



Application of a soil-to-sea modelling framework for coastal eutrophication management

Joao G. Ferreira^{a,b,*}, Leonard Bernard-Jannin^a, Alhambra Cubillo^a,
Joao Lencart e Silva^a, Gerhardus P.J. Diedericks^c, Morten Holtegaard Nielsen^d,
Flemming Gertz^e, Lene Moth^f, Helle Borum^g

^a Longline Environment Ltd, 63, St. Mary Axe, London, W1G 8TB, United Kingdom

^b DCEA, Faculdade de Ciências e Tecnologia, Universidade Nova de Lisboa (NOVA), Quinta da Torre, Monte de Caparica, 2829-516, Portugal

^c Applied Mathematics Division, Department of Mathematical Sciences, Stellenbosch University, Stellenbosch, South Africa

^d Marine Science & Consulting ApS, Peder Lykkes Vej 8, 4. th., Copenhagen S, DK-2300, Denmark

^e SEGES Innovation P/S, Agro Food Park 15, Aarhus N, 8200, Denmark

^f Ringkøbing Municipality, Smed Sørensens Vej 1, Ringkøbing, 6950, Denmark

^g Vestjysk, Herningvej 3-5, Ringkøbing, 6950, Denmark

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ABSTRACT

A soil-to-sea modelling framework was developed for end-to-end simulation of eutrophication in coastal systems. This holistic framework was tested in Ringkøbing Fjord, a 300 km² Danish coastal lagoon classified as a Heavily Modified Water Body (HMWB) under the European Union (EU) Water Framework Directive (2000/60/EC). The framework was used to simulate the watershed loading of nitrogen and phosphorus, exchange of relevant water properties between the fjord and the North Sea, and biogeochemical cycles within the system.

The softshell clam *Mya arenaria* is a key component of the framework, as its filtration activity exerts strong top-down control of phytoplankton biomass; keeping the water sufficiently clear to support extensive submerged aquatic vegetation (SAV). As a result, the primary eutrophication risk shifts from pelagic algal blooms to epiphyte overgrowth of SAV, since elevated nutrient concentrations—no longer expressed as phytoplankton biomass, as a result of clam filtration—instead fuel opportunistic epiphytes that colonise and outcompete the macrophytes.

The model results show that, under reduced *Mya arenaria* filtration, bottom-up control, through a 35% reduction in land-based N-loading, leads to reductions of 37–60% in summer chlorophyll concentration and a decrease in epiphyte risk score of 1–50%. Reducing the N-load by 17.5% reduces summer chlorophyll by 10–29% and decreases the epiphyte risk by 0–34%.

Under current conditions, where high filtration by *Mya arenaria* keeps summer chlorophyll peaks around 15–25 µg L⁻¹, the primary impact of reduced nitrogen loading is expected to manifest as a decrease in epiphyte risk. A catchment model was used to develop stakeholder-driven reduction scenarios; this resulted in a restoration plan for 27 000 ha of wetlands and prioritisation of a new monitoring program for epiphytes.

The end-to-end ecosystem modelling framework was used as a tool to engage managers, stakeholders, and policymakers; this was an effective approach in coastal water management and particularly in the identification of the most effective solutions for water quality improvement.

1. Introduction

Nutrient loading to coastal systems and its consequences on eutrophication of bays, estuaries, and lagoons has been the object of many studies (Bricker et al., 2008; Cloern, 2001; Diaz and Rosenberg, 2008; Riemann et al., 2016; Malone and Newton, 2020; Jørgensen et al.,

2026). The Driver-Pressure-State-Impact-Response (DPSIR) conceptual framework (Elliott et al., 2017; Elliott and O'Higgins, 2020) has been used to develop assessment methodologies such as ASSETS (Bricker et al., 2003; Ferreira et al., 2007), OSPAR-COMPP (OSPAR, 2009; Claussen et al., 2009; Devlin et al., 2007), and HEAT (HELCOM, 2010; Andersen et al., 2010).

* Corresponding author. Longline Environment Ltd, 63, St. Mary Axe, London, W1G 8TB, United Kingdom.

E-mail address: joao@hoomi.com (J.G. Ferreira).

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An overview of the different approaches is given in Ferreira et al. (2011) as part of the guidance for implementation of European Union (EU) water-related directives. Of these, the Water Framework Directive (WFD - 2000/60/EC, European Union, 2000) has led to an EU-wide classification of surface water bodies using a five-category, ecosystem-based approach.

In the case of transitional waters (*sensu* WFD) such as coastal lagoons, a system may be classed as a heavily modified water body (HMWB) if it has substantial modifications to water circulation e.g. due to dykes, weirs, or sluices.

For all surface waters, the WFD stipulates that a classification of *Good* (*Potential*, in the case of HMWB) must be met; if a system has a lower classification, management measures should be implemented to improve its ecological quality—this is evaluated by means of Biological Quality Elements (BQE) such as phytoplankton abundance, biomass, and composition (ABC) and Supporting Quality Elements (SQE) such as nutrient concentration.

The choice of measures is challenging due to the complexity of estuarine and coastal ecosystems; such systems typically have both freshwater and ocean end-members, tidal flats, saltmarsh areas, pelagic and benthic primary production, and a range of secondary producers. In shallow estuaries and lagoons, strong benthic-pelagic coupling means that benthic filter-feeders may exert significant top-down control of pelagic algae, reducing eutrophication.

With respect to drivers and pressures, loading of nutrients and organic matter in most of the EU is no longer primarily of urban origin, but mainly associated with non-point sources (EEA, 2024). These diffuse loads are more difficult to control than point-source emissions; mathematical modelling, in combination with monitoring, is an appropriate approach for managers to determine source apportionment of different types of loads.

However, quantification of catchment pressures *per se* will not provide sufficient management guidance, since it is the drivers that must be modified; furthermore, calculation of changes to nutrient load is not easily translated into effects on a receiving water body such as an estuary or lagoon; these changes to *state*, shown by indicators such as the BQE and SQE used for WFD classification, can only be modelled. This is due to the non-linearity of coastal systems, which vary widely in susceptibility (Bricker et al., 2003), latency, and resilience (Duarte et al., 2009). An example of this is the limitation of phytoplankton blooms in

fast-flushing estuaries (Ketchum, 1954; Ferreira et al., 2005), which may lead to enhanced development of macroalgal blooms on tidal flats.

Due to the inherent complexity of such estuaries and lagoons, soil-to-sea modelling is the only way to analyse the potential effects of management measures on ecosystem quality, albeit constrained by the lack of a paradigm for descriptors such as biodiversity, which are currently a challenge to simulate.

Modelling tools have been widely applied to study eutrophication in coastal systems, both by means of detailed research models (Bricker et al., 2018; Ferreira et al., 2008; Nobre et al., 2010; Peña et al., 2016; Lancelot et al., 2007; Liang et al., 2015) and screening models (Smith et al., 2010; Ferreira et al., 2012).

Although research models typically include loading (often estimated from river discharge and nutrient concentrations) to simulate primary production, frameworks that explicitly integrate catchment modelling are uncommon; this limits the value of such models for policy makers, who must set targets for emissions and therefore need a quantitative understanding of the relation among drivers, pressures, and state, and the variability of these interactions in time and space.

To address this, an end-to-end modelling framework (Fig. 1) was developed and applied to Ringkøbing Fjord, a 300 km² coastal lagoon in western Denmark classified as an HMWB under the WFD. The framework provides a set of coupled models that can address all the elements of DPSIR and therefore provide decision makers with a comprehensive view of different management options; in addition, a framework of this kind also facilitates stakeholder engagement and promotes a consensus approach for policy decisions.

The objectives of this work were to:

1. Build an end-to-end (soil-to-sea) model of Ringkøbing Fjord and its catchment and use it to analyse and interpret the WFD classification for different BQE and SQE;
2. Apply the framework to examine various stakeholder-defined scenarios, with an emphasis on catchment loading measures (bottom-up control) and top-down control of primary production;
3. Provide local actors with a set of tools to simulate key processes within a watershed and adjoining coastal system, exemplifying a holistic approach to watershed management;
4. Support policy decisions for integrated coastal management, regulatory compliance, and ecological and social sustainability.

