ANA MARIA DOMINGOS NOBRE

INTEGRATED ECOLOGICAL-ECONOMIC MODELLING AND ASSESSMENT APPROACH FOR COASTAL ECOSYSTEM MANAGEMENT

Thesis submitted to the Faculty of Sciences and Technology, New University of Lisbon, for the degree of Doctor of Philosophy in Environmental Sciences.

Dissertação apresentada para obtenção do Grau de Doutor em Ciências do Ambiente, pela Universidade Nova de Lisboa, Faculdade de Ciências e Tecnologia.

LISBOA

Agosto 2009

Acknowledgments

Abstract

Over the past few decades, policy-makers have defined new instruments to address coastal ecosystem degradation. Emerging coastal management frameworks highlight the use of the best available knowledge about the ecosystem to manage coastal resources and maintain ecosystem's services. Progress is required, however, in translating data into useful knowledge for environmental problem solving. This thesis aims to contribute to research assessing changes in coastal ecosystems and benefits generated due to management actions (or to the lack thereof). The overall objectives are to assess the ecological and economic impacts of existing management programmes, as well as future response scenarios and to translate the outcomes into useful information for managers.

To address these objectives, three different approaches were developed:

□ <u>A multilayered ecosystem model</u>

A multilayered ecosystem model was developed to simulate management scenarios that account for the cumulative impacts of multiple uses of coastal zones. This modelling field is still at an early stage of development and is crucial, for instance, to simulate the impacts of aquaculture activities on the ecosystem, accounting for multiple farms and their interactions with other coastal activities. The multilayered ecosystem model is applied in this thesis to test scenarios designed to improve water quality and manage aquaculture.

□ <u>An ecological-economic assessment methodology (ΔDPSIR approach)</u>

The Differential Drivers-Pressure-State-Impact-Response (ΔDPSIR) approach further develops the integrated approach by providing an explicit link between ecological and economic information related to the use and management of coastal ecosystems. Furthermore, the ΔDPSIR approach provides a framework to synthesise scientific data into useful information for the evaluation of previously adopted policies and future response scenarios. The ΔDPSIR application is tested using different datasets and scales of analysis, including: (i) assessment of the ecological-economic impacts of the scenarios at the waterbody/watershed level, using the multilayered ecosystem model outputs, and (ii) evaluation of the ecological-economic effects of aquaculture options at the individual aquaculture level, using data from an abalone farm. These are two important scale of analysis for the development of an ecosystem approach to aquaculture.

❑ A dynamic ecological-economic model (MARKET model)

One of the missing links in ecosystem modelling is with economics. The MARKET model was developed to simulate the feedbacks between the ecological-economic components of aquaculture production. This model was applied to simulate shellfish production in a given ecosystem under different assumptions for price and income growth rates and the maximum available area for cultivation. Further application of the MARKET model at a wider scale might be useful for understanding the ecological and economic limitations on global aquaculture production.

This integrated ecological-economic modelling and assessment approach can be further applied to address new coastal management issues, such as coastal vulnerability to natural catastrophes. It can also support implementation of current legislation and policies, such as the EU Integrated Coastal Zone Management recommendation or the development of River Basin Management Plans following the EU Water Framework Directive requirements. On the other hand, the approach can address recurring coastal management needs, such as the assessment of the outcomes of past or on-going coastal management plans worldwide, in order to detect symptoms of the overuse and misuse of coastal ecosystems.

Resumo

Ao longo das últimas décadas, os decisores políticos têm definido novos instrumentos para combater a degradação dos ecossistemas costeiros. Abordagens emergentes de gestão de ecossistemas costeiros salientam o uso do melhor conhecimento disponível sobre o ecossistema para a gestão dos recursos costeiros. Desenvolvimentos são necessários para sintetizar dados em informação relevante para a resolução de problemas ambientais. Esta tese visa contribuir para a investigação sobre a avaliação de alterações nos ecossistemas costeiros e nos benefícios que estes geram devido a medidas de gestão (ou a falta delas). Os objectivos gerais são avaliar os impactes ecológicos e económicos de medidas de gestão adoptadas anteriormente, bem como, de cenários de resposta; e traduzir os resultados em informações úteis para os gestores.

Para atingir os objectivos definidos foram desenvolvidas três metodologias:

    ❑ Um modelo de ecossistema multicamadas

O modelo de ecossistema multicamadas é desenvolvido para simular cenários de gestão que integram os impactes cumulativos dos múltiplos usos das zonas costeiras. Esta é uma área da modelação do ecossistema ainda numa fase inicial de desenvolvimento e crucial para, por exemplo, simular os impactos das actividades aquícolas no ecossistema de forma a incluir a interacção entre diversas unidades de produção e com outras actividades costeiras. O modelo de ecossistema multicamadas é aplicado para testar cenários concebidos para melhorar a qualidade da água e gestão da aquacultura.

    ❑ Uma metodologia de avaliação ecológica-económica (ΔDPSIR)

A metodologia 'Differential Drivers-Pressure-State-Impact-Response' (ΔDPSIR) adiciona uma vantagem à abordagem integrada através da ligação explícita entre informação ecológica e económica relacionada com o uso e gestão de sistemas costeiros. Adicionalmente, o ΔDPSIR fornece uma abordagem para sintetizar os dados científicos em informações relevantes para gestores sobre a avaliação de políticas adoptadas no passado e de cenários para o futuro. A aplicação do ΔDPSIR é testada usando diferentes tipos de dados e escalas de análise, incluindo: (i) avaliação do impacto ecológico-económico dos cenários à escala da massa de água/bacia hidrográfica usando os resultados do modelo multicamadas, e (ii) avaliação dos efeitos ecológico-económicos de diferentes opções da aquacultura a nível de uma unidade de produção individual usando os dados de uma aquacultura de abalone. Estas

são duas escalas de análise importantes para o desenvolvimento de uma abordagem de ecossistema para a aquacultura.

◻ Um modelo ecológico-económico dinâmico (MARKET)

Uma das limitações dos modelos de ecossistema é a ligação com a economia. O modelo MARKET foi desenvolvido para simular o feedback entre as componentes ecológica e económica da produção aquícola. Foi aplicado para simular a produção de bivalves num determinado ecossistema, considerando diferentes pressupostos para as taxas de crescimento de preço e de salários, e para a área máxima disponível para o cultivo. A aplicação do modelo MARKET à escala mais ampla pode ser útil para compreender as limitações ecológicas e económicas da produção de aquacultura a nível mundial.

Esta abordagem integrada ecológico-económica de modelação e avaliação pode ser utilizada para responder a novas questões de gestão das zonas costeiras; tais como a vulnerabilidade a catástrofes naturais. Pode também ser usada para a implementação de legislação e políticas, tais como a recomendação Europeia sobre a Gestão Integrada das Zonas Costeira, ou o desenvolvimento dos Planos de Gestão de Bacia Hidrográfica conforme indicado na Directiva Quadro da Água. Por outro lado, a abordagem desenvolvida pode também responder a necessidades recorrentes dos gestores, nomeadamente avaliar os resultados de planos de gestão costeira já finalizados ou a decorrer, com o intuito de detectar os sintomas visíveis de abuso e mau uso dos ecossistemas costeiros.

Abbreviations

ΔDPSIR, Differential Drivers-Pressures-State-Impact-Response

ASSETS, Assessment of Estuarine Trophic Status model

BOD5, Five-day biochemical oxygen demand

Chl-a, Chlorophyll-a

CZM, Coastal Zone Management

DIN, Dissolved Inorganic Nitrogen

DO, Dissolved oxygen

DPSIR, Drivers-Pressures-State-Impact-Response

DSS, Decision Support Systems

EAA, Ecosystem Approach to Aquaculture

EBM, ecosystem-based management

EC, Eutrophic Condition index of the ASSETS model

EU, European Union

EBM, Ecosystem-Based Management

FO, Future Outlook index of the ASSETS model

GES, Good Environmental Status

GHG, Greenhouse gas

GIS, Geographic Information System

GNP, Gross National Product

GPP, Gross Primary Production

HAB, Harmful algal bloom

ICM, Integrated Coastal Management

ICZM, Integrated Coastal Zone Management

IEA, Integrated Environmental Assessment

IF, Influencing Factors index of the ASSETS model

IMF, International Monetary Fund

IMTA, Integrated Multi-Trophic Aquaculture

LCA, Life-Cycle Assessment

MARKET, Modeling Approach to Resource economics decision-maKing in EcoaquaculTure

MSFD, Marine Strategy Framework Directive

N, Nitrogen

NEEA, USA National Estuarine Eutrophication Assessment

NEP, USA National Estuary Program

NPP, Net Primary Production

P, Phosphorus

PEQ, Population equivalent

PEV, Partial Ecosystem Value

POM, Particulate Organic Matter

PPP, Purchasing Power Parity

RS, Remote Sensing

SAV, Submerged Aquatic Vegetation

SCI, Science Citation Index

SPM, Suspended Particulate Matter

SWAT, Soil and Water Assessment Tool model

TEV, Total Economic Value

TFW, Total Fresh Weight

USA, United States of America

USD, U.S. dollar

UWWTD, Urban Waste Water Treatment Directive

WFD, European Water Framework Directive

WWTP, Wastewater treatment plant

Symbols

ΔDPSIR – economic quantification

VDrivers, Value of the drivers

VDrivers$_{Ecosyst}$, Value of the drivers in the coastal ecosystem

VDrivers$_{External}$, economic value of the activities both in the catchment

VEcosystem, Value of the ecosystem

VImpact, Value of the impact on the ecosystem

VManagement, Economic value of management

VResponse, Value of the response

VDirectUse, Direct use value of the ecosystem

VIndirectUse, Indirect use value of the ecosystem

VNonUse, Non-use value of the ecosystem

VExternalities, Value of the environmental externalities

Simple nutrient mass balance model

Fsea, nutrient source - nutrients from seawater

Fabalone, nutrient source - net nutrient production in the abalone tanks

Ffertilizer, nutrient source - seaweed fertilization

Falgae, nutrient sink - nutrient sinks include seaweed nutrient uptake

Feffluent, nutrient sink - nutrient discharge to the sea

r_{uptake}, nutrient uptake rate

Frecirculation, is the nutrient mass in seaweed effluents that re-enters into the system

Fabalone2algae, is the nutrient mass outflow from the abalone tanks to the seaweed ponds.

e_{uptake}, is the seaweed nutrient removal efficiency (%) that corresponds to the proportion of nutrients removed relative to the available nutrients

MARKET model

SimP, Simulation period

ts, Simulation timestep

ts$_{ecol}$, Ecological timestep

ts$_{econ}$, Economic timestep

Ecological system

μ, Mortality rate

A, Cultivation area

G, Annual growth rate

g, Scope for growth

HB, Harvestable biomass

MaxA, Maximum cultivation area

N, Number of individuals

n$_{seed}$, Seeding density

s, Weight class

sp, Seeding period

tp, Cultivation cycle

w, Ecosystem model seed weight

Economic system

DK, Depreciation of capital

DQ, Desired production

FC, Fixed costs

IKL, Interest on capital loan

K, Capital

L, Labour

LD, Local demand

MC, Marginal costs

MP$_K$ Marginal productivity of capital

MP$_L$, Marginal productivity of labour

MR, Marginal revenue

NP, Net profit

P, Price

Q, Shellfish production

TC$_Q$, Total cost of shellfish production

TC$_{Q+1}$, Total cost of producing one more unit

UVC$_L$ Unit labour cost

VC, Variable costs

VC$_L$, Labour costs

VC$_M$, Maintenance costs

VC$_O$, Other variable costs

d$_f$, Depreciation fraction

d$_p$, Depreciation period

e$_d$, Price elasticity of demand

e$_y$, Income elasticity of demand

m$_f$, Maintenance Fraction

r, Interest rate

R$_{CQ}$, Desired change in production

r$_d$, Demand growth rate

R$_K$, Changes in labour inputs

R$_L$, Changes in labour inputs

r$_{cq}$, annual change rate in production

r$_p$, Price growth rate

r$_y$, Per capita income growth rate

α$_K$, Elasticity of capital

α$_L$, Elasticity of labour

Authorship declaration for published work

Part of the work presented in this dissertation was previously published/submitted as articles in peer-reviewed International journals:

Nobre, A., 2009. An ecological and economic assessment methodology for coastal ecosystem management. Environmental Management, 44(1): 185-204.

Nobre, A.M., Ferreira, J.G., 2009. Integration of ecosystem-based tools to support coastal zone management. Journal of Coastal Research, SI 56: 1676-1680.

Nobre, A.M., Musango, J.K., de Wit M.P., Ferreira, J.G. 2009. A dynamic ecological-economic modeling approach for aquaculture management. Ecological Economics, 68(12): 3007-3017.

Nobre, A.M., Ferreira, J.G., Nunes, J.P., Yan, X., Bricker, S., Corner, R., Groom, S., Gu, H., Hawkins, A.J.S., Hutson, R., Lan, D., Lencart e Silva, J.D., Pascoe, P., Telfer, T., Zhang, X., Zhu, M., Assessment of coastal management options by means of multilayered ecosystem models. Estuarine Coastal and Shelf Science, *In Press*.

Nobre, A.M., Robertson-Andersson, D., Neori, A., Sankar, K., Ecological-economic assessment of aquaculture options: comparison between monoculture and integrated multi-trophic aquaculture. Submitted to Aquaculture.

Nobre, A.M., Bricker, S.B., Ferreira, J.G., Yan, X., de Wit, M., Nunes, J.P., Integrated environmental modelling and assessment of coastal ecosystems, application for aquaculture management. Submitted to Coastal Management.

I hereby declare that as the first author of the above mentioned manuscripts, provided the major contribution to the research and technical work developed, to the interpretation of the results and to the preparation of the manuscripts.

List of contents

List of figures

List of tables

Índice de figuras

Índice de quadros

Chapter 1. Introduction

This chapter presents the frame of reference for the work developed and provides an overview of the thesis. The first part reviews the coastal management challenge and the role of science in addressing emerging coastal zone problems. The second part describes the thesis objectives, presents the study sites used to develop the work and outlines the thesis structure.

1.1   Background

1.1.1 Coastal management challenge: addressing emerging coastal zone problems

Coastal zones comprise important ecosystems (MA, 2005), which generate goods and services with a high economic value (Ledoux and Turner, 2002). As a result, a strip 100 km wide along the coastline contains nearly 40% of the world population and 61% of the gross world product (MA, 2005). Anthropogenic pressures increasingly compromise, directly and indirectly, the important benefits generated by coastal systems (MA, 2005; Costanza and Farley, 2007). The main human threats to coastal areas include: loss of natural habitats, loss in biodiversity and cultural diversity, decline in water quality, vulnerability to global changes such as predicted sea level rise, increased negative impacts of coastal disasters, the diversity of human activities, competition for space and seasonal variations in pressure (Ehler et al., 1997; Fabbri, 1998; Humphrey et al., 2000; MA, 2005; Costanza and Farley, 2007). Therefore, sustainable development of coastal zones constitutes a challenge for stakeholders with a role in coastal management.

Integrated Coastal Zone Management (ICZM)

Policy-makers worldwide have defined policy and legislative instruments to address the emerging coastal zone problems (Clark, 1996; Borja, 2006; Ducrotoy and Elliott, 2006). One of the more widely known and applied is the Integrated Coastal Zone Management (ICZM) approach. ICZM is defined as a dynamic management process that brings together the human and the ecological dimensions to promote the sustainable use, development and protection of coastal zones (Clark, 1996; Olsen, 2003; Forst, 2009). Managers worldwide have adopted ICZM within different contexts: 1) to address specific environmental problems emerged in coastal zones or to manage coastal vulnerability to natural hazards and climate change (Clark, 1996; Krishnamurthy et al., 2008); 2) either at national or local levels, as exemplified by NRMMC (2006) and Lewis III et al. (1999), respectively; 3) following a top-down approach or based on a community-based initiative (Cicin-Sain and Knecht, 1998; Lewis III, et al., 1999; Belfiore, 2000; Kearney et al., 2007). Table 1.1 presents an overview of worldwide coastal management initiatives. Although such synthesis is reductionist about coastal management efforts, it illustrates that ICZM initiatives appeared about four decades ago and that some countries are currently adopting new programmes.

Table 1.1. Overview of major ICZM initiatives worldwide.

	First initiatives		Recent initiatives	
	Date	Programme	Date	Programme
Australia			2003	Framework for a National Cooperative Approach to ICZM
South Australia	1972	Coast Protection Act	1994	Environ. Protection (Marine) Policy
			2000	Environ. Protection (Water Quality) Policy
New South Wales	1979	Coastal Protection Act		
Queensland			1995	Coastal Protection and Management Act
Tasmania			1996	State Coastal Policy
Victoria			1995	Coastal Management Act 1995
Western Australia			2001	CZM Policy
Brazil	1988	Law 7661, establishes the National CZM Plan	2004	Decree 5300, regulates the National CZM Plan and other instruments for ICZM
Canada			1991	Atlantic Coastal Action Program (ACAP)
			1997	Oceans Act
			2002	Oceans Strategy
EU			1996	Demonstration programme on ICZM
			2000	Water Framework Directive (2000/60/EC)
			2002	ICZM Recommendation 2002/413/EC For member states to adopt a National strategy on ICZM.
			2008	Marine Strategy Framework Directive (2008/56/EC)
Baltic Sea			2003	HELCOM ICZM Recommendation 24/10
			2007	The HELCOM Baltic Sea Action Plan
France	1975	Coastal Conservancy (Law of 10 July)	2002	Reform of the Coastal Conservancy's mission (law of 27 February)
	1983	Marine Area Zoning Plan (SMVM) (law of 7 January)		
	1986	Planning, protection, and development of Coastal Space		
IOC member states			1997	Integrated Coastal Area Management (ICAM) programme adopted by the Intergovernmental Oceanographic Commission (IOC)
New Zealand			1994	Coastal Policy Statement (NZCPS)
			2008	Proposed review of NZCPS
USA	1972	CZM Act		
	1987	National Estuary Program (NEP), established by the Water Quality Act of 1987		

The early USA concerted coastal management efforts are stable and in a mature stage (Hershman et al., 1999; Hale, 2000; Gibson, 2003). Hershman et al. (1999) and Humphrey (2000) describe the key features for its success and its shortcomings. Coastal management

3

programmes on a European scale are more recent (Humphrey et al., 2000; Shipman and Stojanovic, 2007). The various EU policies and directives emerged as complementary instruments the most important being (Borja, 2006; Ducrotoy and Elliott, 2006; 2008): the Water Framework Directive (WFD) of 2000, the ICZM recommendation of 2002 and the Marine Strategy Framework Directive (MSFD) of 2008. Table 1.1 shows only a brief sample of the programmes adopted within EU. The individual EU member states have different approaches to coastal management with a variety and complexity of coastal management initiatives and legislations (Gibson, 2003; Rupprecht Consult and IOI, 2006; Shipman and Stojanovic 2007). For detailed coastal management initiatives within and across member states refer to van Alphen (1995), Barragán (2003), Eremina and Stetsko (2003), Pickaver (2003), Veloso-Gomes et al. (2003), Anker et al. (2004), Taveira Pinto and Paskoff (2004), Astron (2005), Enemark (2005), Smith and Potts (2005), DOENI (2006), Rupprecht Consult and IOI (2006), WAG (2007), Deboudt et al. (2008), DEFRA (2008). Clark (1996), Kay et al. (1997), Cicin-Sain and Knecht (1998), Hale (2000) and Krishnamurthy et al. (2008) provide detailed ICZM case studies developed worldwide.

For individual ICZM programmes to evolve, comprehensive evaluations are required. It is important that ICZM program output evaluation is combined with 'state-of-the-coast' information to show, for instance, whether new program goals may be needed and to allow an ICZM program to evolve to an improved version (Olsen et al., 1997; Hershman et al., 1999; Stojanovic et al., 2004; Billé, 2007). However, most of the evaluation efforts focus on measuring the evolution of the ICZM process outputs (Olsen, 2003; Pickaver et al., 2004; Stojanovic et al., 2004; Billé, 2007). Worldwide and independently of maturity of the ICZM process, there is a lack of measurements of its effectiveness, i.e. of the consequent changes in the state of the coastal systems, its resources and associated benefits (Knecht et al., 1996, 1997; Kay et al., 1997; Olsen et al., 1997; Hershman et al., 1999; Humphrey et al., 2000; Billé, 2007; McFadden, 2007). Table 1.2 presents a synthesis of the few studies that evaluated the effectiveness of ICZM programmes. Among other reasons, the difficulty to select criteria to measure performance of the system stands out. The difficulty stems from (i) unclear set of objectives of ICZM, (ii) complexity of coastal ecosystems, and (iii) data requirements (Burbridge, 1997; Stojanovic et al., 2004). Problems for defining a specific set of indicators for all coastal systems are greater at the national or broader level due to different susceptibility and resilience of ecosystems, pressures these are subject and issues to be tackled (Pickaver et al., 2004). The diversity of coastal systems and of the pressures on them require flexibility in the development and implementation of ICZM programmes, which on the other hand call for flexible assessment approaches (Humphrey et al., 2000; Olsen, 2003).

4

Table 1.2. Examples of evaluation of the effectiveness of ICZM programmes.

Programme / Domain	Description	Source
Perceptions of the performance of 24 state CZM programmes in the USA, undertaken in 1993-1995.	Evaluation was based on a survey about perceived performance on four major coastal management issues: (1) protection of coastal resources, (2) management of coastal development, (3) improved public access, and (4) reduction of losses due to coastal hazards.	Knecht et al., 1996
USA National CZM effectiveness study, undertaken in 1995-1997	Objective is to determine success of 5 of the core objectives of the USA CZM Act of 1972: (1) protection of estuaries and coastal wetlands, (2) protection of beaches, dunes, bluffs and rocky shores, (3) provision of public access to the shore, (4) revitalisation of urban waterfronts, and (5) accommodation of seaport development. Although based on limited data it evaluates programme success based "on-the-ground outcomes".	Hershman et al., 1999
Tampa Bay Estuary Program (USA)	The programme includes the definition of specific goals to address the identified issues to be managed. Quantitative criteria were selected to evaluate the program outcomes. These include for instance areal extent of seagrasses and populations of birds.	Lewis III et al., 1999

The development of indicators and tools to evaluate ICZM at different levels is ongoing as analysed by Hoffmann (2009). For instance, Cordah Ltd (2001) and Belfiore et al. (2006) consolidated a suite of indicators developed worldwide for ICZM. At the European level assessment tools are also being developed in a collaboration between managers and the research community (Ducrotoy and Elliott, 2006). An important feature of this effort is the inclusion of measurable indicators as common tools to quantify both the progress of implementation of ICZM and the sustainable development of the coastal zone (Breton, 2006). These worldwide efforts are valuable contributions for making the assessment about the evolution of coastal zones the standard rather than the exception in the ICZM process.

