Estuarine, Coastal and Shelf Science 87 (2010) 43–62

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

Assessment of coastal management options by means of multilayered ecosystem models

Ana M. Nobre[a,*], João G. Ferreira[a], João P. Nunes[b], Xiaojun Yan[c], Suzanne Bricker[d], Richard Corner[e], Steve Groom[f], Haifeng Gu[g], Anthony J.S. Hawkins[f], Rory Hutson[f], Dongzhao Lan[g], João D. Lencart e Silva[h], Philip Pascoe[f], Trevor Telfer[e], Xuelei Zhang[i], Mingyuan Zhu[i]

[a] Institute of Marine Research (IMAR), Centre for Ocean and Environment, New University of Lisbon, DCEA, FCT, Campus da Caparica, 2829-516 Caparica, Portugal
[b] CESAM & Department of Environment and Planning, University of Aveiro, Campus Universitário de Santiago, 3810-193 Aveiro, Portugal
[c] Marine Biotechnology Laboratory, Ningbo University, Ningbo 315211, PR China
[d] NOAA-National Ocean Service, National Centers for Coastal Ocean Science, 1305 East West Highway, Silver Spring, MD 20910, USA
[e] Institute of Aquaculture, University of Stirling, Stirling, FK9 4LA, United Kingdom
[f] Plymouth Marine Laboratory, Prospect Place, The Hoe, Plymouth PL1 3DH, Devon, United Kingdom
[g] Third Institute of Oceanography, Xiamen, 361005, PR China
[h] CESAM & Department of Physics, University of Aveiro, Campus Universitário de Santiago, 3810-193 Aveiro, Portugal
[i] First Institute of Oceanography, 6 Xianxialing Road, Qingdao 266061, PR China

ARTICLE INFO

Article history:
Received 4 June 2009
Accepted 20 December 2009
Available online 4 January 2010

Keywords:
integrated coastal zone management
multilayered ecosystem model
catchment-coastal model
sustainable aquaculture
carrying capacity
SE Asia
China
Zhejiang
Xiangshan Gang

ABSTRACT

This paper presents a multilayered ecosystem modelling approach that combines the simulation of the biogeochemistry of a coastal ecosystem with the simulation of the main forcing functions, such as catchment loading and aquaculture activities. This approach was developed as a tool for sustainable management of coastal ecosystems. A key feature is to simulate management scenarios that account for changes in multiple uses and enable assessment of cumulative impacts of coastal activities. The model was applied to a coastal zone in China with large aquaculture production and multiple catchment uses, and where management efforts to improve water quality are under way. Development scenarios designed in conjunction with local managers and aquaculture producers include the reduction of fish cages and treatment of wastewater. Despite the reduction in nutrient loading simulated in three different scenarios, inorganic nutrient concentrations in the bay were predicted to exceed the thresholds for poor quality defined by Chinese seawater quality legislation. For all scenarios there is still a Moderate High to High nutrient loading from the catchment, so further reductions might be enacted, together with additional decreases in fish cage culture. The model predicts that overall, shellfish production decreases by 10%–28% using any of these development scenarios, principally because shellfish growth is being sustained by the substances to be reduced for improvement of water quality. The model outcomes indicate that this may be counteracted by zoning of shellfish aquaculture at the ecosystem level in order to optimize trade-offs between productivity and environmental effects. The present case study exemplifies the value of multilayered ecosystem modelling as a tool for Integrated Coastal Zone Management and for the adoption of ecosystem approaches for marine resource management. This modelling approach can be applied worldwide, and may be particularly useful for the application of coastal management regulation, for instance in the implementation of the European Marine Strategy Framework Directive.

1. Introduction

Coastal zones provide considerable benefits to society while at the same time human activities exert pressure on coastal ecosystems, therefore threatening those same benefits (Nobre, 2009). To promote the sustainable use of coastal zone resources an ecosystem approach is of considerable value, firstly in understanding the causal relationships between environmental and socio-economic systems, and the cumulative impacts of the range of activities developed in coastal ecosystems (Soto et al., 2008; Nobre and Ferreira, 2009), and secondly to manage coastal resources and biodiversity (Browman and Stergiou, 2005; Murawski et al., 2008). Marine Ecosystem-Based Management (EBM) is an emerging scientific consensus complementary to Integrated Coastal Zone Management (ICZM). EBM

* Corresponding author.
E-mail address: ana@salum.net (A.M. Nobre).

0272-7714/$ – see front matter
doi:10.1016/j.ecss.2009.12.013

highlights the need to use the best available knowledge about the ecosystem in order to manage marine resources, with an emphasis on maintaining ecosystem service functions (Browman and Stergiou, 2005; Murawski, 2007; Murawski et al., 2008). In particular, improved planning and management of aquaculture production is highlighted as one of the sustainability issues related to coastal zone development and management that must urgently be addressed (GESAMP, 2001). Recently, several initiatives have occurred to support the development of an Ecosystem Approach to Aquaculture (EAA), which aims to integrate aquaculture within the wider ecosystem in order to promote the sustainability of the industry (Soto et al., 2008).

Ecosystem modelling is a powerful tool that can contribute the required scientific grounding for the adoption of such an Ecosystem-Based Management approach (Fulton et al., 2003; Greiner, 2004; Hardman-Mountford et al., 2005; Murawski, 2007). Specifically, modelling can be useful to: (1) provide insights about ecological interactions within the ecosystem (Raillard and Ménesguen, 1994; Plus et al., 2003; Dowd, 2005; Grant et al., 2008; Sohma et al., 2008; Dumbauld et al., 2009), (2) estimate the cumulative impacts of multiple activities operating on a given coastal area at an integrated catchment – marine ecosystem scale (Soto et al., 2008), and (3) evaluate the susceptibility of an ecosystem to pressures by means of scenario simulation (Hofmann et al., 2005; Nobre et al., 2005; Roebeling et al., 2005; Marinov et al., 2007; Ferreira et al., 2008a). James (2002), Fulton et al. (2003), and Moll and Radach (2003) have reviewed ecological models used in the simulation of the hydrodynamics and biogeochemistry of aquatic ecosystems. Such models vary widely according to their target application. For instance, aquaculture carrying capacity models can be developed at the farm scale (e.g., Ferreira et al., 2007, 2009; Cromey et al., 2009) or at the ecosystem scale (e.g., Dowd, 2005; Ferreira et al., 2008a). These models can focus on specific features of the environment such as seston biodeposition (Cromey et al., 2009; Weise et al., 2009), or can integrate the ecosystem biogeochemistry (Plus et al., 2003; Dowd, 2005; Grant et al., 2008; Ferreira et al., 2008a). Ecological models can also focus on how the environmental parameters affect the physiology of cultured species (e.g., Raillard and Ménesguen, 1994; Gangnery et al., 2004) or how aquaculture production affects the ecosystem as a whole (e.g., Grant et al., 2008; Weise et al., 2009). The role of models in evaluating the 'disturbances' caused by bivalve mariculture on coastal systems may be especially important in the USA where increasing regulations are in some cases being implemented on the basis of a rather strict interpretation of the precautionary principle, with a consequent restriction of aquaculture activities (Dumbauld et al., 2009). Concurrently, substantial efforts are also ongoing on the simulation of interactions between catchment and coast, for instance the work developed under the EuroCat ('European catchments, catchment changes and their impact on the coast') research project (Salomons and Turner, 2005). The work presented by Artioli et al. (2005), Hofmann et al. (2005) and Nikolaidis et al. (2009) exemplifies the existing modelling approaches including the interface between the biophysical and socio-economic models for the catchment and coastal systems.

Overall, if a model is to contribute to an Ecosystem-Based Management approach, it should integrate the range of key processes relevant to the questions asked, and thus allow simulation of the resulting cumulative impacts of human activities. For instance, to assist in the determination of ecological carrying capacity of aquaculture production, a model must include inputs from the multiple aquaculture farms situated in a given ecosystem and include simulation of other relevant activities, for example those within the catchment area that affect the coastal ecosystem such as agriculture and wastewater discharge and eventual treatment (Soto et al., 2008). Additionally, and particularly important for management, is the use of models for scenario simulation (Roebeling et al., 2005). This practice implies that management-relevant scenarios are developed to test changes in multiple uses or to explore impacts of global environmental changes (Hofmann et al., 2005; Nobre et al., 2005; Marinov et al., 2007; Ferreira et al., 2008a). This type of approach is crucial for EBM and requires close interaction with managers, decision-makers, and ecosystem and resource users (Ledoux et al., 2005; Nunneri and Hofmann, 2005). In addition, ecosystem stakeholders must be able to understand the information that models provide and also contribute information on the issues to be managed, so that model development addresses their particular needs. Ecological modelling was introduced as a management tool in the 1970's (Jørgensen and Bendoricchio, 2001); since then modelling tools have often proven useful in supporting the application and implementation of several legislative and management programmes worldwide, as exemplified in Table 1.

Ongoing research (Raick et al., 2006) is investigating trade-offs between (1) increasingly complex models that provide detailed simulations but require large datasets for model setup/validation (e.g., developed by Marinov et al., 2007) and generate outputs

Table 1
Examples of modelling tools used for the application of legislation and management programmes worldwide.

Legislation/management actions	Model application	Country/region
European Water Framework Directive (WFD, Directive, 2000/60/EC)	Hofmann et al. (2005), Artioli et al. (2005) and Volk et al. (2008)	European Union
CSIRO's Water for Healthy Country 'Floodplain renewal' program	'Landscape toolkit' developed for the management of the coastal strip adjacent to the Great Barrier Reef (Roebeling et al., 2005)	Australia
USA National Estuarine Eutrophication Assessment (NEEA) program	Eutrophication assessment model (Bricker et al., 2003). Also applied outside USA (Whitall et al., 2007; Borja et al., 2008).	USA, Europe and Asia
USA Clean Water Act (CWA).	Calculation of the total maximum daily load (TMDL) of a pollutant that a waterbody can receive and still safely meet water quality standards (EPA, 2008).	USA
Fisheries policy (management of the exploitation of aquatic renewable resources)	- Lobster fishery simulation to explore management options, regulations and the impact of environmental changes (Whalen et al., 2004) - Evolution of the Manila clam population in response to different management measures and to exceptional changes in environmental conditions (Bald et al., 2009).	Canada and France
Nuisance macroalgae blooms management	Combination of remote sensing data and current direction simulation to understand the origin of the world's largest green tide, recorded offshore in the Yellow Sea and along the coast of Qingdao (Liu et al., 2009).	China

which are difficult to synthesise and interpret; and (2) simple models that due to generalisation of processes or resolution may fail to capture important ecosystem features (e.g., McKindsey et al., 2006). A promising intermediate approach, whereby different models running at different scales can be integrated in order to optimize the trade-offs between complex and simple models, has been developed by Ferreira et al. (2008a,b). Model integration can be implemented by (1) coupling offline upscaled outputs of detailed hydrodynamic models with ecological box models (Raillard and Ménesguen, 1994; Nobre et al., 2005; Ferreira et al., 2008a); or (2) explicitly integrating models with different timesteps, which is particularly important if there is a need to take into account feedback between the models, as is the case of ecological-economic simulations (Nobre et al., 2009). The advantages of such an intermediate approach include: (1) running multi-year ecosystem models without the computational limitations reported for detailed models (Grant et al., 2008); (2) fewer data requirements for model setup (Ferreira et al., 2008a); and (3) running coarser models at the end of the modelling chain, that present a higher level of information, which are more suitable to inform decision-makers (Ferreira et al., 2008a), and may be better suited to provide highly aggregated information used to drive management-oriented screening models. The main challenges for model integration include: (1) the model coupling can be time-consuming, given that it implies either processing the model outputs according to the format of the downstream model inputs or understanding the various model architectures for programming the code for communication between models; (2) offline coupling does not allow dynamic feedback between models; and (3) online coupling forces scientists and managers to interact towards a common definition of the problem and the identification of the underlying variables, which often requires a broader understanding of different disciplines. The development of integrative tools that simulate the catchment and the biogeochemistry of coastal waters, including cultivated species, is at an early stage, and there are only a few such simulations of management scenarios at the catchment-coastal scale (e.g., Marinov et al., 2007; Ferreira et al., 2008b).

