorus; oxygen- and temperature-dependent mineralisation of detritus, benthic nitrogen and phosphorus; and oxygen- and temperature-dependent nitrification and denitrification. The sediment module also includes descriptions of permanent burial of organic matter and the model uses oxygen dynamics to model hydrogen sulphide levels.

4.3 Sweden

The model used in Swedish coastal waters is mainly based on the Swedish Coastal zone Model (SCM) that contains a 1-dimensional physical component (Probe) and a biogeochemical model (SCOBI) (Sahlberg 2009), both developed by the Swedish Meteorological and Hydrological Institute (SMHI) (Table 1).

The SCM model primarily describes water exchange and circulation / mixing. These exchanges are calibrated with the help of observational data obtained from 130 hydrographical stations in Swedish coastal waters. The exchanges are assumed to be mainly governed by climatological conditions (e.g. temperature and wind). The vertical resolution of the model is 0.5 m in the uppermost layers, 1 m within the 4-70 m depth interval and 2 m between 70-100 m depth. Below 100 m, the layer thickness increases to 5 m and to 10 m below 250 m (Table 1 and 2).

The biogeochemical components included in the model are: nitrogen, phosphorus and oxygen dynamics as well as a simple representation of plankton dynamics typical for the Baltic Sea. In the pelagic zone, the model can calculate the following variables: one zooplankton group, three functional phytoplankton groups (diatoms, flagellates and cyanobacteria), detritus, dissolved inorganic nitrogen and phosphorus, oxygen, benthic nitrogen and benthic phosphorus. In the model, the following processes are described: phytoplankton assimilation; phytoplankton mortality; nitrogen fixation; zooplankton grazing; zooplankton excretion of detritus, dissolved inorganic nitrogen and phosphorus; oxygen- and temperature-dependent mineralisation of detritus, benthic nitrogen and phosphorus; and oxygen- and temperature-dependent nitrification and denitrification. The sediment module also includes descriptions of permanent burial of organic matter and the model for hydrogen sulphide levels (Table 2).

In the model, mixing and advection of the biogeochemical variables are calculated by PROBE, while SCOBI calculates the biogeochemical process rates for the cycling of elements and the vertical transfer due to the sinking of phytoplankton and detritus.

To run the SCM, atmospheric (meteorological variables and deposition on nitrogen and phosphorus), land (land run-off and point sources, e.g. sewage treatment plants and industries) and also hydrographical / biogeochemical boundary conditions are needed.

Table 2. Brief overview of the physical, chemical and biological components of the 3D models described in this report.

Type	Variable	Model name:	Denmark Mike 3- ECOLAB	The Netherlands Delft3D-GEM	Sweden Swedish Coastal zone Model (SCM)
	Salinity		X	X	X
	Temperature		X	X	X
	Tides		X	X	X
	Wind		X	X	X
	Waves		X	X	X
	Currents		X	X	X
	Water transparency		X	X	
	Sediment transport		X	X	X
	Resuspension		X	X	X
Chemical	River nutrient loads		X	X	X
	Atmospheric inputs		X	X	X
	Inorganic nutrients		X	X	X
	Dissolved organic matter		X	As one pool	As one pool
	Particulate organic Matter		X		
	Oxygen		X	X	X
	Benthic chemistry		X	X	X
Biological	Chlorophyll a		X	X	X
	Phytoplankton		3 groups	4 groups	3 groups
	Primary production		Pelagic and benthic	X	X
	Zooplankton		2 groups	5 groups	1 group
	Benthic vegetation		X	X	
	Benthic fauna		X	X	X

5 Different modelling approaches

Marine ecosystem models are used as management tools to predict and understand ecosystem responses to pressures (e.g. eutrophication, fisheries, pollution) and evaluate environmental health, and the model findings can then be used to develop or update environmental legislation frameworks. In this comparison, we will not directly describe the statistical models used in Danish management as they have been described elsewhere (e.g.(Erichsen et al. 2017) and as none of the other countries use these models, a direct comparison is not possible.