Ecosystem-Based Management (EBM)

Complementary to ICZM, ecosystem-based management (EBM) emerged recently as a scientific consensus that highlights the use of the best available knowledge about the ecosystem in the management of marine resources, with an emphasis on maintaining ecosystem service functions (Browman and Stergiou, 2005; Murawski, 2007; Murawski et al., 2008; Forst, 2009). The EBM approach recognises the need to consider the cumulative impacts of the range of activities that act on the coastal ecosystems for its sustainable management (Halpern et al., 2008). The concept of ecosystem-based approach first appeared

in the 1970's, not specifically related with coastal zones (Slocombe, 1993). Grumbine (1994) and Slocombe (1998) review the origins and principles of EBM and provide lessons for implementing it. An important feature that both authors highlight is that EBM is about integrating environment and development. They emphasise that in the real systems humans are within rather than separated from nature. Slocombe (1998) suggests that an effective EBM (i) starts with a synthesis of information for future research and management, (ii) monitors features to follow changes, (iii) uses local knowledge, and (iv) is practical, and if resources are limited it needs to focus research on knowledge that is meaningful to management. The definition of operational goals is an important challenge for EBM implementation, according to Slocombe (1998). In one of the first references of EBM for coastal zones, Imperial and Hennessey (1996) identified the USA National Estuary Program (NEP) as a promising ecosystem-based approach to managing estuaries. The particularity of NEP is to focus on solutions for problems identified on each estuary (Imperial and Hennessey, 1996). For each estuary is implemented a comprehensive conservation and management plan which contains an action plan to address problems identified and a monitoring programme to measure effectiveness of activities. Furthermore, the plan sets the funding and the institutional context to implement the estuarine programmes. At the European level, there are also several examples of EBM, for example for the Baltic Sea, North Sea and Wadden Sea (Enemark, 2005; HELCOM, 2007; Ducrotoy and Elliott, 2008). In Canada, the Atlantic Coastal Action Program (ACAP) is an ecosystem and community-based approach to integrated planning and management of the environment that has unique features such as the power sharing among stakeholders (McNeil et al., 2006). The Environment Canada launched it in 1991 and the process consists of development and implementation of management plans, partnership building, local involvement and action and scientific research to improve and maintain the environmental integrity of coastal communities (McNeil et al., 2006). The ACAP established an alternative process to environmental and socio-economic management of coastal zones involving interested stakeholders since the beginning to identify problems and solutions. The evaluation of ACAP focuses on the environmental results and consists of accounting the measures adopted and avoided the avoided pressures, e.g., area of enhanced wildlife habitat or weight of mercury eliminated from waste stream. According to Environment Canada, the ACAP is effective on an ecosystem basis (McNeil et al., 2006).

Ecosystem Approach to Aquaculture (EAA)

Sustainable development of mariculture represents a particular challenge for coastal ecosystem and resources managers for the combination of the following reasons (GESAMP, 2001):

- Aquaculture relevance for food security (Ahmed and Lorica, 2002);

- Rapid growth of aquaculture industry (Duarte et al., 2007a) estimated as about 8.8% per annum since 1970 (FAO 2006);

- Generalised concern that the increasing demand for aquaculture can drive coastal degradation, such as habitat loss, pollution, overexploitation of fisheries for fishmeal and oil, due to unsustainable aquaculture practices (MA, 2005);

- Some aquaculture solutions, including those of extractive species (Neori et al., 2004), are advocated for mitigating some of aquaculture's impacts on coastal ecosystems, for instance cultivation of seaweeds and shellfish (Ferreira et al., 2007a; Gren et al., 2009; Stephenson, et al., 2009);

- Aquaculture aesthetic impacts cause conflicts with other users of coastal zones (Dempster and Sanchez-Jerez, 2008; Gibbs, 2009);

- Impacts of aquaculture activities are cumulative among farms and additive to the impacts of other development pressures in the coastal zone, consequently aquaculture development must be addressed beyond the individual farm level, at the ecosystem level (GESAMP, 2001; Ferreira et al., 2008a; Soto et al., 2008);

- The future of the aquaculture industry relies on sustainable coastal development because ultimately it depends on healthy coastal waters (GESAMP, 2001).

For the above-mentioned reasons an ecosystem approach to aquaculture (EAA), integrated with management of other coastal developments, is required for sustaining aquaculture expansion (GESAMP, 2001; FAO, 2007; Soto et al., 2008). According to FAO, EAA is defined as: "An ecosystem approach to aquaculture (EAA) strives to balance diverse societal objectives, by taking account of the knowledge and uncertainties of biotic, abiotic and human components of ecosystems including their interactions, flows and processes and applying an integrated approach to aquaculture within ecologically and operationally meaningful boundaries. The purpose of EAA should be to plan, develop and manage the sector in a manner that addresses the multiple needs and desires of societies, without jeopardizing the options for future generations to benefit from the full range of goods and services provided by aquatic ecosystems." (FAO, 2007).

1.1.2 Role of science for coastal management

The complexity of the phenomena occurring in coastal ecosystems and their management requires the interaction among managers and researchers of a range of disciplines (Fabbri, 1998). The effective integration of science with management is important for better policy formulation and policy-making for achievement of both environmental and development needs and goals (Slocombe, 1993; Peirce, 1998; Turner, 2000; Cheong, 2008). Currently the role of applied environmental science to support coastal management and address legal requirements is increasing (Ducrotoy and Elliott, 2006). In order to communicate science to managers, researchers must follow a problem-oriented approach and distil the outputs into accessible and useful information for managers (Nobre et al., 2005; Dennison, 2008; Hoffmann, 2009). Such an approach calls for the integration of scientific methodologies and disciplines across different scales (IMPRESS, 2003; McFadden, 2007). In particular, the adoption of an EAA poses several challenges to the scientific community (GESAMP, 2001; Soto et al., 2008). For instance, guidance about more sustainable aquaculture options at the farm level (Neori et al., 2004; Robertson-Andersson et al., 2008; Ayer and Tyedmers, 2009) and understanding of cumulative impacts within coastal ecosystem for determination of, for instance, carrying capacity with respect to aquaculture activity (Ferreira, et al. 2008a).

Overall, ecosystem-based tools capable of providing insights about complex ecological processes and interaction with socio-economic systems are valuable to support the sustainable use of high demanded coastal zones. The most commonly applied tools include (Cicin-Sain and Knecht, 1998; Neal et al., 2003): spatial modelling tools, such as geographical information systems (GIS) and remote sensing; catchment and coastal ecosystem modelling; participatory work with stakeholders; integrated environmental assessment, benefit-cost studies and economic valuation. The aim of these tools is to provide information to the decision-making process or its evaluation and not to replace decision-makers (Van Kouwen et al., 2008). The enhanced understanding scientific methodologies provide can be particularly useful in conflict resolution processes inherent to ICZM (Fabbri, 1998; McCreary et al., 2001). The development of integrative tools requires the interaction of all stakeholders in order to ensure (Cicin-Sain and Knecht, 1998; Van Kouwen et al., 2008) that (i) tools address relevant issues for coastal management, and (ii) managers can use the tools and their outputs. The state-of-the-art about ecosystem-based tools is detailed in *Chapter 6*.

There are three major research areas to support ICZM and EBM: (i) increase of knowledge about complex coastal processes, such as the cumulative impacts of coastal zone multiple

pressures, (ii) development of tools to communicate science to managers, and (iii) interaction of coastal environment and socio-economics. These research areas are discussed below.

Increase of knowledge about complex coastal processes

A knowledge gap highlighted as crucial for coastal management is to understand the cumulative impacts of natural and anthropogenic pressures on coastal ecosystem state, and on the goods and services these areas provide (Halpern et al., 2008). Ecological modelling is recognised as an important tool for coastal management, which can contribute for understanding coastal ecosystem processes including the above mentioned research gap (Turner, 2000; Fulton et al., 2003; Greiner, 2004; Hardman-Mountford et al., 2005; Murawski, 2007; Forst, 2009). In particular, more recently the requirement for models at the ecosystem level capable of simulating the cumulative impacts of multiple uses has been highlighted (Fulton et al., 2003; Ferreira et al., 2008a). Nevertheless, modelling approaches that are able to simulate the cumulative impacts of coastal activities on these ecosystems are still at an early stage of development. Such developments are particularly important for determination of ecological carrying capacity required for the sustainable expansion of aquaculture (Ferreira et al., 2008a; Dempster and Sanchez-Jerez, 2008; Soto et al., 2008). *Chapter 2* provides further details about contributions of ecosystem modelling and state-of-the-art of relevant modelling approaches.

Tools to communicate science to managers

Integration and synthesis of complex knowledge from different disciplines into useful information to coastal managers and the public at large is a progressing and challenging field to environmental scientists (Harris, 2002; McNie, 2007; Cheong, 2008). Integrated Environmental Assessment (IEA) methodologies can enhance communication between scientists and policy-makers, since those methodologies aim to present an interdisciplinary synthesis of scientific knowledge (Tol and Vellinga, 1998; Harris, 2002). IEA outcomes normally provide insight regarding complex phenomena, which can guide decision-making and policy development for ecological resources management (Toth and Hizsnyik 1998). The Drivers-Pressure-State-Impact-Response (DPSIR) is a well-known IEA framework (Peirce, 1998) used to communicate science to coastal managers and in particular to bridge the science-management scales gap (Elliott, 2002). *Chapter 3* further reviews the use of IEA frameworks for coastal management. Ecological modelling in particular can benefit from the integration with IEA methodologies to distil the outcomes of complex models into useful information for managers (Nobre et al., 2005). A review about integration of IEA

methodologies with ecological models is presented in *Chapter 4* (*Section 4.1*). Because it involves human interpretation, one of the IEA caveats is subjectivity and dependence on the analyst point of view (Tol and Vellinga, 1998). That is a criticism specifically pointed out to the DPSIR approach (Svarstad et al., 2008). Tol and Vellinga (1998) recommend that for the use of IEA full potential the methodologies for integrating knowledge need to improvement. Specifically for the DPSIR, Svarstad et al. (2008) suggest the expansion of the framework to incorporate social and economic concerns, rather than just report about the state of the environment.

At the EU level several tools are being developed, specifically to support implementation of coastal management related legislation and policy (Ducrotoy and Elliott, 2006). Specific examples include: (i) GIS as a decision support tool to be used in the development of the National Strategy for ICZM of the Catalan coast following the EU recommendation (Sardá et al., 2005), (ii) GIS use for division of ecosystems into homogenous management units as required by the WFD (Ferreira et al., 2006; Balaguer et al., 2008), (iii) tool to assist in the classification of marine angiosperms, one of the WFD biological elements for coastal and transitional waters (Best et al., 2007), (iv) benthic community-based biotic indices to evaluate ecosystem status and condition, in support of WFD implementation (Pinto et al., 2009). Borja et al. (2008) reviews at the worldwide level, existing integrative assessment tools capable to support recent legislation developed in several nations to address ecological quality or integrity.

A particular area where efforts need to be developed is the production of methodologies to assess the impacts of the ICZM initiatives on coastal ecosystems (Olsen et al., 1997), including the changes in the benefits these generate. *Chapter 3* presents existing methodologies that aim to support sound-decision making.

Interaction of coastal environment and socio-economics

Understanding the linkages between the natural and anthropogenic systems is crucial for ICZM and EBM (Turner, 2000; Westmacott, 2001; Boissonnas et al., 2002; Bowen and Riley, 2003; Cheong, 2008). Firstly, the aim of ICZM is to promote the sustainable development of coastal ecosystems including both ecological and socio-economic components. Secondly, coastal management and planning must account for the 'costs' of resource degradation. Finally, the measurement of the effectiveness of ICZM initiatives must screen not only the consequent changes on the ecological state of the ecosystem but also changes of the socio-economic benefits generated in coastal areas. In particular, economic valuation methods are

crucial to account for ecosystem goods and services in decision-making (Boissonnas et al., 2002; Lal, 2003; Farber et al., 2006; Costanza and Farley, 2007).

The DPSIR approach results of the effort to integrate the natural and anthropogenic systems, and to combine science with management (Cheong, 2008). DPSIR is a widely used conceptual framework for integrated coastal management that provides a conceptual scheme of how socio-economic activities interact with the natural systems (Elliott, 2002; Ledoux and Turner, 2002; Bowen and Riley, 2003; IMPRESS, 2003; Bidone and Lacerda, 2004; GTOS, 2005; Hofmann et al., 2005; Scheren et al., 2004; Borja et al., 2006; Nobre, 2009). In simple terms, the DPSIR establishes the link between the human activities ('Drivers'), corresponding loads ('Pressures'), resulting changes of the 'State' of the ecosystem (i.e. the 'Impact') and the actions adopted by the coastal managers and decision-makers (Response). However, this IEA methodology lacks the formal definition of a consistent linkage between ecological and economic indicators over time (Nobre, 2009). The DPSIR approach and the interaction between the natural and anthropogenic systems is further analysed in *Chapter 3*.

Additionally, the inclusion of the economic component in dynamic ecological models is required in order to simulate the feedback between the ecological and socio-economic systems (Bockstael et al., 1995; Nobre et al., 2009). First attempts to integrate the ecological and economic models date back to the 1960's (Westmacott, 2001). Currently integrated ecological-economic modelling is an evolving discipline that has increased recently (Drechsler et al., 2007). Several difficulties exist, such as the difference in scales at which normally these two systems are simulated or analysed (Nijkamp and van den Bergh 1997; Turner, 2000; Drechsler and Watzold, 2007; Nobre et al., 2009). Existing efforts for integration of ecological and economic models are detailed in *Chapter 5*.

1.2   Thesis overview

1.2.1 Objectives

Anthropogenic activity is generating a negative feedback through the significant direct and indirect socio-economic benefits provided by coastal ecosystems; increasing human pressure on coastal zones (Boissonnas et al., 2002) is causing degradation and consequently decreases the benefits that these ecosystems deliver (Bowen and Riley, 2003; MA, 2005; Costanza and Farley, 2007). Emerging coastal management frameworks use the best available knowledge about the ecosystem to manage marine resources and functions (Fluharty, 2005; Murawski, 2007; Forst, 2009). More progress is needed regarding the process of translating data into

useful knowledge for environmental problem solving, and this is also true with regards to coastal zones (Dennison, 2008). Managers and policy-makers require analytical and assessment methodologies capable of (i) generating understanding about coastal ecosystems and their interaction with the socio-economic system, and (ii) synthesising research outcomes into useful information in order to define effective responses and evaluate previously adopted actions (McNie, 2007; Stanners et al., 2008).

This thesis aims to contribute to research on the assessment of changes in coastal ecosystems and in benefits generated due to management actions (or lack of actions). The overall objectives are to (i) assess the ecological and economic impacts of previously adopted policies as well as future response scenarios, and (ii) translate the outcomes into useful information for stakeholders with a management role.

Table 1.3. Key stages of the integrated ecological-economic modelling and assessment methodology development.

Development stage	Methodology	Discipline	Spatial scale	Event analysis
MD[1] & A[2]	Multilayered ecosystem model (MEM)	Natural sciences	Ecosystem (catchment-coastal)	Forecast
MD[1] & A[2]	Ecological-economic assessment methodology (EEAM)	Natural sciences and socio-economics	Ecosystem (catchment-coastal)	Hindcast
	Ecosystem Approach to Aquaculture:			
A[2]	MEM + EEAM	Natural sciences and socio-economics	Ecosystem (catchment-coastal)	Hindcast / Forecast
A[2]	EEAM	Natural sciences and socio-economics	Individual farm	Forecast
MD[1] & A[2]	Dynamic ecological-economic model	Natural sciences and socio-economics	Ecosystem	Forecast

[1]MD – methodology development; [2]A – application of methodology.

The research work is divided into the key stages synthesised in Table 1.3 and described next. The multilayered ecosystem model aims to simulate the cumulative impacts of multiple uses of coastal zones. A coupled ecological-economic assessment methodology is required as a complement to the modelling approach in order to (i) provide useful information for managers about impacts on the coastal environment of previously adopted ICZM programmes based on data surveys as well as about future response scenarios if used together with a simulation

model, and (ii) provide an explicit link between ecological and economic information related to the use and management of a coastal ecosystem within a specific timeframe. To address the challenges of sustainable aquaculture research and management and, specifically, to support an ecosystem approach to aquaculture, the multilayered ecosystem model and the ecological-economic assessment methodology are used both in combination and individually. Finally, the ecological-economic link is dynamically coupled, in order to take into account feedback between the ecological and economic systems.

1.2.2 Study sites

The research work used different study sites (Figure 1.1) for the application of the various methodologies. The rationales for study site selection were the different requirements of each methodology, the characteristics of the study sites and the available dataset, as described next.
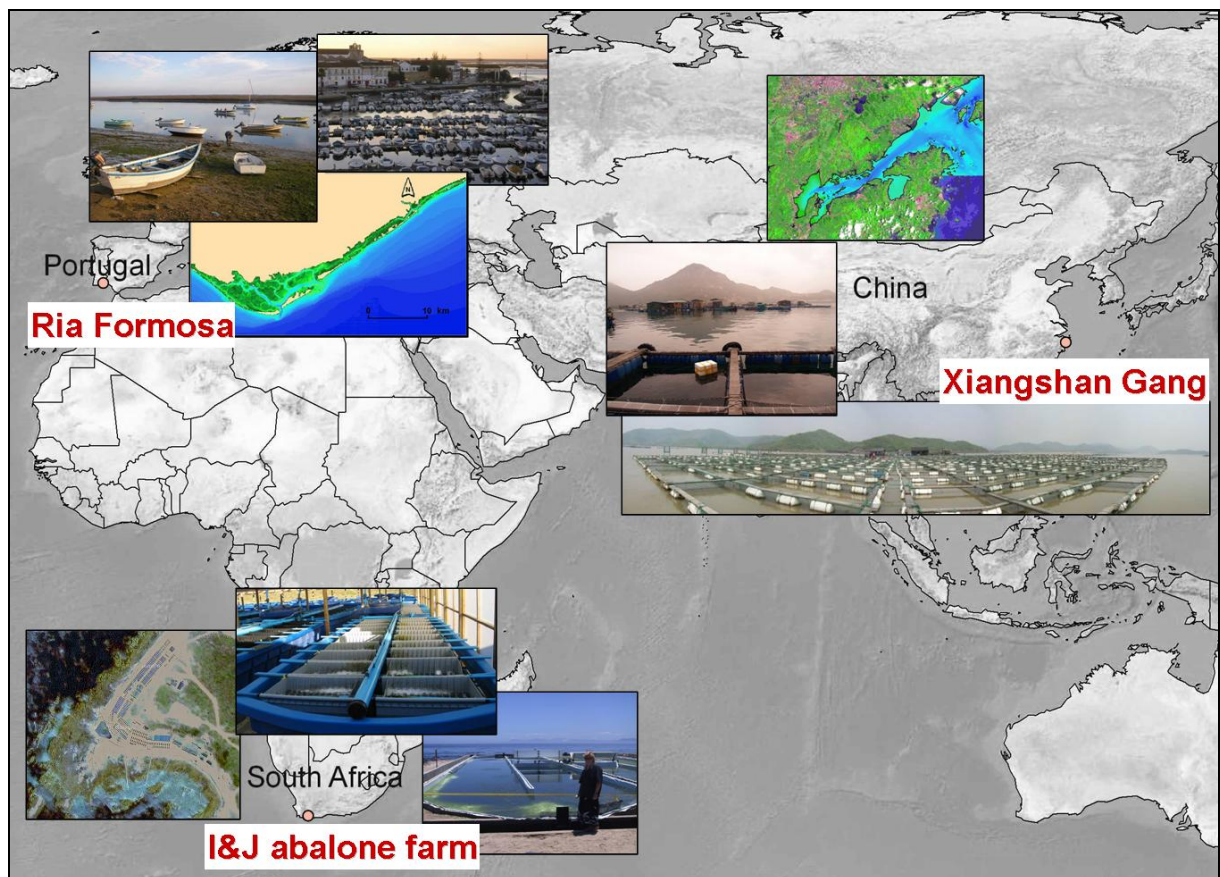
Figure 1.1. Study sites.

❑ The main study site was a Chinese bay, the Xiangshan Gang, with a large amount of aquaculture production and multiple catchment uses. Management efforts to improve water quality are currently under way in this bay. This study site represents a challenge to the local coastal managers because of the multiple uses of the catchment area and marine

ecosystem, such as large aquaculture areas. Xiangshan Gang represents an emblematic coastal ecosystem for the simulation of catchment effects on water quality and aquatic resources. The case study consisted of the simulation of management scenarios that account for changes in multiple uses. Development scenarios, designed in conjunction with local managers and aquaculture producers, included a reduction of fish cages and the treatment of wastewater. The integrated modelling and assessment approach was applied to evaluate the cumulative impacts of the development scenarios on the Xiangshan coastal environment. The model outputs were used to support an ecosystem approach to aquaculture (EAA) in Xiangshan Gang at the waterbody/watershed level. The Xiangshan Gang was also used as a study site to model the explicit link between the ecological and economic systems, in which the MARKET model was applied to simulate shellfish production. The SPEAR research project that took place between 2004 and 2008 (EU Framework VI, INCO-DEV-1 - CT-2004-510706, Ferreira et al., 2008b) provided the means for carrying out advanced integrated modelling work.

❑ A southwest European coastal lagoon, Ria Formosa, which exhibits considerable interaction between the ecological and socio-economic systems, was used as a case study to illustrate the development and application of the ecologic-economic assessment methodology. On the one hand, this coastal zone includes sites of environmental importance recognised by several international conventions and directives. On the other hand, Ria Formosa supports several economic activities that comprise the main source of employment and income in the region (Nobre, 2009). This coastal zone is a well-studied system where accessible datasets are available and other ecosystem-based tools have been applied. As such, Ria Formosa was also used to review ecosystem-based tools used for coastal research and management, including those developed in this thesis.

❑ The third study site is an abalone aquaculture located in South Africa, the Irvine and Johnston (I & J), Cape Cultured Abalone Pty, Ltd, which offers a detailed dataset about farm's ecological and economic performance. This site presented a valuable case study to exemplify the assessment of different aquaculture practices at the individual farm level because the farm recently changed to an IMTA system with macroalgae. In addition, previous research made a unique dataset available that includes not only environmental data from the farm but also its cost structure and revenue information (Robertson-Andersson, 2007; Robertson-Andersson et al., 2008; Sankar, 2009).

The study sites are further detailed in the relevant chapter (Figure 1.2).