In order to contribute to this development, a multilayered catchment-coastal modelling approach is described below, which optimizes these trade-offs through the use of a comprehensive set of models operating at different levels of complexity and geographical scales. China provides an opportunity for an emblematic case study, given that its coastal areas exhibit rapid economic growth (10% average increase of GDP over 1995–2005), which is causing conflict among its multiple uses (Cao and Wong, 2007). Furthermore, Chinese shellfish aquaculture production (including clams, oysters, mussels, scallops, cockles and arkshells) increased at an average annual rate of about 28% since 1990, and in 2007 represented 77% of the world's shellfish production (FAO, 2009). Therefore, integrated management of the Chinese coastal zone is a considerable challenge requiring a comprehensive approach (Cao and Wong, 2007). The key features of the framework presented in this paper are:

(1) Integration of a set of tools at the catchment-coastal scale;
(2) Engagement of stakeholders, i.e. aquaculture producers, local fishery and environmental managers in the modelling process.

The improvements generated by this approach are to allow the examination of different development scenarios by altering variables of both the catchment and coastal systems and to provide insights for managers. These are critical developments for ICZM and EAA given that such models allow for the assessment of cumulative impacts of coastal activities at the ecosystem level. The specific objectives of this work are to (1) develop an integrated coastal management tool for decision-makers; and (2) examine the outcomes of different development scenarios.

2. Methodology

2.1. Study site and data

The Xiangshan Gang (Fig. 1), a large (volume of 3803×10^6 m^3 and area of 365 km^2) Chinese bay, was chosen as a case study. This system (1) encompasses multiple uses of the marine ecosystem and catchment area; (2) is illustrative of Southeast Asian systems and potentially of European and North American systems at a larger scale of coastal resource uses; (3) has proactive stakeholders and management; and (4) has an appropriate and available dataset. The Xiangshan Gang is a long bay (ca. 60 km in length) connected to the East China Sea, with long residence time in the inner bay and middle section of about 80 and 60 days, respectively, for 90% water exchange, and shorter at the mouth of about 7 days for 90% water exchange (Huang et al., 2003).

This embayment has an intensive aquaculture production of shellfish and finfish and is located in an industrialised area South of Shanghai, near the city of Ningbo (with 6 million inhabitants) in Northern Zhejiang Province. Aquaculture production in the Xiangshan Gang has changed considerably over time (Ning and Hu, 2002). In 1987 there was only kelp cultivation, to which molluscan shellfish and shrimp aquaculture were added in the first half of the 1990's. However, due to high shrimp mortalities farmers introduced razor clams in ponds, in order to leverage the ability of filter-feeders to remove particulate waste while producing an additional cash crop in an Integrated Multi-Trophic Aquaculture (IMTA) system. During the second half of the 1990's finfish aquaculture increased considerably. In 1998 the fish cages in the bay were estimated as 18 000, increasing to 67 000 in 2002. Emerging water quality problems in the bay have been associated with the rapid increase in finfish aquaculture: (1) research programmes executed in 2002 measured anoxic layers with an average depth of 20–30 cm and a maximum depth of 80 cm (Ning and Hu, 2002; Huang et al., 2008b); (2) 21 occurrences of harmful algal blooms (HAB) were recorded in 2003 in Xiangshan Gang and the nearby sea area, including 3 occurrences inside the bay that lasted for more than 30 days (SOA, 2006; Zhang et al., 2007). In 2003, local decision-makers reduced the number of the fish cages by 30% (NOFB, 2007) in an attempt to address those environmental problems. Estimates for aquaculture production in 2005–2006 include: 45 000 t shellfish year^{-1} of which 93% is the Chinese oyster *Ostrea plicatula* produced either on ropes or in intertidal areas; 9400 t finfish year^{-1}; and 6700 t year^{-1} pond production of shrimp, crabs and clams.

A detailed description of the bay and its catchment is given in Ferreira et al. (2008b). Table 2 shows a synthesis of the data collated and used in this paper. Data sources included available historical and web-based data complemented by a limited sampling program collected under the EU "Sustainable options for PEople, catchment and Aquatic Resources" (SPEAR) project (Ferreira et al., 2008b) to complement existing data in order to develop the various models. Remote sensing was used to provide catchment land use and aquaculture structure mapping (Table 2).

Water quality data was assimilated into a relational database, used for retrieval of data for ecosystem model setup and evaluation. A geographic information system (GIS-ArcGIS™) was used to store and analyse spatial data, produce thematic maps and generate information for model setup.

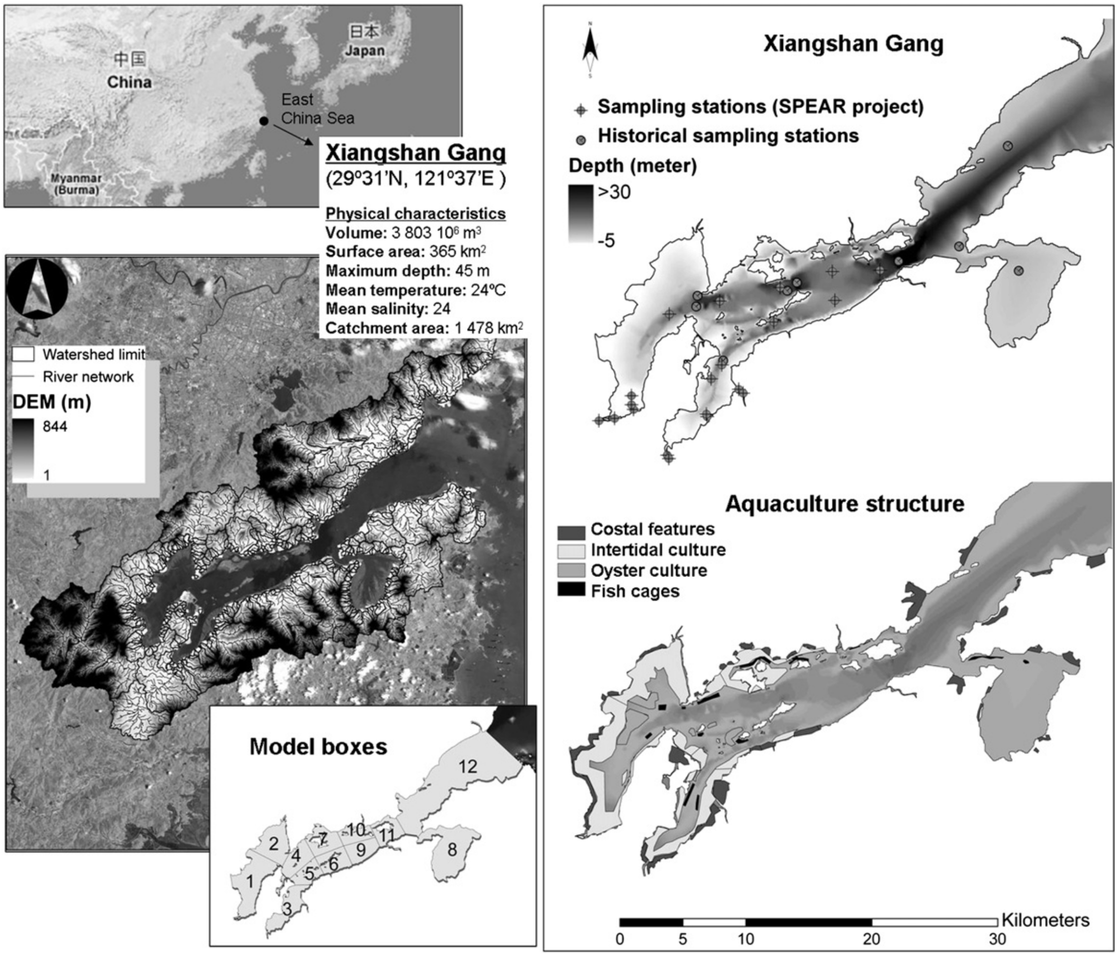


Fig. 1. Xiangshan Gang and catchment area characterisation.

2.2. Multilayered ecosystem model

An integrated ecosystem modelling approach was used (Ferreira et al., 2008b) to simulate the hydrodynamics, biogeochemistry, aquaculture production and forcing functions, such as catchment loading, within Xiangshan Gang. The multilayered approach includes the coupling of several sub-models (Ferreira et al., 2008b) selected following the balance required in the choice of model complexity and structure (Jørgensen and Bendoricchio, 2001): the key state variables and processes to be simulated, such as (1) production of multiple species in polyculture, (2) its effects on the coastal environment and (3) impacts of other catchment-coastal system uses on the water quality and aquaculture resources, were included. However, the multilayered ecosystem model does not include complexity that the dataset cannot validate or that does not significantly contribute to the accurate prediction of drivers for aquaculture; for instance no specific sediment diagenesis sub-model is applied, although this is often appropriate in other ecosystem models (e.g. Simas and Ferreira, 2007). Fig. 2 synthesises the multilayered ecosystem model components, which are detailed below.

The EcoWin2000 modelling platform (Ferreira, 1995) was used to combine (explicitly or implicitly) all the sub-models in order to run the multilayered model. The spatial domain of the Xiangshan Gang model was divided into 12 horizontal boxes and 2 vertical layers (Fig. 1). The division into boxes followed the procedure described in Ferreira et al. (2006) and included a range of criteria: hydrodynamics, catchment loads, water quality and aquaculture structure distribution. EcoWin2000 was set up using a combination of measured data (water quality and aquaculture practice among others) and model outputs (for transport of substances inside the system, from the catchment and exchanged with the sea), as depicted in Fig. 2. The implementation of each sub-model is detailed below and the main equations for state variables are presented in Table 3.

Tables 4 and 5 specify the ecosystem model forcing functions and parameters. The model was run, using a time step of 1 h, for the calibration year (2004), the validation year (standard simulation – June 2005 to June 2006) and a set of different scenarios. Mass conservation in the model was confirmed for the hydrodynamic and biogeochemical components of the ecosystem model by means of a closure analysis for both conservative and non-conservative state variables.

2.2.1. Catchment sub-model

The loading of substances from the Xiangshan Gang watershed was simulated using estimates obtained from the Soil and Water Assessment Tool (SWAT) model (Neitsch et al., 2002). The model was applied to catchment area using data shown in Table 2. The model was calibrated against annual average discharge estimates for the most important rivers in the catchment, using a 30-year model run for a synthetic climate based on the 1961–1990 climatic normal, built with the model's stochastic weather generator. Model performance for water inputs was satisfactory, as indicated by

Table 2
Synthesis of dataset used in the integrated modelling approach for the Xiangshan Gang. Data source: SPEAR project (Ferreira et al., 2008b) unless indicated.

Domain	Parameters
Catchment area	*River water quality data* for years 2005/2006 (monthly sampling): ammonia, nitrate, phosphate, silicate, total nitrogen, total phosphorus, chl-a, temperature, flow rate, salinity, pH, dissolved oxygen. *Land cover ground truth data* collected in 2005: Urban area, paddy fields, dry crop land, burnt land, forest, shrubby area, aquaculture, wetland, shallow water/beach, water and cloud. *Landsat ETM + images (2005/06/28)*, used to create land cover maps following a supervised classification approach (Lillesand and Kiefer, 1999). *Hydrological data*: precipitation, drainage area, river network. *Topographic data* collected during the Shuttle Radar Topography Mission (SRTM), with a resolution of 90 × 90 m (CGIAR, 2005); *Biophysical and agricultural management parameters* following the SWAT database for the most common crop (rice); *Global Zobbler soil maps* with a 2 × 2′ (approx. 3.5 × 3.5 Km) resolution (GRID-Geneva, 2004), parameterized following Batjes (2002). *Urban wastewater discharge*, estimated from the number of inhabitants, using typical per capita wastewater and nutrient generation values (e.g. Economopoulos, 1993).
Meteorological/climate:	*Precipitation data for years 2000/2001* (Liu et al., 2003): total rainfall, ammonia, nitrate, nitrite, phosphate and silicate. *Daily rainfall data for years 2003/2006*: remote sensing using the SSM/I F14 product (Wentz and Spencer, 1998; RSS, 2008). *Daily meteorology for years 2003/2006:* NCEP/NCAR reanalysis for temperature, humidity, wind speed and solar radiation (Kalnay et al., 1996). Climatic normals: calculated using the climate data library maintained by LDEO (2008).
Sea boundary	*Water quality data* for year 2002: Salinity, water temperature, ammonium, nitrate, nitrite, phosphate, dissolved oxygen, chl-a.
Bay (Total 18 stations, Fig. 4)	*Water quality data* for years 2004 (bi-monthly) and Jun05/Jun06 (monthly): Water height, depth, current velocity, water temperature, salinity, ammonia, nitrite, nitrite, organic nitrogen, phosphate, dissolved oxygen, chl-a, particulate organic matter and suspended particulate matter. Samplings include several depths.
Aquaculture data[a]	*Shellfish individual growth experiments*: responses in feeding and metabolism to different combinations of food composition, temperature and salinity. *Shellfish aquaculture production data*: Individual seeding weight, seeding densities, population mortality, harvestable size, total harvest. *Finfish aquaculture for years 2004 and 2005:* (1) Total production; and (2) waste data (Cai and Sun, 2007). *Aquaculture structure mapping*: Landsat visible and infra-red data (2005/06/28) and local maps for ground truthing and to detail smaller aquaculture structures.

