Available online at www.sciencedirect.com

SCIENCE DIRECT®

Journal of Marine Systems 56 (2005) 375–390

JOURNAL OF MARINE SYSTEMS

www.elsevier.com/locate/jmarsys

Management of coastal eutrophication: Integration of field data, ecosystem-scale simulations and screening models

A.M. Nobre[a], J.G. Ferreira[a,*], A. Newton[b], T. Simas[a], J.D. Icely[c], R. Neves[d]

[a]IMAR-CEM, DCEA, FCT, Qta.Torre, 2829-516 Monte Caparica, Portugal
[b]IMAR, FCT, Gambelas Campus, Univ. Algarve, 8000-117 Faro, Portugal
[c]Sagremarisco, Apt 21, 8650-999 Vila do Bispo, Portugal
[d]Instituto Superior Técnico, Av. Rovisco Pais, 1096 Lisboa Codex, Portugal

Received 5 October 2004; accepted 3 March 2005
Available online 17 May 2005

Abstract

A hybrid approach for eutrophication assessment in estuarine and coastal ecosystems is presented. The ASSETS screening model (http://www.eutro.org) classifies eutrophication status into five classes: High (better), Good, Moderate, Poor and Bad (worse). This model was applied to a dataset from a shallow coastal barrier island system in southwest Europe (Ria Formosa), with a resulting score of Good. A detailed dynamic model was developed for this ecosystem, and the outputs were used to drive the screening model. Four scenarios were run on the research model: pristine, standard (simulates present loading), half and double the current nutrient loading. The Ria Formosa has a short water residence time and eutrophication symptoms are not apparent in the water column. However, benthic symptoms are expressed as excessive macroalgal growth and strong dissolved oxygen fluctuations in the tide pools. The standard simulation results showed an ASSETS grade identical to the field data application. The application of the screening model to the other scenario outputs showed the responsiveness of ASSETS to changes in pressure, state and response, scoring a grade of High under pristine conditions, Good for half the standard scenario and Moderate for double the present loadings. The use of this hybrid approach allows managers to test the outcome of measures against a set of well-defined metrics for the evaluation of state. It additionally provides a way of testing and improving the pressure component of ASSETS. Sensitivity analysis revealed that sub-sampling the output of the research model at a monthly scale, typical for the acquisition of field data, may significantly affect the outcome of the screening model, by overlooking extreme events such as occasional night-time anoxia in tide pools.

Keywords: Eutrophication; Estuarine and coastal ecosystems; Shallow lagoon; System analysis; Ecological models; Screening models; Portugal, Ria Formosa

1. Introduction

Models that address eutrophication in estuarine and coastal zones may be broadly divided into two

* Corresponding author. Tel.: +351 21 2948300; fax: +351 21 2948554.
E-mail address: joao@hoomi.com (J.G. Ferreira).

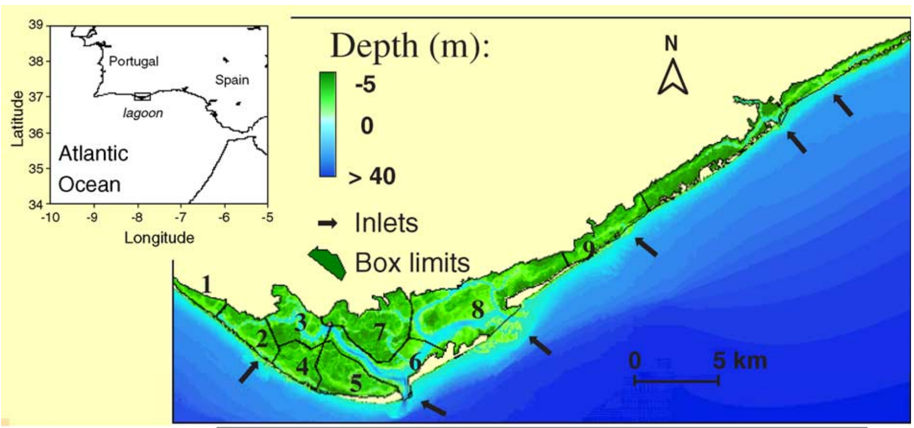
0924-7963/$ - see front matter © 2005 Elsevier B.V. All rights reserved.
doi:10.1016/j.jmarsys.2005.03.003

categories. Screening models constitute the first category, and are designed to provide an overview of trophic status based on a few diagnostic variables, which may include physical and biogeochemical processes (OSPAR, 2001; Tett et al., 2003; Bricker et al., 2003). Models of this type have existed for many years in freshwater (Dillon and Rigler, 1975; Jørgensen, 1976; Vollenweider et al., 1998) and are usually statistical or have simple dynamics. In regions of restricted exchange, screening models have been proposed by, e.g. Bricker et al. (2003) for estuaries, Stigebrandt (2001) for fjords and Tett et al. (2003) for coastal waters. Although these models differ in some of the underlying concepts, they share the following key properties:

1. They provide an integration of many complex processes into a simplified set of relationships and rates;
2. They provide an assessment of the state of a system on the basis of a few measured parameters, using ranges defined on theoretical and/or empirical grounds;
3. They act as a link between data collection, interpretation and coastal management;
4. They are not designed for day-to-day management of a particular water body, but rather are used by managers to provide overviews and to make comparisons.

The second class of models consists of detailed simulations of water quality and ecology, often using many variables and/or high resolution (e.g. Le Gall et al., 2000; Cancino and Neves, 1999; Sohma et al., 2001; Chau and Jin, 2002). Many such models exist, often building on a physical template that describes the hydrodynamics and adding to it a range of processes which are linked to the production of organic matter (Radach and Moll, 1989; Lancelot et al., 1997; Lee et al., 2002; Alvera-Azcarate et al., 2003). Such approaches may be classified as research models, since they are useful tools to study environmental responses to changes in pressure (De Vries et al., 1996; Buzzelli et al., 1999; Moll and Radach, 2003) under specific conditions, but are difficult to interpret and use by non-specialists.

Recent legislation in the EU such as the Water Framework Directive (2000/60/EC), and similar regulatory pressure in the USA, has created a need for assessment tools which may be applied for water body classification, are simple to use, and provide a fair evaluation of quality status across different estuarine and coastal water body types.



Box	1	2	3	4	5	6	7	8	9
Volume (10^6 m^3)	2	7	13	5	8	41	10	50	7
Area (km^2)	2	5	7	5	8	10	10	22	5

Fig. 1. Study site morphology, and EcoWin2000 model boxes. Depths are referenced to tidal datum (negative values are intertidal). The eastern end of the lagoon was not included since it is a distinct hydrographic area.

The aim of this paper is to present a hybrid approach which combines the two model categories described earlier for eutrophication assessment in estuarine and coastal ecosystems, in order to show how simple screening models may be used by managers at a local scale with data and results from complex models.

The objectives of the work were:

1. To calibrate and validate a complex research model using field data;
2. Use both the dataset and model outputs to drive a simple screening model;
3. To define and apply different watershed usage scenarios to force the research model;
4. Use the research model outputs to test the responsiveness of the screening model to these various nutrient loading scenarios.