1.2.3 Thesis outline

In order to attain the objectives described above, the work was developed step by step, as described in *chapters 2 to 6*, followed by a general discussion chapter. This thesis is organised as follows (Figure 1.2):
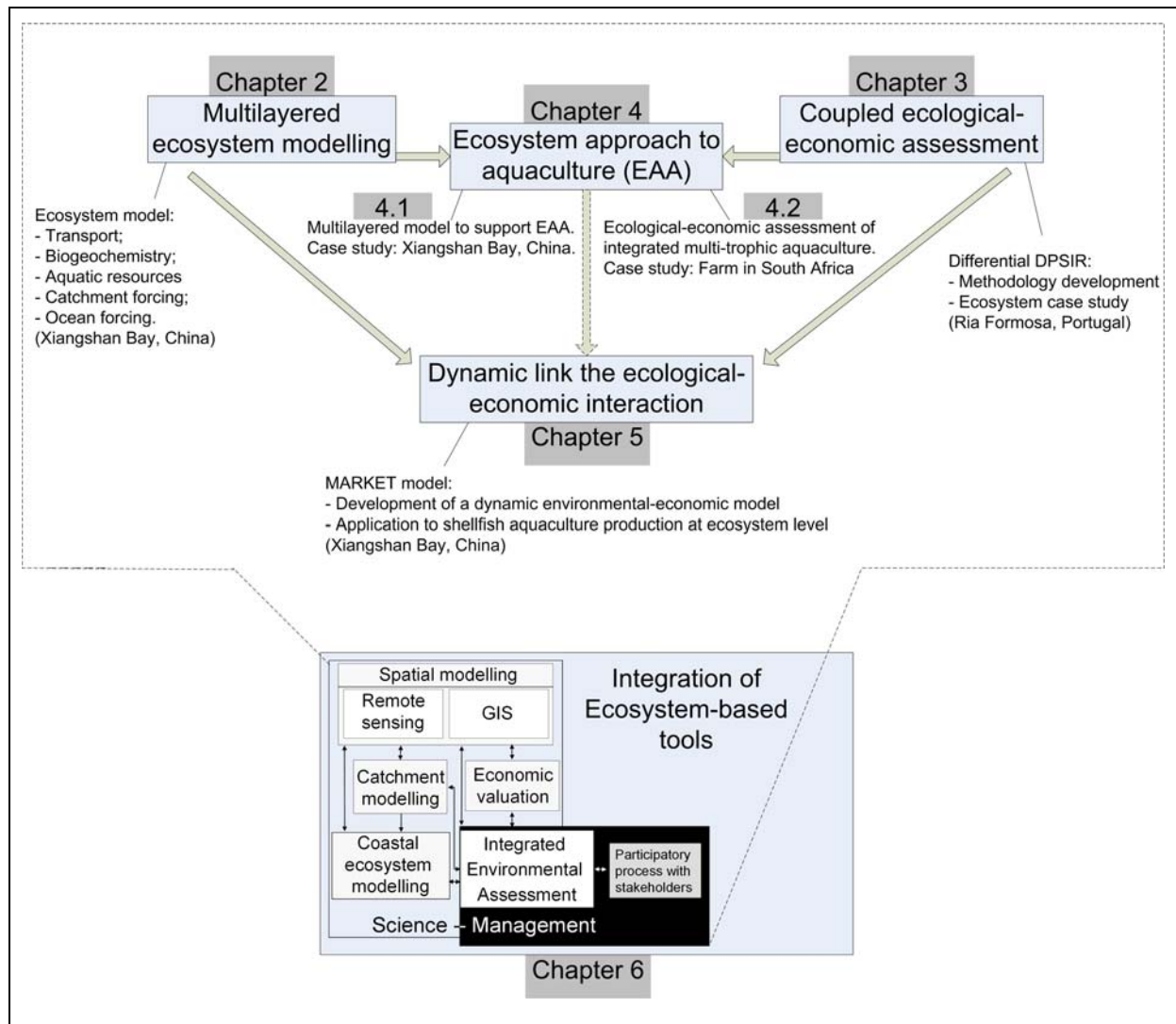
Figure 1.2. Thesis organisation.

Multilayered ecosystem modelling (Chapter 2)

This chapter describes the development and use of the multilayered ecosystem model. The modelling approach combines the simulation of the biogeochemistry of a coastal ecosystem with the simulation of its main forcing functions, such as catchment loading and aquaculture activities. A key feature of the multilayered ecosystem model is the simulation of cumulative impacts in the coastal ecosystem. The model is used to investigate the impacts of different management scenarios and monitoring options on the condition of Xiangshan Gang. This work was developed in collaboration with a multidisciplinary team that provided the required

sub-models for the catchment, hydrodynamic and aquatic resources simulation (Ferreira et al., 2008b).

Integrated ecological-economic assessment (Chapter 3)

This chapter introduces the differential DPSIR (ΔDPSIR) methodology, an adaptation of the general DPSIR approach (Nobre, 2009). The ΔDPSIR includes an explicit linkage between ecology and economy within a specific timeframe. This assessment methodology is developed as a tool to analyse the relationship between the ecosystem state and the use of aquatic resources. The ΔDPSIR aims to provide the scientific-based information required by managers and decision-makers to evaluate the ecological and economic impacts of previously adopted policies, as well future response scenarios, on the coastal environment. The application of the ΔDPSIR is illustrated through an analysis of developments in a southwest European coastal lagoon between 1985 and 1995.

Ecosystem approach to aquaculture (Chapter 4)

This chapter integrates the work developed in *Chapters 2 and 3* to develop an ecosystem approach to aquaculture (EAA). The relevant scales for the EAA application are (Soto et al., 2008) (1) the farm level, (2) the waterbody and respective watershed/aquaculture zone, and (3) the global, market-trade scale. Herein, two case studies are presented to evaluate aquaculture options at the waterbody/watershed level and at the farm level:

4.1 Waterbody/watershed level assessment: evaluation of model scenarios

The application of the ecosystem model outputs and the ΔDPSIR to evaluate development scenarios at the waterbody/watershed level is illustrated in this section.

4.2 Farm level assessment: evaluation of real data

Herein, a detailed dataset is analysed to evaluate aquaculture options at the farm level. The ΔDPSIR is applied to quantify the ecological and economic benefits of shifting from an abalone monoculture to an abalone-seaweed integrated multi-trophic aquaculture (IMTA).

Ecological-economic dynamic modelling (Chapter 5)

The dynamical link between the ecological and economic components is described in this chapter (Nobre et al., 2009). A coupled ecological-economic model has been developed for simulation of aquaculture production. First, the Modelling Approach to Resource economics decision-maKing in EcoaquaculTure (MARKET) was developed as a conceptual framework. Second, the MARKET approach was implemented to integrate an aquatic resources model with an economic model in the context of shellfish production in a Chinese coastal embayment. This work included inputs from an economic team for the definition of the economic functions for the case study (Nobre et al., 2009).

Integration of ecosystem-based tools (Chapter 6)

This chapter presents state-of-the-art ecosystem-based tools used for coastal research and management, including those developed in this thesis (Nobre and Ferreira, 2009). A consolidated demonstration of the application of such tools for coastal management is carried out using the Ria Formosa and catchment area as a case study.

Chapter 2. Multilayered ecosystem modelling

Context

As mentioned in *Chapter 1*, improving knowledge about complex coastal processes requires that the coastal ecosystem be modelled with tools capable of simulating the cumulative impacts of multiple uses. Such developments are still at an early stage but are potentially important for the sustainable expansion of aquaculture. For instance, they could allow for calculation of ecosystem carrying capacity that accounts for effects of multiple farms and other coastal activities on the ecosystem.

Summary

This chapter describes the multilayered ecosystem model and its application to Xiangshan Gang, a Chinese coastal bay with large aquaculture production and multiple catchment uses, where management efforts to improve water quality are underway. This integrated modelling approach combines the simulation of the biogeochemistry of a coastal ecosystem with the simulation of its main forcing functions, such as catchment loading and aquaculture activities. The case study consists of simulation scenarios designed together with local managers and aquaculture producers that account for changes in multiple uses. The integrated modelling approach is applied to simulate the cumulative effects of the reduction of fish cages and treatment of wastewater on the Xiangshan Gang coastal environment.

This chapter corresponds to a manuscript currently in second-stage review in Estuarine Coastal and Shelf Science:

Assessment of coastal management options by means of multilayered ecosystem models, by Nobre, A.M., Ferreira, J.G., Nunes, J.P., Yan, X., Bricker, S., Corner, R., Groom, S., Gu, H., Hawkins, A.J.S., Hutson, R., Lan, D., Lencart e Silva, J.D., Pascoe, P., Telfer, T., Zhang, X., Zhu, M.

Assessment of coastal management options by means of multilayered ecosystem models

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 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: (i) 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), (ii) 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 (iii) 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., 2007a; Cromey et al., 2009; Ferreira 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 Bendoricchi, 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 2.1.

Table 2.1. 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)	Europe
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., *In press*).	Canada and France
Harmful algal blooms (HAB's) 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

Ongoing research (Raick et al., 2006) is investigating trade-offs between (i) 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 which are

difficult to synthesise and interpret; and (ii) 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 optimise the trade-offs between complex and simple models, has been developed by Ferreira et al. (2008a, 2008b). Model integration can be implemented by (i) 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 (ii) explicitly integrating models with different time steps, 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: (i) running multi-year ecosystem models without the computational limitations reported for detailed models (Grant et al., 2008); (ii) fewer data requirements for model setup (Ferreira et al., 2008a); and (iii) 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: (i) 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; (ii) offline coupling does not allow dynamic feedback between models; and (iii) 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:

(i) Integration of a set of tools at the catchment-coastal scale;

(ii) 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 (i) develop an integrated coastal management tool for decision-makers; and (ii) examine the outcomes of different development scenarios.

METHODOLOGY

Study site and data

The Xiangshan Gang (Figure 2.1), a large (volume of 3 803 106 m^3 and area of 365 km^2) Chinese bay, was chosen as a case study. This system (i) encompasses multiple uses of the marine ecosystem and catchment area; (ii) is illustrative of Southeast Asian systems and potentially of European and North American systems at a larger scale of coastal resource uses; (iii) has proactive stakeholders and management; and (iv) 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: (i) 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); (ii) 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; 9 400 t finfish year^{-1}; and 6 700 t year^{-1} pond production of shrimp, crabs and clams.
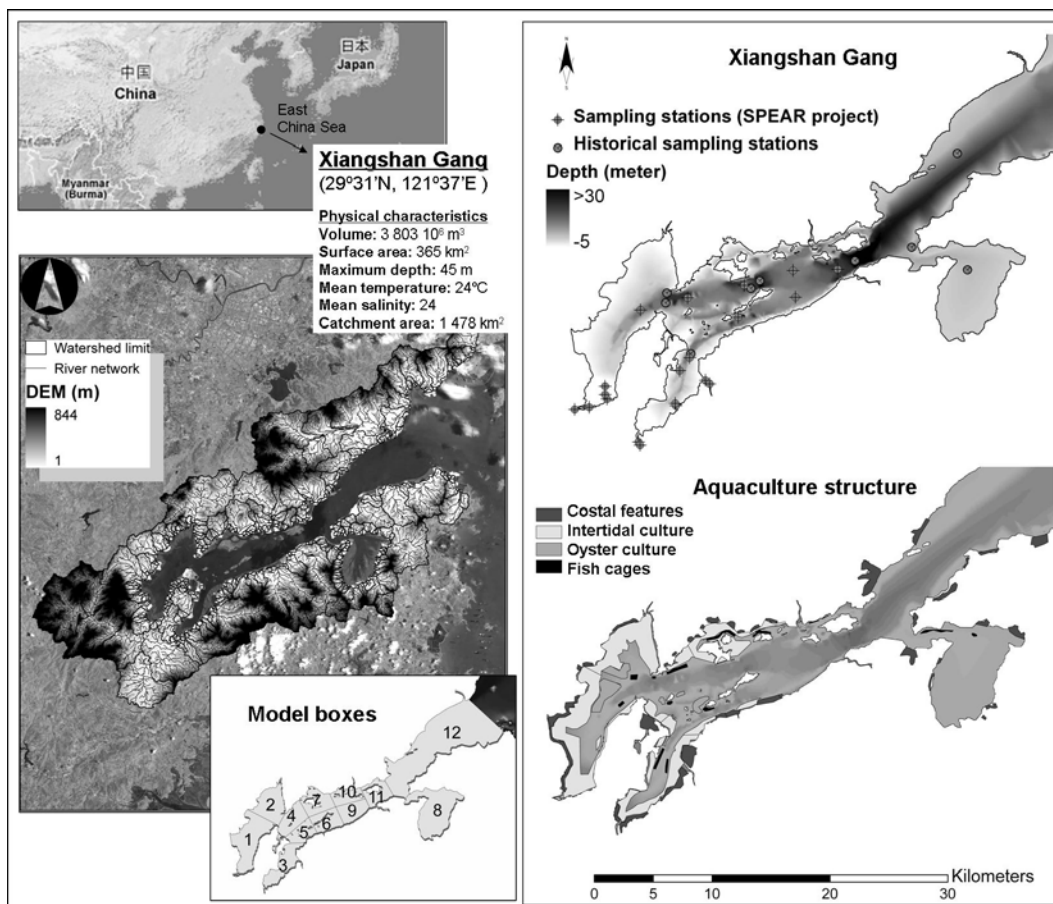
Figure 2.1. Xiangshan Gang and catchment area characterisation.

A detailed description of the bay and its catchment is given in Ferreira et al. (2008b). Table 2.2 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.

Table 2.2. 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, flow rate, temperature, salinity, pH, dissolved oxygen.
	Land cover ground truth data collected in 2005: Urban area, paddy fields, dry cropland, burnt land, forest, shrubby area, aquaculture, wetland, shallow water/beach, water and cloud.
	Landsat ETM+ images (2005/06/28), used to create landcover maps following a supervised classification approach (Lillesand and Kiefer, 2000).
	Hydrological data: precipitation, drainage area, river network.
	Topographic data collected during the Shuttle Radar Topography Mission (SRTM), with a resolution of 90x90 m (CGIAR, 2005);
	Biophysical and agricultural management parameters following the SWAT database for the most common crop (rice);
	Global Zobbler soil maps with a 2x2' (approx. 3.5x3.5 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 (18 stations)	*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.
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: (i) Total production; and (ii) 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.

Remote sensing was used to provide catchment land use and aquaculture structure mapping (Table 2.2). 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^{TM}) was used to store and analyse spatial data, produce thematic maps and generate information for model setup.

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 (i) production of multiple species in polyculture, (ii) its effects on the coastal environment and (iii) 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). Figure 2.2 synthesises the multilayered ecosystem model components, which are detailed below.
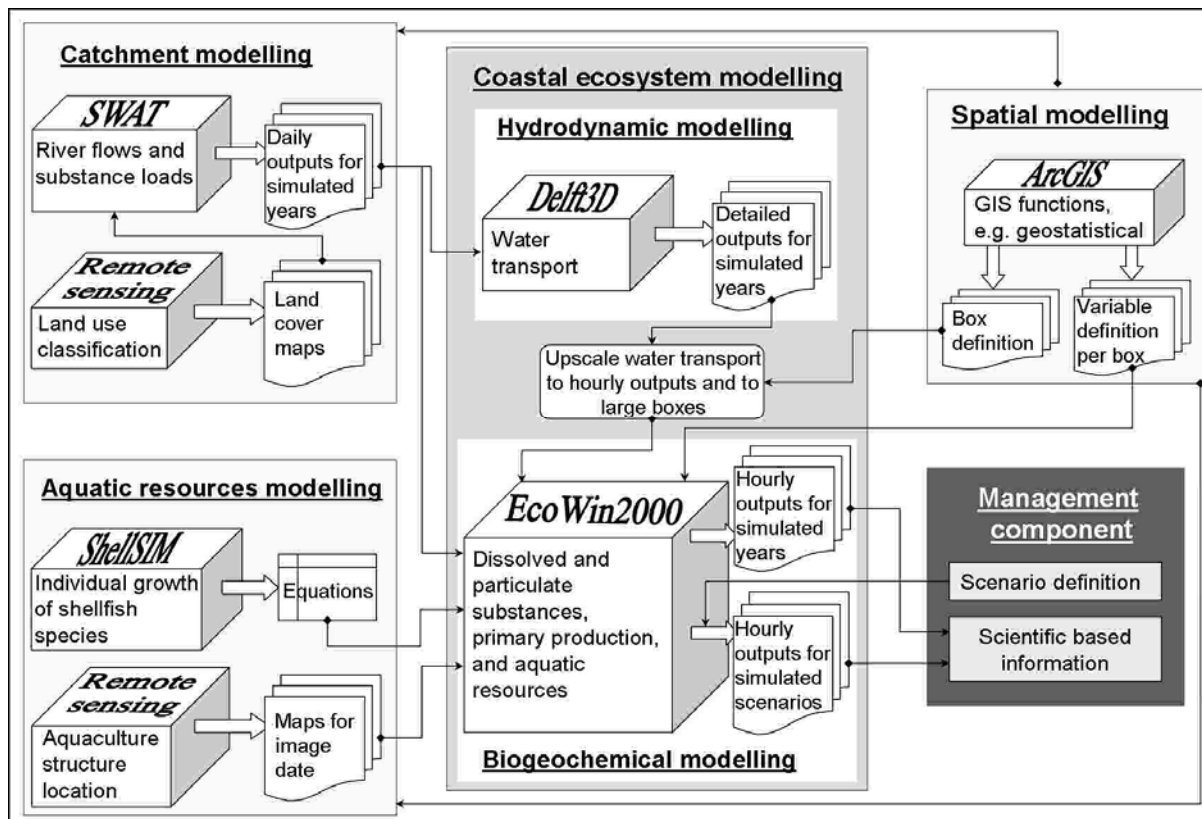
Figure 2.2. Integrated catchment-bay modelling approach for coastal ecosystem management: model components and ecosystem-based tools.

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 (Figure 2.1). 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 Figure 2.2. The implementation of each sub-model is detailed below and the main equations for state variables are presented in Table 2.3.

Table 2.3. 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 $$dSW / dt = PP_t - Qs_t - Ea_t - Ws_t - Qgw_t$$ (1) $(mm^3\ mm^{-2})$ dSW / dt, Rate of change in soil water content 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) $$dN / dt = Fn_t + Rn_t + An_t - PUn_t - Qn_t - Ln_t - Vn_t - Dn_t$$ (2) $(kg\ ha^{-1})$ dN / dt, Rate of change in soil nutrient 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 co-ordinate directions for transport simulation

Aquatic resource sub-model

Shellfish individual growth (Chinese oyster, razor clam, Manila clam and muddy clam)

$$\eta = f(B) \cdot f(POM) \cdot f(SPM) . f(L) \cdot f(T) \qquad (3)$$

η, shellfish scope for growth (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)

$$dS(s,t)/dt = -d[S(s,t) \cdot \eta(s,t)]/ds - \mu(s) \cdot S(s,t) \qquad (4)$$

S, shellfish number of individuals for each weight class s (ind)

η, shellfish scope for growth (g ind^{-1} d^{-1})

μ, mortality rate (d^{-1})

Biogeochemical sub-model

(Equations are presented only for internal processes; transport of state variables and boundary loads are described in the text and in Table 2.4)

Phytoplankton

$$dB/dt = B \cdot (p_{max} \cdot f(I) \cdot f(NL) - r_b - e_b - m_b - S \cdot c_s) \qquad (5)$$

B, Phytoplankton biomass expressed as carbon (μg C L^{-1})

p_{max}, Phytoplankton maximum gross photosynthetic rate (d^{-1})

$f(I)$, Steele's equation for productivity with photoinibition

$f(NL)$, Michaelis-Menten function for nutrient limitation

r_b, Phytoplankton respiration rate (d^{-1})

e_b, Phytoplankton exudation rate (d^{-1})

m_b, Phytoplankton natural mortality rate (d^{-1})

c_s, Shellfish grazing rate (ind^{-1} d^{-1})

Dissolved inorganic nutrients (applied to nitrogen and phosphorus)

$$dN/dt = B \cdot (e_b + m_b) \cdot \alpha + S \cdot e_s + POM \cdot m_{pom} \cdot \varepsilon - B \cdot (p_{max} \cdot f(I) \cdot f(NL)) \cdot \alpha \qquad (6)$$

N, Dissolved inorganic nutrient (nitrogen / phosphorus) (μmol L^{-1})

α, Conversion from phytoplankton carbon to nitrogen units (-)

POM, Particulate organic matter (mg L^{-1})

ε, Conversion from POM dry weight to nitrogen units (-)

m_{pom}, POM mineralization rate (d^{-1})

e_s, Shellfish excretion rate (μmol L^{-1} ind^{-1} d^{-1})

Particulate organic matter

$$dPOM/dt = POM \cdot (e_{pom} - d_{pom}) + S \cdot f_s + B \cdot m_b \cdot \omega - POM \cdot (m_{pom} + p_{pom} \cdot S) \qquad (7)$$

POM, Particulate organic matter (mg L^{-1})

e_{pom}, POM resuspension rate (d^{-1})

d_{pom}, POM deposition rate (d^{-1})

f_s, Shellfish faeces production (mg L^{-1} ind^{-1} d^{-1})

ω, Conversion from phytoplankton carbon to POM dry weight

p_{pom}, Shellfish POM filtration rate (ind^{-1} d^{-1})

Suspended particulate matter

$$dSPM/dt = SPM \cdot (e_{spm} - d_{spm}) + S \cdot f_s - SPM \cdot p_{spm} \cdot S \qquad (8)$$

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})

Table 2.4 and Table 2.5 specify the ecosystem model forcing functions and parameters. The model was run, using a time step of one hour, 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.

Table 2.4. 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 - Figure 2.1 - each divided vertically into 2 boxes) and time step (1 hour).		
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 5ºC and 30ºC respectively.		

Table 2.5. Ecosystem model parameters for Xiangshan Gang standard simulation.

Shellfish population	Number of weight classes		10
	Mortality - μ	Oyster	0.40%
	(% per day)	Clam	0.56%
		Razor	0.20%
		Muddy	0.15%
Shellfish cultivation practice	Seed weight	Oyster	0.2
	(g TFW ind^{-1})	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	2 286 (Boxes 1 to 5, 8, 9, 11, 12)
		Clam	308 (Boxes 1 to 7, 10)
		Razor	313 (Boxes 1 to 6)
		Muddy	187 (Boxes 1 to 3, 5, 6)
	Seeding density (t TFW ha^{-1})	Oyster	0.90
		Clam	0.45
		Razor	0.72
		Muddy	0.82
Phytoplankton growth	Pmax (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

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 2.2. 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 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 2.2. 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 Figure 2.3a 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 Figure 2.3. 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.
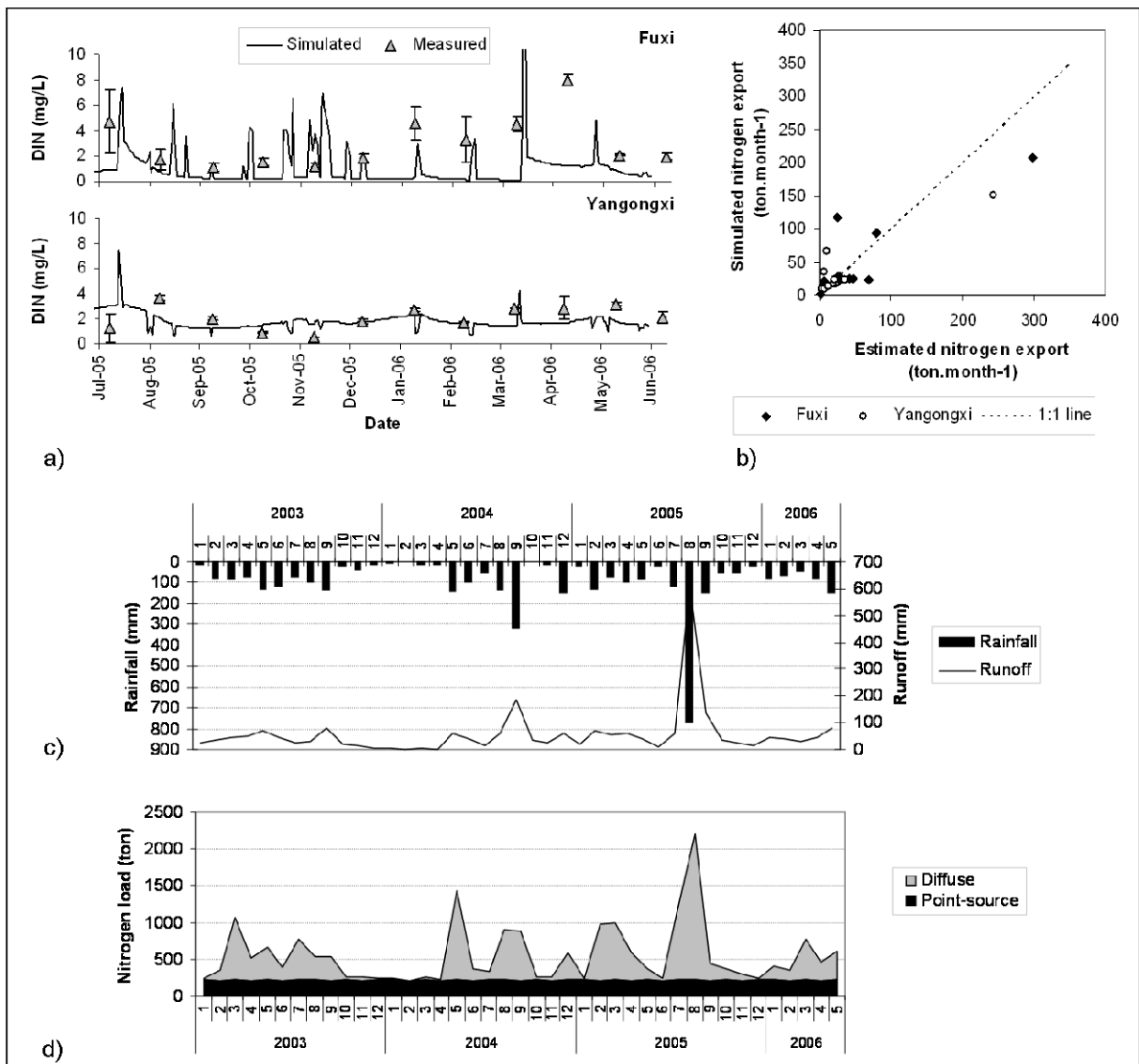
Figure 2.3. Catchment model outputs and comparison with data: a) measured and simulated dissolved inorganic nitrogen (DIN) for Fuxi and Yangongxi rivers; b) estimated and simulated nitrogen export; c) simulated monthly runoff compared with rainfall; and d) nitrogen loads from diffuse and point sources.