One of the first and most vital tasks in modelling is to confirm that the model output reproduces real observed data as exactly as possible. As soon as the basic model structure is determined, this task primarily contains finding adequate parameter values. These parameters include, for instance, growth rates or remineralisation rates, which are essential for the description of the respective biogeochemical process.

In Denmark, the Netherlands and Sweden, models have been used for both deep and shallow coastal systems to describe hydrodynamics, water quality and ecosystem / food web dynamics. The 3-D and 1-D models are fairly easy to understand and interpret and they are all supported by simple user interfaces. The Danish, Dutch and Swedish models all make predictions both on how individual components and / or the whole ecosystem likely respond to changes in pressures (e.g. increased nutrient inputs). The three different biogeochemical models from the countries are similar in that they describe the dynamics of oxygen, dissolved nitrogen (N) and phosphorus (P) (both inorganic and organic N / P) and particulate organic matter consisting of phytoplankton (autotrophs), dead organic matter (detritus) and zooplankton (heterotrophs). The structural differences between the models lie in their resolution, parameterisation and the hydrographical setup (Table 1 and 2). The 1-D model used in Sweden has the advantage that it is less complex and faster to calibrate than the 3-D models used in Denmark and the Netherlands. Contrary to the 1-D model, it can only be applied to one specific place at a time, while the other models take a more complex physical environment into consideration due to their larger spatial domain. This difference may influence the relative model performance if all relevant processes are included and parameterised appropriately (Eilola et al. 2011). The resolution also varies between models; for example, some models have a coarser resolution of coastal waters, which could underestimate the biogeochemical processing in these waters, and do not account for small-scale physical processes, such as turbulent mixing, thus producing imprecise estimates of flow and transport patterns from these waters to adjacent waters (Eilola et al. 2011). Overall, the models used in Denmark, the Netherlands and Sweden seem to contain many of the same processes. This finding is in line with previous inter-comparison studies that showed an overall good comparison between model outputs from the North Sea (Radach & Moll 2006) and the Baltic Sea (Eilola et al. 2011) with field observations. Specifically for the Baltic Sea, the tested models obtained different results, but all were able to capture long-term variability in key variables over the period 1970-2005, and no model performed exceptionally better or worse than the others (Eilola et al. 2011).

The below outlined perspectives describe the main causes for model uncertainties; however, an in-depth analysis of the likely underlying processes causing differences between the models and observations is not given as this is beyond the scope of this report.

Model uncertainty can mainly be divided into four overall categories including the data used for forcing and calibration, the structural setup as well as parametric and boundary condition issues.

Commonly, there is a lack of sufficient data from the local ecosystem for which a given model is used for both the parameterisation, development and verification of model results; though in, for example, Danish waters a relatively good dataset is available for basic variables such as inorganic nutrients and chlorophyll. The environmental data used in a model in itself also include uncertainty due to, for example, measurement errors. Uncertainty in datasets can also occur if, for instance, sampling equipment changes over time and / or if differences in equipment efficiency are not calibrated and corrected for. Furthermore, the environmental characteristics also vary spatial and temporally, so a sufficient amount of quality data needs to be included to represent the system. Thus, natural variability can increase data error and thereby influence model performance.

Ecosystem models are generally very complex and contain many state variables that often are based on average literature values or biogeochemical process rates parameterised from laboratory experiments, which have their limitations (i.e. might not be appropriate in the ecosystem investigated or able to fully cover large / several marine areas). One example is trophic links in ecosystem models, which often build on simple growth and mortality relationships that might not encapsulate all interactions. Generally, there is insufficient process data available for many of the model variables, and this is therefore often seen as the weakest point in an ecosystem models (Fulton et al. 2003). A challenge is therefore to obtain good and homogenous datasets for use in model setup and calibration. One obvious way to advance model development is to target data collection towards modelling needs, which would help to reduce uncertainty and enable models to work in larger or more marine areas. In contrast, statistical models require a dataset with fewer variables, their results are easily verified and they are more appropriate for describing a worse or an average situation. On the other hand, statistical models have the disadvantage compared with mechanistic models that they are “black box” models and do not account for many of the processes and the feedback involved. Consequently, they are less reliable for extrapolation beyond the ranges in the dataset used for estimation.