[a] Spatially distributed aquaculture production data covers only the inner part of Xiangshan Gang. Total production data was extrapolated for the remaining area based proportionally to the box area.

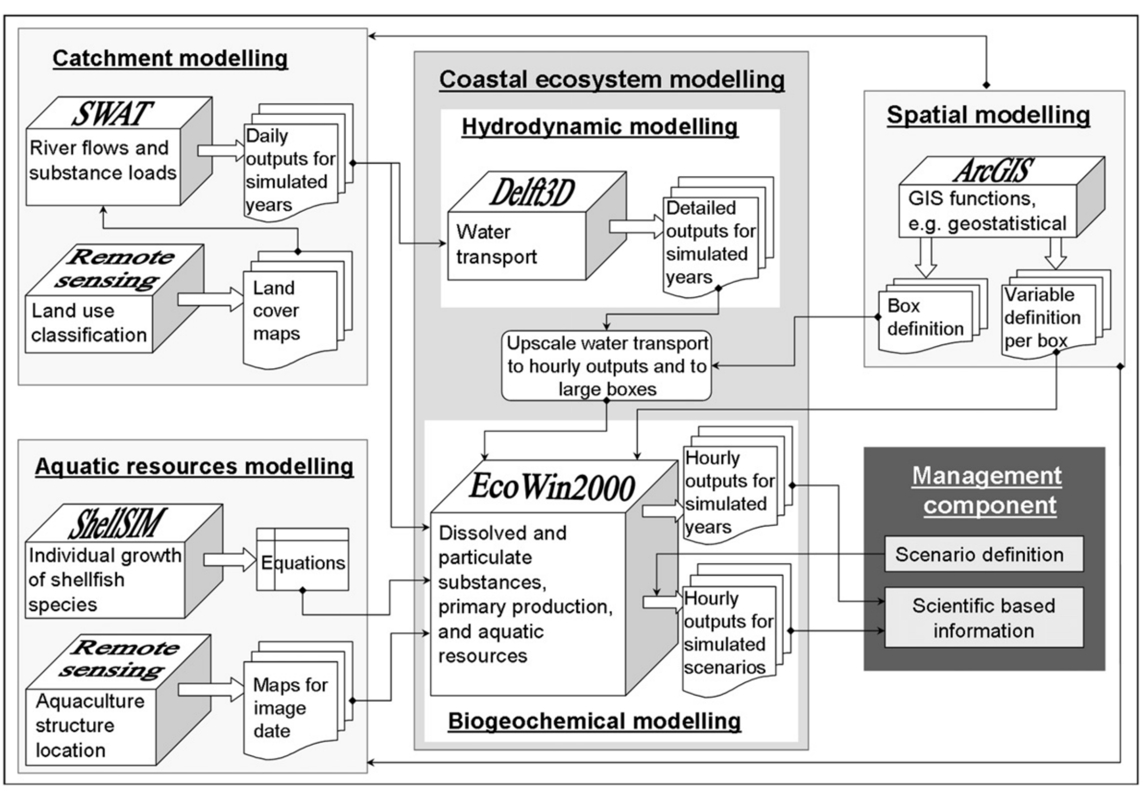


Fig. 2. Integrated catchment-bay modelling approach for coastal ecosystem management: model components and ecosystem-based tools.

Table 3
Main equations for catchment, hydrodynamic, aquatic resources and biogeochemical sub-model state variables.

Catchment processes sub-model (summarized from Neitsch et al., 2002)	Surface water balance $\mathrm{dSW}/\mathrm{d}t = \mathrm{PP}_t - \mathrm{Qs}_t - \mathrm{Ea}_t - \mathrm{Ws}_t - \mathrm{Qgw}_t$ (1) $\mathrm{dSW}/\mathrm{d}t$, Rate of change in soil water content ($\mathrm{mm^3\,mm^{-2}}$) PP_t, Rainfall Qs_t, Surface water runoff Ea_t, Evapotranspiration Ws_t, Exchanges with the deep aquifer Qgw_t, Subsurface water runoff Nutrient export (applied to nitrogen and phosphorus) $\mathrm{dN}/\mathrm{d}t = \mathrm{Fn}_t + \mathrm{Rn}_t + \mathrm{An}_t - \mathrm{PUn}_t - \mathrm{Qn}_t - \mathrm{Ln}_t - \mathrm{Vn}_t - \mathrm{Dn}_t$ (2) $\mathrm{dN}/\mathrm{d}t$, Rate of change in soil nutrient ($\mathrm{kg\,ha^{-1}}$) Fn_t, Fertilization Rn_t, Residue decomposition An_t, Atmospheric fixation (nitrogen only) PUn_t, Plant uptake (including symbiotic fixation for nitrogen) Qn_t, Lateral export (dissolved and particulate) Ln_t, Leaching Vn_t, Volatilization (nitrogen only) Dn_t, Denitrification (nitrogen only)
Hydrodynamic sub-model (WL|Delft-Hydraulics, 1996).	Navier Stokes equations, considering: - hydrostatic, shallow water and Boussinesq assumptions. - orthogonal curvilinear coordinates in the horizontal and terrain following sigma coordinates in the vertical Advection-diffusion equation in three coordinate directions for transport simulation
Aquatic resource sub-model	Shellfish individual growth (Chinese oyster, razor clam, Manila clam and muddy clam) $\eta = f(\mathrm{B}) \bullet f(\mathrm{POM}) \bullet f(\mathrm{SPM}) \bullet f(\mathrm{L}) \bullet f(\mathrm{T})$ (3) η, shellfish scope for growth ($\mathrm{g\,ind^{-1}\,d^{-1}}$) f(B), function of phytoplankton f(POM), function of particulate organic detritus f(SPM), function of suspended particulate matter f(L), function of salinity f(T), function of water temperature Shellfish population growth (Chinese oyster, razor clam, Manila clam and muddy clam) $\mathrm{d}S(s,t)/\mathrm{d}t = -\mathrm{d}[S(s,t) \bullet \eta(s,t)]/ds - \mu(s) \bullet S(s,t)$ (4) S, shellfish number of individuals for each weight class s (ind) η, shellfish scope for growth ($\mathrm{g\,ind^{-1}\,d^{-1}}$) μ, mortality rate ($\mathrm{d^{-1}}$)
Biogeochemical sub-model	Phytoplankton $\mathrm{d}B/\mathrm{d}t = B \bullet (p_{\max} \bullet f(I) \bullet f(\mathrm{NL}) - r_{\mathrm{b}} - e_{\mathrm{b}} - m_{\mathrm{b}} - S \bullet c_{\mathrm{s}})$ (5) B, Phytoplankton biomass expressed as carbon ($\mathrm{\mu g\,C\,L^{-1}}$) $p_{\max}$, Phytoplankton maximum gross photosynthetic rate ($\mathrm{d^{-1}}$) $f(I)$, Steele's equation for productivity with photoinibition f(NL), Michaelis–Menten function for nutrient limitation ($\mathrm{d^{-1}}$) r_{b}, Phytoplankton respiration rate ($\mathrm{d^{-1}}$) e_{b}, Phytoplankton exudation rate ($\mathrm{d^{-1}}$) m_{b}, Phytoplankton natural mortality rate ($\mathrm{ind^{-1}\,d^{-1}}$) c_{s}, Shellfish grazing rate (ind)
(Equations are presented only for internal processes; transport of state variables and boundary loads are described in the text and in Table 4)	Dissolved inorganic nutrients (applied to nitrogen and phosphorus) $\mathrm{d}N/\mathrm{d}t = B \bullet (e_{\mathrm{b}} + m_{\mathrm{b}}) \bullet \alpha + S \bullet e_{\mathrm{s}} + \mathrm{POM} \bullet m_{pom} \bullet \varepsilon - B \bullet (p_{\max} \bullet f(I) \bullet f(\mathrm{NL})) \bullet \alpha$ (6) N, Dissolved inorganic nutrient (nitrogen/phosphorus) ($\mathrm{\mu mol\,L^{-1}}$) α, Conversion from phytoplankton carbon to nitrogen units (-) POM, Particulate organic matter ($\mathrm{mg\,L^{-1}}$) ϵ, Conversion from POM dry weight to nitrogen units (-) m_{pom}, POM mineralization rate ($\mathrm{d^{-1}}$) e_s, Shellfish excretion rate ($\mathrm{\mu mol\,L^{-1}\,ind^{-1}\,d^{-1}}$) Particulate organic matter $\mathrm{dPOM}/\mathrm{d}t = \mathrm{POM} \bullet (e_{\mathrm{pom}} - d_{\mathrm{pom}}) + S \bullet f_{\mathrm{s}} + B \bullet m_{\mathrm{b}} \bullet \omega - \mathrm{POM} \bullet \left(m_{\mathrm{pom}} + p_{\mathrm{pom}} \bullet S\right)$ (7) POM, Particulate organic matter ($\mathrm{mg\,L^{-1}}$) e_{pom}, POM resuspension rate ($\mathrm{d^{-1}}$) d_{pom}, POM deposition rate ($\mathrm{d^{-1}}$) f_{s}, Shellfish faeces production ($\mathrm{mg\,L^{-1}\,ind^{-1}\,d^{-1}}$) ω, Conversion from phytoplankton carbon to POM dry weight p_{pom}, Shellfish POM filtration rate ($\mathrm{ind^{-1}\,d^{-1}}$)

Table 3 (*continued*)

Suspended particulate matter
$$\mathrm{dSPM}/\mathrm{d}t = \mathrm{SPM}\bullet(e_{\mathrm{spm}} - d_{\mathrm{spm}}) + S\bullet f_{\mathrm{s}} - \mathrm{SPM}\bullet p_{\mathrm{spm}}\bullet S \quad (8)$$
SPM, Suspended particulate matter (mg L^{-1})
e_{spm}, SPM resuspension rate (d^{-1})
d_{spm}, SPM deposition rate (d^{-1})
p_{spm}, Shellfish SPM uptake rate (ind^{-1} d^{-1})

a significant correlation between simulated and observed values ($r^2 = 0.92$), low model bias (−5.3%) and high model efficiency (Nash–Sutcliffe efficiency index = 0.91). Simulated annual nitrogen inputs from diffuse agricultural sources (960 t year^{-1}) compared well with an estimate by Huang et al. (2008b) based on export coefficients (900 t year^{-1}).

Following the evaluation for 1961–1990, the model was run for the study period (2004–2006) using climate data described in Table 2. Existing data were not sufficient to evaluate river flow results obtained with SWAT for 2004–2006. However, existing monthly measurements from mid-2005 to mid-2006 of nitrogen (N) and phosphorus (P) in two major rivers – Fuxi and Yangongxi – were compared with model results. As can be seen in Fig. 3a for dissolved inorganic nitrogen (DIN), it is difficult to assess model performance using only these data. In Fuxi, SWAT underestimates measured concentrations, but the measurement dates are consistent with rainfall-induced peaks predicted by SWAT; it is therefore debatable whether measured concentrations represent the average situation or only these short-term peaks. In Yangongxi, the SWAT simulations are more consistent with measurements, due in part to the smaller variability of both. This was also observed for N species and for P. It is also difficult to evaluate the reason behind potential SWAT errors due to the lack of river flow measurements, as an error in nutrient concentration could be due to errors in either the mass of nutrients entering the river or in the river's dilution capacity. To avoid this problem, the simulated export of N was compared with an estimate of exports based on measured nutrient concentrations and simulated river flows. The results are shown in Fig. 3b. SWAT agrees well with the measurement-based estimates, especially in the months with the largest exports; the correlation coefficients (r^2) are 0.72 and 0.84, respectively for Fuxi and Yangongxi. A similar calculation for P shows slightly worse results, with r^2 of 0.51 and 0.83 for the same rivers.

The output from the SWAT model simulation was transformed into daily data series aggregated per box for offline coupling with EcoWin2000 (for both calibration and validation years). In total, the nutrient load entering the bay from the catchment was estimated to be about 11 t d^{-1} of DIN and 2 t d^{-1} of phosphate, of which about 40% of the total loading was diffuse pollution from agriculture and forest litter decomposition (for both DIN and phosphate). The point sources included untreated urban wastewater for ca. 600 000 inhabitants.