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.

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 2.6). 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.

Table 2.6. Amplitude and phase of the harmonic constituents: comparison between observed and simulated values.

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

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 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 hours 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).

Aquatic resource sub-model

The simulated aquatic resources included *Ostrea 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): (i) use of a shellfish individual growth model (ShellSIM - http://www.shellsim.com); (ii) coupling of the individual growth model with a demographic model to simulate the population (Ferreira et al., 1997); (iii) 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 (iv) 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 2.7 there is a statistically significant relationship between the individual model results and observations for shellfish wet weight and shell length. For the simulation of the Manila clam individual growth, the model used in Ferreira et al. (2007a) 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; Nunes et al., 2003; Nobre et al., 2005; Ferreira et al., 2007a) based on a conservation equation (Eq. 4, Table 2.3) 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 derive from the simulation of the individual growth of one animal (Eq. 3,

Table 2.3) in each weight class, thus providing the scope for growth which drives the transition of individuals across weight classes (Eq. 4, Table 2.3). 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 2.5.

Table 2.7. Correlation between measurements and simulation of shellfish individual weight and length, using Pearson product-moment correlation coefficient (r).

	Degrees of freedom	Wet weight (g)		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%

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 2.4) 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 ponds (Table 2.4) were calculated by means of a shrimp growth model (LMPrawn) as described in Ferreira et al. (2008b) and Franco et al. (2006).

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 2.5.

The pelagic variables in the model were calibrated against a historical time series for 2004 (Table 2.2). 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.

Coastal management options simulation

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: (i) a reduction of fish cages corresponding to a 38% reduction in total fish production (Scenario 1); (ii) an extension of wastewater treatment to the entire population (Scenario 2); and (iii) 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.

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 2.8 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:

(i) 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 dilution factors) (Bricker at 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;

(ii) 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;

(iii) Chinese sea water quality standards (National Standard of People's Republic of China, 1997) for DIN and phosphate to assess the compliance with desirable water quality objectives set by decision-makers for the bay; and

(iv) 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.

RESULTS

Ecosystem simulation

Figure 2.3c and Figure 2.3d 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 fertilisation 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.

Figure 2.4 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 boxes 3 and 10 in Figure 2.4. 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: (i) high natural variability of phytoplankton (Rantajärvi et al., 1998), not captured by the sampling window; (ii) phytoplankton dynamics are ruled by complex set of factors difficult to simulate in dynamic ecological models, such as species succession (Arhonditsis et al., 2007); and (iii) 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 (Figure 2.1); 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 i) fish cages, given that fish aquaculture is the major source of this nutrient (Table 2.8) together with the fact that an average annual load is considered due to the lack of temporally detailed data on fish cage loading; or ii) 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 (Figure 2.4).
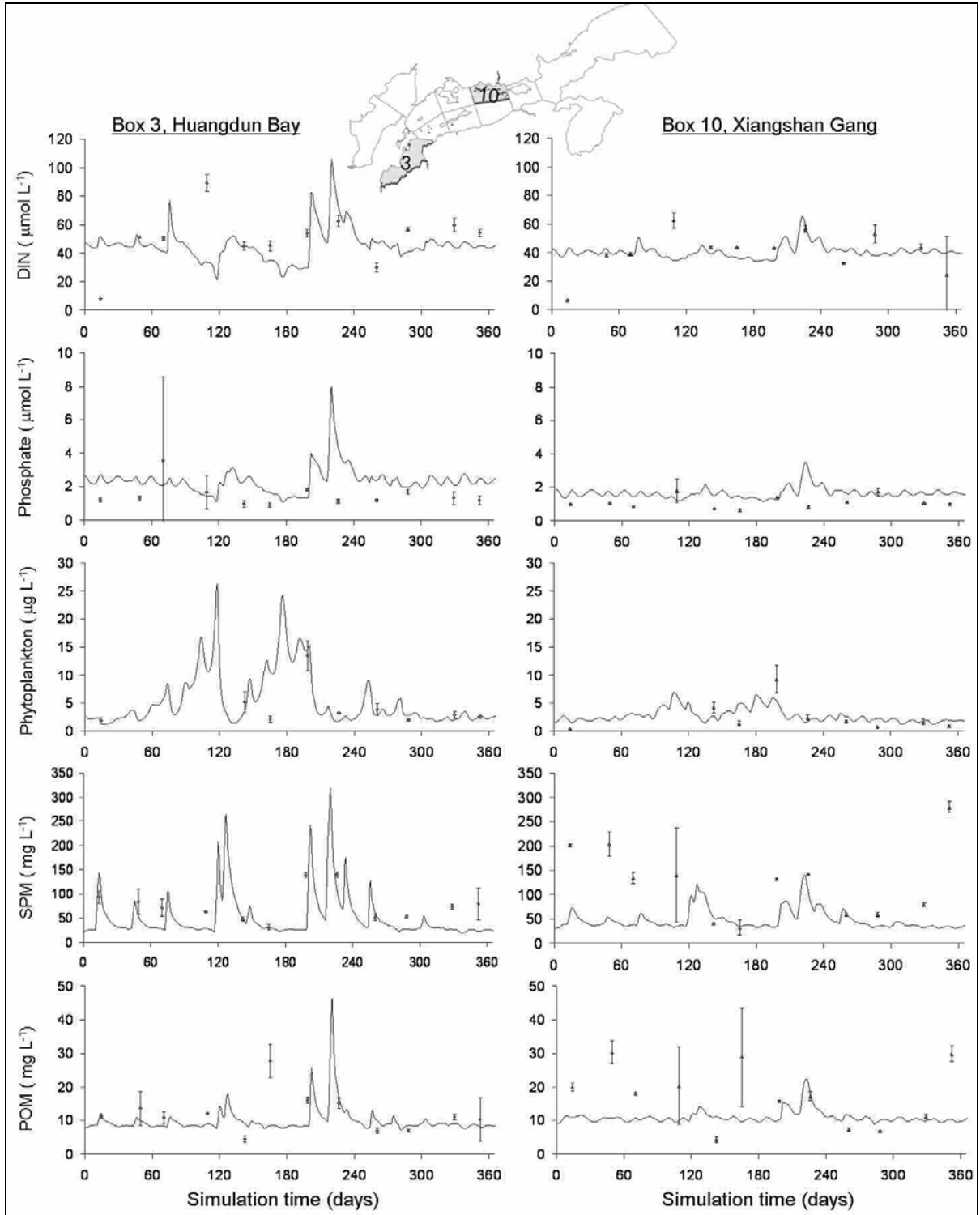
Figure 2.4. Standard simulation outputs for an inner box (Box 3, Huangdun Bay) and a middle box (Box 10), plotted with average daily data (June2005/June2006) and corresponding standard deviation: phytoplankton biomass, dissolved inorganic nitrogen (DIN), phosphate, suspended particulate matter (SPM) matter and particulate organic matter (POM).

41

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 to 5).

Figure 2.6 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.
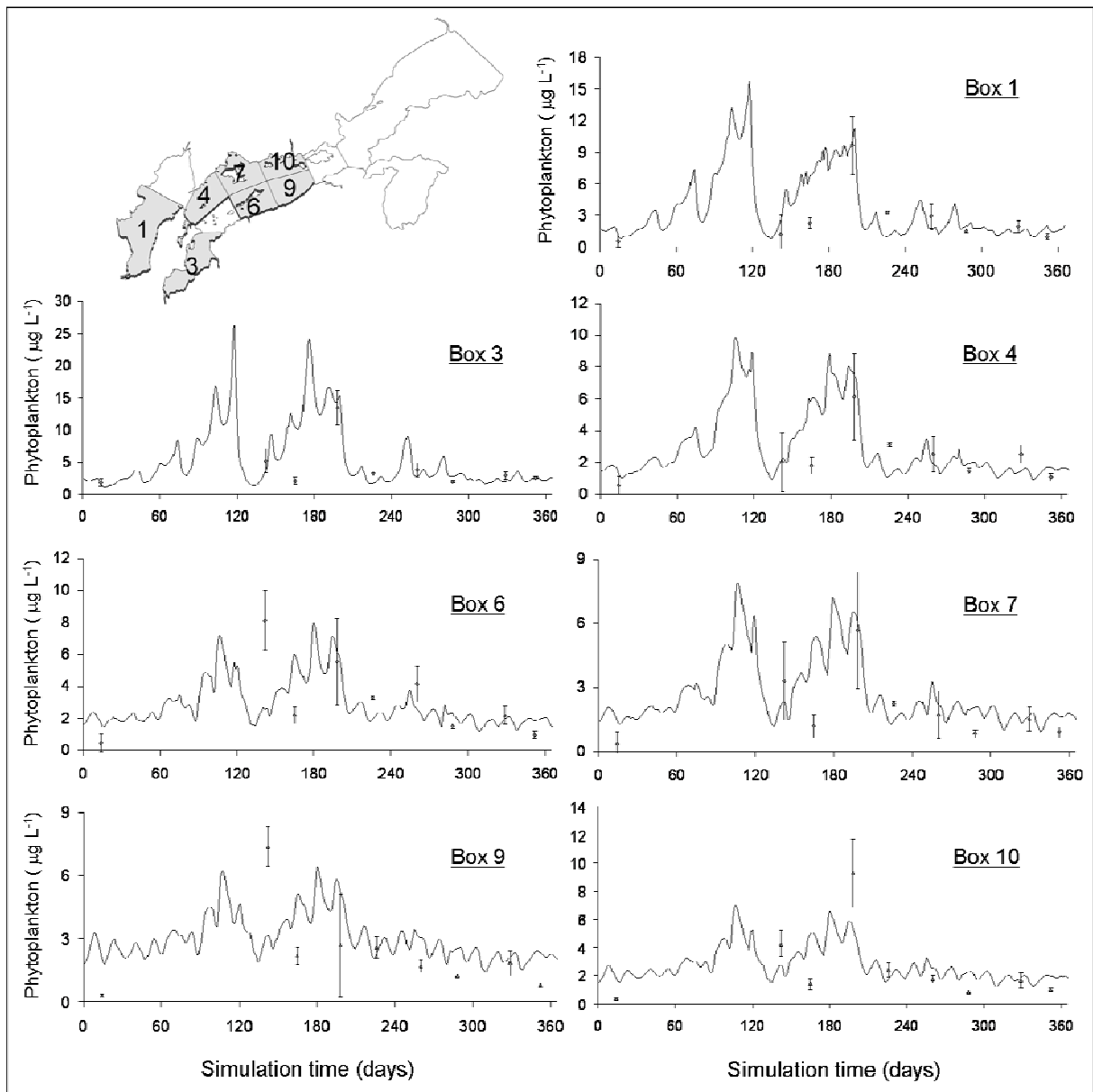
Figure 2.5. Standard simulation outputs for phytoplankton plotted with average daily data (June2005/June2006) and corresponding standard deviation for boxes 1, 3, 4, 6, 7, 9, 10.

Figure 2.6 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. Figure 2.6 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. 2 305 t (oysters account for ca. 1 298 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 (Figure 2.6). As a result, POM uptake by oysters was six-fold higher in Box 11 (3.36 g m^{-2} year^{-1}) than in Box 3 (0.54 g m^{-2} year^{-1}).
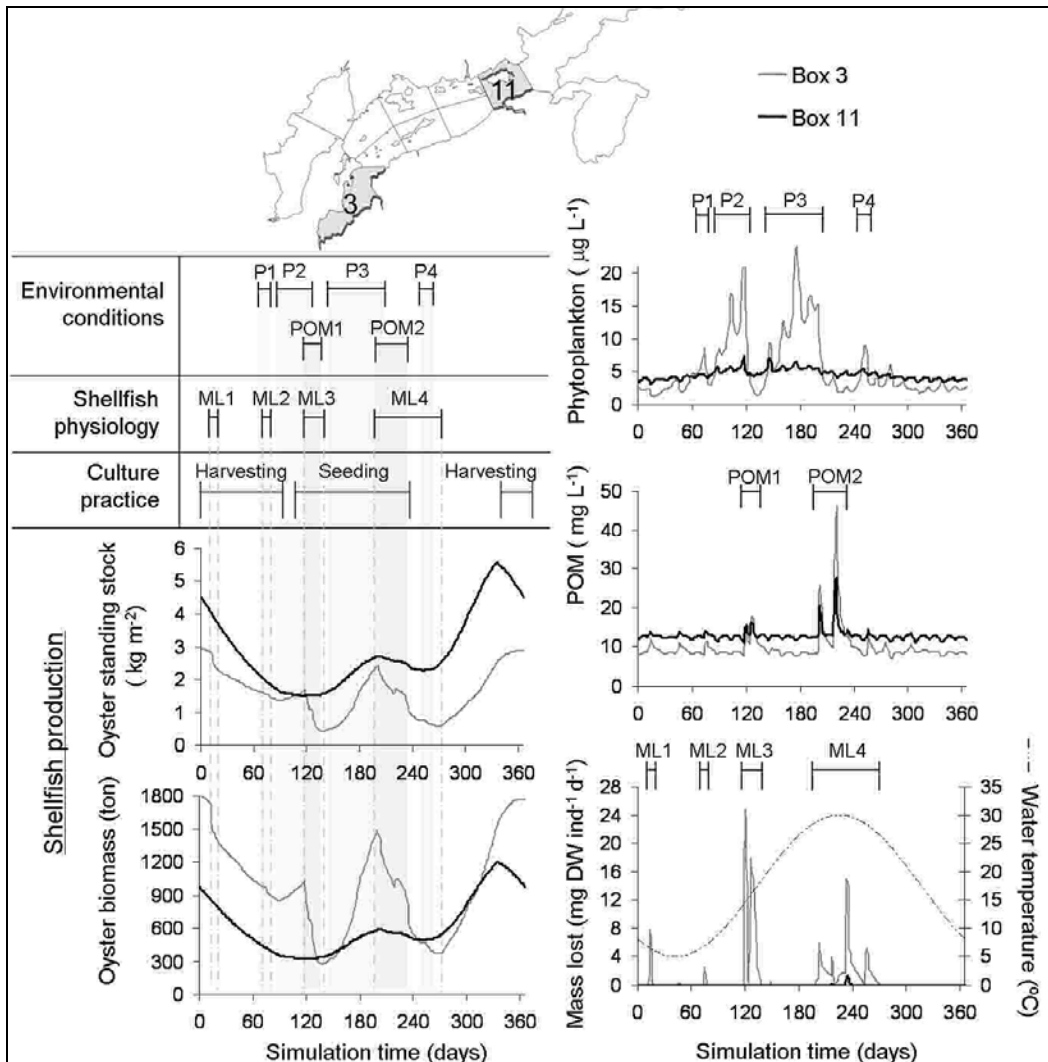
Figure 2.6. Standard simulation outputs for Box 3 (in grey) and Box 11 (in black) for: oyster production (standing stock, total biomass); mass loss due to reproduction, faeces and excretion; and key environmental variables affecting oyster growth, i.e. phytoplankton biomass, particulate organic matter (POM) and water temperature. Peaks are indicated with letters P#, POM#, ML# for phytoplankton, POM and mass loss, respectively. The stripes superimposed in the shellfish production plots indicate the time snapshots that correspond to the peaks, harvesting and seeding.

The effects of the peaks of phytoplankton concentration in Box 3 around days 120 and 180 (peaks P2 and P3, respectively, Figure 2.6) 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, Figure 2.6), because it was cancelled out by the mass lost due to physiological processes (ML4, Figure 2.6), possibly caused by the high temperatures that occur during the ML4 period (Figure 2.6). 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 boxes 3 and 11, 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 boxes 3 and 11, 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 (Figure 2.6).

Overall, the outputs of harvested shellfish compare well with the landings data (Figure 2.7).
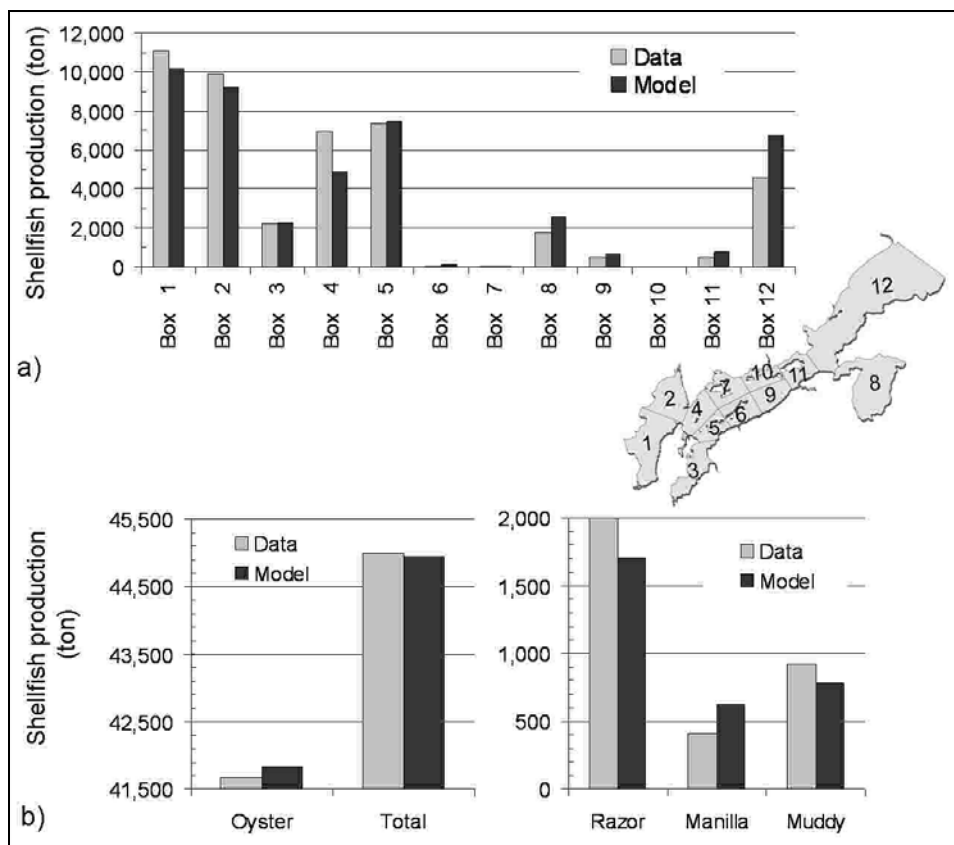
Figure 2.7. Standard simulation outputs for shellfish harvest and comparison with data (in t $year^{-1}$).

Comparison of ecosystem model outputs using different temporal resolutions for the catchment loads (Figure 2.8) 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 Figure 2.8). 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.
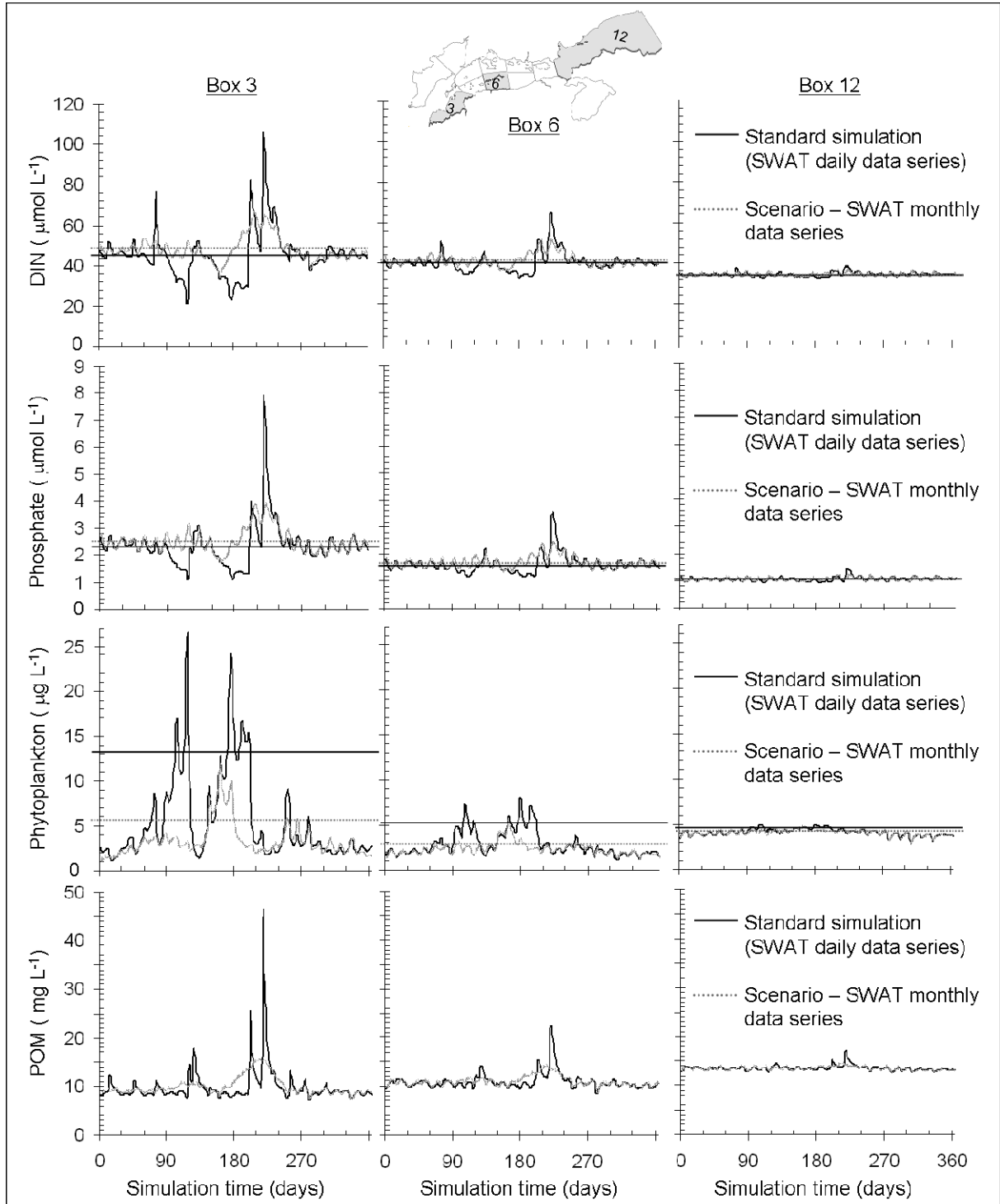
Figure 2.8. Sensitivity analysis of the coastal ecosystem to the temporal resolution of the catchment model outputs for an inner box (Box 3, Huangdun Bay), a middle box (Box 6), and an outer box (Box 12): dissolved inorganic nitrogen (DIN), phosphate, phytoplankton biomass and particulate organic matter (POM). (Straight lines in the plots indicate average value for DIN and phosphate, and 90-Percentile for phytoplankton).