A shortcoming of both mechanistic and statistical models is that they in some cases do not account for functional changes and adaptation in ecosystems. As an example, phytoplankton cells are able to reduce their cell size to adapt to low nutrient levels.

In ecosystem models, hydrodynamics may be problematic without sufficient calibration and validation as energy flows, physical processes and environmental conditions vary between ecosystems. But a hydrodynamic model may be applied to different ecosystems of the same type, provided that sufficient calibration and validation have been carried out.

The spatial and temporal boundaries in a model can also cause problems. A major difficulty is the exchange at the boundaries, where relevant output and input information from other time periods and / or other geographical areas must be included without disturbing the internal model setup. But also weather impacts can be a boundary issue, with large-scale climate systems impacting those at smaller regional scales. To address such climate impacts, many studies have used large-scale climate models and downscaled these to regional levels, which may add uncertainty to the overall model performance.

Model uncertainty is an important issue and should be reported, if possible. As examples, this would both give an estimate of the unknown parameters (empirical-Bayesian analysis) and an expression of uncertainty via a prior probability distribution (Bayesian analysis) (Allen & Somerfield 2009; Cressie et al. 2009; Link et al. 2012).

The chosen model structure may also contribute considerably to uncertainty as some model developers may be more familiar with upper trophic levels (e.g. fish) and thus tend to neglect the role of the lower trophic levels and vice-versa, which could lead to variations in model structure and performance (Essington 2007; Link et al. 2012).

6 Recommendations

Management of marine systems requires comprehensive and robust models to produce realistic scenarios that describe the implications and consequences of policy actions. However, when choosing the appropriate modelling tool, the uncertainties, inclusion and emphasis on certain key processes, mathematics, computational needs, spatial and temporal scales as well as underlying assumptions need to be considered. Also, all models have uncertainties due to limitations in forcing and process descriptions as well as the general poor coverage of field measurements in both time and space. Therefore, different models will generally show varying performance when focusing on particular parameters in different environments, and no model is perfect in all aspects. The choice of the model frameworks will often have to be addressed through expert judgment, with the choice of one modelling tool over others being a trade-off with the optimal solution highly depending upon the issue or the question being addressed (Fulton et al. 2003). Also, models are often described as suitable to offer insights on how characteristics in a given ecosystem might respond to a specific set of conditions and perturbations (Hill et al. 2007) and not to give 100% accurate results.

An important step in the use of models is skill assessment, where the model's ability to capture trends in independent dataset, for instance from different areas or time periods, is tested. One example could be using remote sensing chlorophyll a data for model setup and in situ data for skill assessment. But many times no such independent data are available, which makes a comprehensive and objective assessment of the model performance challenging. In such cases, some of the existing data should be withheld and used solely for the purpose of assessment or validation (e.g. simple statistical analysis) (Allen & Somerfield 2009; Stow et al. 2009).

The choice of an ecosystem model depends strongly on the nature of the issues to be addressed and on the available dataset. Generally, the user should select a model that has the least amount of the necessary complexity but is complex enough to solve the problem and where data are available to support the model setup and verification. However, this is clearly a balance as a very simple model may be easy to understand and quick to run, but it can also miss too much detail, thereby limiting its use. The challenge is therefore to find the best balance that reduces complexity but gives valid and robust prediction (Fulton et al. 2003; Hannah et al. 2010). Ideally, the use of multiple models should be tested to see if similar predictions and outcomes are reached (Fulton et al. 2003). As an example in Danish management, two independent models are used (mechanistic and statistical models), which complement each other, and this gives confidence in the obtained results.

Lastly, it should be remembered that despite the fact that ecosystem models do not always describe accurately the processes of the natural environment, they are still useful in management when all limitations are acknowledged. They provide a valuable ecosystem overview of the problem of interest and they give initial ideas on how ecosystem functioning is affected by anthropogenic or natural factors, and finally they can also be used to indicate current knowledge gaps.