2.2.2. Hydrodynamic sub-model

The transport of substances among boxes and across the ocean boundary was simulated using the upscaled outputs of a detailed three-dimensional hydrodynamic and transport model (Delft3D-Flow – Delft Hydraulics, 2006) (Ferreira et al., 2008b). Delft3D-Flow is well tested software used to generate highly detailed continuous flow fields (Delft Hydraulics, 2006). The model calibration was performed in two major phases. In the first phase, only tidal forcing was used. Variations in tidal forcing were compared against measured water levels to achieve an optimum in harmonic composition of the tidal elevation, followed by adjustment of bottom roughness to reproduce the water velocity characteristics reported by Huang et al. (2003). Overall, the model represented the amplitude of the main harmonic constituents well (Table 6). However, the phase of these constituents was difficult to reproduce due to the imprecise bathymetry data, which hampered the correct estimation of the bay's storage. This limitation is not critical, given that the aim was to predict the contribution of tides to the exchange rather than accurate tidal prediction for navigation purposes. In the second phase, a baroclinic model was developed by including heat and freshwater contributions. In order to define the model boundary conditions, the salinity and temperature dataset was complemented with data from Hur et al. (1999) and Isobe et al. (2004). In this second phase the response of the system was gauged through existing knowledge of circulation as effected by tides and baroclinicity in tidal embayments (Fujiwara et al., 1997; Simpson, 1997). Due to the lack of in situ density and velocity measurements, this procedure was used to tune the model within the theoretically acceptable boundaries for this type of system. The model outputs

Table 4
Ecosystem model forcing functions for Xiangshan Gang standard simulation.

Transport of substances (among boxes and with sea boundary)	Offline assimilation of water fluxes outputs of the detailed hydrodynamic sub-model. The water fluxes were integrated in space and time using the ecosystem model box setup (12 horizontal boxes – Fig. 1 – each divided vertically into 2 boxes) and time step (1 h).		
Catchment loads	Offline assimilation of SWAT model outputs transformed into daily data series aggregated per box.		
Fish cage loads	Total number of cages		69 237
	Production per cage (kg year^{-1})		205
	Food waste (% of feeding)		61%
	Nutrient load per cage (kg year^{-1})	DIN	34
		Phosphate	15
		POM	580
Shrimp loads	Shrimp production (t year^{-1})		700
	N load (kg t^{-1} shrimp year^{-1})		60
	P load (kg t^{-1} shrimp year^{-1})		20
Photoperiod and light energy	Brock model (Brock, 1981)		
Water temperature	Sinusoidal function adjusted to fit observed data with minimum and maximum temperatures recorded as 5 °C and 30 °C respectively.		

Table 5
Ecosystem model parameters for Xiangshan Gang standard simulation.

Shellfish population	Number of weight classes		10
	Mortality – μ (% per day)	Oyster	0.40%
		Clam	0.56%
		Razor	0.20%
		Muddy	0.15%
Shellfish cultivation practice	Seed weight (g TFW ind^{-1})	Oyster	0.2
		Clam	0.5
		Razor	0.5
		Muddy	0.1
	Seeding period	Oyster	April–August
		Clam	May–June
		Razor	April–August
		Muddy	June–September
	Harvestable weight (g TFW ind^{-1})	Oyster	8
		Clam	14
		Razor	11
		Muddy	5
	Harvesting period	Oyster	December–March
		Clam	January–February
		Razor	October–February
		Muddy	November–March
	Aquaculture area (ha) and boxes cultivated	Oyster	2286 (Boxes 1–5, 8, 9, 11, 12)
		Clam	308 (Boxes 1–7, 10)
		Razor	313 (Boxes 1–6)
		Muddy	187 (Boxes 1–3, 5, 6)
	Seeding density (t TFW ha^{-1})	Oyster	0.90
		Clam	0.45
		Razor	0.72
		Muddy	0.82
Phytoplankton growth		P_{max} (h^{-1})	0.2
		Iop (w m^{-2})	300
		Death loss – m_b (d^{-1})	0.01
		Ks DIN (μmol L^{-1})	1
		Ks Phosphate (μmol L^{-1})	0.5
Suspended matter		POM mineralization rate (d^{-1})	0.02
		POM to nitrogen (DW to N)	0.0519
		POM to phosphorus (DW to P)	0.0074

provided a repeatable series of approximately 1 year of flows with which to force transport in the ecosystem model for both the calibration and validation years. The data series length was chosen in order to be as close as possible to an annual cycle (365 days), which is the cycle of simulation of other forcing functions of the ecological model (e.g. light and water temperature). Therefore, the series obtained was 3 days and 10 h longer for 2004. The resulting residual surplus (0.1 m^3 s^{-1} averaged over the bay and 0.7 m^3 s^{-1} at a single box) was artificially subtracted in order to ensure the conservation of the mass. The detailed flow fields were scaled up and converted into a data series of water fluxes between boxes and across the sea boundary with a one hour time step and coupled offline with EcoWin2000 (see e.g., Ferreira et al., 2008a).

2.2.3. Aquatic resource sub-model

The simulated aquatic resources included *O. plicatula* (Chinese oyster), *Sinonvacula constricta* (razor clam), *Tapes philippinarum* (Manila clam) and *Tegillarca granosa* (muddy clam) production. The equations for shellfish aquaculture production were explicitly integrated into the ecosystem model using a four step approach (Ferreira et al., 2008a): (1) use of a shellfish individual growth model (ShellSIM - http://www.shellsim.com); (2) coupling of the individual growth model with a demographic model to simulate the population (Ferreira et al., 1997); (3) integration of the population growth model with an aquaculture practice model which implements the seeding of the population biomass and harvesting of the marketable cohorts for a given production cycle (Ferreira et al., 1997); and (4) use of a multiple-inheritance object-oriented approach (Nunes et al., 2003) to extend to multiple species in polyculture. ShellSIM simulates feeding, metabolism and individual growth in contrasting environments for different shellfish species, as exemplified for *Chlamys farreri* by Hawkins et al. (2002). In the ShellSIM model, removal of particulate organic matter (phytoplankton and detritus) by shellfish is determined through the individual growth models for the bivalves. It is a function of several environmental drivers, including salinity, temperature, suspended particulate matter (SPM) and the food sources themselves, and is additionally driven by allometry. These drivers are used to determine filtration, pre-ingestive selection, ingestion and assimilation. The individual growth model was calibrated for Chinese oyster, razor clam and muddy clam under local conditions (Ferreira et al., 2008b). As shown in Table 7 there is a statistically significant relationship between the individual model results and observations for shellfish total wet weight and shell length in Xiangshan Gang. For the simulation of the Manila clam individual growth, the model used in Ferreira et al. (2007) was applied. The population growth is simulated using a demographic model based on ten weight classes. The demographic model is a widely used model (Ferreira et al., 1997, 2007; Nunes et al., 2003; Nobre et al., 2005) based on a conservation equation (Eq. (4), Table 3) discretised in weight classes. The food (phytoplankton and detritus) removed by the population is scaled for each weight class on the basis of the number of individuals in the class; compliance with the Courant condition is ensured, such that, in the case of numerical instability, the food supply (and therefore the growth potential) is reduced by adjusting the filtration rate. Changes in the population structure

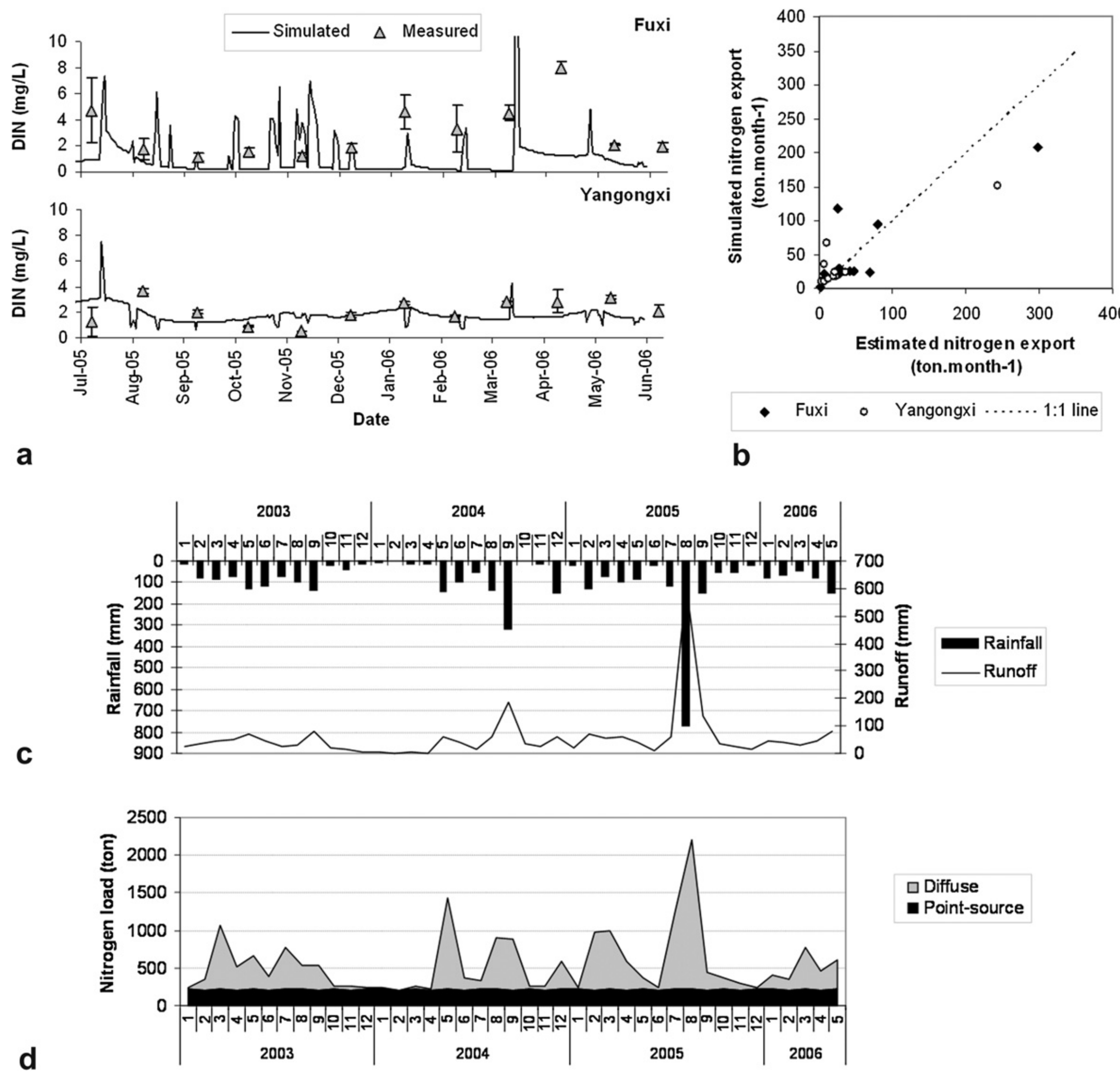


Fig. 3. Catchment model outputs and comparison with data: a) measured and simulated dissolved inorganic nitrogen (DIN) for Fuxi and Yangongxi rivers; b) estimated and simulated nitrogen export; c) simulated monthly runoff compared with rainfall; and d) nitrogen loads from diffuse and point sources.

derive from the simulation of the individual growth of one animal (Eq. (3), Table 3) in each weight class, thus providing the scope for growth which drives the transition of individuals across weight classes (Eq. (4), Table 3). The aquaculture practice model (Ferreira et al., 1997) implements the seeding and harvesting strategies and interacts with the population model by respectively adding and subtracting individuals to the appropriate classes. This modelling approach of the aquatic resources is described in previous applications that simulate polyculture at the ecosystem scale (Nunes et al., 2003; Ferreira et al., 2008a). A synthesis of model parameterization is presented in Table 5.

Both shrimp and fish production were included as forcing functions of the ecosystem model, contributing to dissolved and particulate waste (Ferreira et al., 2008b). The annual fish cage loadings to the Xiangshan Gang (Table 4) were calculated based on the number of fish cages per box; average fish production per cage; food waste; and nutrient load per fish produced, based on dry feed conversion rate (Cai and Sun, 2007). Nutrient loads from the shrimp

Table 6
Amplitude and phase of the harmonic constituents: comparison between observed and simulated values.