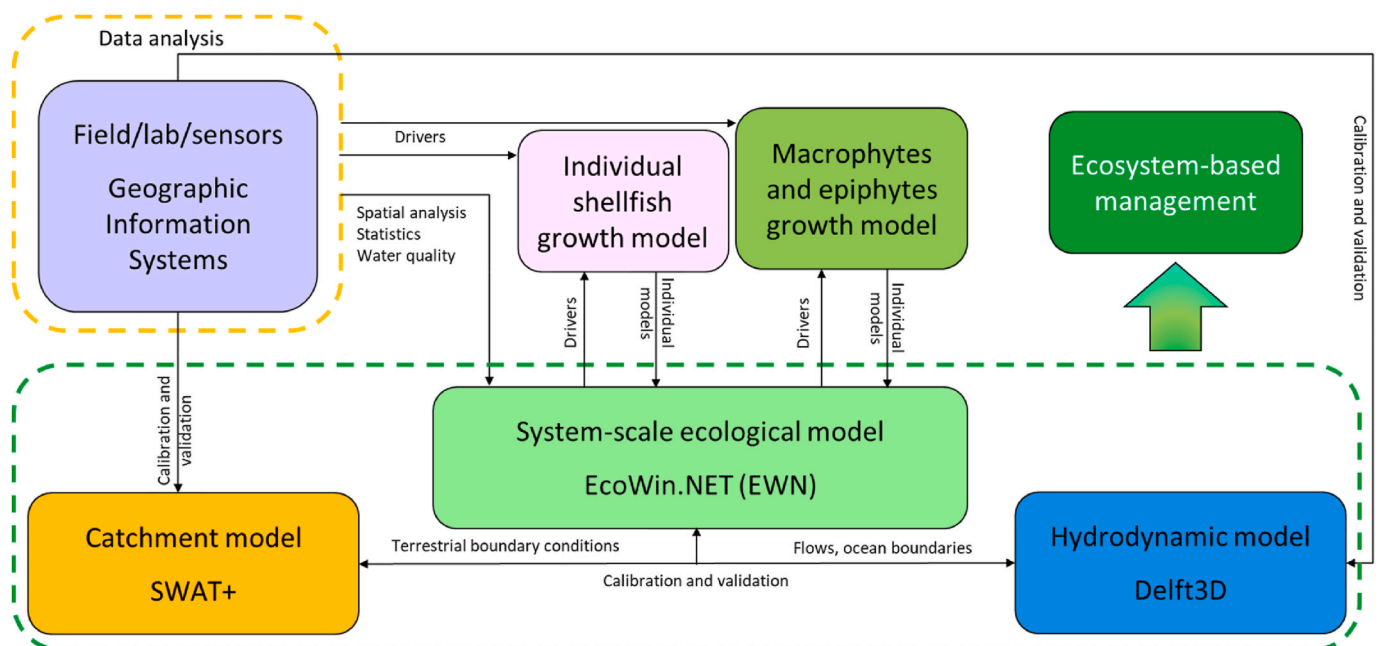


Fig. 1. Conceptual diagram of the Ecological Modelling Framework (EMF).

2. Methods

2.1. Study area

2.1.1. Catchment and loading

The Ringkøbing Fjord catchment is flat (average altitude 40 m) and consists mainly of sandy soils developed on till and outwash deposits (Fig. 2). It covers an area of approximately 3400 km² and includes the Skjern Å river, which is 94 km in length and has a mean flow of 37 m³s⁻¹. Following initial drainage and straightening in the 1960s, the Skjern Å was the subject of a major restoration plan in the early 2000s. Today, its waters are of fairly good quality, as evidenced by the presence of salmon.

The main catchment land use is agriculture, which covers 65–70% of the area and is dominated by pig farms. The remainder is taken up by woodland (15–20%) and other natural areas such as wetlands and heathland.

The catchment is sparsely populated, with a few urban areas, the largest being the towns of Ringkøbing, Brande, Skjern, and part of the town of Ikast. Annual rainfall in the region averages 1026 mm, with the wettest months being September and October and the driest April and May.

2.1.2. Ringkøbing Fjord

The fjord is 30 km long and 10–15 km wide, has an area of 300 km², a maximum depth of 6 m and an average depth of 2 m (Fig. 3) and drains approximately 9 percent of the area of Denmark.

Ringkøbing Fjord has an artificial physical regime governed by a sluice where the water level in the fjord and the water exchange with the North Sea are actively regulated via a lock at Hvide Sande. The water in the fjord is brackish due to a large influx from the Skjern Å river, which drains a large part of West Jutland.

The fjord has a wide range of uses, including recreational activities such as tourism, kite surfing, and commercial and leisure fishing. It is also a biodiversity reserve, RAMSAR site, and includes the Tipperne bird reserve. These uses were put at risk in 2019 with the appearance of a major pelagic algal bloom.

In past decades, the fjord was recurrently eutrophied due to high

nutrient loading from land and the regulated exchange with the North Sea. Sluice operational practices were changed in 1995 to increase salinity in the estuary, reduce nutrient concentrations and algal biomass through flushing, and enhance recovery of submerged aquatic vegetation (SAV). One notable consequence was the significant recruitment of the softshell clam *Mya arenaria*, which thrives in the fjord provided the salinity exceeds the reproductive threshold of 10 psu, and whose high clearance rate led to a strong drawdown in phytoplankton biomass in the following years (Nielsen et al., 2005).

Mya arenaria stops filtering below 6 psu and its clearance rate increases significantly above 14 psu. The top-down control on phytoplankton biomass exerted by *Mya* is thus strongly dependent on both the rainfall and sluice management regime.

Different species of SAV are present in Ringkøbing Fjord, including *Zostera marina*, *Ruppia* sp., and *Potamogeton* sp. In a shallow coastal lagoon such as this one, the relative distribution of these species will be a function of physical factors such as salinity and underwater light climate. The former is mainly governed by the sluice dynamics, with respect to seawater supply, and by river discharge, in terms of fresh-water inputs. The latter is a function of bathymetry, but also of light attenuation in the water column. Light attenuation is mainly controlled by suspended particulate matter (SPM), which under eutrophic conditions can include a significant contribution from pelagic algae.

Since the fjord connection to the ocean is artificial, there are no tidal dynamics driving resuspension of sediment, which occurs mainly through wind-induced bed shear stress. The other factor that conditions light attenuation is phytoplankton biomass, resulting from the balance between primary production and grazing by consumers such as the softshell clam.

The interactions among the various ecological components of Ringkøbing Fjord are shown in Fig. 4 and reflect the non-linear dynamics that characterise many estuarine ecosystems with strong pelagic-benthic coupling. As an example, a high biomass of *Mya* will strongly limit pelagic algae even if nutrient concentrations are elevated, and the improved water clarity will enhance SAV production. Concurrently, higher nutrient availability may trigger development of undesirable epiphytes that benefit from both increased SAV substrate and a more opportunistic response to elevated dissolved inorganic nitrogen (DIN)

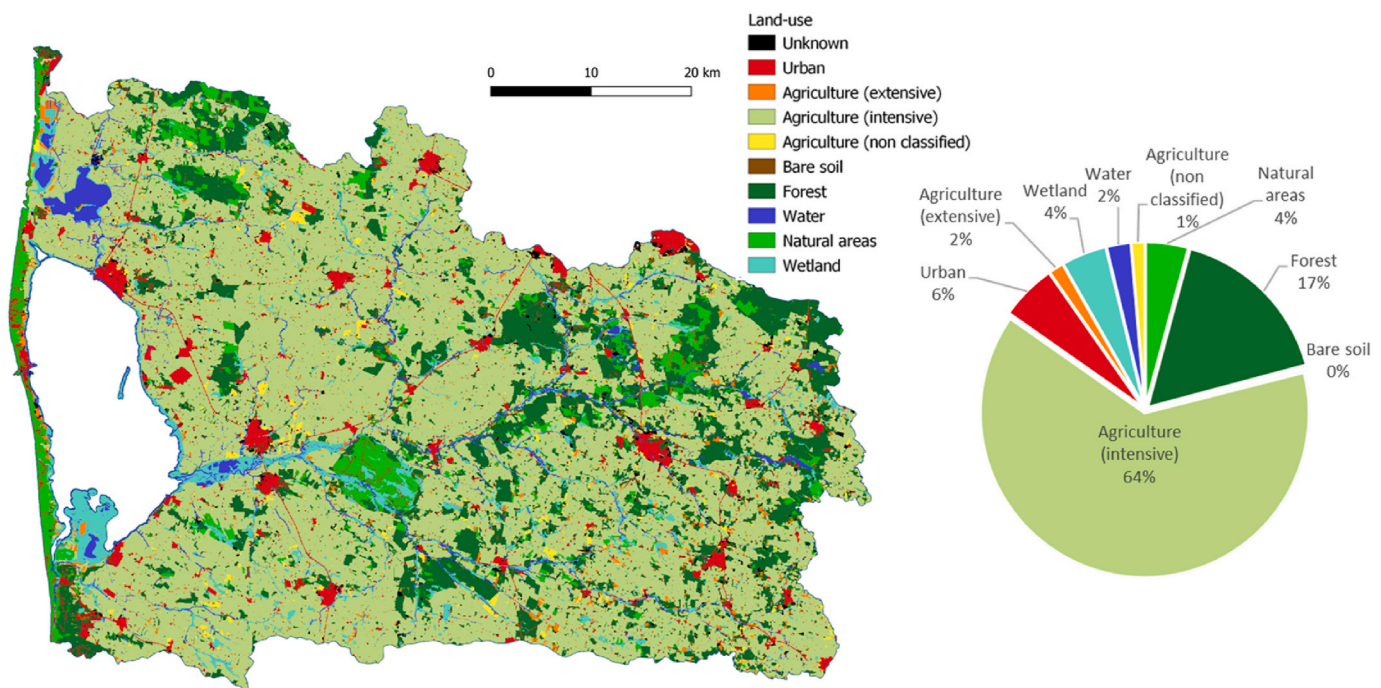


Fig. 2. Land-use (left) and soil (right) maps used to create the hydrological response units for the Soil and Water Assessment Tool + (SWAT+) model.