2. Methods

2.1. Study site

The models were applied to a shallow (mean depth: 1.5 m) lagoon located in a sheltered coastal area of Southern Europe. The Ria Formosa is a hypersaline barrier island lagoon system in Portugal, connected to the ocean by six inlets (Fig. 1). The semi-diurnal tidal exchange is significantly greater than the residual

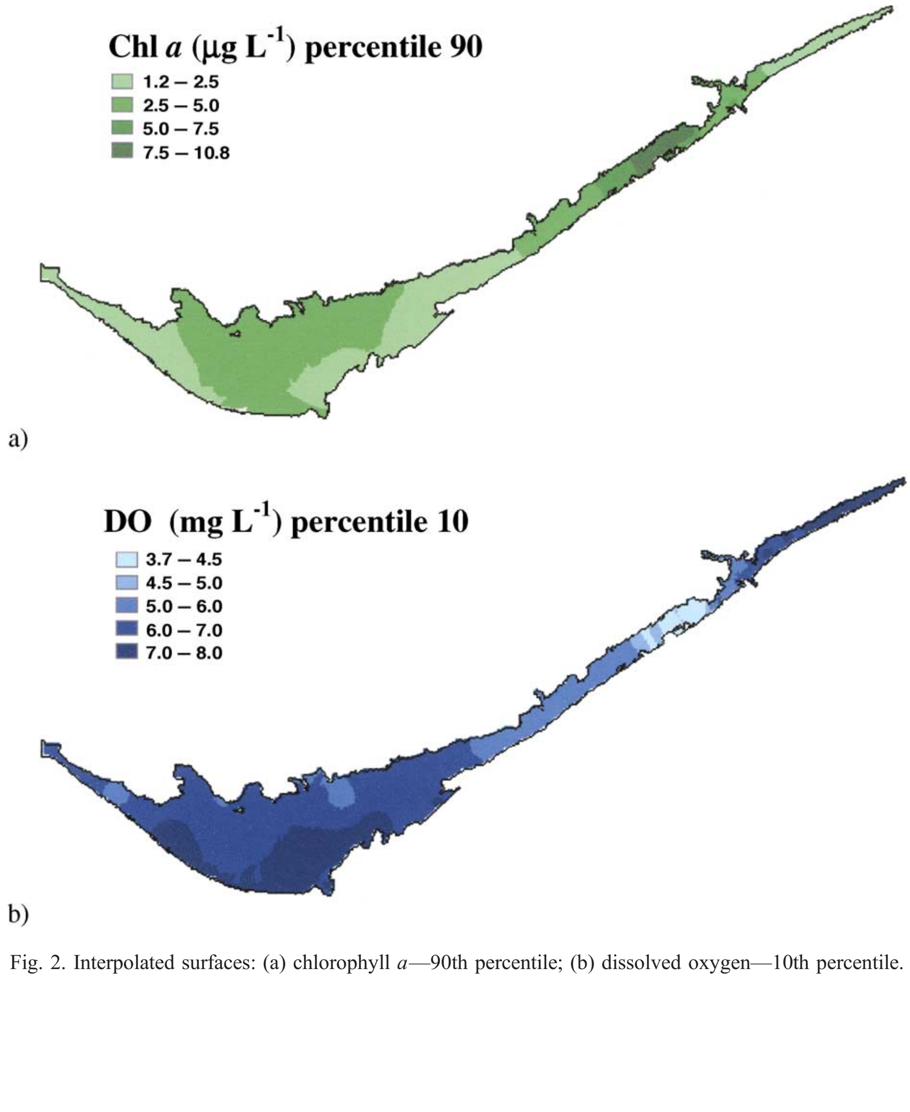


Fig. 2. Interpolated surfaces: (a) chlorophyll *a*—90th percentile; (b) dissolved oxygen—10th percentile.

volume and freshwater inputs are negligible. The main sources of nutrients are point-source discharges from a population of 150 000 inhabitants. The Ria has a wide range of uses, including tourism, extraction of salt and sand, fisheries, and aquaculture. Clam (*Ruditapes decussatus*) aquaculture provides a (total fresh weight) yield of 8000 t year^{-1}.

The pelagic primary production within the lagoon is strongly limited by the fast water turnover (Ketchum, 1954; Le Pape and Menesguen, 1997; Valiela et al., 1997). The spatial distribution of the pelagic variables is shown in Fig. 2. The combination of nutrient peaks, shallow water, large intertidal area and short water residence time (approximately 1 day) results in benthic eutrophication symptoms such as large macroalgal blooms (Coffaro and Sfriso, 1997; Deegan et al., 2002). The maximum values of macroalgal biomass observed in the Ria Formosa reach about 2 kg DW m^{-2}.

A full description of the study site is presented in Brotas et al. (1990), Sprung (1994), Sobral and Widdows (1997), Falcão and Vale (1998) and Newton et al. (2003).

2.2. Data acquisition and analysis

A historical dataset for Ria Formosa was assembled from the work of Newton et al. (2003), and references therein, and complemented by a detailed data acquisition program (Tett et al., 2003).

The data program was designed according to the modelling requirements and aimed to resolve tidal and seasonal variability. Ocean boundary conditions were established through periodic sampling campaigns over an annual cycle. Synoptic water quality sampling was carried out at a range of stations from the inlets to the inner lagoon and included pristine areas and locations in the vicinity of point-source nutrient discharges. Data on biomass, spatial coverage and productivity of the main species of opportunistic macroalgae (*Ulva* sp. and *Enteromorpha* sp.) were obtained, using a variety of techniques ranging from in situ and laboratory incubation experiments (Serpa, 2005) to geographical information system (GIS) analysis of the bathymetry. The targeted sampling on both ocean boundary conditions and internal processes was complemented by the deployment of continuous data loggers.

All data were stored in a relational database (Table 1). The database was exploited to obtain initial and boundary conditions for model state variables, derive relationships for implementation of processes (e.g. to determine limiting nutrients) and for model calibration and validation.

2.3. Research model concept and implementation

The modelling domain was divided into nine boxes (Fig. 1) and an ecological model (EcoWin 2000, Ferreira, 1995) was developed to simulate physical and biogeochemical state variables for multi-year runs. The water fluxes between boxes and across the ocean boundaries were calculated by integration of results obtained using a 3D model, which uses hydrostatic and Boussinesq approximations (Martins et al., 2001). This model (MOHID) has been used for a wide range of coastal and estuarine systems, e.g. the

Table 1
Data details (sampled and historical)

Stations	Parameters	Samples	Results
69	165	97 021	139 932
Number of campaigns	Date	Parameters[a]	
152	1984–2002, discontinuously as part of 17 different research programmes	*Physical* Atmospheric pressure; Water height; Depth; Wind speed; Wind direction; Air temperature; Dew temperature; Current velocity; Water temperature; Radiation; Salinity; pH; Suspended particulate matter. *Nutrients* Ammonia; Nitrite; Nitrate; Phosphate; Silicate; Total organic nitrogen; Total organic phosphorus; Total N; Total P. *Others* Dissolved oxygen; Chlorophyll *a*; Phaeopigments; Primary production; Photosynthetic efficiency.	

[a] Different sampling programmes measured different parameters in various areas of the Ria Formosa.

Western Scheldt and Gironde estuaries (Cancino and Neves, 1999), the Sado estuary (Martins et al., 2001) and Ria Formosa (Silva et al., 2001, 2002). The model was used with only one vertical layer, behaving as a 2D depth-integrated model, with a grid of 140 000 cells and a 5-s timestep. A spring–neap tide period was simulated using MOHID, and the water fluxes were integrated in time and space, using box boundaries defined by GIS on the hydrodynamic model bathymetry. The nine box ecological model (hereafter termed *research model*) assimilated these outputs offline by cyclically running the spring–neap tide period data with a 30-min timestep over a 4-year period. Fig. 3a shows the volume variation in four boxes of the research model, obtained by upscaling the water fluxes given by the hydrodynamic model. The relation obtained between tidal height determined from harmonic constants and the box volume calculated using the model (Fig. 3b) is significant ($P<0.001$).