Constituent	Difference between model and observed	
	Amplitude (m)	Phase (°)
O1	0.04	32
K1	0.07	−90
N2	0.03	103
M2	0.1	−83
S2	−0.13	45
MO3	−0.01	−54

Table 7
Correlation between simulated and measured individual shellfish total wet weight and shell length, using Pearson product–moment correlation coefficient (r). Data were measured at regular intervals through a normal culture period in Xiangshan Gang.

	Degrees of freedom	Individual total wet weight(g)		Individual shell length(mm)	
		r	Level of confidence	r	Level of confidence
Chinese oyster	2	0.926	90%	0.958	95%
Razor clam	3	0.999	99%	0.942	98%
Muddy clam	4	0.951	95%	0.977	95%

ponds (Table 4) were calculated by means of a shrimp growth model (LMPrawn) as described in Ferreira et al. (2008b) and Franco et al. (2006).

2.2.4. Biogeochemical sub-model

The biogeochemical model was developed using EcoWin2000 to simulate the following biogeochemical state variables: salinity, dissolved nutrients, particulate matter and phytoplankton (Ferreira, 1995; Nunes et al., 2003; Nobre et al., 2005; Ferreira et al., 2008b). Simulated DIN and phosphate concentrations were used for calculation of the nutrient limiting phytoplankton growth. The sub-models described previously were used to simulate the shellfish aquaculture production, the catchment loads, and the transport of water and substances among boxes and across the sea boundary. Ocean boundary conditions and atmospheric loadings were derived from historical data and defined as average annual values. Due to the lack of synoptic data for the setup of the ocean boundary, seawater quality data for 2002 was used for both calibration and validation. Seasonal data for nutrients contained in the rainwater were used to determine the average annual atmospheric load of N and P to the bay. The parameterization of the model for Xiangshan Gang is presented in Table 5.

The pelagic variables in the model were calibrated against a historical time series for 2004 (Table 2). Due to lack of historical data, the annual average of the validation year was used for SPM and particulate organic matter (POM). The model was run for the validation year using the same parameters employed for the calibration year but adjusting the data series for forcing functions and the initial conditions to simulate the period from June 2005 to June 2006. Model performance was evaluated by comparing the model outputs of the standard simulation with the water quality and aquaculture production data for the validation period.

2.3. Coastal management options simulation

2.3.1. Definition of scenarios

The development scenarios were defined as a result of the participatory work among stakeholders carried out during the SPEAR project (Ferreira et al., 2008b). Several stakeholder meetings were held involving modellers, local fishery and environmental managers and aquaculture producers. The capabilities of the modelling tools to support catchment and aquaculture management were explained to the local managers and producers. In addition, the issues of concern to the local managers and producers were discussed with the modelling team. The participatory work among stakeholders culminated with a clear set of scenarios defined by the Xiangshan Gang managers and aquaculture producers. The scenarios to be simulated by the multilayered ecosystem modelling framework comprise: (1) a reduction of fish cages corresponding to a 38% reduction in total fish production (Scenario 1); (2) an extension of wastewater treatment to the entire population (Scenario 2); and (3) a simultaneous reduction of fish cages and extended wastewater treatment (Scenario 3). These scenarios are important for the evaluation of nutrient abatement strategies defined by managers to improve water quality in Xiangshan Gang. From a management perspective, the scientific assessment of such scenarios also provides guidelines/grounding for future aquaculture policy and for eutrophication control.

Using SWAT model outputs with different timesteps an additional scenario was run to test the consequences of different temporal resolution of forcing functions on simulated results. Monitoring of substance loadings from the adjacent catchment area are often used as forcing for coastal ecosystem models. However, this is often restricted to a few locations within the watershed and to a few sampling occasions over the year. In this work the use of SWAT model enabled the application of detailed forcing in space and time for catchment loads and to test the sensitivity analysis of the coastal ecosystem to the temporal resolution of the catchment model outputs. The scenario includes running the standard simulation using monthly, rather than of daily, data series of the SWAT model outputs. This scenario exemplifies how the multilayered ecosystem model can be used in the future to further explore a larger research issue about monitoring data requirements and optimal temporal resolutions to use in the models.

2.3.2. Development scenario implementation and interpretation

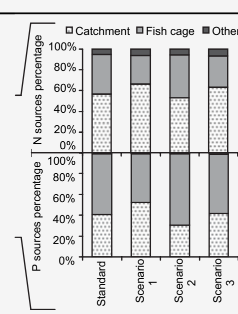
The reduction of fish cages (scenarios 1 and 3) was implemented assuming that the decrease in nutrient loading is proportional to the decrease in fish production. The impact of wastewater treatment (scenarios 2 and 3) on the exports of N, P and sediment from urban areas was calculated following Burks and Minnis (1994). Table 8 synthesises the corresponding substance loading used to simulate each scenario.

A comparison of the results obtained for the different scenarios was performed and the interpretation of the outcomes was guided by means of:

(1) Influencing Factors (IF) from the ASSETS eutrophication model (Bricker et al., 2003) to interpret the influence of catchment and aquaculture loads on eutrophication; The IF index calculates the pressure on the system as a combination of the nutrient loading with the system susceptibility to eutrophication (flushing and

Table 8
Scenario definition (percentage changes compared with standard simulation are shown in brackets and italics).

Setup		Standard	Scn 1	Scn 2	Scn 3
No. fish cages		69 237	42 927	69 237	42 927
% of standard simulation			62%	100%	62%
Treated wastewater (million inhabitants)		0	0	0.6	0.6
Total loads (t d^{-1})	DIN	18.9	16.2 (*−14%*)	17.5 (*−8%*)	14.7 (*−22%*)
	Phosphate	5.0	3.9 (*−22%*)	4.2 (*−15%*)	3.1 (*−37%*)
	POM	451.7	410.1 (*−9%*)	413.8 (*−8%*)	372.1 (*−18%*)
Influencing Factors (IF) [a]	N	75%	72%	73%	70%
	P	94%	92%	93%	90%
Boxes with changes			1–5, 7–12	1,3,8,9,12	1–5, 7–12



[a] The Influencing Factors comprise the catchment and aquaculture nutrient loads, this index is calculated from Bricker et al. (2003).

dilution factors) (Bricker et al., 2008). Bricker et al. (2003, 2008) calculates the relative magnitude of the different sources considering inputs from watershed (manageable anthropogenic sources) and ocean (background sources) boundaries. For the IF application to Xiangshan Gang, aquaculture and watershed are together considered manageable anthropogenic sources. Details on the IF calculation are provided by Bricker et al. (2003) and a computer application is freely available online (http://www.eutro.org/register) to perform the calculations;

(2) The threshold of chl-a 90-percentile values as defined in the ASSETS model (Bricker et al., 2003) to assess the level of expression of the phytoplankton symptom;

(3) Chinese seawater quality standards (National Standard of People's Republic of China, 1997) for DIN and phosphate to

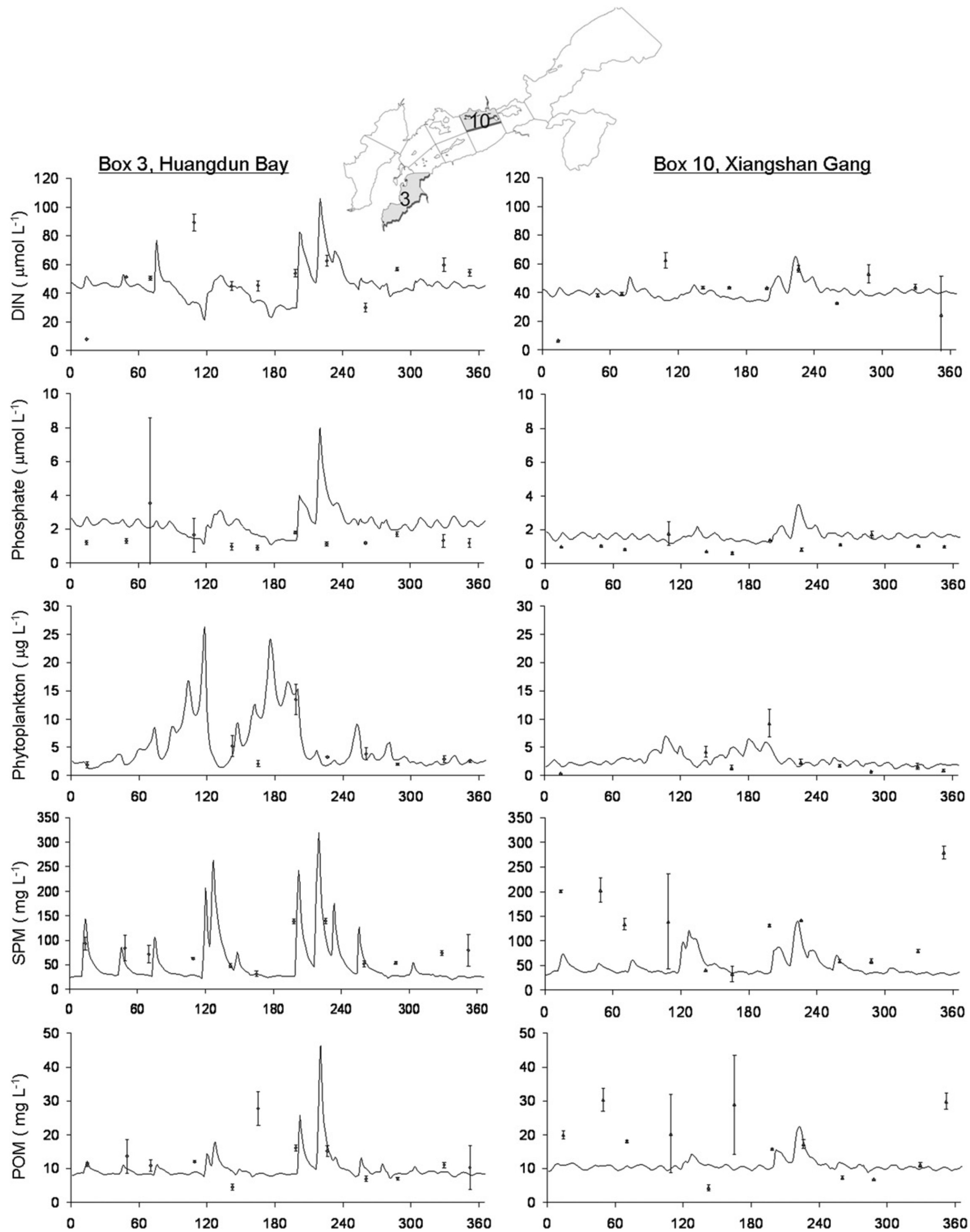


Fig. 4. Standard simulation outputs for an inner box (Box 3, Huangdun Bay) and a middle box (Box 10), plotted with average daily data (June2005/June2006) and corresponding standard deviation: phytoplankton biomass, dissolved inorganic nitrogen (DIN), phosphate, suspended particulate matter (SPM) matter and particulate organic matter (POM).

assess the compliance with desirable water quality objectives set by decision-makers for the bay; and

(4) Shellfish productivity, given as the ratio of total weight of shellfish harvested to total weight of seed, also known as average physical product (APP, Jolly and Clonts, 1993), to interpret the changes in the ecosystem use due to scenario implementation.

3. Results

3.1. Ecosystem simulation

Fig. 3c and d show catchment model results for runoff and N loading into Xiangshan Gang, from diffuse (agricultural) and point (urban sewage) sources. N inputs have two annual peaks, in early spring and early summer, which can be related to both the fertilization of rice (which is harvested twice per year in this region) and the annual rainfall and runoff patterns. This pattern was also found for particulate matter and P loads. The large input peak in August 2005 is an exceptional occurrence, mostly caused by typhoon Matsa on August 5th. The major sources for N, according to the model results are urban sewage discharges (56%); agricultural, namely fertilization in rice crops (27%); and rangelands, mostly detritus decomposition from forests (17%). P followed a similar pattern, with 60% coming from urban sewage discharge and the remainder from agricultural and natural sources.