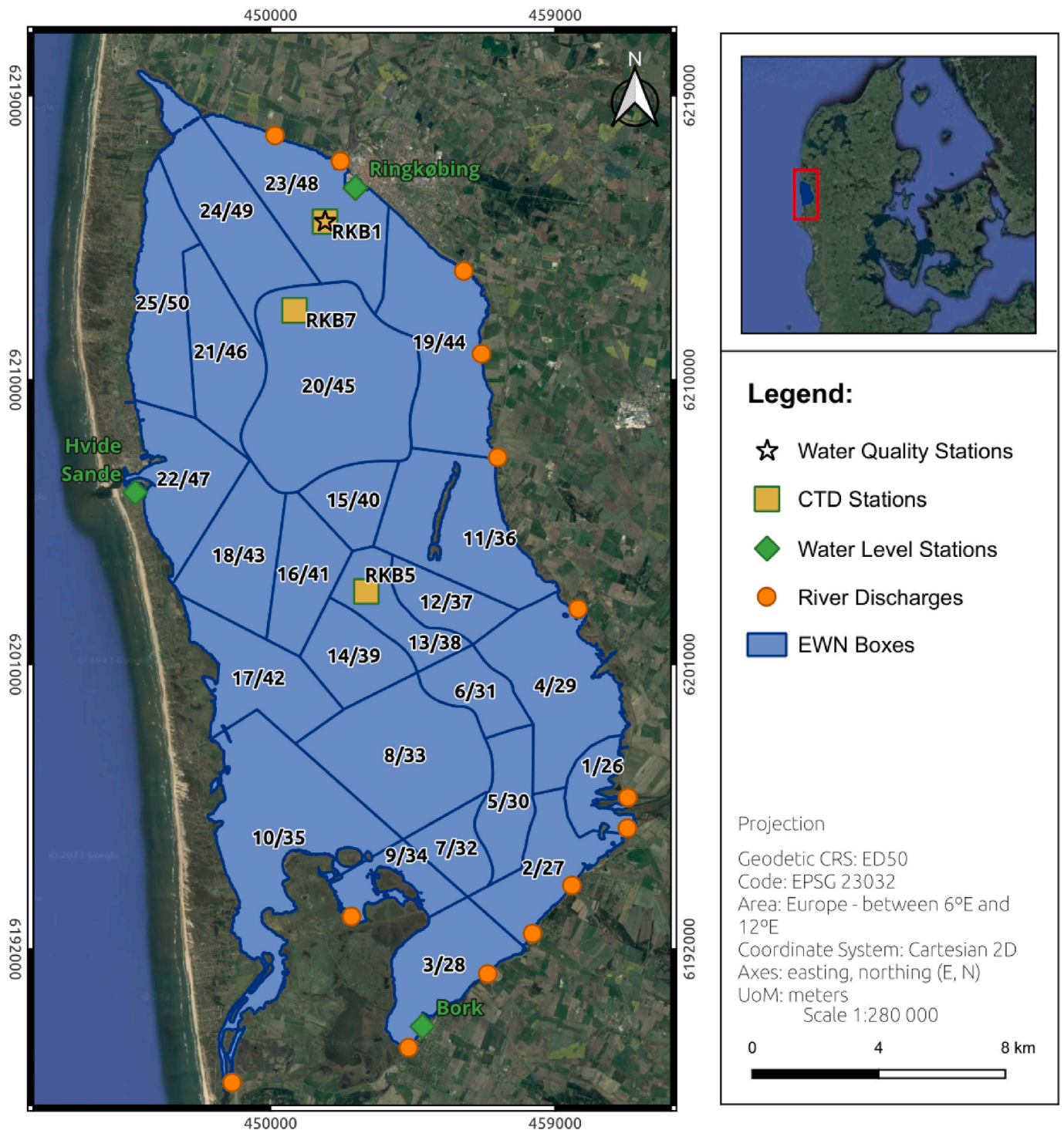


Fig. 3. Ringkøbing Fjord showing sampling stations, river discharges, and EcoWin (EWN) model boxes (labelled 1-25 for boxes in the upper layer of the water column and 26-50 for the lower layer).

and phosphate concentrations.

2.2. General modelling framework

The general framework developed for this work is shown in Fig. 1, together with examples of the models and other components it integrates. The modelling framework addresses: (i) nutrient loading from land; (ii) water circulation and water properties within Ringkøbing Fjord and the exchange with the North Sea through the sluice; (iii) growth of

bivalves inside the fjord; (iv) growth of macrophytes and associated epiphytes; (v) biogeochemistry of Ringkøbing Fjord, including the role of pelagic algae, SAV, and benthic filter-feeders on setting ecosystem status.

Each model has a number of uses *per se* and addresses different management challenges, and the linkages among models allow the Ecological Modelling Framework (EMF) to be leveraged for improved decision-support (Table 1).

A number of these framework components have been previously

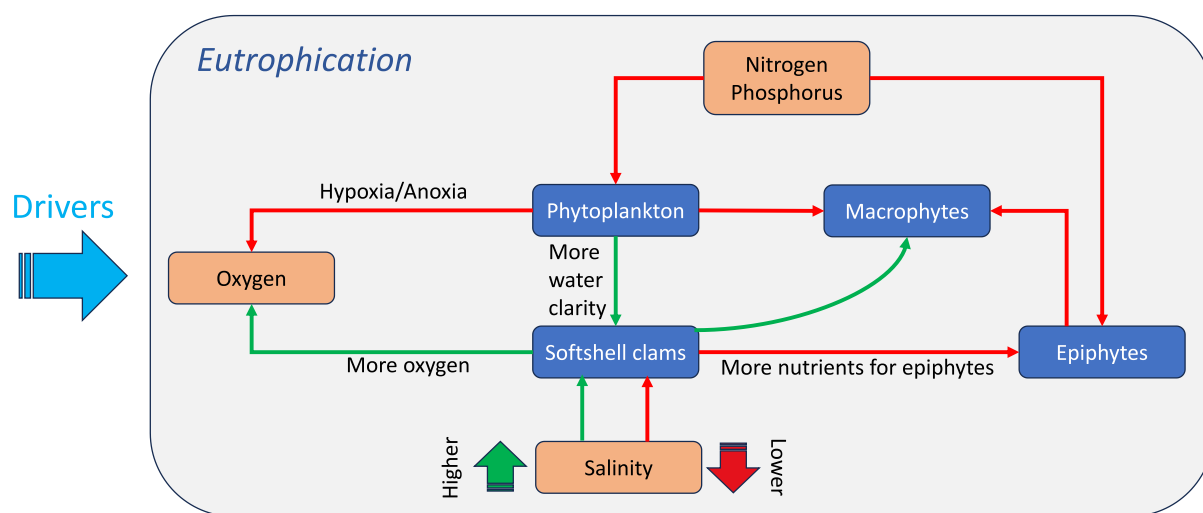


Fig. 4. Conceptual diagram illustrating interactions of key ecosystem components in Ringkøbing Fjord.

Table 1
Models and scope of application.

Name	Acronym	Scope
Soil and Water Assessment Tool + 3D hydrodynamic model	SWAT+ Delft3D	Catchment loading of water, nutrients, organic matter, and suspended sediment 3D detailed circulation within the fjord
Bivalve individual growth model	AquaShell	Individual growth and environmental effects of <i>Mya arenaria</i> (softshell clam)
Vegetation growth model	AquaFronD	Growth of macrophytes and associated epiphytes.
Ecosystem carrying capacity model	EcoWin. NET	Ecosystem-scale model for the fjord, including relevant biogeochemistry

described in the literature (Nobre et al., 2010; Nunes et al., 2011; Bricker et al., 2018; Ferreira et al., 2023); further details are provided in the Supplementary Material section, together with calibration and validation outputs for the study area.

2.3. Data collection and model setup

2.3.1. Catchment data

2.3.1.1. Land use data. Land use was based on a Danish national map for the Third River Basin Management Plan (Styrelsen for Grøn Arealomlægning og Vandmiljø, 2025). The land use map (CLC, 2012) was reclassified to match the Soil and Water Assessment Tool (SWAT+) land use classes (Fig. 2).

The agricultural area was subdivided according to data collected by the Ministry of Agriculture and compiled by SEGES Innovation for the Ringkøbing Fjord catchment area (see Supplementary Material, Table S1). In addition, information on drained areas has been provided by the different municipalities of the Ringkøbing Fjord catchment and was included in the model setup.

2.3.1.2. Soil data. The soil input data were based on the national topsoil texture map with a 250 m grid resolution derived from 45 000 soil samples and interpolated using ordinary kriging (Greve et al., 2007).

2.3.1.3. Wastewater data. Data on nutrient (Total N and Total P) content as well as discharge flow were collected from utility companies and major businesses with outlets to Ringkøbing Fjord for a 5-year period from 2017 to 2022. These data have been incorporated as point sources input into the SWAT+ catchment model.

2.3.1.4. Climate data. Daily precipitation data were obtained from the national 10 km grid (Scharling, 2012) and corrected with the Danish Meteorological Institute (DMI) precipitation correction model with help from the Geological Survey of Denmark and Greenland (GEUS). Data on daily minimum and maximum air temperatures, relative humidity, wind speed, and solar radiation were obtained from the 10 km DMI dataset.

2.3.1.5. Streamflow and water quality data. The streamflow simulation was calibrated against data collected at a selection of hydrometric stations with over 5 years of daily data for the period 2012–2020. 12 to 18 measurements per year were available for the same station used for flow calibration and validation. Measured parameters include nitrite + nitrate ($\text{NO}_2 + \text{NO}_3$) load, total nitrogen (TN) load, phosphate (PO_4) load, total phosphorus (TP) load, and suspended sediment load (SPM).

2.3.2. Hydrodynamic data

Water level data were used to calibrate and validate the hydrodynamic model. Three stations record the water level in the fjord: Hvide Sande, next to the sluice, Ringkøbing in the northern part, and Bork in the southern part of the fjord (Fig. 3).

Conductivity, temperature, and depth (CTD) profiles were also used to calibrate and validate the hydrodynamic model. Fortnightly profiles are available for three stations in the fjord (Fig. 3).