The research model includes both pelagic and benthic state variables (see, e.g. Nunes et al., 2003 and references therein) and explicitly accounts for land-based nutrient inputs, calculated using population equivalents (PEQ) for urban effluents and GIS for estimating diffuse watershed contributions. The main model equations are given in Table 2. Only nitrogen was considered to be limiting for primary production, since the median Redfield ratio (in atoms, 621 samples, over a 5-year period) is 10.6. Where appropriate, biogeochemical equations were used to describe individual growth (e.g. Hawkins et al., 2002), and then coupled to a population dynamics model, as described, e.g. by Nunes et al. (2003) for multi-species aquaculture. This approach was used for simulating clam seeding, growth and harvest, and was also used to simulate the growth of opportunistic macroalgal species. In both the clam and macroalgal population dynamics models five weight classes were used (Table 2; Eq. (7)).

Growth of individual algae was forced by the underwater light climate, tidal immersion period (Alvera-Azcarate et al., 2003), and by nutrient availability, using a cell quota model following Solidoro et al. (1997) by means of Eqs. (4) and (5) (Table 2). Data for initial conditions, calibration and validation were obtained in the Ria Formosa as described in Section 2.2. For the seaweed modelling, biomass data, productivity–light intensity (P–I) curves and cellular composition of *Enteromorpha* and *Ulva* species sampled in Ria Formosa were complemented with results from Pedersen and Borum (1996, 1997), Lotze et al. (1999), Solidoro et al. (1997) and Vergara et al. (1998).

One of the key impacts of macroalgal growth driven by nutrient enrichment in shallow water tidal systems is a high variability in dissolved oxygen in intertidal areas, at a very fine spatial scale. The research model described herein is unable to simulate oxygen fluctuations at such a fine scale, because of the box size and strong flushing of the system. Even a detailed hydrodynamic model with, e.g. a grid size of 100 m would consider 1 ha boxes, which would be an insufficiently fine scale to simulate local oxygen deficiency, production of H_2S and benthic mortality. However, this symptom is an important entry point for applying the ASSETS screening model, and in order to address this, a small-scale tide pool model was developed and driven using boundary conditions determined from the research model. The tide pool model is a proxy for the larger intertidal areas within system boxes, but not for the larger system boxes as a whole.

For each box, a small (10 m × 10 m) shallow tide pool was defined for simulating the dissolved oxygen budget. The tidal elevation was used to determine the period of the tidal cycle when the pool was connected to the channel (see, e.g. Alvera-Azcarate et al., 2003). During this period, oxygen was freely exchanged between the pool and the adjoining large box, parameterised by means of an exchange coefficient E (day^{-1}). During the latter part of the ebb and early flood, the pool was disconnected from the main channel, and considered to maintain a constant volume.

Oxygen exchange at the air–water interface was simulated after Chapra (1997) over the entire tidal cycle (Eq. (1)) and the oxygen flux due to photosynthesis and respiration of macroalgae was determined based on the net productivity per unit area for all seaweed classes, as described above. This tide pool model was used to examine fluctuations of dissolved oxygen under different conditions of nutrient loading.

$$\frac{\partial O}{\partial t} = \frac{0.108 U_{\mathrm{w}}^{1.64}}{H}\left(\frac{S_c}{600}\right)^{0.5} \cdot (O_{\mathrm{s}} - O) \qquad (1)$$

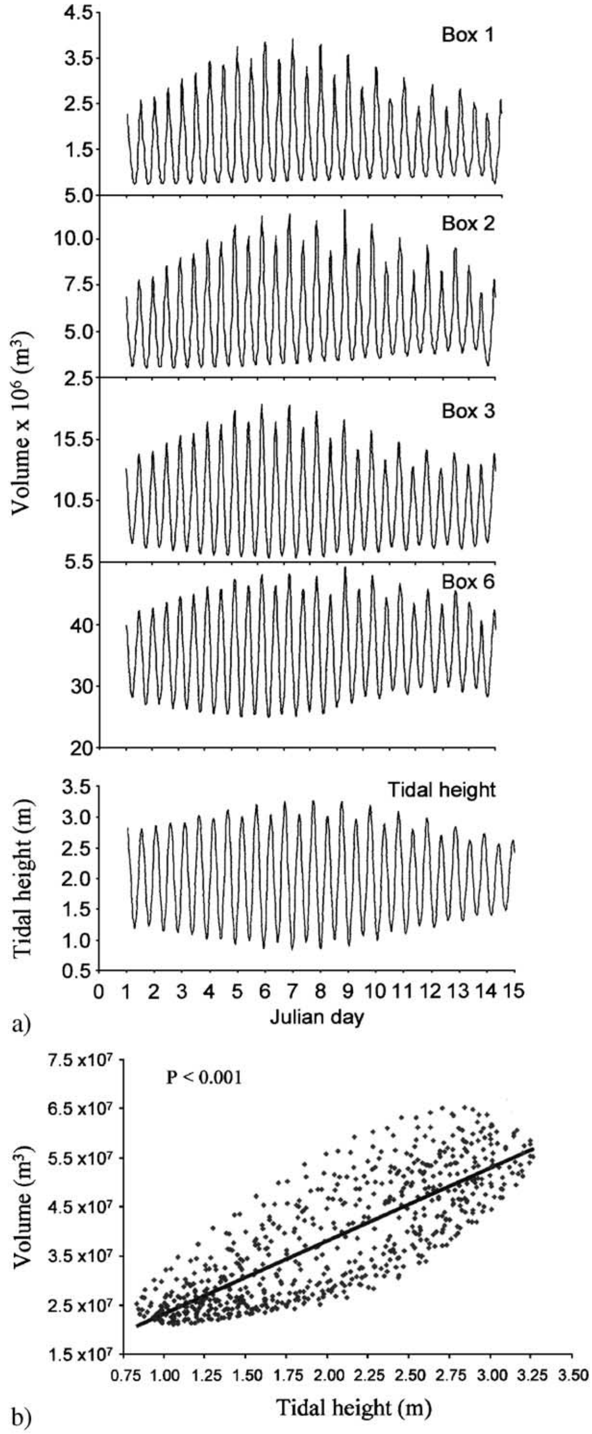


Fig. 3. Comparison of water exchanges calculated using fluxes from the hydrodynamic model with tidal elevation determined from harmonic constants at the tide gauge in the main port area of the Ria Formosa.

Table 2
Main equations of the EcoWin2000 research model

Phytoplankton			
$\frac{dB}{dt} = B \cdot (p_{\max b} \cdot f(I) \cdot f(N) - r_b - e_b - m_b - c_c C)$			(1)
B	Phytoplankton biomass (expressed as carbon)	μg C L^{-1}	
$p_{\max b}$	Phytoplankton maximum gross photosynthetic rate	day^{-1}	
$f(I)$	Steele's equation for productivity with photoinhibition		
$f(N)$	Michaelis–Menten function for nutrient (N) limitation		
r_b	Phytoplankton respiration rate	day^{-1}	
e_b	Phytoplankton exudation rate	day^{-1}	
m_b	Phytoplankton natural mortality	day^{-1}	
c_c	Bivalve (C) grazing rate	ind^{-1} day^{-1}	
Dissolved inorganic nitrogen			
$\frac{dN}{dt} = \alpha \cdot B \cdot (e_b + m_b) + e_c \cdot C - \alpha \cdot B(p_{\max b} \cdot f(I) \cdot f(I) \cdot f(N)) - p_{\max}$			(2)
N	Dissolved inorganic nitrogen	μmol L^{-1}	
α	Conversion to nitrogen units		
e_c	Bivalve (C) excretion	μmol L^{-1} ind^{-1}	
$p_{\max}$	Potential macroalgal production	day^{-1}	
Particulate matter			
SPM	Suspended particulate matter, used as a forcing function in the model from field measurements	mg L^{-1}	
POC	Particulate organic matter, calculated with an empirical equation derived from the % of POC in SPM	mg C L^{-1}	
Bivalve (Ruditapes decussatus) individual scope for growth			
$SFG = FR - (Pf + F + MR)$			(3)
SFG	Bivalve scope for growth	g day^{-1}	
FR	Filtration rate	g day^{-1}	
Pf	Pseudofaeces production	g day^{-1}	
F	Faeces production	g day^{-1}	
MR	Metabolic rate	g day^{-1}	
Macroalgal scope for growth (nutrient cell quota (4), production (5) and biomass (6))			
$\frac{dQ}{dt} = \frac{\mu_{\max} S}{k_s + S} \cdot \frac{q_{\max} - Q}{q_{\max} - q_{\min}} - p_{\max} \cdot \frac{Q - q_{\min}}{Q - k_c} \cdot Q$			(4)
$P = p_{\max} \cdot \frac{Q - q_{\min}}{Q - k_c}$			(5)
$\frac{dM}{dt} = p_{\max} \cdot \frac{(Q - q_{\min})}{(Q - k_c)} \cdot M - (r_m + e_m) \cdot M$			(6)