Fig. 4 shows the results of the coastal ecosystem model for the pelagic variables in an inner location (Box 3) and a location in the middle of the bay (Box 10), which represents the outermost box with sampling data. The ecosystem model outputs for DIN and phytoplankton compare reasonably well with collected data, as exemplified for in Fig. 4. The DIN peak observed around day 120 in Box 3 is not reproduced, possibly due to an underestimation of the loads for that period (from catchment or from aquaculture) or due to a local phenomenon that does not represent the average for the box. In contrast, the model outputs for phytoplankton exhibit peaks not seen in the data. In particular, the sampling point immediately before day 180 shows a very low value for phytoplankton, whereas the model simulates high phytoplankton concentrations. A combination of three factors can justify this occurrence: (1) high natural variability of phytoplankton (Rantajärvi et al., 1998), not captured by the sampling window; (2) phytoplankton dynamics are

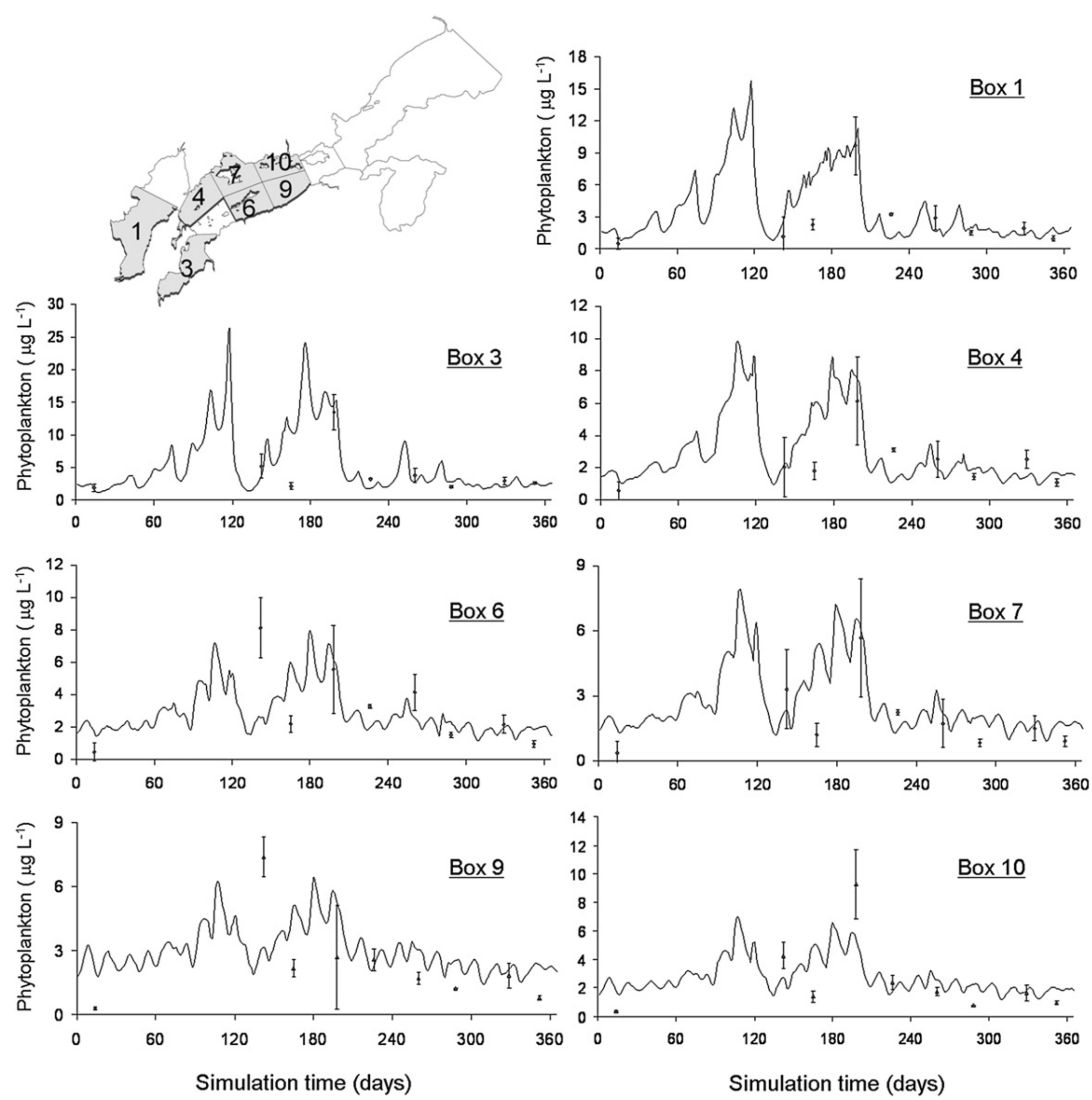


Fig. 5. Standard simulation outputs for phytoplankton plotted with average daily data (June2005/June2006) and corresponding standard deviation for boxes 1, 3, 4, 6, 7, 9, 10.

ruled by complex set of factors difficult to simulate in dynamic ecological models, such as species succession (Arhonditsis et al., 2007); and (3) the model outputs represent an uniform value for a box, and thus cannot account for the variability in that area, given that for most boxes data coverage for validation includes only one sampling station (Fig. 1); for box 3 in particular there are 2 stations, the remaining stations are for rivers or from the historical dataset used for calibration). With regard to phosphate, the data do not indicate a particular pattern, and in general the model overestimates observed phosphate concentrations. This might be due to an overestimate of phosphate loads from either 1) fish cages, given that fish aquaculture is the major source of this nutrient (Table 8) together with the fact that an average annual load is considered due to the lack of temporally detailed data on fish cage loading; or 2) from the catchment, which as described previously had a performance which was less good than that obtained for DIN load estimates. Model outputs of SPM and POM in Box 10 did not represent the observed variability whereas in the inner box (Box 3) the model outputs reproduced the trends shown by the data points (Fig. 4). A possible explanation is that the temporal resolution of SPM and POM values being used to force the ocean boundary was not sufficient to represent the variability in the adjacent boxes. As such, a time series should be used instead of the annual average ocean concentration. In the inner boxes the marine influence was reduced and catchment inputs of POM and SPM were more important, thus the daily inputs provided by the catchment model provided the appropriate forcing. Nevertheless, this limitation is not likely to significantly affect the simulation of aquaculture production, given that 83% of the bivalves are produced in the inner boxes (boxes 1–5).

Fig. 5 provides an overview of the model agreement with measured data for all boxes with sampling stations, using phytoplankton as an example, given that this is a critical model variable. Overall, the phytoplankton results compare reasonably well with measured data.

Fig. 6 shows the simulation of shellfish production and the respective key environmental drivers for shellfish growth. Oysters were used as an example since this species accounts for 93% of the total shellfish production. Fig. 6 also shows the mass loss calculated based on the net energy lost due to physiological processes. The energy balance accounts for the energy ingested, energy lost as faeces, energy excreted, the heat loss and the energy loss due to reproduction (Ferreira et al., 2008b). Model results are presented for an inner box (Box 3) with a total shellfish production of ca. 2305 t

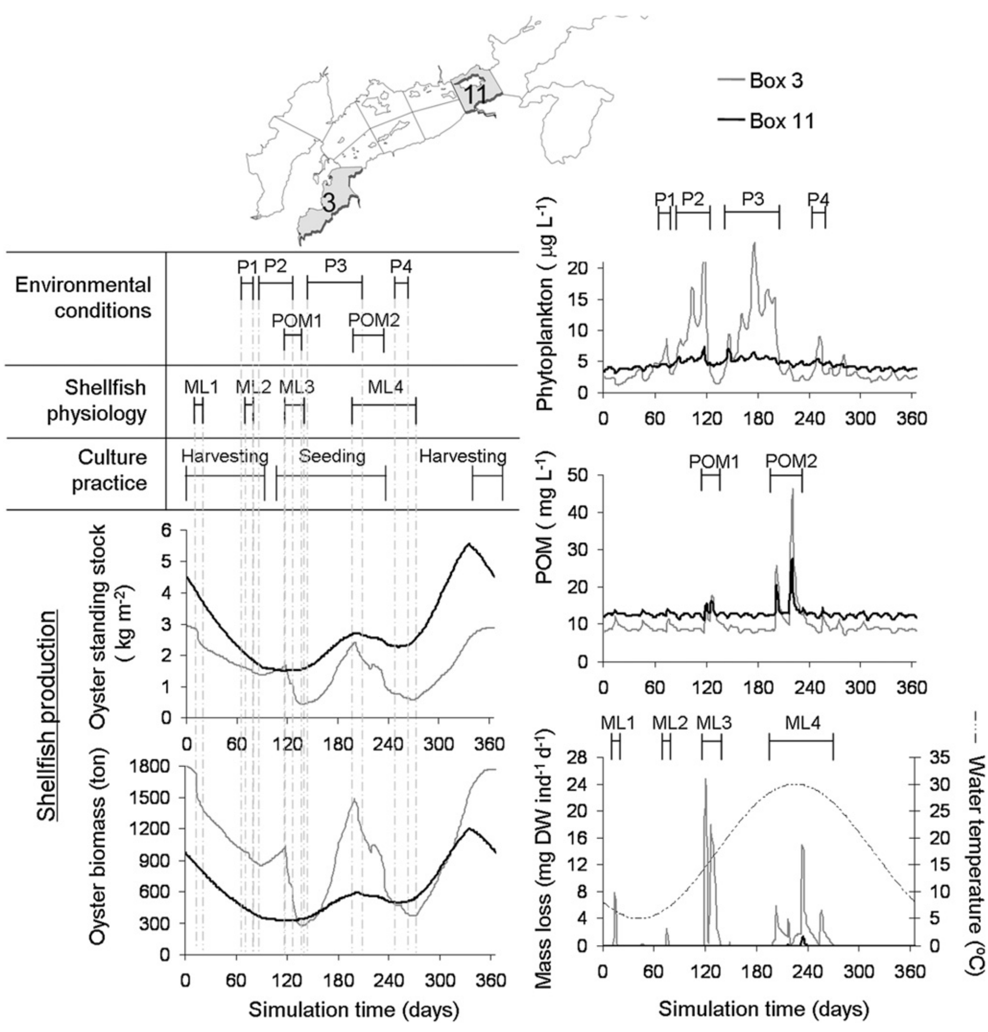


Fig. 6. Standard simulation outputs for Box 3 (in grey) and Box 11 (in black) for: oyster production (standing stock, total biomass); mass loss (ML) due to reproduction, faeces and excretion; and key environmental variables affecting oyster growth, i.e. phytoplankton biomass (P), particulate organic matter (POM) and water temperature. Peaks are indicated with letters P#, POM#, ML# for variables phytoplankton biomass, POM and shellfish mass loss, respectively. The stripes superimposed in the shellfish production plots indicate the time snapshots that correspond to the peaks, harvesting and seeding.

(oysters account for ca. 1298 t) and a box near the sea boundary (Box 11) with a total shellfish production of ca. 741 t (all oyster). The oyster standing stock was generally higher in Box 11 than in Box 3, possibly due to the higher POM availability registered in most of the year in Box 11 (Fig. 6). As a result, POM uptake by oysters was about six-fold higher in Box 11 ($3.36\ g\ m^{-2}\ year^{-1}$) than in Box 3 ($0.54\ g\ m^{-2}\ year^{-1}$). The effects of the peaks of phytoplankton concentration in Box 3 around days 120 and 180 (peaks P2 and P3, respectively, Fig. 6) are visible through the increase of shellfish biomass and standing stock. This effect was not noticeable for the smaller peak that occurs after day 240 (peak P4, Fig. 6), because it was cancelled out by the mass lost due to physiological processes (ML4, Fig. 6), possibly caused by the high temperatures that occur during the ML4 period (Fig. 6). On average, phytoplankton concentration was higher in Box 3: annual average values ca. 5.5 μg Chl-a L^{-1} and 3.4 μg Chl-a L^{-1} in, respectively; the average phytoplankton uptake was also higher in Box 3: ca. 38.6 $g\ C\ m^{-2}\ year^{-1}$ and 36.0 $g\ C\ m^{-2}\ year^{-1}$ in, respectively. Possibly, these differences of phytoplankton consumption among both boxes were much smaller than differences in POM uptake given that higher phytoplankton availability in Box 3 was counteracted by a higher shellfish production in that box which led to resource partitioning among cultivated animals (Fig. 7).

Overall, the outputs of harvested shellfish compare well with the landings data (Fig. 7).

Comparison of ecosystem model outputs using different temporal resolutions for the catchment loads (Fig. 8) indicated that using monthly instead of daily catchment inputs led to significantly different outcomes, especially for the inner boxes (as illustrated for Box 3 in Fig. 8). In general, the biogeochemical model could not reproduce observed peaks in DIN, phosphate, phytoplankton and POM when the monthly SWAT inputs were used. As a consequence, for example, the calculation of the percentile 90 chl-a value changed from ca. 13 μg Chl-a L^{-1} to 6 μg Chl-a L^{-1} in Box 3 and from ca. 5 μg Chl-a L^{-1} to 3 μg Chl-a L^{-1} in Box 6. In the outer box, there were no significant changes in the 90-Percentile chl-a value.