45

Development scenarios

The scenarios tested simulate different nutrient loads entering into the bay. Table 2.8 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 2.8). For P the contribution was higher, ranging from 94% in the standard simulation to 90% in scenario 3 (Table 2.8). 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 2.8).

Table 2.8. 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

In general, model outcomes indicate that the effects of changes implemented in the scenario simulations were mostly visible in the inner boxes. Figure 2.9 shows the model outputs for (i) an inner box (Box 3 – Huangdun Bay), where the reduction of fish cages (in scenarios 1 and 3, Table 2.8) and the reduction of nutrient loads from wastewater discharge (in scenarios 2 and 3, Table 2.8) were implemented; (ii) a middle box (Box6) where no direct changes were implemented; and (iii) an outer box (Box 12), where, as for Box 3, a reduction of fish cages

and nutrient loads from wastewater was tested (Table 2.8). The changes simulated in the three scenarios were less evident for the outer box for DIN, phosphate, phytoplankton, shellfish harvest and shellfish productivity (Figure 2.9), possibly due to the exchanges with the ocean boundary.
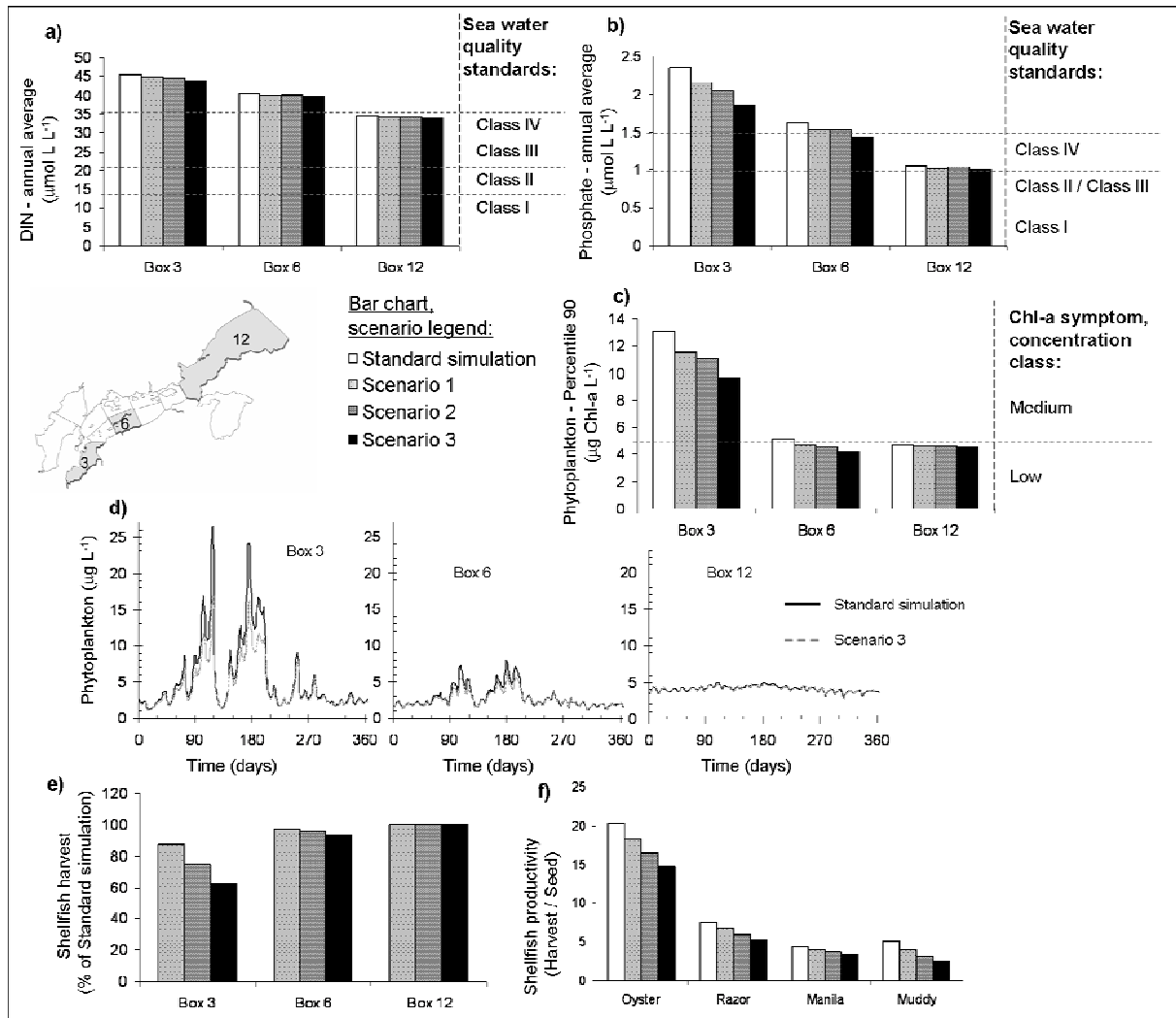
Figure 2.9. Scenario simulation outputs for an inner box (Box 3, Huangdun Bay), a middle box (Box 6), and an outer box (Box 12): dissolved inorganic nitrogen (DIN), phosphate, phytoplankton biomass, harvested shellfish and shellfish productivity (calculated as the ratio of total weight of shellfish harvested to total weight of seeding).

The reduction of nutrient loads in any of the scenarios resulted in very small changes in bay DIN concentration for any of the boxes (Figure 2.9a). There was a higher impact of nutrient load reduction on the simulated phosphate concentration (Figure 2.9b), probably because this was the substance with higher decrease (Table 2.8). 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

47

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 (Figure 2.9b) 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 boxes 6 and 12 the effects of nutrient load reduction were possibly dissipated (Figure 2.9c and d). Figure 2.9c shows the phytoplankton 90-percentile value for different boxes and scenarios. 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. boxes 6 and 12 in Figure 2.9b), 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 boxes 6 and 10, 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 (Figure 2.9f). Figure 2.9e 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 (Figure 2.10) showed that in general productivity levels were lower in boxes 1 to 7 (inner) and higher in boxes 8 to 12 (outer).
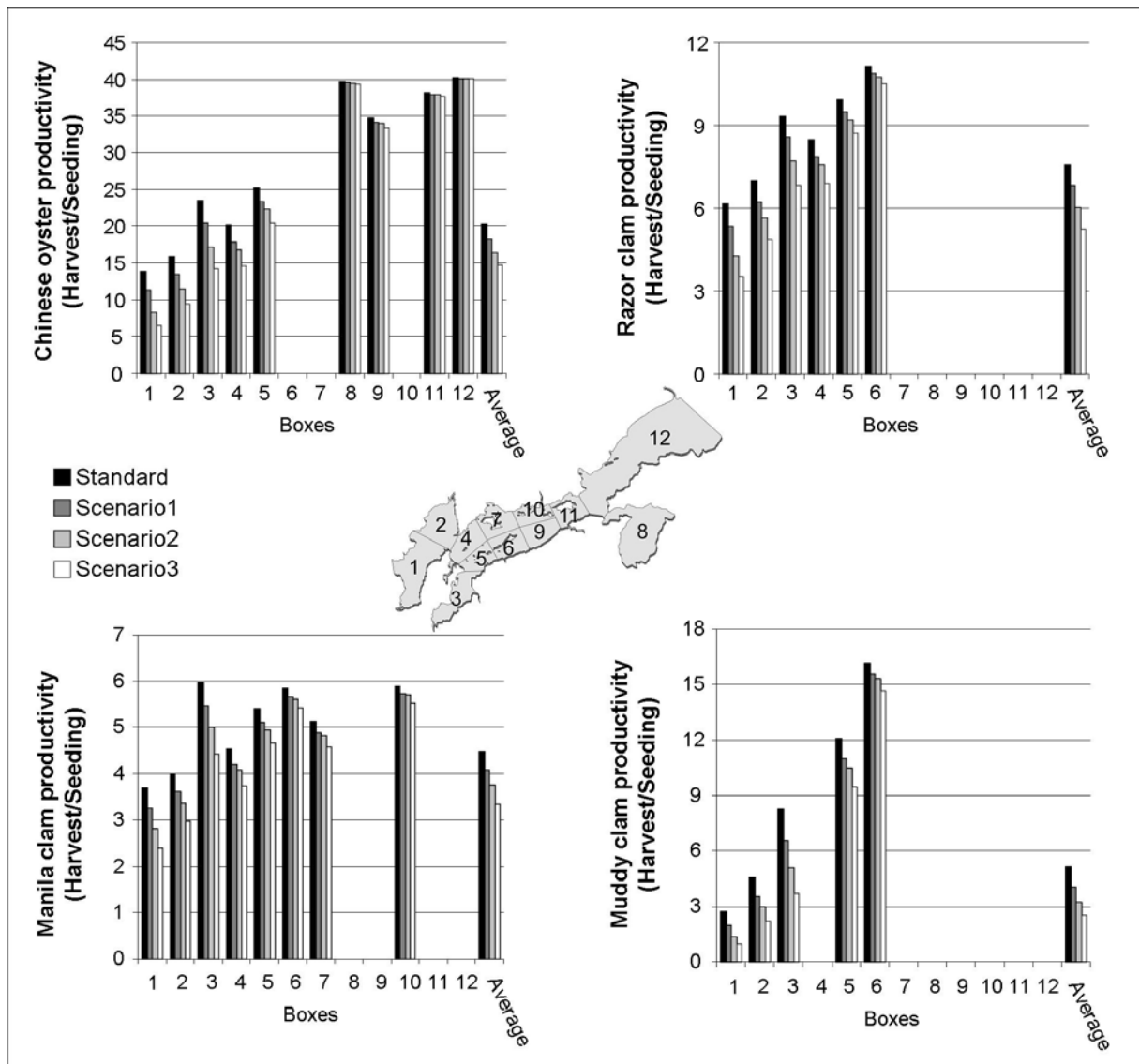
Figure 2.10. Shellfish productivity, calculated as the ratio of total weight of shellfish harvested to total weight of seeding.

DISCUSSION

The modelled nutrient load reduction had no significant effect on the water quality of the Xiangshan Gang according to Chinese Sea water 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 boxes. In the scenarios that test a decrease of these substances (Table 2.8), shellfish production decreases (Figure 2.9e,f and Figure 2.10). The estimated total loss of harvested shellfish was between 4 600 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 1 500 thousand Euro. Those effects are predicted to be more evident in the inner section of the Xiangshan Gang because of: (i) higher water residence times, in the range of 60 to 80 days; and (ii) higher competition for food resources given that cultivation areas in boxes 1 to 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 Figure 2.10, 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.

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.

Chapter 3. Integrated ecological-economic assessment

Context

The preceding chapter describes the development of an integrated ecosystem model and its application for simulating scenarios designed together with local managers to test potential measures to improve water quality in a Chinese embayment.

Further efforts are needed to translate the complex model results into knowledge useful for managers. Likewise, after implementing a set of measures, either to address a specific problem or in the context of a broader ICZM programme, managers need to assess the effectiveness of their actions.

Summary

This chapter presents a methodology to provide scientific-based information required by managers and decision-makers to evaluate previously adopted policies as well as future response scenarios. The method described here consists of an adaptation of the Drivers-Pressure-State-Impact-Response methodology, named differential DPSIR (ΔDPSIR). The ΔDPSIR approach further develops the multilayered ecosystem model by explicitly linking ecological and economic information related to the use and management of a coastal ecosystem within a specific timeframe. The application of ΔDPSIR is illustrated through an analysis of developments in a southwest European coastal lagoon between 1985 and 1995. The results are presented herein. Furthermore, the methodology was made available online at http://www.salum.net/ddpsir/.

This chapter corresponds to the published manuscript:

Nobre, A., 2009. An ecological and economic assessment methodology for coastal ecosystem management. Environmental Management, 44(1): 185-204.

(For consistency with published version this chapter is written in American English)

An ecological and economic assessment methodology for coastal ecosystem management

INTRODUCTION

Coastal zones are important areas that provide provisioning, regulating and recreational services to coastal populations and have a high economic value (Costanza et al., 1997; Ledoux and Turner, 2002). Boissonnas et al. (2002) estimate that the services provided by coastal environments and wetlands make up 43% of the world's ecosystem services. However, the benefits that these ecosystems generate are threatened by society's own activity. Population settlement in coastal areas is responsible for increasing pressure on these ecosystems (Boissonnas et al., 2002), resulting in severe consequences, such as (1) eutrophication related problems (Bricker et al., 2003; Ferreira et al., 2007b), (2) degradation of natural habitat areas (Cicin-Sain and Belfiore, 2005; Ortiz-Lozano et al., 2005), and (3) water quality degradation and sedimentation due to non-sustainable aquaculture production (Gibbs, 2004; Bondad-Reantaso et al., 2005). Negative changes in natural systems directly feed back on the socio-economic system that relies on the coastal ecosystem's goods and services (Bowen and Riley, 2003). This can result in (1) economic losses, as exemplified by Islam and Tanaka (2004) for the fisheries industry and by Lipton and Hicks (2003) for recreational fishing, or (2) an increase in the negative impacts of coastal disasters (Costanza and Farley, 2007).

Managers and policy-makers face the challenge of adopting responses to reverse the general trend of coastal ecosystem degradation and biodiversity loss. New legislative and policy instruments have been defined worldwide over the past few decades (Table 3.1). In order for decision-makers to gain insight into the performance of their responses, the management and science scales paradox should be addressed (Nijkamp and van den Bergh, 1997; Elliott, 2002). This implies the need for the application of scientific methodologies across different scales to enable understanding of ecosystem behavior (IMPRESS, 2003; Ferreira et al., 2005). Additionally, research results must be aggregated across a broader scale so that they may be useful to managers and they must integrate with the social sciences (Turner, 2000; Boissonnas et al., 2002; Lal, 2003). Bridging this scale gap requires integrated methodologies, such as the Driver-Pressure-State-Impact-Response (DPSIR) framework (Luiten, 1999; Ledoux and Turner, 2002; Bowen and Riley, 2003).

Table 3.1. Legislative and policy instruments adopted worldwide for coastal ecosystem management.

Domain	Legislative and policy instruments
United States of America	Coastal Zone Management Act of 1972
	National Estuary Program established in 1987 by amendments to the Clean Water Act of 1972
	Harmful Algal Bloom and Hypoxia Research and Control Act of 1998
Oceania	New Zealand Coastal Policy Statement of 1994
	Commonwealth Government's Coastal Policy of 1995
	Australia's Oceans Policy of 1998
Europe	European Water Framework Directive (WFD) of 2000
	Recommendation of the European Parliament and of the Council concerning the implementation of Integrated Coastal Zone Management of 2002
	Proposal for a Marine Strategy Directive of 2005
China	Measures of management on utilization of sea areas of 2001
	Law on prevention of marine pollution and damage from marine construction projects of 2006
Global	Millennium Ecosystem Assessment of 2005

The DPSIR framework is a widely used method. For instance, it was adopted in a guidance document (IMPRESS, 2003) for the application of the European Water Framework Directive (WFD). According to this document: (1) driver is an anthropogenic activity that may have an environmental effect, (2) pressure is the direct effect of the driver, (3) state is the condition of the water body resulting from both natural and anthropogenic factors, (4) impact is the environmental effect of that pressure, and (5) response is the measure taken to improve the state of the water body.

In order to contribute to the development of approaches that explicitly establish the link between ecological and economic assessment for coastal zone management, the present article proposes a new version of the DPSIR framework, herein named differential DPSIR (ΔDPSIR). The aim of the ΔDPSIR approach is to screen the ecological and economic evolution of an ecosystem during a given time period (Δt) that is relevant from a management perspective (response implementation period). This approach includes an analysis of the drivers, pressures and state before and after the response. The impact on the ecosystem (positive or negative) corresponds to the changes of state during the study period, Δt.

One of the advantages of this methodology is the explicit inclusion of a timeframe for the ecological-economic evaluation. This is a relevant consideration in impact assessment since an impact, by definition, implies a change in the ecosystem and thereby must include the analysis of at least two points in time. Another purpose of this methodology is to assess the differential value of indicators of ecosystem health and economic components. The use of differential instead of absolute values is particularly important in regard to ecosystem benefits. Absolute values for ecosystem benefits are not widely accepted since their valuation depends on subjective perceptions of the environment and so will be highly dependent on factors such as wealth and education (Oglethorpe and Miliadou, 2001). In addition, absolute classification does not account for the natural variability of the environmental component (Baan and van Buuren, 2003) that is inherent to different ecological regions.

The objectives of this article are to formalize the ΔDPSIR approach, provide guidelines for application of this new approach and to illustrate its implementation using a case study.

METHODOLOGY

ΔDPSIR framework

In the ΔDPSIR framework, an ecosystem is analyzed in accordance with the stresses to which it is subjected. The ΔDPSIR approach proposes a structured framework to apply already existing methodologies and tools for quantification of both ecological and economic variables. As shown in Figure 3.1, application of the ΔDPSIR framework can be divided into three stages.

Figure 3.1. ΔDPSIR conceptual model: characterization (stage 1), quantification (stage 2), and overview (stage 3) stages.

Characterization stage

The objective of stage 1 is to define the scope and aims of the study.

As schematized in Figure 3.2, this includes identification of: (1) the most relevant management issues in a given coastal ecosystem, (2) adopted management actions if using hindcast analysis or the defined action scenarios to be adopted in case of a forecast analysis, and (3) definition of the study period.
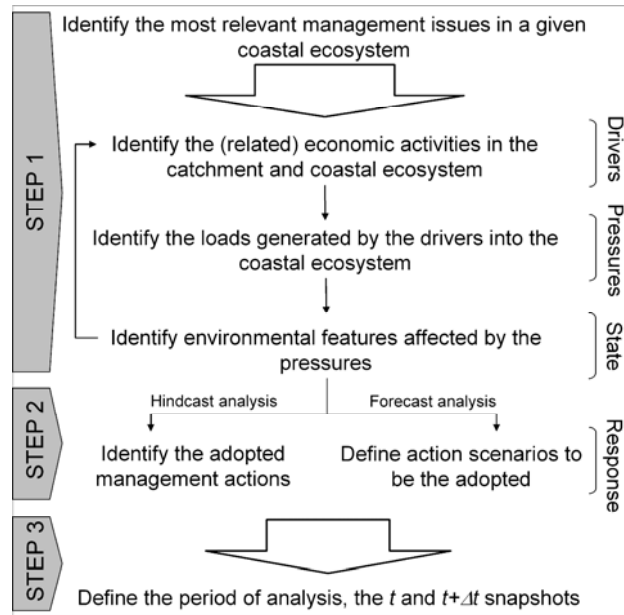
Figure 3.2. Schematic representation of the characterization stage of the ΔDPSIR approach.

Quantification stage

Stage 2 includes quantification of the ecological and economic variables (Figure 3.3). The ecological assessment is carried out on two different information scales: the research and management levels. The research level provides more complete information. This information is synthesized into useful information for the non-scientific community at the management level. The ΔDPSIR economic assessment constitutes a cost-benefit analysis to evaluate a given management response from environmental and socio-economic perspectives.
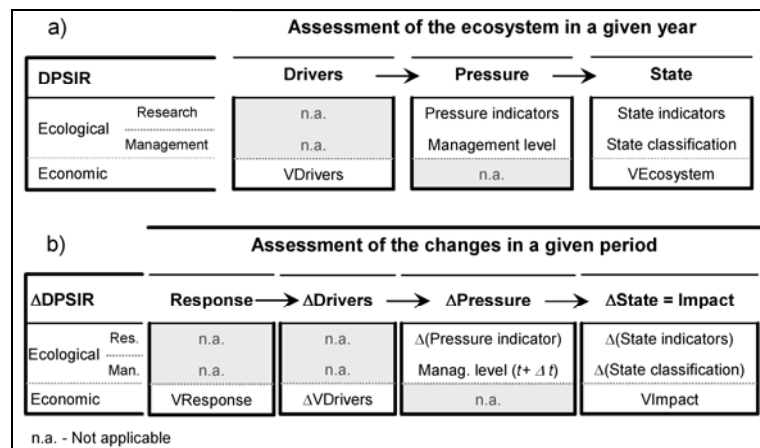
Figure 3.3. Schematic presentation of the quantification stage of the ΔDPSIR approach. a) Assessment in a given year and b) assessment of the changes in a given period.

The first step of the quantification stage is assessment of the ecosystem during at least two specific years or two scenarios (at the beginning, *t*, and at the end of the study period, *t+Δt*). This includes economic quantification of the drivers, ecological quantification of the pressures and ecological and economic quantification of the state of the ecosystem (Figure 3.3a). The second step is to assess the changes that occur during the response implementation period, either (1) due to the adoption of a set of management actions in a hindcast analysis, or (2) resulting from scenario simulation in a forecast analysis. This step (Figure 3.3b) includes quantification of changes in the drivers, pressures and state, which corresponds to the impact, and economic quantification of the response.

A number of well-tested methodologies and tools are available to carry out quantification of both ecological and economic variables. For example, regarding environmental monitoring of coastal ecosystems, there are a number of indices and indicators (NZME, 1998; NAP, 2000; Crawford, 2003; IMPRESS, 2003; La Rosa et al., 2004; Martinez-Cordero and Leung, 2004; Rogers and Greenaway, 2005) as well as screening models to distil fine resolution data into management information (McAllister et al., 1996; Bricker et al., 2003; Nobre et al., 2005; Ferreira et al., 2006). A number of methodologies and studies for economic valuation also exist to estimate the economic value of ecosystem goods and services either in or out of the market, as exemplified by Bower and Turner (1998), Anderson et al. (2000), Ledoux and Turner (2002), Nunes and van den Bergh (2004), Allen and Loomis (2006), Birol et al. (2006), and Eom and Larson (2006).

Overview stage

The purpose of stage 3 is to synthesize an application of the ΔDPSIR. It includes (1) quantification of the net value of the cost-benefit analysis regarding the management of a given coastal ecosystem during a specific time period, and (2) evolution of the ecological pressure and state indicators. Figure 3.4 exemplifies the type of integrated ecological-economic analysis that can be accomplished after quantifying the various components of the ΔDPSIR model for several years. The scenarios shown in the graphs represent a meaningful subset of hypothetical situations for ecosystem management and ecosystem evolution.

These scenarios show whether the ecosystem has been used in a sustainable way (Figure 3.4a); a given ecosystem is being overexploited (Figure 3.4b), which represents a typical scenario in which the economic system is limited by its pressures on the ecosystem state (Nijkamp and van den Bergh 1997); the restoration/remediation measures are effective (Figure 3.4c); or there is no evidence of management action (Figure 3.4d).
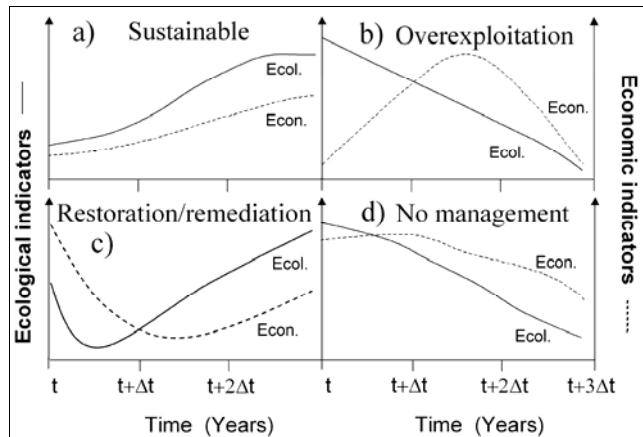
Figure 3.4. Scenarios for the evolution of ecological and economic indicators: a) sustainable scenario, b) overexploitation scenario, c) restoration/remediation scenario, d) no management scenario.