Table 2 (*continued*)

Macroalgal scope for growth (nutrient cell quota (4), production (5) and biomass (6))			
P	Macroalgal production	day^{-1}	
M	Macroalgal biomass	g dw	
Q	Cell quota	mg N g dw^{-1}	
$\mu_{\max}$	Maximum DIN uptake rate from the water column	mg N g dw^{-1} day^{-1}	
S	DIN concentration	μmol L^{-1}	
k_s	Half-saturation constant for DIN	μmol L^{-1}	
$q_{\max}$	Maximum DIN cell quota	mg N g dw^{-1}	
$q_{\min}$	Minimum DIN cell quota	mg N g dw^{-1}	
k_c	Nitrogen cell quota for growth	mg N g dw^{-1}	
r_m	Macroalgal respiration rate	day^{-1}	
e_m	Macroalgal exudation rate	day^{-1}	
Population dynamics model (applied to bivalves and macroalgae)			
$\frac{\partial S(s,t)}{\partial t} = -\frac{\partial [S(s,t) \cdot \eta(s,t)]}{\partial s} - \mu(s) \cdot S(s,t)$			(7)
S_s	Number of individuals for each class weight s	ind	
η	Organism scope for growth	g day^{-1}	
μ_s	Mortality rate for the sth class	day^{-1}	

where: O: dissolved oxygen (mg L^{-1}); U_w: wind speed (m s^{-1}); S_c: Schmidt number (taken to be 500 for oxygen in water); H: pool depth (m); O_S: saturation concentration of dissolved oxygen (mg L^{-1}).

The EcoWin2000 ecological modelling platform (Ferreira, 1995) was used to implement the research model simulating the principal biogeochemical processes of pelagic and benthic eutrophication. Water and pelagic state variables were redistributed within the Ria Formosa and exchanged with the ocean using the flows calculated from the hydrodynamic model, and appropriate forcing was imposed at the land and ocean boundaries for salinity, nutrients and phytoplankton.

The model steady state conditions were verified through mass balance closure for conservative and non-conservative state variables, for the hydrodynamic and biogeochemical components, respectively. The model was calibrated using a dataset from 1988 and validated for the reference period of 2001, which is considered herein to represent present conditions, i.e. the standard model.

Multiyear simulations were performed with the research model for several pressure scenarios. The standard model (1S) was set up for simulating the present anthropogenic nitrogen input, 40 kg N ha^{-1} year^{-1}, i.e. the nutrient loading defined as the land boundary condition. Changes in the standard nutrient loads were simulated for decreasing (0.5S, half the load), and increasing (2S, double the load) anthropogenic pressure, as well as for pristine conditions (0S, no land-derived load). The pristine scenario is important for the definition of reference conditions for water quality management, as required, e.g. by EU Directive 2000/60/EC (Water Framework Directive).

2.4. Screening model

The ASSETS screening model (Bricker et al., 2003) was chosen as an integrated approach for eutrophication assessment. This model, together with its predecessor (National Estuarine Eutrophication Assessment—NEEA, Bricker et al., 1999), has been applied to a wide range of estuarine and coastal systems both in the U.S and EU (Bricker et al., 1999; Ferreira and Bricker, 2004), and is available at http://www.eutro.org. The model provides an overall classification of the system by aggregating the results of three diagnostic indices (Bricker et al., 2003): an index of pressure (Overall Human Influence, OHI), a symptoms-based evaluation of state (Overall Eutrophic Conditions, OEC), and an indicator of management response (Definition of Future Outlook, DFO). The OHI uses a simple mass balance model based on land nutrient loading and system susceptibility (equations to determine the OHI are shown in Bricker et al., 2003). The OEC is calculated by aggregating primary and secondary eutrophication symptoms, using a combination matrix (Fig. 4). The symptoms are evaluated using a logical decision process (Bricker et al., 2003) applied to the variables chlorophyll a, epiphytes and macroalgae for the Primary Symptoms Method (PSM) and dissolved oxygen, submerged aquatic vegetation (SAV) loss and nuisance and toxic blooms for the Secondary Symptom Method (SSM). The DFO is determined based on an assessment of the susceptibility of the system and its foreseeable evolution and is graded into five classes (from better to worse): Improve High, Improve Low, No Change, Worsen Low and Worsen High.

The calculation of the pressure component in ASSETS required developments to accommodate the

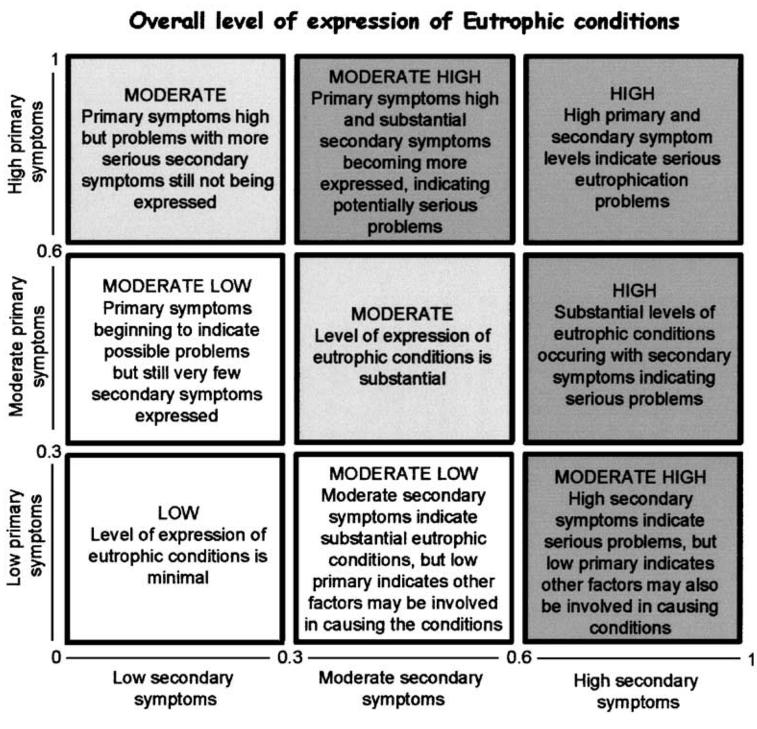


Fig. 4. ASSETS matrix for calculation of Overall Eutrophic Condition (OEC).

negligible freshwater inputs to fully saline coastal systems or inverse estuaries. A simplified tidal prism approach replaced the flushing time method used in the calculation of the susceptibility of the OHI index (Bricker et al., 2003).