3.2. Development scenarios

The scenarios tested simulate different nutrient loads entering into the bay. Table 8 presents the N, P and POM loading into the bay from catchment and aquaculture sources for each scenario. The Influencing Factors from aquaculture and catchment loads on the bay's nutrient concentration ranged from 75% in the standard simulation to 70% for scenario 3, for N (Table 8). For P the contribution was higher, ranging from 94% in the standard simulation to 90% in scenario 3 (Table 8). These results, according to the categories defined in the ASSETS model (Bricker et al., 2003), indicated that for N and P there was a Moderate High and a High class, respectively, for the portion of nutrients from anthropogenic sources compared with those coming from the sea. Therefore, there is the potential for a significant reduction of nutrients through management. The major contribution of nutrients was from catchment loading and from fish cages for nitrate and phosphate, respectively, for any of the scenarios tested (Table 8).

In general, model outcomes indicate that the effects of changes implemented in the scenario simulations were mostly visible in the inner boxes. Fig. 9 shows the model outputs for (1) an inner box (Box 3 – Huangdun Bay), where the reduction of fish cages (in scenarios 1 and 3, Table 8) and the reduction of nutrient loads from wastewater discharge (in scenarios 2 and 3, Table 8) were implemented; (2) a middle box (Box6) where no direct changes were implemented; and (3) an outer box (Box 12), where, as for Box 3, a reduction of fish cages and nutrient loads from wastewater was tested (Table 8). The changes simulated in the three scenarios were less evident for the outer box for DIN, phosphate, phytoplankton, shellfish harvest and shellfish productivity (Fig. 9), possibly due to the exchanges with the ocean boundary.

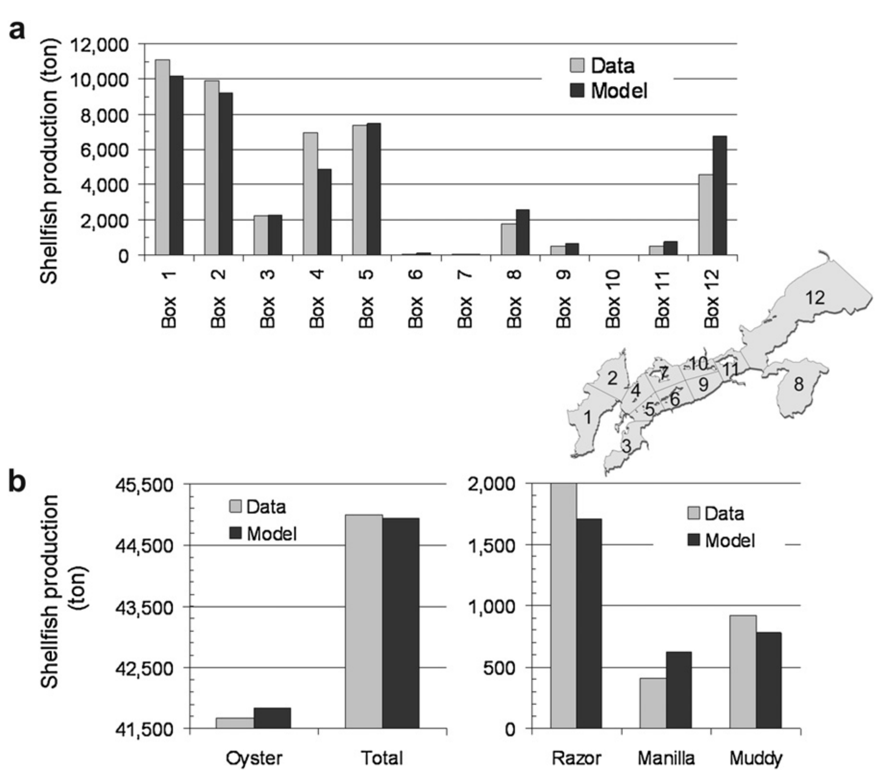


Fig. 7. Standard simulation outputs for shellfish harvest and comparison with data (in $t\ year^{-1}$).

The reduction of nutrient loads in any of the scenarios resulted in very small changes in bay DIN concentration for any of the boxes (Fig. 9a). There was a higher impact of nutrient load reduction on the simulated phosphate concentration (Fig. 9b), probably because this was the substance with higher decrease (Table 8). Changes in phosphate concentration ranged from −8% to −21% in Box 3 and from −2% to −6% in Box 12 when comparing the scenarios with the standard simulation. The expected causes for the phosphate overestimation in the standard simulation, i.e., overvaluation of fish cage and catchment loads, also apply to the simulated scenarios; as such it is likely that this source of error does not affect the predicted range of change of phosphate concentration from the standard simulation compared to scenarios. Despite the fact that no direct changes were simulated in any of the scenarios for Box 6, model outputs (Fig. 9b) also indicated changes of phosphate concentration (between −6% and −12%), possibly as a result of the transport between boxes. Both DIN and phosphate were present in high concentrations and, on average phosphate was the limiting nutrient for the phytoplankton growth for every scenario and in every box.

According to the Chinese seawater quality standards for nutrient concentration parameters (National Standard of People's Republic of China, 1997), water quality in Xiangshan Gang is classified on average as being above the limit of Class IV, meaning poor quality. Given that the model overestimates phosphate concentration, these standards were also calculated for the sampled water quality data, which confirms the results of poor water quality.

The most pronounced changes in phytoplankton concentration occurred in the inner boxes; in the effects of nutrient load reduction were possibly dissipated (Fig. 9c and d). Fig. 9c shows the phytoplankton 90-percentile value for different boxes and scenarios.

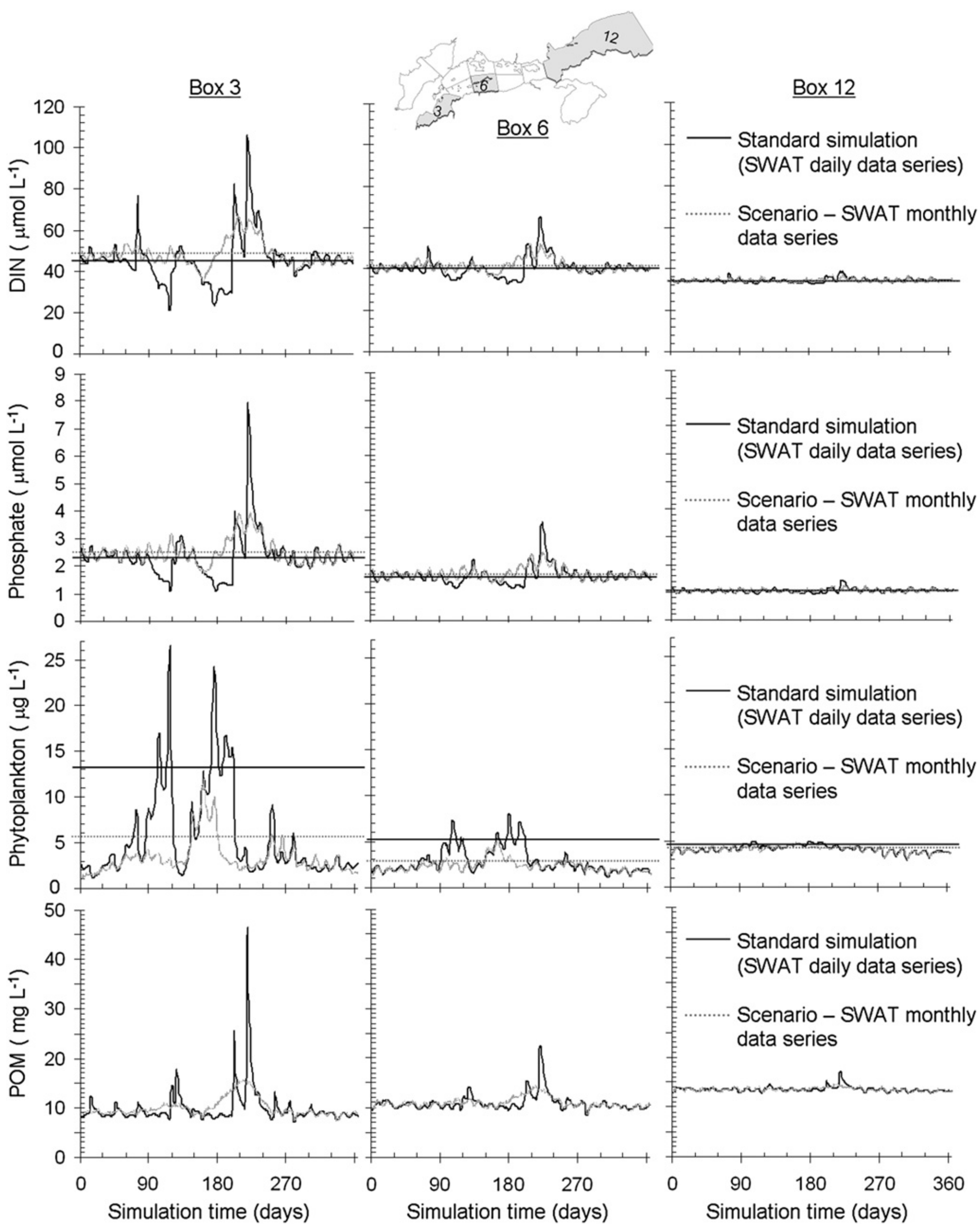


Fig. 8. Sensitivity analysis of the coastal ecosystem to the temporal resolution of the catchment model outputs for an inner box (Box 3, Huangdun Bay), a middle box (Box 6), and an outer box (Box 12): dissolved inorganic nitrogen (DIN), phosphate, phytoplankton biomass and particulate organic matter (POM). (Straight lines in the plots indicate average value for DIN and phosphate, and 90-Percentile for phytoplankton).

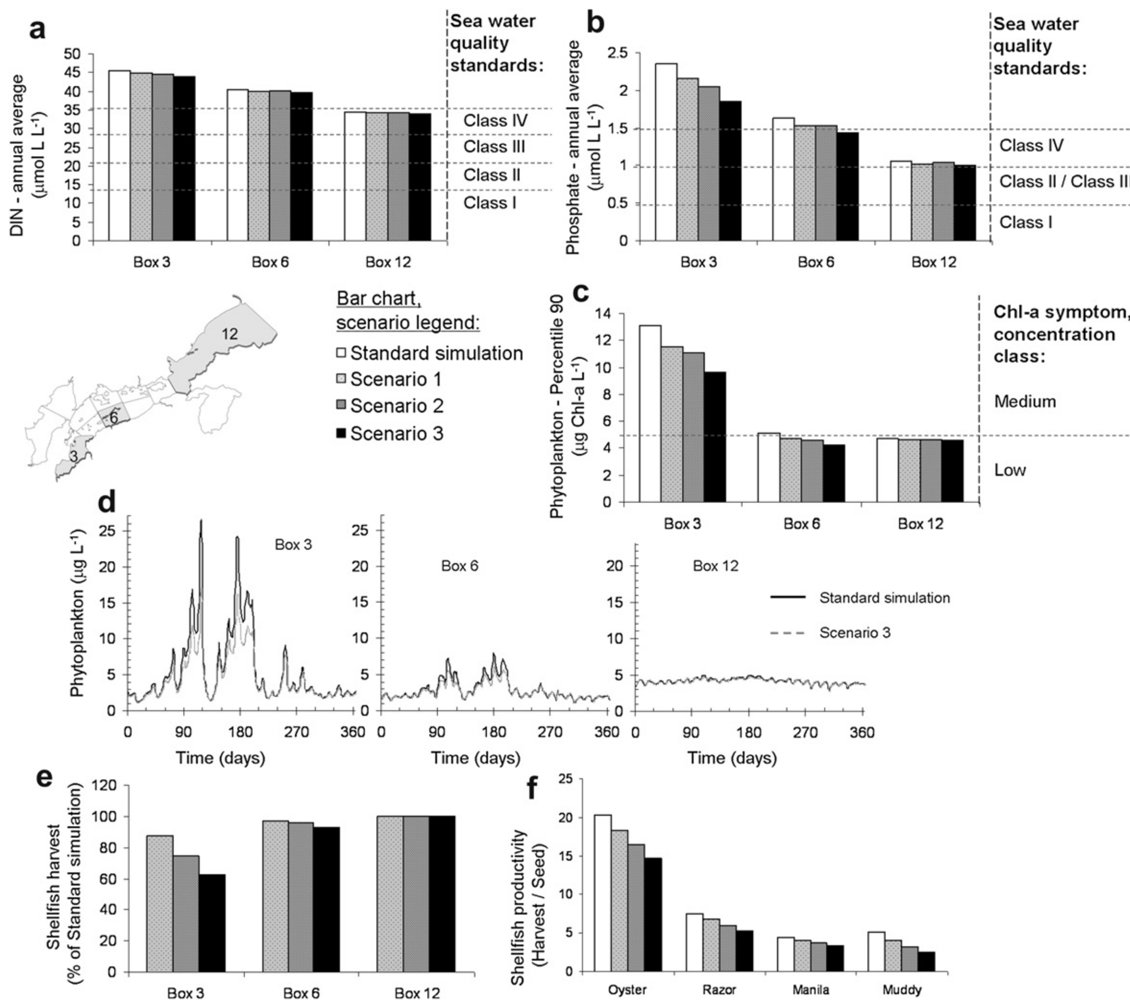


Fig. 9. Scenario simulation outputs for an inner box (Box 3, Huangdun Bay), a middle box (Box 6), and an outer box (Box 12): dissolved inorganic nitrogen (DIN), phosphate, phytoplankton biomass, harvested shellfish and shellfish productivity (calculated as the ratio of total weight of shellfish harvested to total weight of seeding).