2.3.3. Water quality and benthic data

Water quality data from 1985 are only available for one station, RKB1 (Fig. 3) at monthly or fortnightly intervals. Some data also exist for the softshell clam and for seagrasses (see Supplementary Material).

2.3.4. Model box definition

Ringkøbing Fjord was divided into simulation areas or model boxes (Fig. 3) for ecological modelling with EcoWin. Details of the box division procedure are given in the Supplementary Material.

2.4. Hydrological modelling

SWAT+ takes into account factors such as land cover, soil types, land management practices, and climate data to model how these factors impact water quality and quantity, and was developed to provide more robust and flexible capabilities for addressing contemporary water resource management challenges.

2.4.1. Model validation

The catchment model was calibrated against daily streamflow and

biweekly-to-monthly loads of NO₃+NO₂, TN, PO₄, TP and SPM: 17 of 20 performance criteria were satisfactory to very good following Moriasi et al. (2015), with Nash-Sutcliffe efficiency of 0.48–0.78 and percent bias of –4.8% to 17%. Because lower-frequency water quality sampling can make standard criteria misleading, observed and simulated load time series were also compared visually, and the model was deemed fit for purpose for simulating nutrient and SPM loads entering the fjord. Further details of SWAT+ setup, calibration, and validation are provided in the Supplementary Material (see Figs. S1–S3, Tables S1–S2).

2.5. Hydrodynamic modelling

The Delft3D-Flow model (D3D, Deltares, 2010; Lesser et al., 2004) was used to simulate Ringkøbing Fjord and the connection of the lagoon to the North Sea. The fjord was modelled by means of curvilinear grids using terrain-following vertical coordinates and advanced turbulence-closure methods, thus improving the simulation of stratification and mixing processes in shallow water. Details on the computational grid, bathymetry, and model calibration and validation are given in the Supplementary Material (see Figs. S4–S6).

2.5.1. Model parametrization

The hydrodynamics in the fjord are driven by freshwater inflow from rivers, inflow and outflow of salt water through the sluice at Hvide Sande, and wind. The discharge of water through the sluice was calculated using the zero-dimensional model described by Nielsen et al. (2005). This model uses the water-level differences in the North Sea and fjord to determine the discharge through the sluice. These discharges are then used in Delft3D-Flow as sources and sinks.

River flow is included for the Skjern Å River and a total of 13 other smaller rivers and inflows using the SWAT+ simulated discharge. Hourly wind speed and wind direction were also provided to the model from DMI's meteorological station at Hvide Sande.

Bottom friction was modelled with a Chézy coefficient obtained from the White-Colebrook formulation defined by the Nikuradse roughness length set to $k_s = 0.1$ m.

2.5.2. Model validation

The hydrodynamic model was calibrated against water level and salinity. Water level skill was excellent at all stations except Bork in the calibration year (very good), with RMSE below 10 cm (<10% of range) and negligible bias. The seasonal salinity cycle was well represented at all three stations in both years, with minor underestimation (<2 psu); stratification intensity and range were reproduced, though not always in the correct location. The model was deemed fit for purpose for simulating water exchanges between EWN boxes. Further details are available in the Supplementary Material (Figs. S4–S6).

2.5.3. Individual bivalve model

An individual growth model for the softshell clam *Mya arenaria* was developed and integrated into the EcoWin.NET ecological model, with the aim of simulating bivalve biomass in the fjord, sustainable carrying capacity, and environmental externalities (eutrophication abatement and biodeposition) of the bivalve population at the fjord scale.

The model was developed using the AquaShell™ framework, which has been successfully used to simulate shellfish feeding, metabolic expenditure, growth, reproductive effort, and overall mass balance of many bivalve species (see e.g. Ferreira and Bricker, 2016; Cubillo et al., 2023).

The key features of the individual growth model are: (i) simulation of physiological functions that determine the change in individual weight and shell length; (ii) integration of relevant physical and biogeochemical elements such as temperature, salinity, chlorophyll, and organic detritus; and (iii) provision of environmental feedbacks of shellfish growth, such as the depletion of phytoplankton and detritus, excretion of dissolved nitrogen, and oxygen consumption. See Supplementary

Material (Figs. S7–S11) for further details.

2.6. Modelling of macrophytes and epiphytes

2.6.1. Background and rationale

The relative distribution of benthic macrophytes in Ringkøbing Fjord depends on physical factors such as salinity and underwater light climate. The former is mainly governed by the sluice dynamics, with respect to seawater supply, and river discharge, in terms of freshwater inputs. The latter is a function of bathymetry, but also of light attenuation in the water column, which is mainly a function of SPM.

Distribution and abundance of macrophytes is one of the BQE of the WFD and therefore a component that must be included in an ecosystem model of Ringkøbing Fjord, where benthic vegetation is a key feature. Furthermore, nutrient loading is a driving force of both pelagic and benthic primary production, and the inclusion in EcoWin of a benthic primary production model helps to understand the partitioning of this load between the two components (Fig. 4).

The final consideration is the addition of epiphytic vegetation in the benthic macrophyte model. Epiphytes are considered a symptom of eutrophication (e.g. Bricker et al., 2003) and therefore an undesirable feature of coastal ecosystems.

The implementation of a macrophyte model in EcoWin aimed to (i) simulate above- and below-ground biomass of macrophytes based on the key physiological processes; (ii) simulate nitrogen removal due to benthic primary production; (iii) develop a risk-based approach to colonisation of macrophytes by epiphytes; and (iv) introduce two key indicators (macrophytes and epiphytes) for eutrophication assessment into the modelling framework.

2.6.2. Macrophyte model

A review of existing models for the macrophyte species relevant to Ringkøbing Fjord revealed a paucity of work in this area. The most promising model was developed by Bocci et al. (1997) for *Zostera marina* and later adapted by Plus et al. (2003) for *Zostera noltii*. Other work in this field has been reported by Wetzel and Neckles (1986) and Bulthuis (1987).

No adequate models were found for other macrophyte species, so the Bocci et al. (1997) model was chosen for this work since (i) it is specific to *Zostera marina*, a keystone species found in Ringkøbing Fjord; and (ii) it includes parameterisation for Danish waters.

The conceptual model is shown in Fig. 5 and contains three state variables: shoot biomass, root biomass, and internal nitrogen quota. The sum of shoot and root biomass is the stand biomass, and the model output is dry weight per unit area (g DW m^{-2}).

Gross primary production is limited by four drivers: nitrogen, temperature, light, and available space. In the Ringkøbing Fjord EcoWin model, the first three drivers are derived from the biogeochemical model and available space is a calibration parameter based on the typical maximum biomass per unit area. See Supplementary Material (Fig. S12) for additional details of the modelling approach.

2.7. Ecological modelling

2.7.1. Model development

The ecological model for Ringkøbing Fjord was developed using the well-tested EcoWin.NET (EWN) ecosystem-scale model (Ferreira, 1995).

EWN simulates ecological processes in aquatic systems, including eutrophication in both the water column and tidal flats, saltmarsh dynamics, and production and environmental effects of aquaculture. Over the last two decades, EWN has been used as part of a modelling framework, integrating detailed hydrodynamic models such as Delft3D, the Princeton Ocean Model, and the Regional Ocean Modeling System (ROMS). The model incorporates the relevant physics to simulate water circulation, a full representation of catchment loading, including both point and diffuse sources of water and nutrients, and simulates a

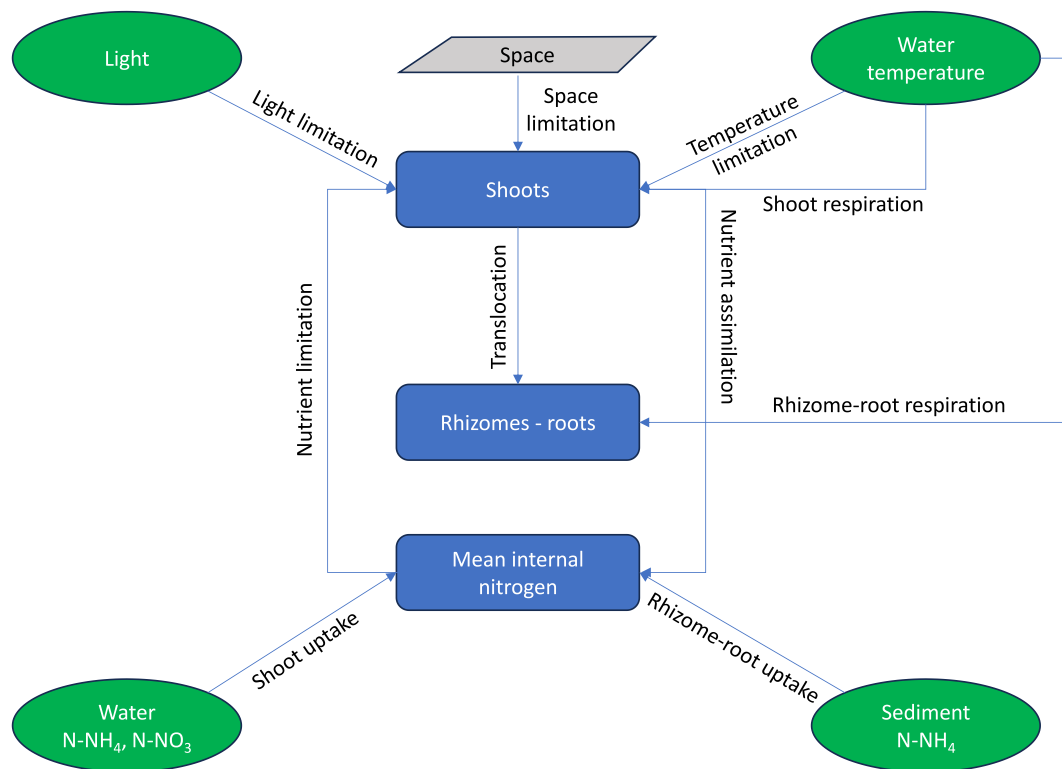


Fig. 5. Conceptual diagram of macrophyte model (adapted from Bocci et al., 1997).

comprehensive set of biogeochemical processes, dealing with nutrient cycling, primary and secondary production, and addressing both the pelagic and benthic compartments of the system.