2.5. Coupling of research and screening models

Model coupling was implemented by using the research model outputs to drive the screening model. The results of the screening model State Component (OEC) were calculated using: (a) field data, and (b) standard model results. The pressure scenarios were simulated with the research model and the corresponding trophic condition evaluated with the screening model. The screening model results were compared, and the responsiveness of the model to changes in pressure was tested.

The State component of ASSETS was applied to the research model outputs using the OEC index. The 90th percentile for chlorophyll *a* and 10th percentile for dissolved oxygen were calculated using a monthly random sample of the research model outputs to reproduce the typical sampling frequency of field data used for ASSETS. The classification within the symptom level of expression categories was carried out using the thresholds defined in ASSETS (Bricker et al., 2003) for chlorophyll *a* and dissolved oxygen. The research model outputs of macroalgal growth were evaluated using the results of the highest weight class (16–20 g total fresh weight) as a proxy, since this is considered to be the most problematic, due to the smothering of benthic fauna and seagrasses. Larger seaweeds are also more easily detached (Peckol and Rivers, 1996; Lehvo and Back, 2001), potentially leading to a wider distribution of eutrophication symptoms. Quantitative ranges for macroalgae are by definition hard to define, because this benthic eutrophication symptom may be expressed in a variety of ways, including excessive areal coverage and/or biomass, low oxygen problems, smothering of macrofauna, etc. The opportunistic nature of nutrient-related macroalgal blooms causes additional difficulties for quantification. This symptom is therefore classified heuristically on the

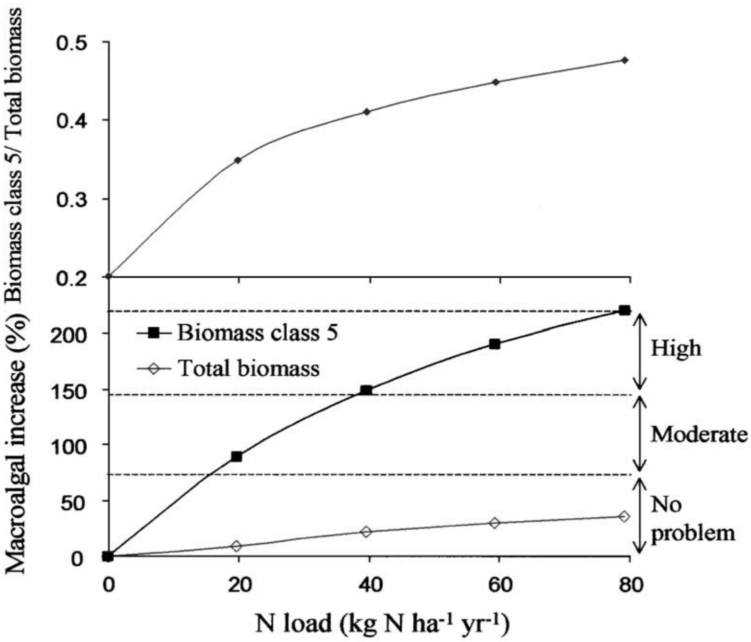


Fig. 5. Research model results of macroalgal growth in box 1 as a function of different nutrient loads: (a) variation of the ratio: biomass class 5/ total biomass; (b) % increase of total biomass and of biomass class 5 divided into the OEC symptom expression classes (class 5 corresponds to the highest weight class).

basis of the research model results of macroalgal growth in box 1 (where the worst symptoms occur) as a function of different nutrient loads (Fig. 5b).

Epiphytes, SAV, and nuisance and toxic blooms are not simulated in the research model, therefore the values used in the scenarios were those obtained for the OEC application to field data.

The sensitivity of the screening model to input data was tested by changing the frequency of the research model outputs used to drive the OEC. The screening model results were tested for the 2S scenario using a monthly subset of research model results (during the day, neap or spring tide) and the complete dataset.

The full ASSETS score combining pressure, state and response was determined by:

1. Calculating the pressure (OHI) values for each scenario;
2. Using the research model outputs to determine state (OEC) as described above;
3. Determining the response (DFO) as described below.

The DFO was estimated for the standard (1S) model according to the methodology presented in the screening model description, based on information provided in the management plans for the coastal lagoon and drainage basin. For the scenarios, the DFO was determined heuristically using the standard (1S) model as a reference. The different loading scenarios are interpreted as a result of a management process: e.g. the 2S scenario would be the result of a Worsen Low response option (e.g. urban and industrial development) taking the 1S case as a reference, which would lead to a doubling of nutrient loads. A snapshot of the system taken at the 2S stage would provide the OHI and OEC values simulated in the research model that are used to drive the screening model. Any corrective management action taken at that stage would not have immediate effects due to the lag between response, change in pressure and modification of state, and additionally, considering Worsen Low as the DFO rating at the 2S stage has precautionary value.

3. Results and discussion

3.1. Application of the research model—field data simulation

Fig. 6 shows the research model validation results for phytoplankton and dissolved inorganic nitrogen, two key variables in eutrophication assessment. The results shown are for an inner box (box 1) which exchanges water and pelagic properties only with box 2, and for a box with highly energetic exchange connected to the ocean (box 2). The general pattern and timing of the annual production cycle is well

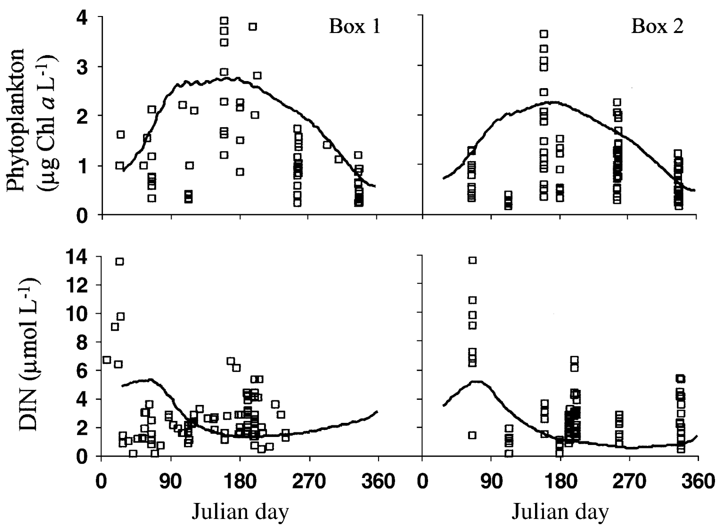


Fig. 6. Research model validation. Lines correspond to standard model simulations and points to mean field data.

reproduced by the research model; however, the model underestimates the dissolved inorganic nitrogen (DIN) measured in the system, which may be due to point-source inputs not being considered in the model. The measured phytoplankton data indicate that the model slightly overestimates this variable, although the results are within the standard deviation of the data.

3.2. Application of the screening model using field data and the standard model simulation

The ASSETS grade obtained using the field data was Good, based on the Moderate Low conditions obtained for pressure and state, given by OHI and by OEC, respectively (Table 3).

Although the research model results slightly overestimate observed phytoplankton data, similar results were obtained for the PSM using the available dataset and using the standard model outputs (Table 3). In both cases the percentile 90 value for chlorophyll *a* was below the screening model threshold for the Low category, 5 μg chl *a* L^{-1}, which corresponds to value of 0.25, as shown in Table 3.

The resulting OEC classification using both the field data and the standard model results was Moderate Low. Identical values for OHI and DFO were used, resulting in an overall score of Good (Table 3).