7 References

Allen JI, Somerfield PJ (2009) A multivariate approach to model skill assessment. Journal of Marine Systems 76(1-2): 83-94

Blauw AN, Los HFJ, Bokhorst M, Erftemeijer PLA (2008) GEM: a generic ecological model for estuaries and coastal waters. Hydrobiologia 618(1): 175-198

Cressie N, Calder CA, Clark JS, Ver Hoef JM, Wikle C.K. (2009) Accounting for uncertainty in ecological analysis: the strengths and limitations of hierarchical statistical modelling. . Ecological Applications 19: 553–570

Eilola K, Gustafsson BG, Kuznetsov I, Meier HEM, Neumann T, Savchuk OP (2011) Evaluation of biogeochemical cycles in an ensemble of three state-of-the-art numerical models of the Baltic Sea. Journal of Marine Systems 88(2): 267-284

Erichsen AC, Timmermann K, Christensen JPA, Kaas H, Markager SS, Møhlenberg F (2017) Development of models and methods to support the Danish River Basin Management Plans. Scientific documentation. . In: Scientific reports. p 191

Essington TE (2007) Evaluating the sensitivity of a trophic mass-balance model (Ecopath) to imprecise data inputs. Canadian Journal of Fisheries and Aquatic Sciences 64: 628–637

Fulton EA, Smith ADM, Hjohnson CR (2003) Effect of complexity on marine ecosystem models. Marine ecology progress series 253: 1-16

Hannah C, Vezina A, John MS (2010) The case for marine ecosystem models of intermediate complexity. Progress in Oceanography 84(1-2): 121-128

Hill SL, Watters GM, Punt AE, McAllister MK, Quéré CL, Turner J (2007) Model uncertainty in the ecosystem approach to fisheries. Fish and Fisheries 8: 315-336

Jørgensen SE (2008) Overview of the model types available for development of ecological models. Ecological modelling 215: 3-9

Link JS, Ihde TF, Harvey CJ, Gaichas SK, Field JC, Brodziak JKT, Townsend HM, Peterman RM (2012) Dealing with uncertainty in ecosystem models: The paradox of use for living marine resource management. Progress in Oceanography 102: 102-114

Los FJ, Villars MT, Van der Tol MWM (2008) A 3-dimensional primary production model (BLOOM/GEM) and its applications to the (southern) North Sea (coupled physical–chemical–ecological model). Journal of Marine Systems 74(1-2): 259-294

Piroddi C, Teixeira H, Lynam CP, Smith C, Alvarez MC, Mazik K, Andonegi E, Churilova T, Tedesco L, Chifflet M, Chust G, Galparsoro I, Garcia AC, Kämäri M, Kryvenko O, Lassalle G, Neville S, Niquil N, Papadopoulou N, Rossberg AG, Suslin V, Uyarra MC (2015) Using ecological models to assess ecosystem status in support of the European Marine Strategy Framework Directive. Ecological Indicators 58: 175-191

Radach G, Moll A (2006) Review of three-dimensional ecological modelling related to the North Sea shelf system. Part II: Model validation and data needs. Oceanography and Marine Biology - Annual Review 44: 1–60

Rombouts I, Beaugrand G, Fizzala X, Gaill F, Greenstreet SPR, Lamare S, Le Loc'h F, McQuatters-Gollop A, Mialet B, Niquil N, Percelay J, Renaud F, Rossberg AG, Féral JP (2013) Food web indicators under the Marine Strategy Framework Directive: From complexity to simplicity? Ecological Indicators 29: 246-254

Sahlberg J (2009) The Coastal Zone Model. In: SMHI Repoerts Oceanography No.98.

Stow CA, Jolliff J, McGillicuddy DJ, Jr., Doney SC, Allen JI, Friedrichs MA, Rose KA, Wallhead P (2009) Skill Assessment for Coupled Biological/Physical Models of Marine Systems. J Mar Syst 76(1-2): 4-15

Troost TA, de Kluijver A, Los FJ (2014) Evaluation of eutrophication variables and thresholds in the Dutch North Sea in a historical context — A model analysis. Journal of Marine Systems 134: 45-56

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