The design of the EWN application makes it straightforward to make changes to loads from land or ocean for scenario analysis, and the object-oriented formulation means that objects such as shellfish can be turned on or off to examine the role of top-down control by filter-feeders on eutrophication.

EWN simulates a single representative year (2017), compared against fortnightly observations from the fjord's only water quality station (RKB1, Fig. 3) and against the broader 2010–2019 record to assess whether this simulated year falls within typical interannual variability.

2.7.2. Model validation

Ecological models incorporate many processes, reflecting the complexity of coastal ecosystems (e.g. Butenschön et al., 2016); as a consequence, calibration and validation are inherently complex.

EWN is calibrated and validated using four different approaches:

1. Calibration of external model components and validation of these against measured data;
2. Calibration and validation of internal model components based on measurements and/or declarative data;
3. Calibration and validation of internal model parameters based on measurements and literature;
4. Development of bespoke objects that extend the EWN model library.

For the Ringkøbing Fjord model, these components are part of the EMF shown in Fig. 1, i.e. the SWAT+, Delft3D, AquaFronD, and AquaShell models, dealing with hydrology and nutrient loading, circulation, and macrophyte and shellfish growth respectively.

Water transport in the upscaled EWN model was validated against Delft3D outputs using two metrics: volume variation over an annual cycle and comparison of water residence time by releasing a conservative tracer in the EWN model boxes. The salinity of the different boxes

was then compared both to measured data and hydrodynamic model outputs.

No direct data were available to validate the eelgrass and softshell clam submodels; the approach used was to assess the performance of the individual models with respect to growth and environmental effects by comparing to literature data (e.g. for eelgrass: Bulthuis, 1987; Bocci et al., 1997; for clams: Brousseau, 1979; Emerson et al., 1988; Riisgård et al., 2003; Zwartz, 1991) and by closing the mass balance for key variables such as biomass and nitrogen. The integration of these individual models into EWN is first tested by using constant environmental drivers (temperature, salinity, and food) in both EcoWin and the applications used for running the individual models to ensure a perfect match.

Simulated dissolved inorganic nitrogen concentrations reproduce the observed seasonal cycle well, declining to near 0 mg L⁻¹ between March and May and rising to 1.5–2 mg L⁻¹ between September and February.

POM measurements distinguish a summer period (1–4 mg L⁻¹) from a winter period (generally >5 mg L⁻¹); the model reproduces this seasonal pattern, with occasional outliers attributed to resuspension during strong wind events.

Chlorophyll is less well reproduced by the model, which tends to overestimate peaks in some years; this partly identified a need for targeted data acquisition, both for improved spatial resolution of pelagic state variables but also for partitioning of pelagic and benthic primary and secondary production.

Overall, the behaviour of the EMF was considered acceptable for its main function, i.e. the analysis of nutrient reduction scenarios.

Further details on the calibration and validation of the models integrating the EMF are provided in the Supplementary Material section (see Figs. S13–S14). Information is also presented for each component of the model framework on key assumptions with respect to datasets, forcing functions, processes, and parameters.

2.7.3. Epiphyte risk index

Epiphytes are considered to be a symptom of eutrophication,

although they are not part of the Assessment of Estuarine Trophic Status (ASSETS) model (see e.g. Ferreira et al., 2007) and are not included in the WFD. In both cases, the main reason for not including this symptom was the difficulty in performing a quantitative assessment; allied to this, the definition of a classification scheme is non-trivial.

There are not enough data on spatial and temporal variation of epiphyte biomass in Ringkøbing Fjord and its correlation with nutrient availability, but it is recognised that three drivers are important in conditioning epiphyte biomass: (i) substrate: epiphytes can only exist if they have a plant substrate to attach to (i.e. benthic macrophytes). Therefore, a low or residual stand biomass leads to low or negligible epiphyte biomass; (ii) underwater light climate: for macrophyte biomass to be sufficient for epiphyte colonisation, the light climate must be suitable for macrophyte growth. In Ringkøbing Fjord, this means that phytoplankton biomass (and thus chlorophyll concentration) must not be too high. This can occur either through bottom-up control (low nutrient concentration) and/or top-down control (phytoplankton filtration) by bivalves. As algal biomass increases, if there is still sufficient light at the bottom to allow for primary production, opportunistic epiphytes tend to outcompete perennial vegetation such as *Zostera*; and (iii) nutrient concentration: if there is sufficient light, low nutrient concentrations allow macrophytes to outcompete epiphytes; however, in a situation where the light climate is appropriate and chlorophyll is low due to top-down control, elevated concentrations of nutrients will create appropriate conditions for opportunistic epiphytes to outcompete macrophytes.

A risk assessment index was implemented in EcoWin based on these factors (Fig. 6), using stand biomass (a proxy for substrate availability), pelagic chlorophyll (a proxy for light availability to the benthos), and nutrient concentration (a proxy for loading) as indicators.

The indicators, thresholds, and combinations used are shown in the Supplementary Material (see Table S3). In practice, the resulting index is a management-oriented decision-support tool that synthesises the outputs of a complex model into categories.

3. Results and discussion

3.1. The standard model

The Ringkøbing Fjord ecosystem model was defined as the standard or baseline and was then used to analyse the present-day ecosystem status. Fig. 7 and Figs. S15 and S16 (Supplementary Material) show the average summer (May–September) chlorophyll concentrations and average winter concentrations of DIN. The results indicate that summer

chlorophyll is lower in the bottom boxes, as expected due to top-down control by bivalves.

Summer chlorophyll is relatively homogeneous across all boxes and no spatial patterns can be identified except for lower values in box 1 due to freshwater inputs from the Skjern River. Summer chlorophyll in all model boxes with the exception of boxes 8 and 20 is below the upper threshold for good ecological potential ($8.4 \mu\text{g L}^{-1}$, Timmermann et al., 2021). Winter DIN is homogeneous across all top and bottom boxes (Fig. S16, Supplementary Material), which is characteristic of a vertically well-mixed system. Winter DIN is slightly higher in the southern part of the fjord due to the higher nutrient load from the Skjern Å (EWN box 1).

The model was also used to examine the maximum biomass of macrophytes and epiphytes (Fig. S17, Supplementary Material). Biomass values are highest in the shallow boxes receiving land inputs (boxes 26, 27, 29, 34, 36, 37, and 44).

The model also provides information on the individual weight of *Mya arenaria* across all the boxes (Fig. S18, Supplementary Material). Growth is higher in the boxes near the sluice due to higher salinity.

3.2. Scenarios and stakeholders

The catchment model was used to assess the impact of various scenarios, including wetland restoration and land use changes, on nutrient exports to the fjord. In addition, the fjord model was used to simulate the effects of changes in nutrient loading from land and the impact of top-down control by shellfish on the fjord ecosystem. During the model setup and scenario development phase, meetings were held with the local water board to: (i) gather local input for the model setup; (ii) foster ownership of the model and its results; and (iii) ensure alignment with local priorities. The local water board made the final decisions for prioritising different scenarios.

Over the past decades, Danish regulators set a target of 80% reduction in P load and then prioritised the reduction of nitrogen loading to coastal systems, with the objective of reducing coastal eutrophication, enhancing water transparency, and restoring eelgrass (e.g. Riemann et al., 2016). Nevertheless, there is evidence that phosphorus may be the limiting nutrient in some Danish waterbodies at certain periods of the year (e.g. Conley et al., 2009).

As a result of legislative requirements, particularly the EU River Basin Management Plans (RBMP) mandated by the WFD (Miljøministeriet, 2011; Styrelsen for Vand-og Naturforvaltning, 2016; Styrelsen for Grøn Arealomlægning og Vandmiljø, 2025), the primary interest of local stakeholders from the agricultural community was on

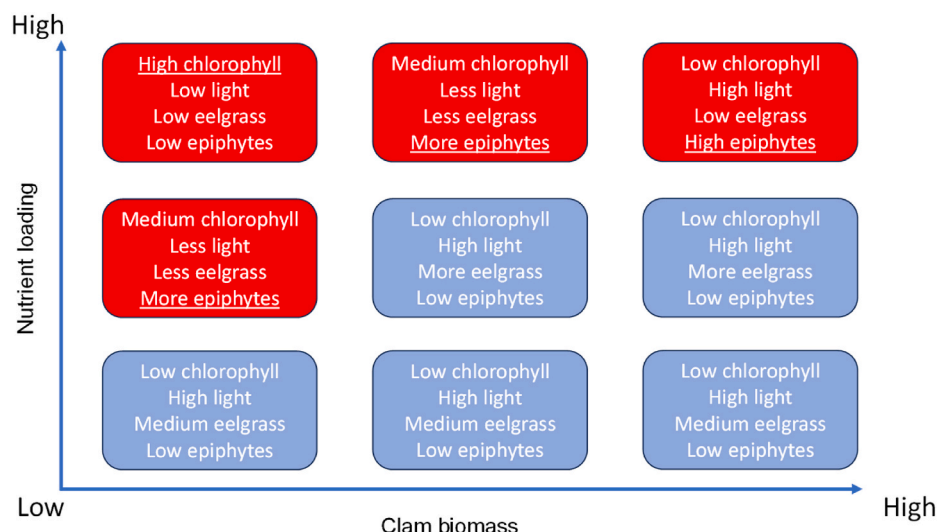


Fig. 6. Matrix approach implemented for risk assessment in EcoWin based on nutrient loading and clam biomass.