3.3. Application of research and screening models to different nutrient loading scenarios

Table 4 presents the research model results obtained for the main primary producers (phytoplankton and macroalgae) for scenarios 0.5S, 1S and 2S. The increase of nutrient loads from pristine conditions to double the standard loads resulted in a 38% phytoplankton biomass increase, with a maximum (percentile 90) value of 2.6 μg chl *a* L^{-1} (Table 4). Phytoplankton growth is limited by the fast water turnover that flushes the cells (Ketchum, 1954). This limitation has been reported for many systems (Brest Bay, France: Le Pape and Menesguen, 1997; Waquoit Bay, USA: Valiela et al., 1997; San Antonio Bay, USA: MacIntyre and Cullen, 1996). Chlorophyll *a* has a Low OEC grade for all the research model scenarios (Table 5).

The occurrence of excessive macroalgal growth in conditions of increasing nutrient pressure is commonly seen in shallow coastal systems with low residence times (Peckol and Rivers, 1996; Solidoro et

Table 3
ASSETS application to field data and to the research model outputs for the 1S scenario

	Index	Method	Parameter	Value	Level of expression	Index result	ASSETS score
Field data	OHI[a]	Nutrient inputs based on susceptibility		0.32		Moderate Low	Good
	OEC[b]	PSM[c]	Chlorophyll *a*	0.25	0.57 Moderate	Moderate Low	
			Epiphytes	0.50			
			Macroalgae	0.96			
		SSM[d]	Dissolved oxygen	0	0.25 Low		
			SAV loss	0.25			
			Nuisance and toxic blooms	0			
	DFO[e]	Future nutrient pressure		Future nutrient pressures decrease		Improve Low	
Standard model (1S)	OHI[a]	Nutrient inputs based on susceptibility		0.32		Moderate Low	Good
	OEC[b]	PSM[c]	Chlorophyll *a*	0.25	0.57 Moderate	Moderate Low	
			Epiphytes	0.50			
			Macroalgae	0.96			
		SSM[d]	Dissolved oxygen	0	0.25 Low		
			SAV loss	0.25			
			Nuisance and toxic blooms	0			
	DFO[e]	Future nutrient pressure		Future nutrient pressures decrease		Improve Low	

[a] OHI—Overall Human Influence index.
[b] OEC—Overall Eutrophic Condition index.
[c] PSM—Primary Symptoms Method.
[d] SSM—Secondary Symptoms Method.
[e] DFO—Determination of Future Outlook index.

Table 4
Research model results of nutrient pressure effects on macroalgal biomass, phytoplankton biomass and gross primary production (GPP): values and percentage increase compared with 0S scenario

Scenario	Annual load to the system (kg N ha^{-1})	DIN (μmol L^{-1})		Phytoplankton biomass (μg chl *a* L^{-1})		Macroalgae (16–20 g) biomass (g DW m^{-2})		GPP (g C m^{-2} year^{-1}) Phytoplankton		GPP (g C m^{-2} year^{-1}) Macroalgae	
0.5S	20	3.7	3%	2.0	8%	242	18%	33	19%	144	18%
1S	40	4.7	30%	2.2	17%	280	36%	38	38%	166	37%
2S	80	5.3	48%	2.6	38%	303	48%	48	76%	204	68%

al., 1997; Valiela et al., 1997; Havens et al., 2001; Martins and Marques, 2002). There is a marked increase in macroalgal biomass for the highest weight class when compared to the total algal biomass, as the DIN load is increased (Fig. 5). This is an important result, indicating that although the overall seaweed biomass does not increase significantly under higher nutrient loads, the larger algae, responsible for smothering of benthic fauna and seagrasses, and also more easily detached (Peckol and Rivers, 1996; Lehvo and Back, 2001), become dominant in the system.

In order to apply the ASSETS screening model to the research model results, the percentage increase for

Table 5
ASSETS application to the 0S, 0.5S and 2S scenarios (differences from 1S, see Table 3, in italics)

	Index	Method	Parameter	Value	Level of expression	Index result	ASSETS score
Pristine (0S)	OHI	Nutrient inputs based on susceptibility		*0*		*Low*	*High*
	OEC	PSM	Chlorophyll *a*	0.25	*0.25 Low*	*Low*	
			Epiphytes	0.50			
			Macroalgae	0			
		SSM	Dissolved oxygen	0	0.25 Low		
			SAV loss	0.25			
			Nuisance and toxic blooms	0			
	DFO	Future nutrient pressure		Future nutrient pressures decrease		*Improve High*	
Half the present load (0.5S)	OHI	Nutrient inputs based on susceptibility		*0.19*		*Low*	Good
	OEC	PSM	Chlorophyll *a*	0.25	0.42 Moderate	Moderate Low	
			Epiphytes	0.50			
			Macroalgae	0.5			
		SSM	Dissolved oxygen	0	0.25 Low		
			SAV loss	0.25			
			Nuisance and toxic blooms	0			
	DFO	Future nutrient pressure		Future nutrient pressures decrease		*Improve*	
Double the present load (2S)	OHI	Nutrient inputs based on susceptibility		*0.49*		*Moderate*	*Moderate*
	OEC	PSM	Chlorophyll *a*	0.25	0.57 Moderate	Moderate Low	
			Epiphytes	0.50			
			Macroalgae	0.96			
		SSM	Dissolved oxygen	0	0.25 Low		
			SAV loss	0.25			
			Nuisance and toxic blooms	0			
	DFO	Future nutrient pressure		*Future nutrient pressures increase*		*Worsen Low*	

OHI—Overall Human Influence index; OEC—Overall Eutrophic Condition index; PSM—Primary Symptoms Method; SSM—Secondary Symptoms Method; DFO—Determination of Future Outlook index.

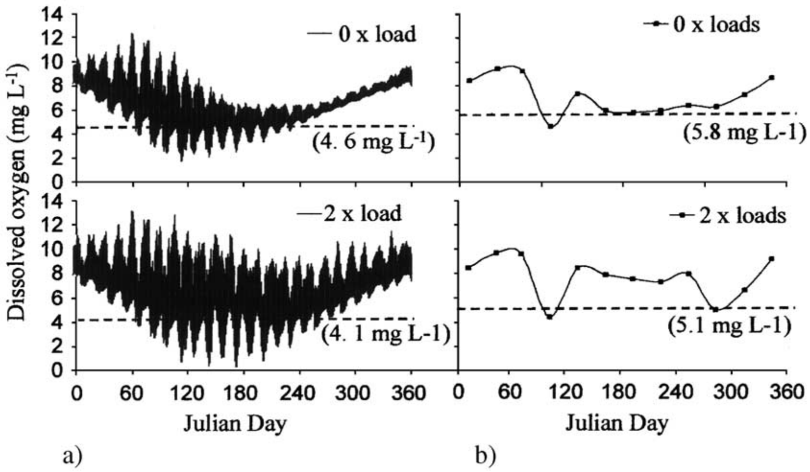


Fig. 7. Dissolved oxygen results for the research model and percentile 10 value (in dashed lines) for 0S and 2S scenarios in box 1: (a) complete dataset, and (b) monthly sub-sampling.

the highest seaweed weight class was divided linearly into three categories (Fig. 5b), ranging from pristine conditions (0S) to maximum DIN loading (2S). The results obtained for OEC are shown in Table 5.