The next section includes a detailed description of the application of the ΔDPSIR including: the characterization stage, the ecological and economic assessment of the quantification and overview stages, and an explanation of the spatial and temporal scopes. The ΔDPSIR application guidelines are illustrated using a case study.

Case Study: site and data description

The study site for application of the ΔDPSIR approach was Ria Formosa (Figure 3.5), a shallow, well flushed coastal lagoon with large intertidal areas located in Southwest Europe. This lagoon has an average depth of less than 2 m and a short residence time of about one day. There is considerable interaction between the ecological and economic systems of this coastal lagoon. This ecosystem is classified as a Natural Park by the Portuguese legislation (D.L. 373/87) and is considered an area of high ecological value, given that it has been recognized by several international conventions (RAMSAR, Cites, Bonn), EC Council Directives (Birds Directive-79/409/EEC; Habitats Directive-92/43/EEC) and it is also included in the Natura2000 network. The Ria Formosa and its catchment support several economic activities that represent the main source of employment and income in the region.

The most important economic activities are extensive bivalve aquaculture, fish aquaculture, salt production, tourism, manufacturing, agriculture and livestock. Furthermore, the local economy includes traditional activities important to cultural preservation, as in the case of salt production.
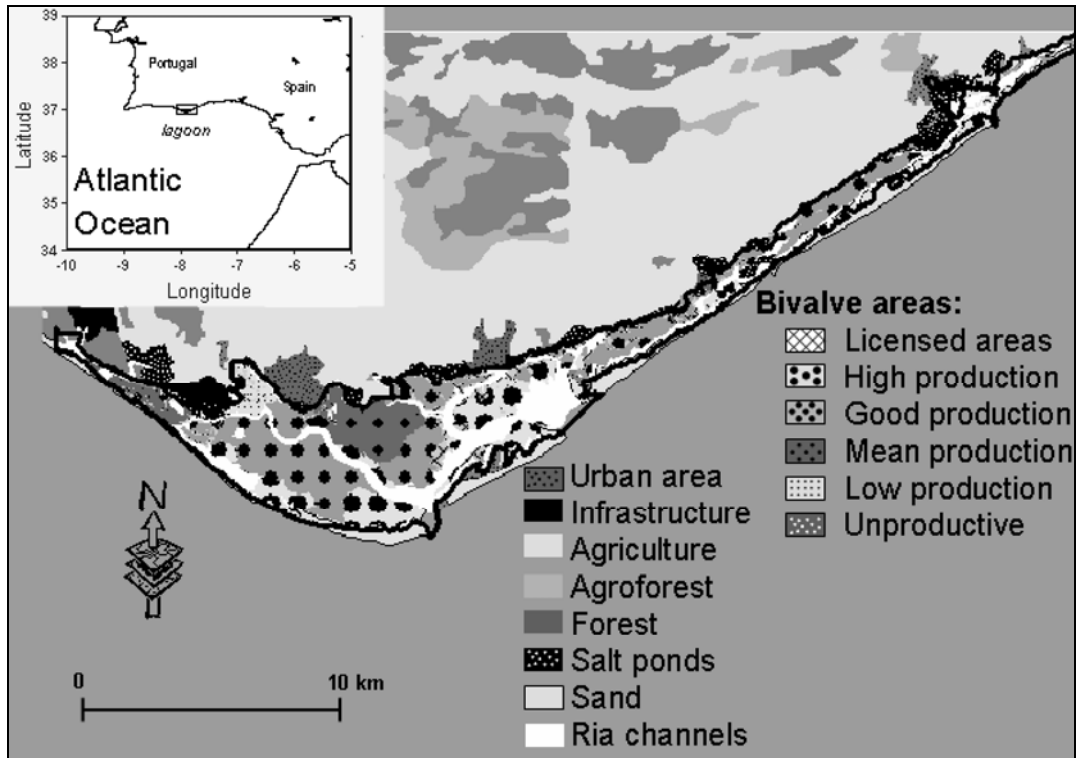
Figure 3.5. Land use and occupation in Ria Formosa and its catchment area.

Data collection and analysis

For the quantification of ΔDPSIR variables, a wide range of diverse data must be collected and analyzed. The main sources of data (Table 3.2) were official documents and statistics produced by institutes with roles in the lagoon's management, such as the Portuguese Institute of Statistics, scientific literature about Ria Formosa consolidated in Nobre et al. (2005) and unpublished literature from the University of Algarve. All economic values presented herein have been converted into constant year 2000 Euros using the general index of consumer prices for Portugal.

Table 3.2. Data description.

ΔDPSIR component		Indicator / variable	Data source
Drivers	Aquaculture production	Cultivated area (bivalve beds, fish ponds) x production rate x market price	POPNRF, LAA, INE, UAlg
	Fisheries	Fisheries production statistics	DGP
	Salt production	Salt production statistics	POPNRF
	Tourism	Average tourist expenditure x total number of tourists in major cities of Ria Formosa area	INE, DGT
	Agriculture/Livestock	Gross added value	INE, PBH
	Manufacturing industry	Gross added value	POPNRF, INE
Pressures	Population loads	WWTP estimates; PEQ and daily discharge per PEQ	POPNRF, PBH
	Agriculture/livestock diffuse loads	Land cover charts and coefficient of nutrient loss per area and per type of land use	PBH, CORINE
	Industry waste water discharge	Population equivalents and coefficients of organic loads per population equivalents	UAlg, PBH
	Livestock point source	Number of animals and PEQ per animal	PBH, INE
State	Eutrophication symptoms	Macroalgal simulated growth, dissolved oxygen simulated in the intertidal area	Nobre et al. 2005
	Bivalve growth	Bivalve production rates	LAA
		Official monitoring data of water quality in the bivalve production areas	IPIMAR
Response	Response costs	Detailed planned actions for the period 1985-1990; expenditure on wastewater treatment for the period 1991-1994	POPNRF, PBH
Impact (*VExternalities*)	Estimates on reduction of organic loads	Determination of loads not treated: -PEQ of population not served by WWTP -PEQ of livestock that generate point source pollution -PEQ of industry	POPNRF, PBH
		Price of implementation and maintenance of commercial compact WWTP designed for less than 500 PEQ	PLA
	Reduce shellfish parasite infection	Cost for screening of seed infection	IPIMAR
		Cost for buying certified seeds	LAA
	Monitoring of the *VExternalities* actions	Group of 4 persons for monitoring and implementing the several actions	Salary tables

PEQ - population equivalent; WWTP - waste water treatment plant. Data source abbreviations: CORINE - data from CORINE Land Cover project; DGPA - data from the fisheries and aquaculture ministering office; DGT - data from the tourism ministering office; INE - data from Portuguese Institute of Statistics; IPIMAR - data from the Institute of Fisheries, Research and Sea; LAA - local aquaculture association; UAlg - unpublished undergraduate thesis from University of Algarve; PBH - drainage basin management plan (MAOT, 2000); PLA - private company that commercialises compact WWTPs; POPNRF - management plan of Ria Formosa Natural Park (SNPRCN, 1986).

Characterization stage of the ΔDPSIR

The first step of the characterization stage is to create an overview of the issues to be managed, including identification and description of the main drivers, consequent pressures and the most relevant environmental features that might be affected (Figure 3.2). Identification of the main driving forces is generally made by use of local knowledge.

To support this task, the use of lists can be helpful. A guidance document for the implementation of the WFD is a good example of such a list (IMPRESS, 2003). In addition, the guidance document shows the connection between the expected pressures and impacts related to each driver, which is important for the analysis of the components of ΔDPSIR that follows. Table 3.3 presents examples of pressure indicators for the most common drivers in a catchment-coastal ecosystem and the most common water quality indicators that contribute to an understanding of the interaction between ecosystem changes and their main driving forces.

Table 3.3. Correspondence between most common drivers with respective pressure indicators and with ecological state indicators.

Pressure indicators	Aquaculture	Fisheries	Agriculture	Cattle production	Industry (General)	Hydropower	Forestry	Habitat conservation	Navigation	Urbanisation	Tourism / recreation	State indicators
Sediments/ SPM/ POM loads	⊗ ↳	⌐	⊗ ↳	⊗ ↳	⊗ ↳		⊗	⌐		⊗ ↳		Toxic contaminants in aquatic plants and animals
Antibiotic loads	⊗ ↳	⌐	↳	↳	↳		↳	↳		↳	⌐	Phytoplankton concentration
Fishing effort	⌐	⊗ ↳	↳	↳	↳					↳	⌐	Toxic algal blooms
Loss of habitat area	⊗ ↳	⌐	↳	↳	↳		↳	↳	⊗	⊗ ↳	⊗	Nutrient concentrations
Resource partitioning	⊗ ↳	⌐	↳	↳	↳	↳	↳	↳		↳	⌐	Dissolved oxygen
Production effort	⊗ ↳	⌐	↳	↳	↳		↳	↳		↳	⌐	Macroalgae (species, biomass, density)
Fish production	↳	⊗	↳	↳	↳			↳				Shellfish health
N/P load	⊗	↳	⊗	⊗	⊗					⊗		Fish stocks
Pesticides	⌐	⌐	⊗	↳	↳					↳	⌐	Faecal bacteria
Loads of toxic substances	⌐	⌐			⊗							Plankton concentration
Faecal bacteria inputs	↳			⊗	⊗ ↳				↳	⊗	↳	Contaminant in the sediments
Water temperature increase	↳	↳				⊗		↳	↳	↳	↳	Habitat area and condition
N/P load reduction	↳	⌐	↳	↳	↳		⊗ ↳	⊗ ↳	↳		⌐	Turbidity / Secchi depth
Area of protected area	↳							⊗				Sediment anoxia
Intertidal area loss	⊗ ↳	⌐					↳		⊗	⊗	⌐	Water temperature
Intertidal mean height	⊗								⊗ ⌐	⊗ ⌐	⌐	Coastal erosion
Ship traffic									⊗			

⊗ Correspondence between each driver and respective pressure indicators. ↳ Ecological state indicator affected by the corresponding driver; ⌐ State indicator that affects the corresponding driver; ↳ State indicator that is affected and that affects the corresponding driver.

The next step is identification of response actions that have been adopted in the case of a hindcast analysis or definition of management action scenarios in the case of a forecast analysis (Figure 3.2). If the ecosystem is subject to management policies, a set of laws (e.g. Urban Waste Water Treatment Directive, UWWTD, 91/271/EEC or Nitrates Directive, ND, 91/676/EEC), policies (e.g. DOENI, 2006) or economic instruments (e.g. Romstad, 2003; Zylicz, 2003; Hatton MacDonald et al., 2004) are planned and enforced by managers or policy makers according to the evaluation of the state of the ecosystem. These responses can be targeted at the catchment region, the coastal ecosystem or both, and have a precautionary or remedial nature. For ecosystems that are not subject to management, a null response must be considered.

Finally, during the characterization stage, the appropriate timeframe for application of the ΔDPSIR must be defined (Figure 3.2). The evaluation can be either a hindcast analysis (in order to assess changes that have already occurred in the ecosystem due to a given management response) or a forecast analysis (in order to predict changes that result from the simulation of management scenarios). Definition of the appropriate time period for the analysis must take into account the possibility that some of the pressures might only be evident in impacts with a time lag of several years (IMPRESS, 2003).

In the Ria Formosa case study, the characterization stage was elaborated based on the Management Plan of Ria Formosa Natural Park (SNPRCN, 1986), local community knowledge and the scientific literature. The most outstanding issues are symptoms of benthic eutrophication and high clam mortality. Further details, including definition of the period of analysis, are provided in the results section.

Ecological assessment of the ΔDPSIR

The ecological assessment is done by quantifying (1) the loads (pressure), (2) the biogeochemical quality of the ecosystem (state), and (3) their changes (ΔPressure and ΔState, i.e., impact) due to a given management response to a given problem or research topic (Figure 3.3).

Pressure

Since the same magnitude of pressure is likely to produce different effects in different ecosystems (e.g. due to susceptibility), it is important to identify the significant pressures (those that are likely to affect the ecosystem state). The quantification of pressures is dealt with at the research level, for example, by determining the annual load of nitrogen from the

catchment to the coast and its spatial and temporal distribution (Nikolaidis et al., 1998; Grizzetti et al., 2003; Plus et al., 2003; Yuan et al., 2007).

While the above-mentioned indicators are useful at the research level, they can be quite uninformative for managers who might not know whether a given load of nitrogen is high or low for a particular ecosystem. To be useful for coastal managers, the provided information must describe what is manageable from the catchment-coastal perspective, i.e., the relative ratio of manageable to unmanageable nutrient loads (pressure management level). For complete pressure quantification, the management level can also include pressures outside the scope of the catchment-coastal area. For example, atmospheric loads can represent a significant percentage of nutrient inputs (Bower and Turner, 1998), and those must be balanced with loads from the catchment-coastal drivers. Simple models, such as Overall Human Influence (OHI) (described in Bricker et al., 2003), can be adapted for this quantification. Other examples of pressure indicators, at both research and management levels, are given in Table 3.4.

Table 3.4. Example of pressure indicators at the research and management levels for the loss of natural habitat areas.

Level	Pressure indicator	Units	Remarks
Research	Habitat loss per activity	$(m^2 .yr^{-1})$	In case of habitat re-establishment, habitat loss is negative
Management	Magnitude of habitat loss $= \frac{Existing Habitat Area}{Target Habitat Area}$	(-)	Depending on restoration objectives the *TargetHabitatArea* is calculated based on the pristine or potential habitat area

In order to focus on the identified management issues in Ria Formosa, the most relevant pressures for analysis are nutrient discharges. These pressures were quantified using the information provided in Table 3.2.

State

The state assessment is made at two levels, one applies the state indicators to the research level and the other aggregates these indicators into information for managers (e.g., Nobre et al., 2005) using screening models (e.g., Bricker et al., 2003) or other methods to provide state classifications, which are useful to managers. Table 3.5 shows examples of ecosystem state classification tools that aggregate indicators in a simple range of classes that are meaningful to managers. These tools may include screening models, such as the Assessment of Estuarine Trophic Status (ASSETS) model and water quality standards (e.g., European Union Council Directives; United States, EPA water quality standards; China, Sea water quality standards).

Table 3.5. State classification tools used to inform managers

"Issue" to manage	Classification tool	Indicators used	Classes
Eutrophication	ASSETS (Bricker et al., 2003 ; Ferreira et al., 2007b)	Chl *a*, macroalgae, dissolved oxygen (DO), loss of submerged aquatic vegetation nuisance and toxic blooms	5 classes
Bathing water quality	Council Directive 76/160/EEC of 8 December 1975 concerning the quality of bathing water	Total and faecal coliforms, faecal streptococci, salmonellas, enterovirus, mineral oil, pH, colour, mineral oils, surfactant, phenol, transparency, DO, residuals and fluctuating substances, ammonium, nitrate, phosphate, kjeldahl nitrogen, pesticides, heavy metals .	2 classes: in conformity or not.
Aquaculture water quality - shellfish	Council Directive 79/923/EEC of 30 October 1979 on the quality required of shellfish waters	pH, temperature, colour, total suspended solids, salinity, DO, petrol hydrocarbon, organohalogenated compounds, heavy metals, biotoxins, faecal coliforms.	2 classes: in conformity or not.
Aquaculture environmental quality – fish cages	Fish farming manual – regulation and monitoring of marine cage fish farming in Scotland (SEPA, 2005)	*Water quality:* DO, DAIN, DAIP, chl *a*, zinc, copper, sulphide, pH, suspended solids, salinity, trace metals, oil, fat, grease, litter. *Sediments:* number of taxa, abundance of polychaetes, Shannon-Weiner Diversity, Infaunal Trophic Index, Beggiatoa, feed pellets, teflubenzuron, copper, zinc, free sulphide, organic carbon, redox potential, loss on ignition.	2 classes: acceptable or unacceptable levels.

For the application of ΔDPSIR in Ria Formosa, the ecosystem state is analyzed for eutrophication symptoms and bivalve production based on the identified major management issues.

Eutrophication Symptoms

In the Ria Formosa lagoon, it is appropriate to analyze eutrophication benthic symptoms, such as macroalgal growth and dissolved oxygen in shallow areas (Nobre et al., 2005). The use of ecological modeling was considered because there is no available data to quantify macroalgal growth (Ferreira et al., 2003) or dissolved oxygen in intertidal areas (Nobre et al., 2005) for either t or $t+\Delta t$. The results of an ecosystem model previously applied to Ria Formosa were used (Nobre et al., 2005; Figure 3.6). The simulated macroalgal biomass and dissolved oxygen results were analyzed at the management level using the ASSETS eutrophication assessment model (Bricker et al., 2003; Ferreira et al., 2007b).
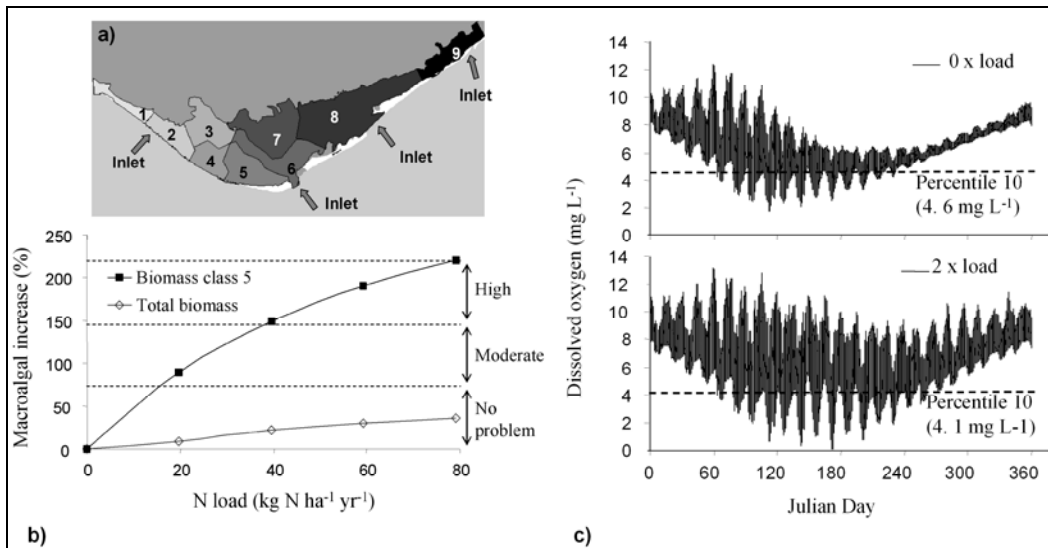
Figure 3.6. Ria Formosa ecological model results from Nobre et al. (2005). a) model boxes, b) macroalgal growth as function of nutrient loads, c) dissolved oxygen concentration

Bivalve Production

The evolution of bivalve production over Δt was analyzed using the production rates and water quality in bivalve cultivation areas (Table 3.2). No long-term research on bivalve growth or production rates covering the study period was available. Production was estimated using knowledge from local aquaculture associations. It is very common to have qualitative information that must be sorted through in environmental evaluations (Nijkamp and van den Bergh, 1997).

ΔPressure, ΔState and Impact State

The quantification of changes over the period of analysis (Δt) is given by the difference between the value of the indicator at t and at $t+\Delta t$ (Figure 3.3b). The pressure management level is an exception, since its objective is to provide information about what is manageable at $t+\Delta t$ rather than the difference over Δt. Changes in Δt allow one to ascertain:

(1) The direction of changes in the state of the ecosystem (i.e., the impact) using the changes in the state classification results at the management level, if they exist, or using the changes in the state indicator results at the research level;

(2) The evolution of the pressure component during the response implementation period through changes in the pressure indicators results (research level);

(3) The changes in pressures most likely to produce the target changes in state through the pressure management level at $t+\Delta t$.

As good practice and whenever data exist for the *Δt* period, an analysis should be performed to ensure that the data at *t* and *t+Δt* are not outliers in any of the data series.

In Ria Formosa, the evolution of pressures was analyzed based on changes in the nutrient loads. The impact on the ecosystem was characterized based on changes in bivalve production rates and on changes in the model simulation results for macroalgal growth and dissolved oxygen.

Economic assessment of the ΔDPSIR

The ΔDPSIR economic assessment is a cost-benefit analysis that evaluates a given coastal zone management response from an environmental catchment-coastal perspective. It includes the calculation of the variables shown in Table 3.6.

Table 3.6. Economic assessment variables of the ΔDPSIR

Variables	Objective	Stage
Value of the drivers (*VDrivers*)	Estimates the economic value of the activities both in the catchment (*VDrivers$_{External}$*) and in the coastal ecosystem (*VDrivers$_{Ecosyst}$*)	
Value of the ecosystem (*VEcosystem*)	Aims to estimate the economic value provided by its goods and services	Quantification
Value of the response (*VResponse*)	Includes the direct costs of the actions incurred during the response period (*Δt*).	
Value of the impact on the ecosystem (*VImpact*)	Intends to capture the changes in the economic value of the ecosystem during *Δt*.	
Economic value of management (*VManagement*)	Provides the net value of the cost benefit analysis in *Δt*.	Overview

Ecosystem valuation must encompass a wide range of goods and services provided by nature, not just the direct market values (Emerton and Bos, 2004). As such, the value of the ecosystem should be given in terms of the total economic value (TEV), which includes direct use, indirect use and non-use values (Turner et al., 2000). It is important to note that ecosystem valuation is an exercise with many limitations, including the complexity and nonlinearity of ecosystems, which makes it difficult to compute an objective and holistic TEV (Nijkamp and van den Bergh, 1997; Emerton and Bos, 2004). To accommodate the limitations of TEV calculation, ΔDPSIR considers two possible approaches to economic assessment: a complex and a simple approach, as shown in Figure 3.7.

If the goods and services provided by an ecosystem are well known and a complete dataset exists or can be gathered for the valuation, the complex approach should be adopted. According to that approach the *VEcosystem* is given by the TEV (Figure 3.7). If it is not possible to calculate the full extent of the TEV, the economic assessment is simplified and the *VEcosystem* is computed using the partial ecosystem value (PEV) (Figure 3.7) The PEV is a simplification of the *VEcosystem* and is given by *VDrivers_{Ecosyst}*.
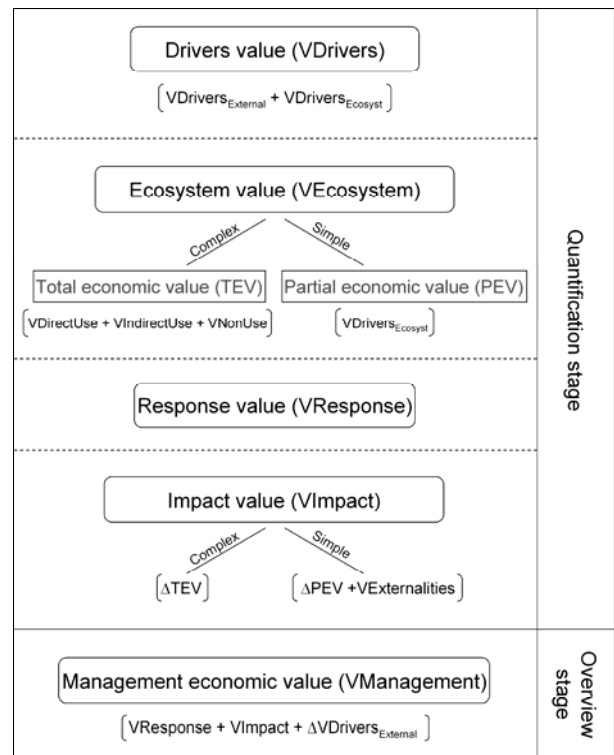
Figure 3.7. ΔDPSIR economic assessment.