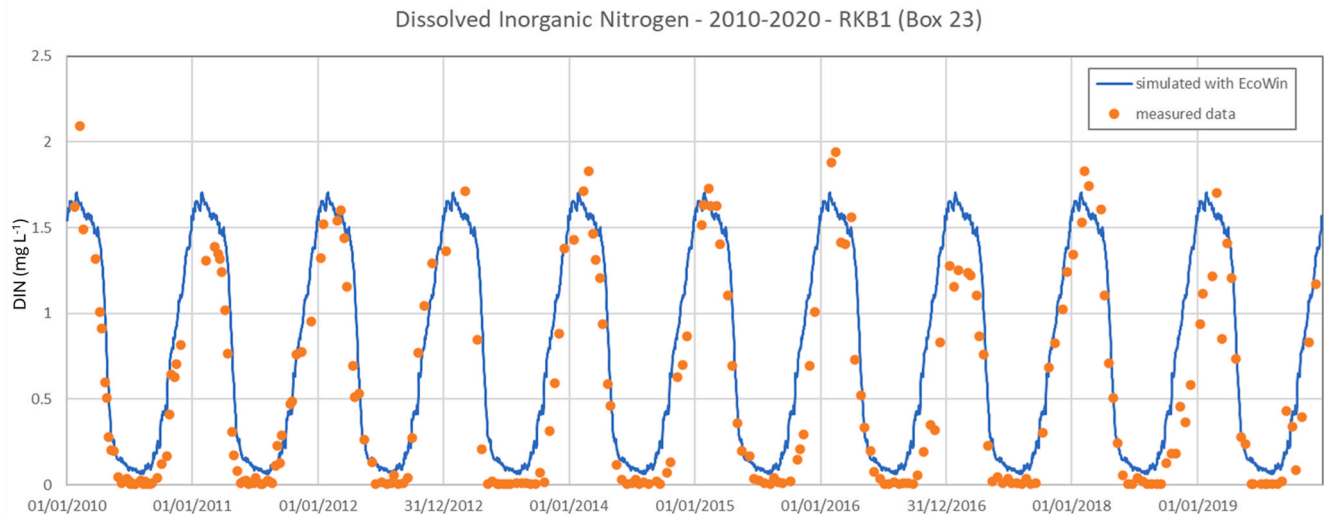


Fig. 7. Measured and modelled dissolved inorganic nitrogen (DIN) at station RKB1 (Box 23).

the consequences of N control on eutrophication in the fjord, and the efficacy of source-control management measures.

For these reasons, although EWN simulates the sources and sinks of both nutrients, the model emphasis was placed on the effects of nitrogen reduction.

An analysis of DIN and phosphate data in the fjord suggests that nitrogen becomes limiting during the summer, the period when elevated chlorophyll concentrations typically occur, due to an increase in both photosynthetically active radiation (PAR) and photoperiod duration.

In addition, most of the P is reversibly bound to iron oxides on resuspended fine sediment; soluble reactive phosphorus is a median 10.8% of total phosphorus at station RKB1. DIN:PO₄ is therefore an overestimate of the true N:P ratio, while DIN:TP is a floor, since TP includes refractory and organic fractions. Data for DIN:TP ratios between 2010 and 2020 show the fjord is N limited between May and October, reinforcing the modelling choice of N limited primary production.

These aspects led to a set of scenarios associated to changes in N load associated with modified agricultural practices that include reconstructed wetlands, together with changes in wastewater discharge.

3.2.1. Catchment loading

The impact of the following 6 scenarios on TN exports was examined using SWAT+. The resulting change in state (sensu DPSIR) was then assessed with the EMF.

1. *No Farming*: removal of all agricultural activities in the catchment, replaced by natural grassland. This indicates the magnitude of changes that could be expected by ceasing agricultural activities in the catchment;
2. *Full Wetlands*: replacement of the entire floodplain area by wetlands. This represents the maximum nutrient retention achievable by restoring all potential wetlands in the catchment;
3. *60% Wetlands*: replacement of 60% of the floodplain area (excluding urban areas) with wetlands;
4. *50% Wetlands*: replacement of 50% of the floodplain area (excluding urban areas) with wetlands;
5. *40% Wetlands*: replacement of 40% of the floodplain area (excluding urban areas) with wetlands;
6. *No Wastewater*: no wastewater discharges within the Ringkøbing Fjord catchment.

Table 2 shows the main component of the TN balance simulated in the catchment for the baseline and the different scenarios. The annual fluxes include fertiliser input, leaching to groundwater, removal in groundwater, removal in wetlands, exports with surface runoff and exports to the fjord.

These fluxes correspond to the annual average for the simulated period (2012–2019). In the baseline situation, 4518 t of TN are exported to the fjord, with a significant part (>12 000 t) retained in groundwater and a smaller fraction (about 100 t) removed in wetlands. These results

Table 2

Simulated annual fluxes (average for 2012–2019) of total nitrogen (TN) in tons for the Ringkøbing Fjord catchment for the baseline situation and the 6 simulated scenarios. Reduction compared to baseline in % and wetlands area in ha are also shown.

Scenario	Fertiliser inputs	Wastewater discharge	Leaching to groundwater	Removed groundwater	Removed wetlands	Export with runoff	Exports to Ringkøbing Fjord	Export reduction compared to baseline (%)	Wetland area (ha)
0 – Baseline	45 485	34	17 769	12 686	101	168	4518		4031
1 – No farming	0	34	7742	5243	51	77	2248	50%	4031
2 – Full wetland	37 836	34	14 354	10 722	2470	115	1191	74%	53 937
3 – 60% wetlands	41 005	34	15 839	11 833	1428	140	2425	46%	32 029
4 – 50% wetlands	41 781	34	16 185	11 992	1105	154	2797	38%	26 966
5 – 40% wetlands	42 557	34	16 530	12 164	947	149	3158	30%	21 903
6 – No wastewater	45 485	0	17 769	12 691	101	168	4507	0%	4031

highlight the importance of nutrient retention in groundwater for the catchment, accounting for 71% of the TN leaching to the groundwater. The TN exports with runoff constitute a minimal portion of the overall exports to the fjord, indicating that most of the flux is routed through the groundwater.

For the *No farming* scenario, where all farming areas are converted to natural grasslands, fjord exports decrease by 50% compared to the current situation. This reduction is lower than when maintaining farming activities and converting the whole floodplain area to wetlands (*Full wetland* scenario), in which case TN reduction reaches 74%. This apparently counterintuitive result—a lower reduction in TN export when fertiliser application ceases—can be explained by two factors: (i) the conversion of cropland to grassland not only eliminates fertiliser inputs but also replaces the crops that are harvested, leading to the removal of nitrogen stored in the system. In contrast, dead biomass in grasslands returns nitrogen to the soil, retaining it in the system; (ii) The limited fertiliser application rates in current cropland lead to efficient nutrient uptake by plants, thereby minimizing leaching from fertilisers. This is evident in the fact that, although the reduction in fertiliser inputs is substantial (45 000 t), the simulated decrease in TN leaching to groundwater is only around 10 000 t TN.

The *Full Wetland* scenario, resulting in a 74% decrease in TN exports to the fjord, is unrealistic but serves to illustrate the expected maximum potential reduction in TN exports from the catchment. Scenarios involving the conversion of a percentage of the floodplain to wetlands present more achievable and realistic values. The area of restored wetland exhibits a relatively linear relationship with the reduction in TN exports, enabling the estimation of the required wetland area to achieve specific objectives. For instance, to fulfil a targeted decrease of 1647 t, as outlined in the Third River Basin Management Plan (*Styrelsen for Grøn Arealomlægning og Vandmiljø, 2025*), about 27 000 ha, or 50% of the floodplain area, would need to be converted to wetlands. Half of this reduction (824 t) could be achieved with roughly 27% of the floodplain (14 700 ha) converted to wetlands.

The *No wastewater* scenario shows that wastewater discharges represent less than 1% TN export and are therefore not significant.

Wetland restoration appears to be a viable mitigation strategy to reduce nutrient exports from the Ringkøbing Fjord catchment. While the model does not pinpoint specific areas as especially effective for wetland implementation, priority should be given to locations draining large agricultural areas over those draining smaller forested areas. These results have limitations as the model operates at the catchment scale and does not offer site-specific outputs. Therefore, the figures presented herein serve as an indicator of the potential scale of TN retention that could be achieved. Site-specific studies must be performed to better assess the potential TN retention, taking into account the local characteristics of each site.

For all these scenarios, and consequently for the downstream analysis of changes in state in the fjord proper, it is important to understand the limitations inherent in any ecological modelling framework—these run across the various component models, including such challenges as accurate prediction of precipitation and atmospheric radiation in future years, uncertainty and variability in model parameters such as rates of nutrient uptake, and interannual variation in the mass of key state variables.