The ASSETS results (Table 5) show that OEC is responsive to nutrient increase. These changes are essentially due to the variations obtained in the level of expression for macroalgal growth. The level of expression for chlorophyll *a* did not change in the different scenarios, which is a consequence of the high dilution potential of the system, and of the hydrodynamic limitations to bloom development. The overall ASSETS scores determined for the various scenarios are shown in Table 5. The OHI ranges from 0% at pristine conditions to 49% for the 2S case. The present DFO rating for Ria Formosa is Improve Low, but for the pristine (0S) scenario DFO was heuristically set as Improve High, for 0.5S as Improve Low, and for double the nutrient loading (2S) it was considered to be Worsen Low. The integrated ASSETS scores do well as a global indicator for changes in the system, with the pristine classification at the very top end of the High range of values defined by Bricker et al. (2003), and a worsening tendency from Good (0.5S, 1S) to Moderate in the 2S scenario.

The research model results for the 0S scenario and for the 2S scenario show that the increase in nutrient loading leads to high variability of the dissolved oxygen concentration in the intertidal pools (Fig. 7a.). However, the research model results obtained from the monthly data points hide this variability (Fig. 7b.), since the percentile 10 values of dissolved oxygen results were all above the 5 mg L^{-1} threshold defined for biological stress (Bricker et al., 1999) as shown in

Table 6
ASSETS application to the complete dataset of the 2S scenario (differences from this scenario using the monthly outputs in italics)

	Index	Method	Parameter	Value	Level of expression	Index result	ASSETS score
Complete dataset 2S	OHI	Nutrient inputs based on susceptibility		*0.49*		*Moderate*	*Poor*
	OEC	PSM	Chlorophyll *a*	0.25	0.57 Moderate	*Moderate*	
			Epiphytes	0.50			
			Macroalgae	0.96			
		SSM	Dissolved oxygen	*0.46*	*0.46 Moderate*		
			SAV loss	0.25			
			Nuisance and toxic blooms	0			
	DFO	Future nutrient pressure		*Future nutrient pressures increase*		*Worsen Low*	

Fig. 7b, resulting in the screening model scores presented in Tables 5 and 6.

The sensitivity analysis performed on the screening model simulation of the increasing pressure shows that using a detailed dataset there is a change in the OEC classification from Moderate Low to Moderate (Tables 5 and 6). The comparison between the screening model results shows that use of the higher frequency yields a decrease of the percentile 10 for dissolved oxygen (Fig. 7b) to below the 5 mg L^{-1} threshold defined for biological stress.

4. Conclusions

This work provides a link between data acquisition, complex models that are useful for detailed analysis of a particular system, and generic screening models designed for comparative eutrophication assessment. Because this type of screening model generally uses a reference dataset, system developments can only be evaluated either by spatial comparisons or by using a timeline for a particular system.

The use of such a reference dataset to validate a detailed model, which reproduces the key variables and processes of interest, allows for scenarios to be examined through simulation of management options that affect pressure. As a rule, the outputs of this type of model are too complex to be useful for management, but may be synthesized into a screening model that provides a set of simple indices. These outputs provide a benchmark for evaluation of potential outcomes by decision-makers.

There is an additional benefit to the use of research models in this way, since relevant state variables that have not been measured may be included in simulations, as exemplified by the determination of dissolved oxygen in tide pools. ASSETS currently uses only the oxygen concentration in the water column, and in highly energetic shallow systems such as the Ria Formosa, values will always be in the *No Problems* class. The use of complex models to refine the assessment procedure may give an indication of which facultative variables should be considered for specific water bodies. This is key to the development of screening models that are capable of correctly assessing eutrophication status across a range of physical types. Since the development of typology is seen as the next step in improving the performance of models such as ASSETS, by reducing misclassification, research models may also play a role in helping to define scientifically meaningful thresholds to manage against.

Acknowledgements

The authors would like to thank INAG and the EU OAERRE (EVK3-CT1999-0002) project for supporting this work, and M. Falcão, D. Serpa, J.D. Lencart, M. Sprung and C. Vale for data on the Ria Formosa. We are grateful to S.B. Bricker for helpful discussions on the application of ASSETS and for comments on the text, and to Ed Dettmann and an anonymous reviewer for helpful suggestions on a first draft of the manuscript.

References

Alvera-Azcarate, A., Ferreira, J.G., Nunes, J.P., 2003. Modelling eutrophication in mesotidal and macrotidal estuaries. The role of intertidal seaweeds. Estuarine, Coastal and Shelf Science 56 (4), 1–10.

Bricker, S.B., Clement, C.G., Pirhalla, D.E., Orlando, S.P., Farrow, D.R.G., 1999. National Estuarine Eutrophication Assessment. Effects of Nutrient Enrichment in the Nation's Estuaries. NOAA–NOS Special Projects Office, 1999.

Bricker, S.B., Ferreira, J.G., Simas, T., 2003. An integrated methodology for assessment of estuarine trophic status. Ecological Modelling 169, 39–60.

Brotas, V., Ferreira, A., Vale, C., 1990. Oxygen profiles in intertidal sediments of Ria Formosa (S. Portugal). Hydrobiologia 207, 123–129.

Buzzelli, C.P., Wetzel, R.L., Meyers, M.B., 1999. A linked physical and biological framework to assess biogeochemical dynamics in a shallow estuarine ecosystem. Estuarine, Coastal and Shelf Science 49, 829–851.

Cancino, L., Neves, R., 1999. Hydrodynamic and sediment suspension modelling in estuarine systems: Part II. Application to the Western Scheldt and Gironde estuaries. Journal of Marine Systems 22, 117–131.

Chapra, S.C., 1997. Surface Water-Quality Modelling. McGraw-Hill, NY.

Chau, K.W., Jin, H., 2002. Two-layered, 2D unsteady eutrophication model in boundary-fitted coordinate system. Marine Pollution Bulletin 45 (1–12), 300–310.

Coffaro, G., Sfriso, A., 1997. Simulation model of *Ulva rigida* growth in shallow water of the Lagoon of Venice. Ecological Modelling 102 (1), 55–66.

Deegan, L.A., Wright, A., Ayvazian, S.G., Finn, J.T., Golden, H., Merson, R.R., Harrison, J., 2002. Nitrogen loading alters seagrass ecosystem structure and support of higher trophic levels. Aquatic Conservation: Marine and Freshwater Ecosystems 12, 193–212.

De Vries, I., Philippart, C.J.M., DeGroodt, E.G., Van Der Tol, M.W.M., 1996. Coastal eutrophication and marine benthic vegetation: a model analysis. In: Schramm, W., Nienhuis, P.H. (Eds.), Marine Benthic Vegetation, Ecological Studies, vol. 123. Springer-Verlag, pp. 79–113.

Dillon, P.J., Rigler, F.H., 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. Journal of the Fisheries Research Board of Canada 32, 1519–1531.

Falcão, M., Vale, C., 1998. Sediment–water exchanges of ammonium and phosphate in intertidal and subtidal areas of a mesotidal coastal lagoon (Ria Formosa). Hydrobiologia 374, 193–201.

Ferreira, J.G., 1995. ECOWIN—an object-oriented ecological model for aquatic ecosystems. Ecological Modelling 79, 21–34.

Ferreira, J.G., Bricker, S.B., 2004. Application of the ASSETS model to US and European estuaries. Website for Assessment of Estuarine Trophic Status, NOAA/IMAR, http://www.eutro.org/syslist.aspx.

Havens, K.E., Hauxwell, J., Tyler, A.C., Thomas, S., McGlathery, K.J., Cebrian, J., Valiela, I., Steinman, A.D., Hwang, S.J., 2001. Complex interactions between autotrophs in shallow marine and freshwater ecosystems: implications for community responses to nutrient stress. Environmental Pollution 113, 95–107.

Hawkins, A.J.S., Duarte, P., Fang, J.G., Pascoe, P.L., Zhang, J.H., Zhang, X.L., Zhu, M.Y., 2002. A functional growth model of responsive suspension-feeding and growth in bivalve shellfish, configured and validated for the scallop *Chlamys farreri* during culture in China. Journal of Experimental Marine Biology and Ecology 281, 13–40.