In the simple approach, the value of the environmental externalities are internalised in the differential component of the economic assessment (in *VImpact*). As shown in Figure 3.7, *VImpact* is given by the changes in the economic value of the activities that rely on the ecosystem and by the value of the environmental externalities. This approach ensures that the environmental degradation that is not captured in PEV is included in the differential component. The choice of whether to use the complex or simple approach depends on the specific case study objectives, available data and available resources for further data collection.

To compare t and $t+\Delta t$, all economic values calculated in the ΔDPSIR approach must be converted into constant prices. If the assessment to be made is a hindcast analysis, an inflation rate (such as the general consumer index) can be used to convert past values (t) into present values ($t+\Delta t$). If the assessment is a forecast analysis, an appropriate discount rate can be applied to convert future values $t+\Delta t$) into present values (t), as discussed by Chee (2004), Field (1997), Ledoux and Turner (2002), and Tol et al. (1996). When comparing ΔDPSIR results across countries, it is necessary to normalize economic values by use of the purchasing power parity.

Details related to the calculation of *VDrivers, VEcosystem, VResponse, VImpact* and *VManagement* are provided in Table 3.7 and Table 3.8 for the complex and simple approaches, respectively.

Table 3.7. ΔDPSIR complex economic approach.

Value of the Drivers (*VDrivers*)		
$VDrivers = VDrivers_{External} + VDrivers_{Ecosyst}$		**Eq. 3.1**
VDrivers	Production value of the drivers in *t*	€
$VDrivers_{External}$	Value of the drivers in the catchment in *t*	€
$VDrivers_{Ecosyst}$	Value of the drivers in the coastal ecosystem in *t*	€
Value of the ecosystem (*VEcosystem*), given by the total economic value (TEV)		
$VEcosystem = TEV = VDirectUse + VIndirectUse + VNonUse$		**Eq. 3.2**
VEcosystem	Benefits generated from the ecosystem in *t*	€
TEV	Total economic value of the ecosystem in *t*	€
VDirectUse	Direct use value of the ecosystem in *t*	€
VIndirectUse	Indirect use value of the ecosystem in *t*	€
VNonUse	Non-use value of the ecosystem in *t*	€
Value of the impact on the ecosystem (*VImpact*)		
$VImpact = ΔVEcosystem$		**Eq. 3.3**
VImpact	Economic value of the impact on the ecosystem in *Δt*	€
ΔVEcosystem	Changes of the value of the ecosystem in *Δt*	€
Economic value of management (*VManagement*)		
$VManagement = VResponse + VImpact + ΔVDrivers_{External}$		**Eq. 3.4**
VManagement	Economic value of management in *Δt*	€
VResponse	Value of the response in *Δt*	€
$ΔVDrivers_{External}$	Changes of the value of the drivers in the catchment in *Δt*	€

Table 3.8. ΔDPSIR simple economic approach.

Value of the Drivers (*VDrivers*)		
Same as for complex approach – Eq. 3.1, Table 3.7.		
Value of the ecosystem (*VEcosystem*), given by the partial ecosystem value (*PEV*)		
$VEcosystem = PEV = VDrivers_{Ecosyst}$		**Eq. 3.5**
PEV	Partial ecosystem value, corresponds to the $VDrivers_{Ecosyst}$ in *t*, instead of *TEV*	€
Value of the impact on the ecosystem (*VImpact*)		
$VImpact = ΔPEV + VExternalities$		**Eq. 3.6**
VImpact	Economic value of the impact on the ecosystem in *Δt*	€
ΔPEV	Changes of the partial ecosystem value in *Δt*	€
VExternalities	Value of the environmental externalities in *Δt*	€
Economic value of management (*VManagement*)		
In the simple approach, Eq. 3.4 (Table 3.7) can be rewritten as:		
$VManagement = VResponse + ΔVDrivers + VExternalities$		**Eq. 3.7**

Value of the drivers

The value of the drivers (*VDrivers*, Eq. 3.1 in Table 3.7) includes: (1) the value of economic activities that impact a coastal ecosystem but are not sustained by it (*VDrivers$_{External}$*); and (2) the value of activities that depend on the coastal ecosystem (*VDrivers$_{Ecosyst}$*). *VDrivers* corresponds to the sum of the production values for these activities. If such information is not available, the value of the economic activities may be estimated based on proxies, such as the number of workers or the level of output/production. In the case of agriculture, this can be estimated based upon the cultivated area and the average production value per area. The specific method will depend on the available information, but the same method must be applied for both *t* and *t+Δt*. Also, when comparing results between ecosystems, it is important to verify which of the approaches was used. *ΔVDrivers* is given by the difference between *VDrivers$_t$* and *VDrivers$_{t+Δt}$*. In a hindcast application of the ΔDPSIR, this value is calculated based on data; in a forecast analysis, *VDrivers$_{t+Δt}$* is calculated based on scenario predictions, such as those provided by Ferreira et al. (2007a) for aquaculture productivity and by Lipton and Hicks (2003) for recreational fishing. In some cases, especially in forecast analyzes, *ΔVDrivers* can represent a measure of opportunity costs, as in the following conceptual example: If the ΔDPSIR is employed to analyze a set of response actions designed to improve water quality in a coastal ecosystem, which includes the reduction of economic activities in the catchment; then *ΔVDrivers$_{External}$* (which measures the reduction of the economic value of these activities) corresponds to the opportunity cost of the management strategy.

In Ria Formosa, the drivers listed in Table 3.9 were considered. Whenever possible, official statistics for production values were used to determine the activity economic value; however, proxies were used for cases of aquaculture production and tourism (as described in Table 3.2).

Value of the ecosystem

There are a number of well-known techniques that may be used to calculate components of TEV: market prices, production function approaches, surrogate market approaches, cost-based approaches and stated preference approaches (Emerton and Bos, 2004). Söderqvist et al. (2004) provided several case studies that exemplify how several economic components of the ΔDPSIR approach may be calculated. In cases where it is possible to calculate the TEV, the complex approach for ΔDPSIR economic assessment is used (Eq. 3.2 in Table 3.7). Subjectivity of the existing valuation methods for calculating components of the TEV is known (Chee, 2004; Driml, 1997; Nunes and van den Bergh, 2001); however, their systematic

application to the same ecosystem over time can lead to an objective differential value of the ecosystem (*ΔVEcosystem*). Nevertheless, it is often not possible to calculate the full TEV.

For cases in which the full TEV cannot be calculated, application of the simple ΔDPSIR approach is recommended (Figure 3.7). Following this approach, *VEcosystem* is given by *VDrivers$_{Ecosyst}$*, and is therefore named the PEV (Eq. 3.5 in Table 3.8). Non-market natural capital is accounted for in the differential component and is assimilated in *VImpact*, through determination of the value of the environmental externalities during the study period.

Due to data limitations, the simple ΔDPSIR approach was applied to Ria Formosa (Eq. 3.5 in Table 3.8). *VDrivers$_{Ecosyst}$* was calculated considering the activities listed in Table 3.9 that depend on the coastal ecosystem.

Value of the response

The costs of implementing a response must be carefully defined so as not to duplicate costs already included in *ΔVDrivers*. For example, if construction of waste water treatment plants (WWTP) is among the response actions for improving water quality and these costs are already included in the drivers, they should not be included in *VResponse*. The Tillamook Bay National Estuary Project Action Plans (TBNEP, 1999) provides a good example of the items that should be included in *VResponse*, especially in the case of restoration scenarios. If the ΔDPSIR approach is applied to a forecast analysis, i.e., to test different management scenarios, *VResponse* can be calculated based on models (Brady, 2003; Gren and Folmer, 2003).

For the Ria Formosa case study, the response costs (*VResponse*) were calculated based on data described in Table 3.2.

Value of the impact on the ecosystem

In the ΔDPSIR framework, the aim of *VImpact* is to give a quantified measure of the changes in the economic value of an ecosystem, including the market and non-market value of natural capital. Theoretically, the most straightforward way to calculate *VImpact* would be to estimate the TEV at *t* and *t+Δt* and calculate the difference between the two values (ΔDPSIR complex approach). However, as explained previously, in some cases it is only possible to estimate the economic value of activities that depend on the coastal ecosystems. Despite this, managers require information regarding changes in the ecosystem that are not captured in the market. An alternative approach is proposed (ΔDPSIR simple approach) where the value of the environmental externalities (*VExternalities*) is calculated. For instance, if there was a decrease

in the state of the ecosystem during Δt and the pressures are known, the cost of the necessary actions to avoid the environmental degradation or the costs to replace the loss of ecosystem functions should be calculated (*VExternalities*). Examples of estuarine, coastal and marine ecosystem restoration actions are given by Elliott et al. (2007). For cases in which man-made capital is not able to compensate for functions provided by the ecosystem or critical natural thresholds are irreversibly reached (Ledoux and Turner, 2002), *VExternalities* must be flagged and marked as not determinable (n.d.). The *VExternalities* calculation varies from case to case. For example, if the observed changes in state correspond to an increase of eutrophication symptoms due to urban wastewater discharges, the costs of building or improving an already existing WWTP would be a proxy for *VExternalities*. In addition, if the symptoms were also due to agricultural runoff, the costs could be estimated by modeling approaches, such as the simulated scenarios for nutrient load reduction presented by Elofsson (2003).

The two options for *VImpact* calculation are as follows: the complex approach (Eq. 3.3 in Table 3.7), where it is calculated based on the change in TEV, which accounts for changes in the direct, indirect and non-use values; the simple approach (Eq. 3.6 in Table 3.8), where it is calculated based on the change in PEV, plus the value of the environmental externalities (*VExternalities*). A positive *VImpact* value reflects a positive economic impact of the response on the ecosystem, and vice versa.

The economic component of the impact (*VImpact*) in Ria Formosa was calculated based on changes in PEV and the value of the environmental externalities (*VExternalities*). For the calculation of *VExternalities* in Ria Formosa, a list of the actions required to avoid the most relevant negative ecological changes was developed. The costs for implementing these actions were calculated and used to compute the *VExternalities*, based on data described in Table 3.2.

Economic value of management

The economic value of management (*VManagement*, Eq. 3.4 in Table 3.7) provides an overall balance between (1) the direct costs of the response actions (*VResponse*), (2) the changes in the value of the ecosystem (*VImpact*), and (3) the impacts to the local economic activities in the catchment ($\Delta VDrivers_{External}$). This economic variable is quantified in the overview stage (stage 3) of the ΔDPSIR and aims to provide a synopsis of the ΔDPSIR economic assessment.

Spatial and Temporal Scope

The analysis of ecological processes in the coastal zone generally implies temporal resolutions of seconds to days in scientific studies and much longer (years) for management purposes. In the ΔDPSIR framework, the ecosystem is analyzed for a given year and the changes are evaluated after the response implementation period, which normally spans several years. The economic and ecological analyzes for a given year are made using data or simulation results with yearly, monthly, daily or even smaller time steps, depending on the available data. The difference in timescales can be addressed by upscaling the detailed results into the relevant scales of the upstream processes, as exemplified by Nobre et al. (2005): (1) for simulating the transport of substances in large-scale ecosystem models, and (2) to distil the model results into information for managers using screening models (e.g. McAllister et al., 1996; Bricker et al., 2003).

The spatial extent includes processes that occur in the catchment and their effects on coastal ecosystems. The results for managers should be presented at a coarse scale for the entire ecosystem or divided into large bodies of water, as required by several management instruments (Ferreira et al., 2006). This requires a scientific background that ranges from very detailed hydrodynamics (resolved with a temporal resolution of seconds and with a spatial resolution of a grid of millions of cells) to less detailed ecological resolution.

RESULTS AND DISCUSSION

Characterization stage

The period between 1985 and 1995 corresponds to the implementation period of a set of actions defined in the Management Plan of Ria Formosa Natural Park (SNPRCN, 1986). During these years, a significant number of WWTPs were built or improved (15 out of a total of 27). The most important management issues identified in the Ria Formosa regarding that period are seasonal variation of the local human population and a decrease in clam stocks. Tourism during the high season increased by 100% and 150% the resident population in 1985 and 1995, respectively, which made management and operation of WWTP difficult (MAOT, 2000). The decrease in clam stocks in the mid 1980s resulted from the appearance of the parasite *Perkinsus atlanticus* (Azevedo, 1989). The decreased production affected the socio-economy of the local population, since the local clam species (*Ruditapes decussatus*) is a highly valued commodity (Matias et al., 2008), of which Ria Formosa contributes ca. 90% of

Portuguese production. In addition, bivalve aquaculture in Ria Formosa is responsible for the direct employment of up to 10 000 people according to unofficial estimates (Campos and Cachola, 2006). In regard to water quality, the major concerns are (1) the upper reaches of the lagoon channels, where water turnover is substantially lower than in the main channels (Nobre et al., 2005), and (2) benthic eutrophication symptoms, such as excessive macroalgal growth, which occurred as a result of nutrient peaks, large intertidal areas and short water residence times (Nobre et al., 2005).

The period and snapshots considered for the ΔDPSIR analysis in Ria Formosa were:

(1) For t: the annual average for the period 1980–1985;

(2) For Δt: the period between 1985 and 1995;

(3) For $t+\Delta t$: the annual average for the period 1995–1999.

For t and $t+\Delta t$ the average of a period is used instead of a given year due to data limitations.

A list of drivers, pressures and state indicators studied in Ria Formosa is shown in Table 3.9. Although the main component of fish landings in Ria Formosa is obtained outside the lagoon from the open coastal water, this activity is also included in the drivers because it is carried out by local fishermen and the lagoon provides the channels and port for commercialization.

Table 3.9. Characterization of the drivers, state indicators and pressure indicators in Ria Formosa

Pressure indicators	Bivalve aquaculture	Fish aquaculture	Fisheries	Salt production	Tourism	Agriculture/ Livestock	Manufacturing industry	State indicators
Population waste water loads					⊗ ↳	↳	↳	Macroalgal growth
Agriculture/ livestock diffuse loads	⌐					⊗		Dissolved oxygen in shallow areas
Livestock point source	⌐					⊗		Bivalve production rates
Industrial waste water	⌐				↳	↳	⊗ ↳	Water quality in bivalve production areas

⊗ Correspondence between each driver and respective pressure indicators. ↳ Ecological state indicator affected by the corresponding driver; ⌐ State indicator that affects the corresponding driver; ↳ State indicator that is affected and that affects the corresponding driver.

For fish aquaculture, no causal effects between pressures and state were considered given that most of the farmers practise extensive small-scale aquaculture in old saltpans.

Quantification stage

Drivers

According to the characterization of drivers in Ria Formosa and its catchment (Figure 3.8), the most relevant economic activity is bivalve production, which represented 74% of total production in 1980–1985 and 55% in 1995–1999.

In general, there was a decrease in drivers' production, labor force and area in Ria Formosa and its drainage basin, as shown in Figure 3.8 and Table 3.10. The significant reduction in drivers' production between 1985 and 1995 (-299 million Euros) was mostly due to the decrease in bivalve productivity during this period (approximately -66%).
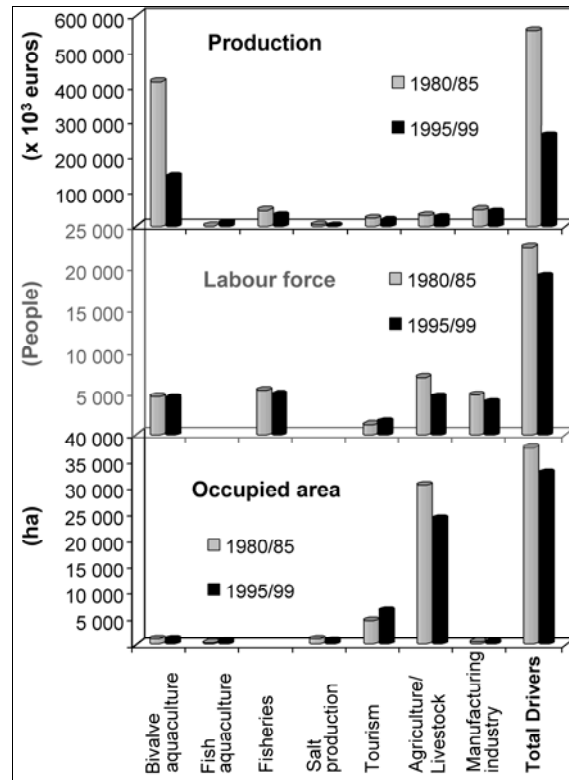
Figure 3.8. Driver production, labor force and occupied area in Ria Formosa and its catchment.

Table 3.10. Quantification of drivers in Ria Formosa and its catchment (changes between 1985 and 1995).

	Δt, 1980/1985 – 1995/1999					
	People		Area		Production	
		(%)	(ha)	(%)	(x10³ €)	(%)
Ecosystem	**- 350**	**- 3%**	*1 750*	*26%*	**- 287 500**	**- 60%**
Bivalve aquaculture	- 200	- 4%	0	0%	- 271 000	- 66%
Fish aquaculture	n.av.	n.av.	200	80%	8 200	1 640%
Fisheries	-500	- 10%	n.ap.	n.ap.	- 15 100	- 34%
Salt production	n.av.	n.av.	- 400	- 38%	- 3 300	- 85%
Tourism	350	29%	1 950	44%	- 6 300	- 29%
Catchment area	**- 3 250**	**- 28%**	**- 6 500**	**- 21%**	**- 11 500**	**- 15%**
Agriculture/Livestock	- 2 350	- 34%	- 6 350	- 21%	- 6 300	- 21%
Manufacturing industry	- 900	- 19%	- 150	- 33%	- 5 200	- 11%
Total drivers	**- 3 600**	**- 16%**	**- 4 750**	**- 13%**	**- 299 000**	**- 54%**

n.av. – not available; n.ap. – not applicable.

Pressure

The nutrients generated in the catchment area are shown in Figure 3.9a. The diffuse loads generated by agriculture and extensive livestock are much higher than the wastewater loads that enter the lagoon directly. However, the potential pressure that the diffuse loads could exert in the lagoon is limited by the seasonal nature of freshwater discharges (Nobre et al., 2005). The significant reduction of nutrients between 1980–1985 and 1995–1999 (Figure 3.9a) was due to the reduction in agriculture and extensive livestock production areas.

Figure 3.9b shows the organic loads (expressed by the biochemical oxygen demand (BOD5) parameter) and the corresponding population equivalents (PEQ). The BOD5 parameter is shown to provide a comparison measure between population wastewater discharge and the loads generated in the catchment area due to intensive livestock and industry.
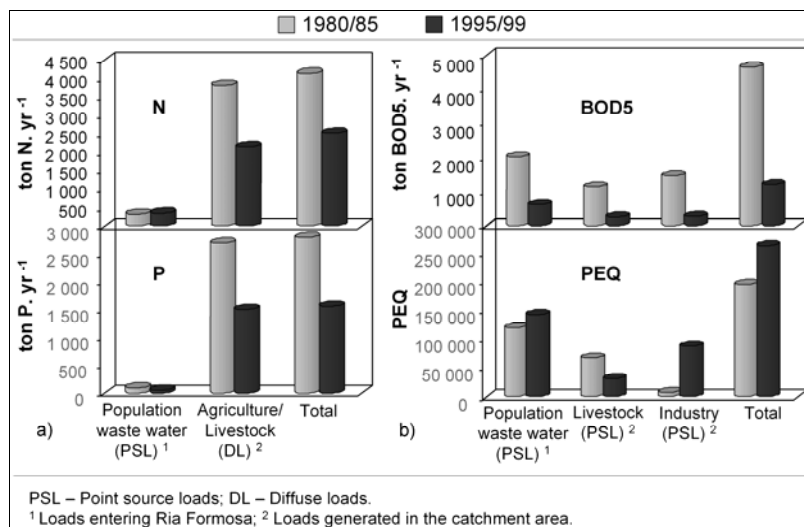
Figure 3.9. Pressure quantification: a) nitrogen (N) and phosphorus (P) loads generated by the drivers, and b) biochemical oxygen demand (BOD5) and population equivalents (PEQ) of the drivers' wastewater.

The estimated loads (shown in Figure 3.9) indicate that these pressures are managed because, although there was an increase from 1980–1985 to 1995–1999 in the population equivalent (196 400 PEQ to 264 600 PEQ), there was a significant decrease in the generated organic load (4 750 ton BOD5 year^{-1} to 1 240 ton BOD5 year^{-1}). It is important to note that a number of WWTPs were built during this period. This generally causes a reduction of loads for the entire catchment; however, there may be localized increases in nutrient loads in areas surrounding the WWTP outlets. With respect to nitrogen inputs into the coastal ecosystem, the direct loads from point sources did not change significantly during Δt (inputs are estimated as 315 ton N year^{-1}).

State

The state of the ecosystem was assessed for the presence of eutrophication symptoms using results from an ecological model of the Ria Formosa (Nobre et al., 2005). The focus was on a problematic area of the lagoon (Box 1 in Figure 3.6a) where the loads increased due to the construction of a WWTP with an outlet in this area. The nutrient load in 1995–1999 corresponds to the standard simulation of Nobre et al. (2005) with an average load of 40 kg N ha^{-1} year^{-1}. In 1980–1985, the direct inputs to this area were 88% less than the levels in 1995–1999. The ecological model indicates that the nutrient load increase causes the macroalgal growth of the larger mass class from less than 50% to about 150% (Figure 3.6b). The model results for dissolved oxygen indicate that the intertidal pools in Ria Formosa are potentially problematic, with dissolved oxygen tenth-percentile values below the threshold defined by Bricker et al. (2003) for biological stress, independent of the nutrient loads (Figure 3.6c). However, a reduction in nutrient loads causes a decrease in the frequency of low dissolved oxygen events, which decreases the biological stress for bivalves in the intertidal areas (Figure 3.6c).

The provided bivalve production rates (ratio between the harvested biomass and seeding biomass) were used to build Figure 3.10a. The collected information indicates a decrease in the production rates from 1980–1985 to 1995–1999 due to the appearance of *Perkinsus atlanticus*. The standard harvest in 1980–1985 was four times the seeding biomass, while in 1995–1999, harvest was as low as one-half of the seeding biomass and the standard harvest was only three times the seeding biomass. The water quality monitoring data (compiled in Figure 3.10b) indicate that the bivalve production areas are overall in good microbiological condition.
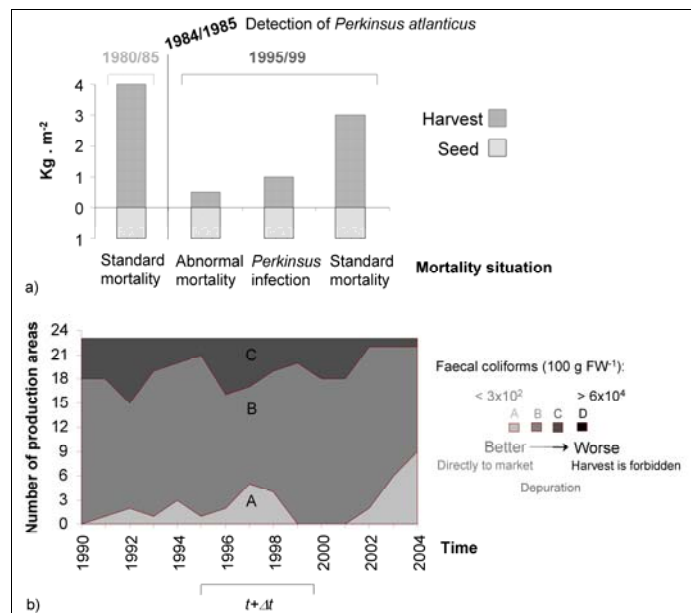
Figure 3.10. Data used for state quantification of bivalve production: a) estimated production rates given by a local aquaculture association and b) classification of bivalve production areas based on annual average values of faecal coliforms.