As a consequence, the results obtained for these scenarios should not be interpreted as strictly deterministic, but reflect a degree of uncertainty that is part of any model framework. Quantitative changes in eutrophication symptoms are clearly observed, but the management emphasis should be on the general predictions and underlying trends.

3.2.2. Eutrophication in Ringkøbing Fjord

The ecosystem modelling framework (EMF) for Ringkøbing Fjord is a valuable management tool for assessing different scenarios for eutrophication status and legislative compliance, from source control of nutrient emissions to complementary approaches such as top-down

control by bivalves. If diffuse sources such as agriculture are a significant part of nutrient loading, shellfish can play an important role in providing regulatory ecosystem services (*Ferreira and Bricker, 2019*) and offset eutrophication symptoms, not least because of the cost of controlling non-point emissions (*Stephenson et al., 2010*).

The three scenarios tested include top-down control of eutrophication simulated by the removal of shellfish. The other scenarios represent bottom-up control of eutrophication with two levels of reduction in nutrient loads from the catchment to Ringkøbing Fjord. The first is a 35% reduction, corresponding to the reduction in TN targeted by the RBMP. The other considers half of this reduction (17.5%). The indicators chosen to analyse the response of the ecosystem, and the rationale for these choices, are presented in *Table 3*.

3.2.2.1. Bottom-up control of eutrophication. Nutrient inputs to the fjord increase primary production and potentially lead to eutrophication, which would result in a degradation of water quality and ecosystem health and impact economic activities and other uses of the fjord. The EMF was applied to analyse various nutrient loading scenarios, and their effect on chlorophyll concentrations. Bottom-up control represents the effect of source control on primary production, while maintaining the shellfish population in the system. Due to limitations in *Mya arenaria* spatial abundance data, the EMF does not fully capture the top-down filtration effect under high-density conditions, producing chlorophyll peaks around $60 \mu\text{g L}^{-1}$ higher than those currently observed in the fjord ($15\text{--}25 \mu\text{g L}^{-1}$). The results below therefore represent a reduced-filtration state, comparable to periods of low bivalve filtration such as those observed before the *Mya* invasion and in specific years such as 2005 and 2019 (*Petersen et al., 2008, Fig. 8*).

Bottom-up control of chlorophyll is shown in *Fig. 8* for Box 13, considering three loading levels. By reducing the river load by 35% (including nutrients and suspended matter), a reduction in summer chlorophyll occurs in all EWN boxes, ranging from 37 to 60% ($0.2\text{--}4.2 \mu\text{g L}^{-1}$). This reduction ranges from 10 to 29% when the load is reduced by 17.5% ($0.1\text{--}1.7 \mu\text{g L}^{-1}$). While the high percentage could indicate a significant decrease in summer chlorophyll, this reduction occurs for already low concentrations, so that the absolute change is not very high, averaging $1.4 \mu\text{g L}^{-1}$. By contrast, the limit set by the WFD for achieving good ecological potential is $8.4 \mu\text{g L}^{-1}$.

Bottom-up control appears stronger at peak values (*Fig. 8*). Bottom-up control of eutrophication by pro rata reduction of river loads is significant across all the fjord as shown in *Table 4*. Bottom-up control of macrophyte and epiphyte biomass is also significant in some model boxes. A reduction of the river load affects the amount of nutrients available for macrophyte and epiphyte growth.

The epiphyte risk score for all boxes for different loading levels is shown in *Fig. S19* (Supplementary Material). Overall, the epiphyte risk score decreases with decreasing load, although this is not the case for the boxes with the lowest epiphyte risk score, which correspond to boxes in the middle of the fjord that are generally deeper and have less macrophyte biomass due to lower light availability. While the decrease in epiphyte risk score is relatively homogeneous in the other boxes, a few boxes seem to be more reactive to changes in load in terms of the decrease in the epiphyte risk score, such as boxes 7 and 8, located in the southern part of the fjord. Finally, under current high-filtration

Table 3
Indicators and rationale for analysing different loading scenarios.

Variable	Proxy for	Rationale
Chlorophyll (May–September average)	Phytoplankton biomass	Primary symptom of eutrophication, used in WFD
Dissolved inorganic nitrogen (winter average)	Nitrogen in the fjord	Causative factor of eutrophication, WFD supporting quality element (SQE)
Epiphyte risk score (average score)	Epiphyte biomass	Eutrophication symptom

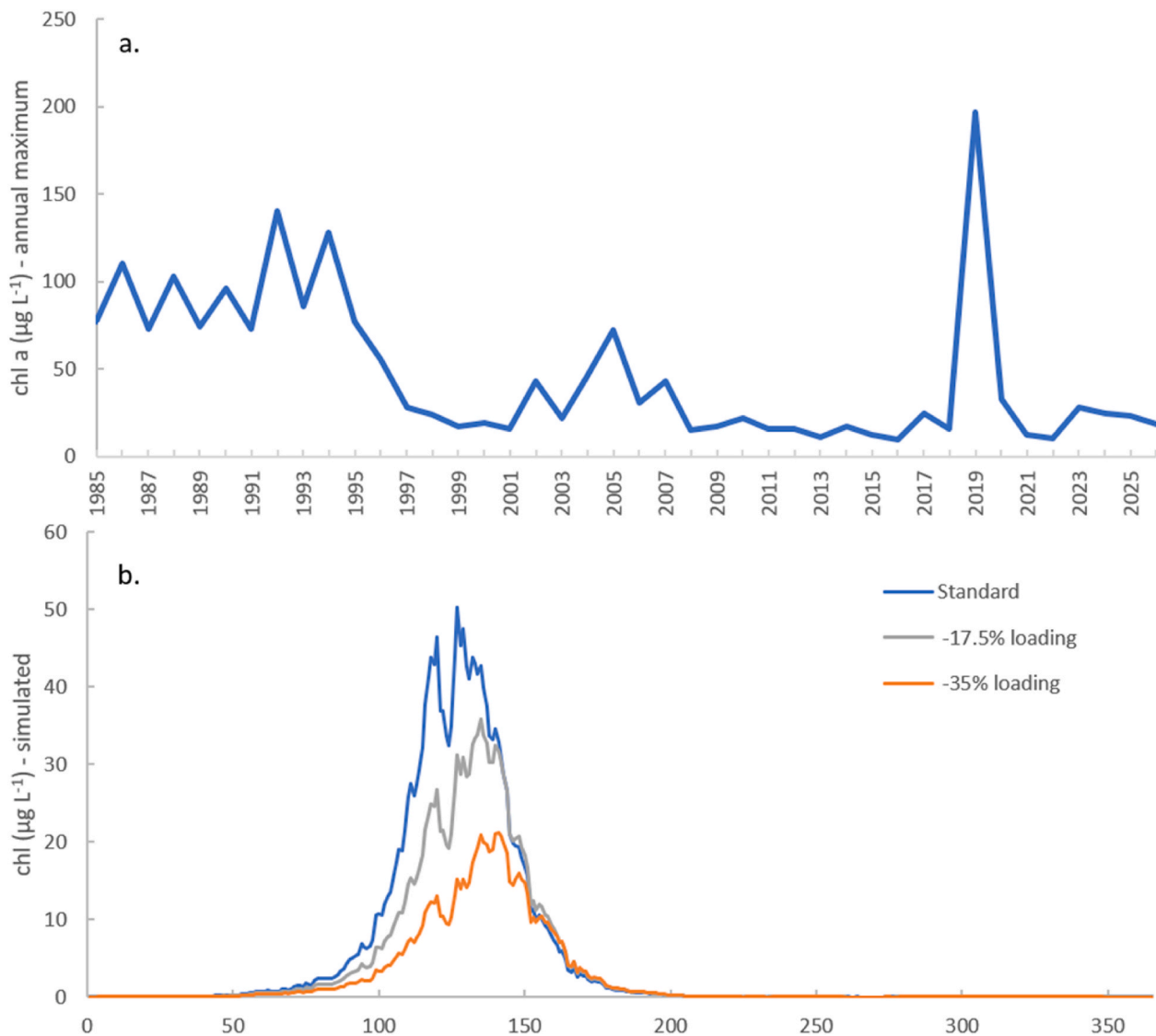


Fig. 8. (a.) Observed annual maximum chlorophyll-a concentrations at Station 1, Ringkøbing Fjord. (b.) Modelled chlorophyll-a response in Box 13 (central Ringkøbing Fjord) across three nutrient loading levels.

conditions, where *Mya arenaria* keeps summer chlorophyll peaks close to 15–25 $\mu\text{g L}^{-1}$, the primary effect of bottom-up nutrient control is expected to manifest especially through reductions in epiphyte risk rather than phytoplankton biomass.

3.2.2.2. Top-down control of eutrophication. Higher salinity in the fjord due to changes in sluice management led to regime shift. *Mya arenaria* is the keystone secondary producer (Petersen et al., 2008), outcompeting other benthic filter-feeders and zooplankton; consequently, it was the only grazer included in the EMF.

Analysis of chlorophyll concentrations with and without shellfish was carried out using the EcoWin model. Fig. 9 shows the difference in chlorophyll depending on whether or not shellfish are active in the system for a selection of upper and lower boxes and suggests that the filtration of phytoplankton and organic detritus contributes significantly to the control of eutrophication.