Jørgensen, S.E., 1976. A eutrophication model for a lake. Ecological Modelling 2, 147–162.

Ketchum, B.H., 1954. Relation between circulation and planktonic populations in estuaries. Ecology 35 (2), 191–200.

Lancelot, C., Rousseau, V., Billen, G., Eeckhout, D.V., 1997. Coastal eutrophication of the southern bight of the North Sea: assessment and modelling. NATO Advanced Research Workshop on Sensitivity of North Sea, Baltic Sea and Black Sea to anthropogenic and climatic changes, Nov. 1995. NATO series, 15 pp.

Lee, J.Y., Tett, P., Jones, K., Jones, S., Luyten, P., Smith, C., Wild-Allen, K., 2002. The PROWQM physical–biological model with benthic–pelagic coupling applied to the northern North Sea. Journal of Sea Research 48, 287–331.

Le Gall, A.C., Hydes, D.J., Kelly-Gerreyn, B., Slinn, D.J., 2000. Development of a 2D horizontal biogeochemical model for the Irish Sea DYMONIS. ICES Journal of Marine Science 57, 1050–1059.

Lehvo, A., Back, S., 2001. Survey of macroalgal mats in the Gulf of Finland, Baltic Sea. Aquatic Conservation: Marine and Freshwater Ecosystems 11, 11–18.

Le Pape, O., Menesguen, A., 1997. Hydrodynamic prevention of eutrophication in the Bay of Brest (France), a modelling approach. Journal of Marine Systems 12, 171–186.

Lotze, H.K., Schramm, W., Schories, D., Worm, B., 1999. Control of macroalgal blooms at early developmental stages: *Pilayella littoralis* versus *Enteromorpha* spp. Oecologia 119, 46–54.

MacIntyre, H.L., Cullen, J.J., 1996. Primary production by suspended and benthic microalgae in a turbid estuary: time-scales of variability in San Antonio Bay, Texas. Marine Ecology. Progress Series 145, 245–268.

Martins, I., Marques, J.C., 2002. A model for the growth of opportunistic macroalgae (*Enteromorpha* sp.) in tidal estuaries. Estuarine, Coastal and Shelf Science 55 (2), 247–257.

Martins, F., Neves, R., Leitão, P., Silva, A., 2001. 3D modelling in the Sado estuary using a new generic vertical discretization approach. Oceanologica Acta 24 (1), s51–s62.

Moll, A., Radach, G., 2003. Review of three-dimensional ecological modelling related to the North Sea shelf system—Part 1: models and their results. Progress in Oceanography 57, 175–217.

Newton, A., Icely, J.D., Falcao, M., Nobre, A., Nunes, J.P., Ferreira, J.G., Vale, C., 2003. Evaluation of eutrophication in the Ria Formosa coastal lagoon, Portugal. Continental Shelf Research 23 (17–19), 1945–1961.

Nunes, J.P., Ferreira, J.G., Gazeau, F., Lencart-Silva, J., Zhang, X.L., Zhu, M.Y., Fang, J.G., 2003. A model for sustainable management of shellfish polyculture in coastal bays. Aquaculture 219, 257–277.

OSPAR, 2001. Draft Common Assessment Criteria and their Application within the Comprehensive Procedure of the Common Procedure. Meeting of the eutrophication task group (ETG), London, 9–11 October 2001. OSPAR convention for the protection of the marine environment of the North-East Atlantic.

Peckol, P., Rivers, J.S., 1996. Contribution by macroalgal mats to primary production of a shallow embayment under high and low nitrogen-loading rates. Estuarine, Coastal and Shelf Science 43, 311–325.

Pedersen, M.F., Borum, J., 1996. Nutrient control of algal growth in estuarine waters. Nutrient limitation and the importance of nitrogen requirements and nitrogen storage among phytoplankton and species of macroalgae. Marine Ecology. Progress Series 142, 261–272.

Pedersen, M.F., Borum, J., 1997. Nutrient control of estuarine macroalgae: growth strategy and the balance between nitrogen requirements and uptake. Marine Ecology. Progress Series 161, 155–163.

Radach, G., Moll, A., 1989. State of the art in algal bloom modelling. In: Lancelot, C., Billen, G., Barth, H. (Eds.), Water Pollution Research Report 12—Eutrophication and Algal Blooms in North Sea Coastal Zones, the Baltic and Adjacent Areas: Prediction and Assessment of Preventive Actions. Commission of the European Communities Press, Brussels, Belgium, pp. 115–149.

Serpa, D., 2005. Macroalgae (*Enteromorpha* spp. and *Ulva* spp.) Primary Productivity in the Ria Formosa Lagoon. MSc thesis, 75 pp. http://www.megamar.org/.

Silva, A., Leitao, R., Santos, R., Ciavola, P., 2001. Simplified modelling approaches to support the evaluation of site-specific relevant processes: Ria Formosa case study. ELOISE 4th Open Science Workshop, p. 175.

Silva, A.J.R., Leitão, P.C., Leitão, J.C., Braunschweig, F., Neves, R., 2002. Ria Formosa 3D hydrodynamic model. A contribution for the understanding of the Faro–Olhão inlet processes. Littoral 2002, The Changing Coast. EUROCOAST, Portugal, pp. 197–207.

Sobral, P., Widdows, J., 1997. Influence of hypoxia and anoxia on the physiological responses of the clam *Ruditapes decussatus* from southern Portugal. Marine Biology 127 (3), 455–461.

Sohma, A., Sekiguchi, Y., Yamada, H., Sato, T., Nakata, K., 2001. A new coastal marine ecosystem model study coupled with hydrodynamics and tidal flat ecosystem effect. Marine Pollution Bulletin 43 (7–12), 187–208.

Solidoro, C., Pecenik, G., Pastres, R., Franco, D., Dejak, C., 1997. Modelling macroalgae (*Ulva rigida*) in the Venice lagoon: model structure identification and first parameters estimation. Ecological Modelling 94, 191–206.

Sprung, M., 1994. Macrobenthic secondary production in the intertidal zone of the Ria Formosa—a lagoon in southern Portugal. Estuarine, Coastal and Shelf Science 38, 539–558.

Stigebrandt, A., 2001. FJORDENV—a water quality model for fjords and other inshore waters. Report C40. Earth Sciences Centre, Göteborg University, Göteborg.

Tett, P., Gilpin, L., Svendsen, H., Erlandsson, C.P., Larsson, U., Kratzer, S., Fouilland, E., Janzen, C., Lee, J.Y., Grenz, C., Newton, A., Ferreira, J.G., Fernandes, T., Scory, S., 2003. Eutrophication and some European waters of restricted exchange. Continental Shelf Research 23, 1635–1671.

Valiela, I., McClelland, J., Hauxwell, J., Behr, P.J., Hersh, D., Foreman, K., 1997. Macroalgal blooms in shallow estuaries: controls and ecophysiological and ecosystem consequences. Limnology and Oceanography 42 (5), 1105–1118.

Vergara, J.J., Sebastian, M., Pérez-Lloréns, J.L., Hernández, I., 1998. Photoacclimation of *Ulva rigida* and *U. Rotundata* (Chlorophyta) arranged in canopies. Marine Ecology. Progress Series 165, 283–292.

Vollenweider, R.A., Giovanardi, F., Montanari, G., Rinaldi, A., 1998. Characterisation of the trophic conditions of marine coastal waters with special reference to the NW Adriatic Sea: proposal for a trophic scale, turbidity and generalised water quality index. Environmetrics 9, 329–357.