Considering thresholds defined in the ASSETS model, this eutrophication symptom is classified as Medium in Box 3 for any scenario. In the middle and outer boxes the phytoplankton concentrations were lower and 90-percentile values fell in the limit between the Low and Medium classes (e.g. in Fig. 9b), possibly due to higher seawater renewal. For Box 6, the small decrease of phytoplankton due to nutrient load reduction resulted in a shift of the phytoplankton 90-percentile value from Medium in the standard scenario to Low in any of the scenarios. In Box 12, the phytoplankton 90-percentile value falls within the Low class for all the scenarios.

Overall, the simulated actions had a limited positive impact on the water quality in the bay. There was an improvement in the chl-a classification from Medium to Low with the implementation of every scenario in Box 6 and with implementation of scenarios 2 and 3 in Box 7. Regarding DIN concentration, there was a reduction in Box 8 following the implementation of every scenario, which lowers the ranking to Class IV (poor). There was also a reduction of phosphate concentration in Scenario 3 that lowers the classification of this variable to Class IV (poor) in, and to Class II/III in Box 12.

For all scenarios, the model predicted a decrease of shellfish productivity for each cultivated species when compared with the standard scenario (Fig. 9f). Fig. 9e indicates that the shellfish production decrease was more significant in the inner box (Box 3, 12–37% corresponding to less 286–864 t year^{-1}), whereas in the outer box (Box 12) no significant changes occurred (0.1–0.2% corresponding to less 8–16 t year^{-1}). A more detailed examination of the shellfish productivity in each box and scenario (Fig. 10) showed that in general productivity levels were lower in boxes 1–7 (inner) and higher in boxes 8–12 (outer).

4. Discussion

The modelled nutrient load reduction had no significant effect on the water quality of the Xiangshan Gang according to Chinese Seawater quality thresholds for nitrate and phosphate. Improvements in phytoplankton concentration were limited to some areas of the bay. Therefore, the model suggests that the proposed scenarios will not achieve the management goals they were designed for. From an eutrophication perspective, there remains a Moderate High to High proportion of nutrient loads from the catchment and fish cages to Xiangshan Gang that need to be managed. Future work using this multilayered ecosystem model includes the definition of further scenarios, using the SWAT model to assess how different land use management practices may impact the bay. Likewise, future scenarios might include the adoption of different aquaculture practices such as described by Ayer and Tyedmers (2009) to decrease the wastes from fish cages. The model outputs indicated that the nutrients and POM provided by fish cages and wastewater are sustaining shellfish growth in the inner

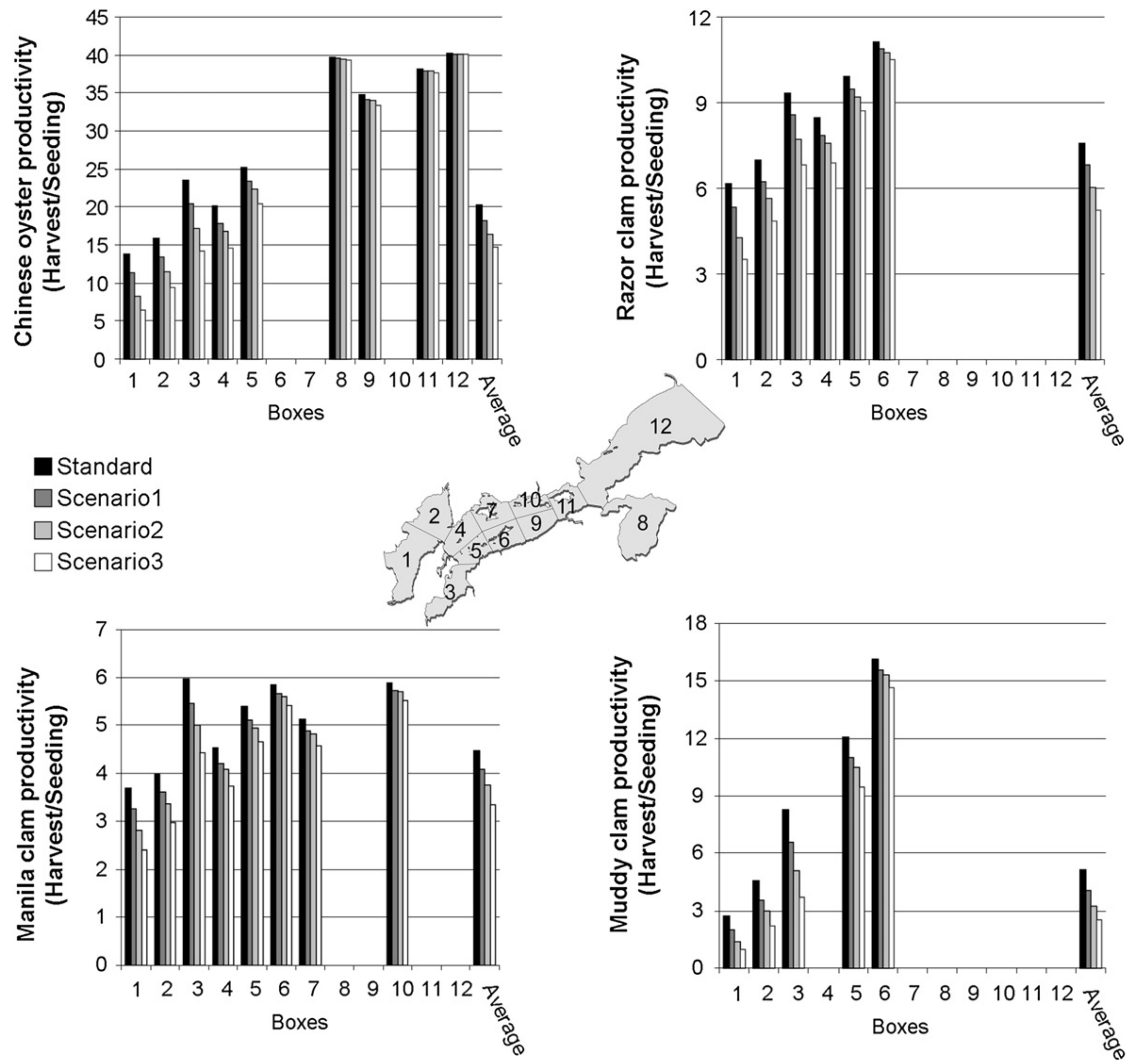


Fig. 10. Shellfish productivity, calculated as the ratio of total weight of shellfish harvested to total weight of seeding.

boxes. In the scenarios that test a decrease of these substances (Table 8), shellfish production decreases (Fig. 9e,f and Fig. 10). The estimated total loss of harvested shellfish was between 4600 t year^{-1} and 12 700 t year^{-1}, corresponding to a relative decrease in the range of 10–28%, and to a loss of annual revenue between 555 and 1500 thousand Euro. Those effects are predicted to be more evident in the inner section of the Xiangshan Gang because of: (1) higher water residence times, in the range of 60–80 days; and (2) higher competition for food resources given that cultivation areas in boxes 1–5 represented 89% of the total shellfish cultivation area, whereas these boxes accounted only for 34% of the total bay area.

As such, and based on the analysis in Fig. 10, it is advisable to reallocate part of the shellfish culture towards the mouth of the embayment, in particular for the Chinese oyster and muddy clam. Such measures should be adopted in parallel to the reduction of substance loading into the bay in order to minimize the reduction in shellfish production. Notwithstanding, it is suggested that a cost-benefit analysis should be carried out to analyse the economic and environmental viability of alternative sources of income for the local community that might compensate for any decrease in aquaculture activities. A combined environmental and economic strategic assessment is even more important given that the Xiangshan Gang area is considered as a key area to promote sustainable development of the Ningbo municipality. Planning includes a balance between its protection and its use, to take advantage of ecological and marine resources (Ningbo Municipal People's Government, 2006). Expected uses include the entertainment and tourism industries, modern fishing and international logistics such as harbour activities.

The multilayered ecosystem model presented in this paper can be used to simulate further nutrient and aquaculture management scenarios in Xiangshan Gang, and in particular to test varying nutrient loads from catchment and aquaculture sources in order to determine the nutrient load level required to meet water quality targets for the bay. On this basis, an indication of the various management options available for such load reduction and corresponding costs could be provided.

Although harmful algal blooms are a severe problem in the Xiangshan Gang and adjacent ocean (ZOFB, 2008), due to the complex and uncertain causes of HAB and the chaotic nature of these events (Huppert et al., 2005; Huang et al., 2008a) HAB simulation is not included in the ecosystem model. While some observations indicate that many red tides originate in the East

China Sea, some have developed inside the bay (Long et al., 2008; ZOFB, 2008). Severe economic losses were associated with these incidents, either as a result of shellfish and finfish mortalities due to toxic algae or to interdiction of seafood sales from the affected areas (ZOFB, 2008). The increase of HAB's in China since 2000 may be associated with an increase of fish cages (Wang, 2002), but given the uncertainty about causes of HAB's in Xiangshan Gang, it is speculative whether a reduction of nutrient discharge might cause a reduction of the occurrence of HAB's inside the bay and a consequent reduction in aquaculture closure time due to toxin contamination and/or death of cultivated organisms. A clear understanding about the origin and the triggering mechanisms of the HAB's in the Xiangshan Gang is required for determining the management possibilities. Monitoring of HAB events is recommended, in particular research about causative and sustaining factors for HAB, which can be applied for managing aquaculture sites subject to these events (Babaran et al., 1998).

The comparison of ecosystem model results using different temporal resolutions for the catchment loads illustrates the importance of the SWAT catchment model in providing a temporally distributed estimate of water and nutrient loadings from catchments into coastal systems, for different outlets. These issues should be further explored. A detailed sampling program together with the catchment modelling should be used to guide on the amount of catchment monitoring data and temporal resolution to use in coastal ecosystem models. Likewise, similar research should be carried out for ocean boundary conditions and aquaculture loads.

5. Conclusions

The outcomes obtained for Xiangshan Gang indicate that multilayered ecosystem models can play a key role in Integrated Coastal Zone Management and for the adoption of an ecosystem-based approach to marine resource management. The present case study also indicates that the integration of ecosystem-based tools can be used to fill data gaps, improve the temporal/spatial detail of the setup datasets, and provide guidance to monitoring programmes.

The multilayered ecosystem modelling approach is appropriate to support management of coastal and estuarine systems worldwide including the assessment of cumulative impacts of activities developed in these zones. Overall, the modelling approach presented in this paper can be helpful for the implementation of legislation and other regulatory instruments. For instance, it can contribute towards the implementation of the European Marine Strategy Framework Directive (Directive, 2008/56/EC), for analysing scenarios designed to achieve the 'good environmental status' (GES) in coastal waters.

To maximize the potential benefits of multilayered ecosystem models, a natural development is the application of aggregated results in simple screening models for management, and the coupling of this kind of ecological model to socio-economic models, in order to more effectively address the interactions between natural and social systems.

Acknowledgments

The authors would like to thank four anonymous reviewers for their valuable comments on an earlier draft. We are grateful to the EU project SPEAR – Sustainable options for PEople, catchment and Aquatic Resources (INCO-CT-2004-510706) for funding this research. Part of the work was supported by the Portuguese Foundation for Science and Technology (FCT) Ph.D. scholarship (SFRH/BD/25131/2005) to A.M. Nobre. The authors are grateful to C. Saurel for valuable comments.

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