When shellfish are 'turned off' in EcoWin, i.e. removed from the system, chlorophyll increases significantly throughout the fjord. The change in chlorophyll is always more pronounced in the lower boxes than in the upper boxes because the shellfish are part of the benthos. Although chlorophyll generally increases when shellfish are switched off in the model, in Box 20, delta-chlorophyll briefly falls to negative values

(–10 to –15) at days 120–130 before returning to near-zero values. This transient reversal is due to a lag effect: the chlorophyll peak is both reduced and delayed when shellfish are switched on compared to when they are turned off. When shellfish are on during days 120–130 the peak is still rising while the peak when shellfish are off has already begun to decline, leading to a negative delta-chlorophyll.

The impact of top-down control by shellfish on a set of indicators for a selection of boxes with contrasting settings is shown in Table 5. In the absence of *Mya arenaria*, summer chlorophyll values would be multiplied by 12 on average in the lower boxes and by 2 in the upper boxes. The presence or absence of shellfish has no impact on winter DIN.

The top-down control of chlorophyll by bivalves also affects other symptoms of eutrophication, such as the presence of epiphytes. When shellfish are removed, the chlorophyll increases, reducing light penetration and therefore macrophyte growth, which is reduced by a factor of two in 16 of the 25 bottom boxes. As a result, the epiphyte risk score also decreases since the available substrate is considerably reduced.

4. Conclusions

This work highlights the role of end-to-end mathematical modelling in supporting management of coastal systems. Ringkøbing Fjord is a

Table 4
Impact of bottom-up control on a set of indicators for a selection of bottom boxes with contrasting settings.

Indicator	Scenario	Box 29	Box 45	Box 47
Summer chl a ($\mu\text{g L}^{-1}$)	Standard loading	1.3	1.3	1.6
	–17.5% loading	1.0	1.1	1.2
	–35% loading	0.7	0.7	0.8
Winter DIN (mg L^{-1})	Standard loading	1.8	1.6	1.6
	–17.5% loading	1.5	1.3	1.2
	–35% loading	1.1	0.96	0.90
Macrophyte + epiphyte biomass (g DW m^{-2})	Standard loading	708	42	373
	–17.5% loading	708	42	336
	–35% loading	707	42	375
Epiphyte risk score	Standard loading	3.42	1.10	2.70
	–17.5% loading	3.26	1.09	2.39
	–35% loading	3.02	1.06	2.21

good example of the complex interactions that occur in ecosystems with strong benthic-pelagic coupling and the challenges associated with defining and quantifying appropriate measures for compliance with regulatory requirements. Both bottom-up and top-down control of primary production can play important roles in reducing primary symptoms of eutrophication such as elevated chlorophyll concentrations and overgrowth of epiphytes on SAV, and integrated modelling frameworks can be useful in analysing synergistic effects and helping to achieve balanced management solutions.

As an example, the disappearance of softshell clams from Ringkøbing Fjord in 2019 due to reduced salinity contributed to the spectacular algal bloom observed that year. The model framework also helped establish that the potential for nitrogen reduction was higher for wetlands than for removal of land from arable production, leading to stakeholders

prioritising wetlands as the main measure to reduce nutrient loading. In general, models play a key role in identifying data shortfalls and informing monitoring plans. As a consequence of this work, a new monitoring programme for epiphytes began in June 2025 and will lead to improvements in the epiphyte risk index, in particular through local tuning of thresholds and potential incorporation of other indicators, but in the present form it already allows for some interesting scenario analyses with respect to changes in nutrient loading to Ringkøbing Fjord and a better understanding of measures leading to seagrass recovery.

Since the softshell clams play an important role in eutrophication control, providing a key ecosystem service to the lagoon, the management community also recognised the need for better data on stocking density, production, spatial and temporal variability, and physiology of *Mya arenaria*.

The integration of both the natural system, through the application

Table 5
Impact of top-down control on a set of indicators for a selection of bottom boxes.

Indicator	Scenario	Box 29	Box 45	Box 47
Summer chl a ($\mu\text{g L}^{-1}$)	With <i>Mya arenaria</i>	1.3	1.3	1.6
	No top-down control	16.9	13.8	12.7
	Difference (%)	+1191	+955	+709
Winter DIN (mg L^{-1})	With <i>Mya arenaria</i>	1.8	1.6	1.6
	No top-down control	1.8	1.6	1.5
	Difference (%)	+1	+1	+1
Macrophyte + epiphyte biomass (g DW m^{-2})	With <i>Mya arenaria</i>	708	42	373
	No top-down control	705	33	53
	Difference (%)	–1	–22	–86
Epiphyte risk score	With <i>Mya arenaria</i>	3.42	1.10	2.70
	No top-down control	3.30	1.08	1.17
	Difference (%)	–4	–1	–57

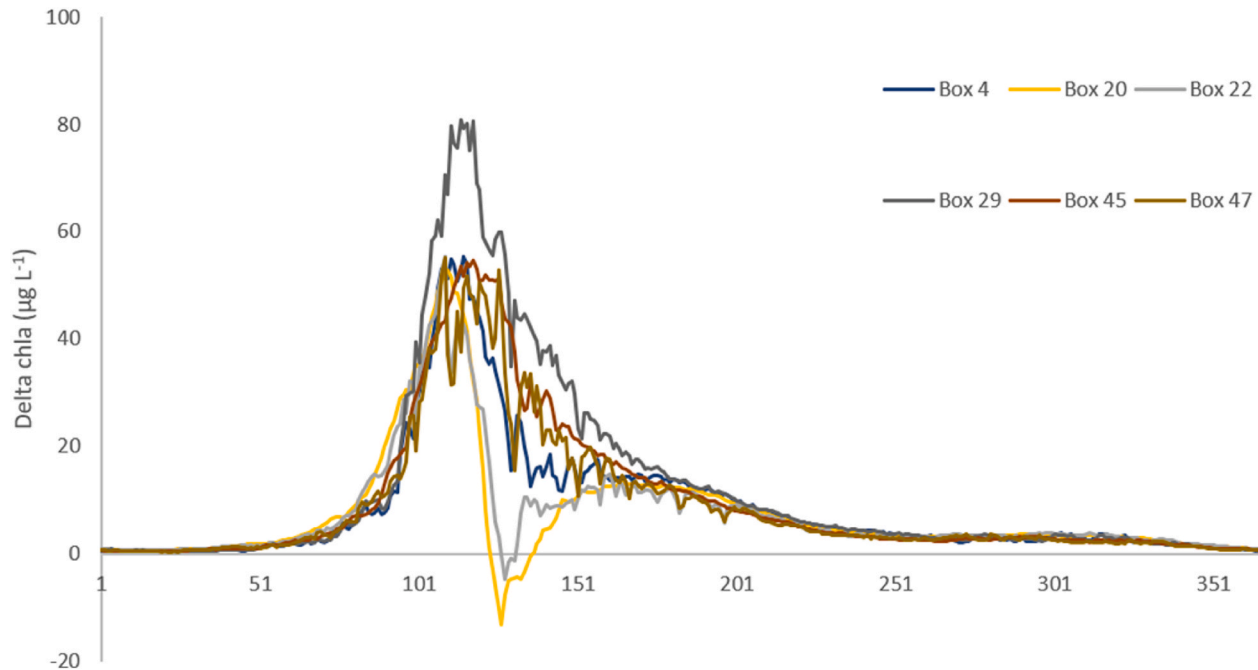


Fig. 9. Top-down control of eutrophication by *Mya arenaria* in six EWN boxes in Ringkøbing Fjord.

of a robust soil-to-sea ecological modelling framework, and the social system, through close involvement of stakeholders, resulted in trust and ownership of the key outcomes. The resulting plan has thus received widespread support from the community—this is a fundamental step in effecting change and ensuring successful policy implementation.

Models need to balance four key properties: generality, realism, accuracy, and simplicity. The EMF presented in this work simplifies reality by focusing on key state variables, while maintaining acceptable accuracy for management support. We have kept parameter tuning to a minimum to ensure that the framework has broad applicability.

Our approach preserves the numerical requirements and stability of the component models, particularly with respect to temporal and spatial constraints, and the ecological model itself is optimised for speed. The 50-box 3D model, running 9 ecological objects containing 60 state variables and 20 forcing functions and connected to 16 boundaries (14 of which with the SWAT+ hydrological model), performs a decadal run with a 15-min timestep in under 3 min.

All the models that integrate the framework can be deployed in any coastal system—all have been widely applied in such systems throughout the world. The innovation of this application is in coupling drivers and pressures from land with interactions at the ocean end-member to predict state and impact in an estuary or bay and to support management response.

The EMF can thus be generalised across a wide range of coastal systems (including all those that are part of the WFD), from Mediterranean lagoons to macrotidal estuaries, as long as the key processes are adequately resolved.

CRedit authorship contribution statement

Joao G. Ferreira: Writing – review & editing, Writing – original draft, Supervision, Software, Project administration, Methodology, Conceptualization. **Leonard Bernard-Jannin:** Writing – review & editing, Software, Methodology. **Alhambra Cubillo:** Writing – review & editing, Software, Methodology, Conceptualization. **Joao Lencart e Silva:** Validation, Software, Methodology, Data curation. **Gerhardus P. J. Diedericks:** Software, Methodology, Data curation. **Morten Holtegaard Nielsen:** Software, Methodology, Investigation, Formal analysis, Data curation. **Flemming Gertz:** Writing – review & editing, Supervision, Methodology, Funding acquisition, Conceptualization. **Lene Moth:** Validation, Project administration, Funding acquisition. **Helle Borum:** Validation.

Declaration of competing interests

The authors, whose names are listed immediately below, declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.ecss.2026.110291>.

Data availability

The authors do not have permission to share data.

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