Following Eq. 3.5 (Table 8), the value of ecosystem benefits for the drivers was used as an economic measure for the state component. Between 1980–1985 and 1995– 1999, a 60% decrease in the PEV (corresponding to -287.5 million Euros) was observed (Table 3.10). This significant decrease is mainly explained by the reduction in bivalve production, which represents 75% and 55% of the value of the drivers at t and $t+\Delta t$, respectively.

Response

Several actions were planned in SNPRCN (1986) for the response implementation period: load reduction, industrial process improvement, aquatic resources quality improvement, sustainable tourism, environmental education, technical and scientific research, and agriculture-related actions. The cost of the actions adopted during the response implementation period was estimated from the data sources shown in Table 3.2 and corresponds to a *VResponse* of -175.9 million Euros.

Impact

The state of the ecosystem worsened from 1980–1985 to 1995–1999, specifically evidenced by: (1) ecological model results indicating an approximate 100% increase in macroalgal growth in certain regions, and (2) abnormal clam mortality caused by infection with *P. atlanticus*.

These negative ecological changes were considered in the calculation of VExternalities. The reduction of organic loads was the action identified to prevent excessive macroalgal growth. This was estimated based on the data described in Table 3.2. The estimated cost for the period of analysis Δt (10 years) was 26.5 million Euros. The main factor indicated to be responsible for abnormal clam mortality was infection with *P. atlanticus*. Outbreaks of this parasite are triggered by temperature and salinity. Furthermore, stressful conditions (like low dissolved oxygen) cause an increase in bivalve mortality due to *P. atlanticus* (Lenihan et al., 1999). This parasite affects shellfish worldwide (Goggin and Barker, 1993) and there are no known eradication methods. However, several management actions can be taken to reduce infection intensity and prevalence. These actions are listed in Table 3.11 along with the respective implementation costs for the 1985 to 1995 period. Given the uncertain nature of these estimates, three scenarios were considered for calculation of the extra cost of certified seeds.

Table 3.11. Possible management action costs necessary to avoid abnormal clam mortality.

Abnormal clam mortality due *P. atlanticus*		
Actions to reduce infection intensity and prevalence	Costs in 2000 constant prices	**(x 10³ €)**
Good screening of seed infection	Examination and sampling costs:	**41**
Extra cost for buying certified seeds	Scenario	Cost
	Δt1 All seeds from outside RF	**250 000**
	Δt2 Seed scarcity in RF	**100 000**
	Δt3 No extra costs	**0**
Lower population densities	No implementation costs:	**0**
Improve D.O. in clam beds (same action as for macroalgal growth)	Reduction of nutrient loads:	**26 500**
Optimise conditions in depuration plants	No implementation costs:	**0**

The salary of the workforce (estimated as 0.48 million Euros for *Δt*) required to monitor the implementation of management actions was also included for calculation of *VExternalities*. Estimates of *VExternalities* are presented in Figure 3.11 for the three action scenarios considered in Table 3.11.

The economic value of the impact was estimated using Eq. 3.6 (Table 3.8) and is presented in Figure 3.11 for the three scenarios of *VExternalities*. *VImpact* assumed a negative value due to the reduction in *VDrivers$_{Ecosyst}$* and the negative value of *VExternalities*. The analysis of economic impact indicates that expenditure of the value estimated in *VExternalities* may have reduced the economic loss of bivalve production, which, according to Table 3.10, represents 94% of the estimated losses of *VDrivers$_{Ecosyst}$*.

VExternalities represents between 49% (in scenario *Δt1*) to 9% (in scenario *Δt3*) of *VImpact* (Figure 3.11). These estimates indicate that if 49% (in scenario *Δt1*) to 9% (in scenario *Δt3*) of the value of the economic impact had been spent, the loss of ecological value during *Δt* may have been reduced to a range between 51% (in scenario *Δt1*) and 91% (in scenario *Δt3*).
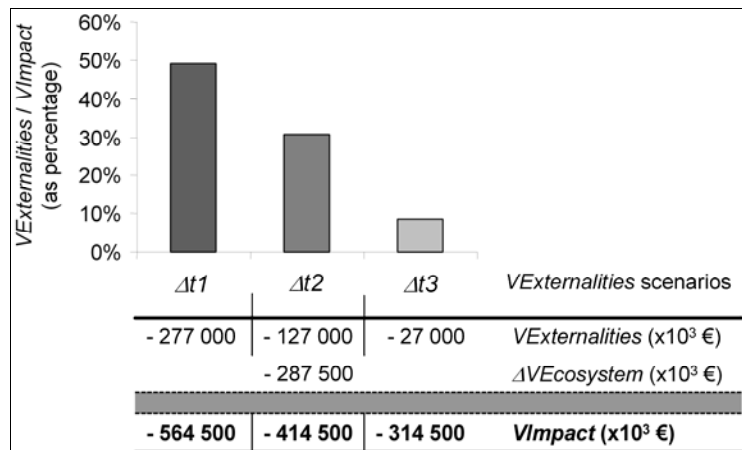
Figure 3.11 ΔDPSIR results in *Δt*: *VImpact* for the three scenarios considered for the calculation of *VExternalities* as defined in Table 3.11.

Overview stage

Figure 3.12 shows the components of *VManagement* according to Eq. 3.7 in Table 3.8. These values show the economic balance, which includes direct costs related to ecosystem management and changes in the economic value of the drivers, both in the catchment and in the ecosystem, and the costs that should have been expended to potentially avoid the ecological changes.
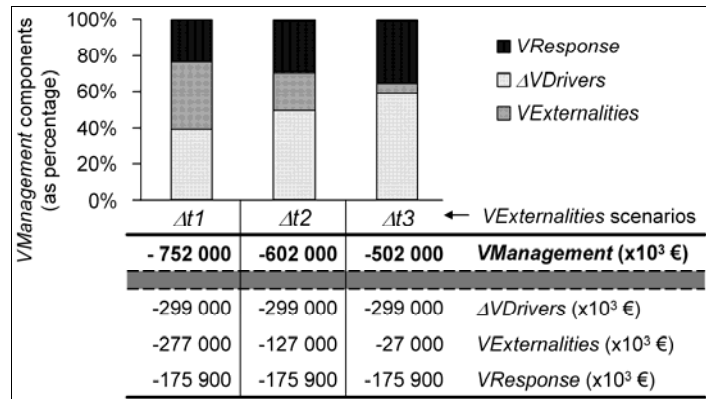
Figure 3.12. ΔDPSIR results in *Δt*: *VManagement* for the three scenarios considered for the calculation of *VExternalities* as defined in Table 11.

VManagement is presented in Figure 3.12 for the three scenarios of *VExternalities* for the extra cost of buying certified seeds, as shown in Table 3.11. The estimated values indicate that there was a negative trend for all components of the economic value of management (Figure 3.12). *VManagement* ranged between -752 million Euros and -502 million Euros, depending on the scenario of *VExternalities*.
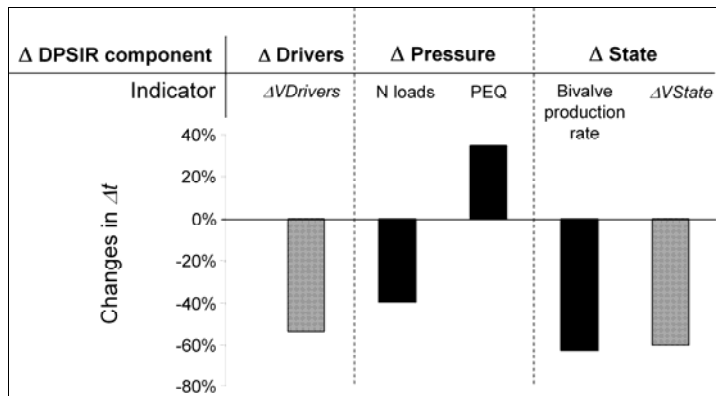
Figure 3.13. ΔDPSIR synthesis: ecological and economic changes in drivers, pressure and state.

The integrated application of ΔDPSIR to Ria Formosa is shown in Figure 3.12 and Figure 3.13. There was a significant management response between 1985 and 1995, with the purpose of reducing nutrient pressures from the catchment on the coastal ecosystem.

The corresponding costs (*VResponse*) represent about six times to half of the value of the environmental externalities (depending on the scenario considered for calculating *VExternalities*). Nevertheless, the response actions did not prevent the negative ecological and economic impacts to the ecosystem: a decrease in the bivalve production rate and an increase in macroalgal growth in the problematic areas. These negative ecological impacts correspond to a significant decrease of *VDrivers$_{Ecosyst}$* estimated at -287.5 million Euros. The negative

economic impacts represent 80–220% of the response cost, depending on the scenario considered for the value of the environmental externalities.

In the Ria Formosa case the main reason for loss of ecosystem economic value is a parasite that significantly decreased the bivalve production rate and for which there are no eradication methods. Research still needs to be conducted in order to determine if the introduction of the Japanese clam (*Ruditapes philippinarum*) is responsible for the appearance of this parasite (Campos and Cachola, 2006). Either way, it is important to note that although there are no eradication methods several measures could have been adopted to mitigate these negative impacts. These cost estimates represent 49–9% of the value of the impact (Figure 3.11). This result indicates that the costs of those measures could have potentially avoided (1) in the worst case, a loss of 51% of the economic impact, and (2) in the best case, a loss of 91% of the economic impact.

The conclusions of the ΔDPSIR application to developments in Ria Formosa between 1985 and 1995 are particularly important for future management actions. For instance, given the significant decrease of provisioning services due to the decrease in clam production, the consequent socio-economic impact for local communities estimated herein, and the fact that the local clam species is being displaced by nonindigenous species with impacts to biodiversity (Campos and Cachola, 2006), it is advisable to invest in the appropriate management of bivalve aquaculture, such as hatcheries programs to reduce the limitations on local clam seeds (Matias et al., 2008). These insights suggest revision of the proposal for the new Management Plan of Ria Formosa Natural Park (ICN, 2005), which only allocates 1.9% of planned total budget for bivalve related actions. Future applications of the ΔDPSIR can provide guidance on the definition of management strategies for the Ria Formosa. As a starting point, management options for current environmental and socio-economic concerns of local stakeholders, such as dredging operations, changes in bivalve cultivation practice, changes in salt marsh areas and change of number and efficiency of WWTPs could be evaluated (Duarte et al., 2007b). In further applications of ΔDPSIR in Ria Formosa it is advisable to include pressure indicators related to the bivalve cultivation practice, in particular with the seeding procurement.

CONCLUSIONS

The ΔDPSIR framework is a powerful tool for integrated coastal management that can support realistic decision-making that accounts for the value of the environmental externalities. It can be particularly useful for the evaluation of management and policy scenarios according to cost and effectiveness criteria. The ΔDPSIR approach includes key concepts for an integrated ecosystem analysis, namely: (1) explicit simulation of the link between ecological and economic systems, and (2) inclusion of a temporal component for comparison of the ecosystem in t and $t+\Delta t$, which is crucial for the assessment of the benefits and impacts of the ecosystem. This approach allows for an analysis of the economic consequences of changes in environmental quality. In addition, the ΔDPSIR approach may stimulate discussion of possible links between management and science, which is required for sound decision-making and contributes to a better understanding of the management/science scale paradox.

Application of the ΔDPSIR was illustrated through an analysis of developments in the Ria Formosa coastal lagoon between 1985 and 1995. The value of economic activities dependent on the lagoon suffered a significant reduction (ca. –60%) over that period, mainly due to a decrease in bivalve production. During that decade the pressures from the catchment area were managed (ca. 176 million Euros), mainly through the building of WWTP's. Nevertheless, the ecosystem state worsened with respect to abnormal clam mortalities due to a parasite infection and to benthic eutrophication symptoms in specific problematic areas. The negative economic impacts during the decade were estimated between -565 and -315 million Euros of which 9–49% represent the cost of the environmental externalities. The evaluation of past developments suggests that future management actions should focus on reducing the limitation on local clam (*Ruditapes decussatus*) seeds, with positive impacts expected for both the socio-economy of the local population as well on biodiversity.

The ΔDPSIR should be applied to a range of ecosystems with different problems and different levels of monitoring in order to ascertain its usefulness and to compare the results. Further validation of this approach is necessary to verify its ability to consistently translate ecological and economic outputs into information that is useful to managers. Furthermore, future work should include the development of a social component for the ΔDPSIR in order to monitor changes in social indicators, such as employment and per capita income, which result from implementation of the policies or changes in economic activity.

Chapter 4. Ecosystem approach to aquaculture

Context

The aquaculture industry is an important socio-economic activity: (i) it is one of the fastest growing animal food-producing industries (6.1 % increase from 2004 to 2006 – FAO, 2009), (ii) it contributes for food security, particularly in developing countries (Ahmed and Lorica, 2002; Kaliba et al., 2007), and (iii) it can generate employment and other economic benefits for local communities (Ahmed and Lorica, 2002; Kaliba et al., 2007). Aquaculture is expected to increase to meet increasing demand for fish, given the strong likelihood that wild fisheries will remain stagnant (FAO, 2009). However, aquaculture production is slowing (FAO, 2009). The development of sustainable aquaculture calls for an integrated ecosystem approach (FAO, 2007; Soto et al., 2008), known as the ecosystem approach to aquaculture (EAA, as explained in *Chapter 1*). EAA considers three scales of analysis: (i) the waterbody/watershed level, (ii) the individual farm level, and (iii) the global market-trade level.

Summary

This chapter uses the methodologies developed in *chapters 2 and 3* to illustrate their application to EAA at two of the relevant levels of analysis. The first part of this chapter uses the ΔDPSIR approach to carry out an ecological-economic assessment, at the waterbody/watershed level, of the scenarios simulated with the multilayered ecosystem model. The second part provides a detailed ecological-economic analysis of aquaculture options at the farm level, by means of the ΔDPSIR.

4.1 Waterbody/watershed level assessment: evaluation of model scenarios

Context

The analysis of aquaculture at the ecosystem level is essential, primarily, because of the feedbacks between this industry and coastal ecosystem. For instance, production depends on the good condition of the ecosystem, which aquaculture itself may compromise (GESAMP, 2001; Islam, 2005). Second, an ecosystem-level analysis is required because coastal ecosystems are characterised by complex ecological interactions and are subject to a multiplicity of driving forces generated in the catchment, inside the waterbody and also from the sea boundary (Ferreira et al., 2008a).

Summary

Chapter 2 presents the multilayered ecosystem model for simulating the cumulative impacts of multiple uses of coastal zones. *Chapter 3* describes the ΔDPSIR methodology developed for the ecological-economic assessment of the effectiveness of coastal management actions. In this chapter, the multilayered ecosystem model and the ΔDPSIR are applied together, with the objective of synthesising the scenario simulation outputs into useful information for sustainable aquaculture development at the ecosystem level. Furthermore, this chapter extends the ΔDPSIR testing carried out in *Chapter 3* by using it to evaluate management scenarios, as opposed to evaluation of past management responses by means of data analysis.

Integrated environmental modelling and assessment of coastal ecosystems, application for aquaculture management

INTRODUCTION

Sustainable use and development of coastal areas represent a challenge to coastal managers as detailed in the *Chapter 1*. As discussed previously, in *chapters 1 and 2*, ecological modelling is a powerful tool to assist coastal management. In many integrated environmental assessments (IEA's), not specifically related with coastal management, a large effort focused on development of simulation models (Peirce, 1998). IEA consists on the interdisciplinary synthesis of scientific knowledge to provide insight regarding complex phenomena, namely to guide on decision-making and policy development to address environmental problems and for ecological resources management (Peirce, 1998; Toth and Hizsnyik, 1998). There are many possible roles for models in IEA, as for instance: (i) understanding behaviour of complex system; (ii) scenarios analysis; and (iii) quantifying uncertainty of integrated assessment (Peirce, 1998). A wide range of models exist whereby the spatial and temporal resolutions, the features of the system included in the model and the level of detail at which they are simulated, the computational and numeric complexity are adapted to the specific needs of each case (as further discussed in *Chapter 2*). Integrated assessment and modelling is required in order to adopt a comprehensive analysis of the system including its feedbacks (Harris, 2002). To be useful for managers, independently of the complexity degree of model, the generated outputs need to be translated into meaningful information to managers and other stakeholders (Harris, 2002; Nobre et al., 2005). For instance while there are many coastal ecosystem models, few efforts exist for communicating its output for managers with concrete solutions for coastal problems. For that purpose a multidisciplinary approach that synthesises scientific-based information to managers is required.

IEA frameworks are normally applied for organizing and structuring information to facilitate analysis and assessment of environmental data (Stanners et al., 2008). IEA can also be used to analyse and synthesise ecological models into meaningful information for coastal managers. For instance, Liu et al. (2008) proposes a framework to make integrated modelling efforts useful for managers; whereby one of the steps includes assessing and comparing impacts of defined scenarios. The Drivers-Pressure-State-Impact-Response (DPSIR) is a conceptual IEA approach widely used for management of coastal systems. The DPSIR was previously used to

guide on reporting of catchment-coastal ecosystem models to compare nutrient loading scenarios (Artioli et al., 2005; Hofmann et al., 2005; Salomons and Turner, 2005). This conceptual approach establishes a causal link between (Elliott, 2002, Borja et al., 2006, Stanners et al., 2008; Nobre, 2009): human activities (Drivers), the direct effects that these generate (Pressures), the resulting condition of the ecosystems at a given moment in time (State), the variation of the State of the ecosystem as a result of the Pressures during a given time period (Impact), the management actions and policies that cause a change of the Drivers (Response). In particular, the differential DPSIR (ΔDPSIR – developed in *Chapter 3*) provides an explicit link between the ecological and economic quantification of the D-P-S-I-R components. Other set of IEA approaches is targeted to assess specific environmental issues. For instance, the increase of visible effects of coastal eutrophication worldwide enhanced the need for assessment tools for management of eutrophication process (Vidal et al., 1999). The ASSETS approach (Bricker et al., 2003; 2008) exemplifies an eutrophication assessment model applied worldwide (Whitall et al., 2007; Borja et al., 2008). This is one of US governmental tool to guide on eutrophication management in about 140 coastal systems over the entire US coast (Bricker et al., 2008).

The sustainability of aquaculture industry is a current challenge for managers of coastal ecosystems and of aquatic resources. While better management practices must be implemented at the farm level the analysis of the aquaculture impacts must be carried out at the ecosystem level; mainly because the individual farm effects are cumulative in relation to other farms in the same ecosystem and to other coastal activities (GESAMP, 2001; Soto et al., 2008). The use of simulation models coupled with IEA approaches can be particularly important for mariculture managers to adopt the emerging concept of an ecosystem approach to aquaculture (EAA – FAO, 2007; Soto et al., 2008). Firstly, simulation models provide understanding about coastal ecosystems and interactions with aquaculture activities. For instance McKindsey et al. (2006) and Ferreira et al. (2008a) illustrate the importance of using ecosystem models for determining aquaculture carrying capacity. Particularly important are integrated modelling approaches, such as the multilayered ecosystem model developed in *Chapter 2* that allows for the assessment of cumulative impacts of coastal activities at the ecosystem level. Secondly, the IEA approaches are useful to distil the generated outputs to managers and to compare the impacts estimated due to simulated management scenarios. For instance, the simulated effects of changes of nutrient loading into a shallow lagoon are synthesised concerning eutrophication status using the ASSETS screening model (Nobre et al., 2005). The nutrient loading into coastal systems from fish wastes represents an important

aquaculture related problem to be tackled, since not only is it an eutrophication source compromising water quality but also limits the expansion of aquaculture itself (Islam, 2005).

Therefore, the application of a similar approach to investigate impacts of scenarios designed to manage fish aquaculture wastes is highly relevant for EAA. Furthermore, the use of an IEA approach such as the ΔDPSIR can extend the scenario analysis by assessing both the ecological and economic impacts of the management options.

The overall aim of this chapter is to present an approach that couples coastal ecosystem modelling with integrated environmental assessment methodologies. The focus of this work is to support the development of an ecosystem approach to aquaculture management including interactions with substance loading from the watershed. The integrated environmental modelling and assessment approach is illustrated using the Xiangshan Gang, China. The simulated scenarios defined by the bay stakeholders as described in *Chapter 2* are herein analysed and compared using the ΔDPSIR developed in *Chapter 3*. The objectives of this work are to: (i) assess the eutrophication condition of a coastal bay and analyse the impacts of simulated scenarios on bay eutrophic state; (ii) assess the ecological and economic impacts of the management scenarios; and (iii) provide information about the adoption of an EAA and about options for sustainable coastal management of the bay.

METHODOLOGY

General approach

The integrated environmental modelling and assessment approach consists in using IEA methodologies to evaluate ecosystem model outputs for different scenarios (Figure 4.1). For this work the outputs of the multilayered ecosystem model developed in *Chapter 2* are used. The model runs include simulation of scenarios that comprise changes of aquaculture and catchment pressure.

These scenarios represent the actions that managers of the Chinese bay want to test as future responses. The integrated assessment of the model outputs is made by means of (i) an eutrophication assessment model (ASSETS model - Bricker et al., 2003), to classify the overall eutrophic condition of each development scenario; and (ii) a differential version of the Drivers-Pressure-State-Impact-Response (ΔDPSIR – Nobre, 2009) to compare the ecological and economic performance of each scenario with the standard simulation.
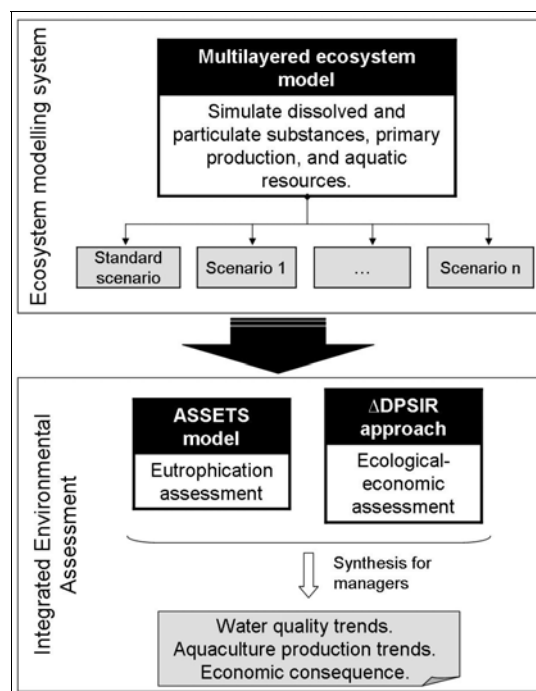
Figure 4.1. Diagram of the integrated environmental modelling and assessment approach for coastal ecosystems.

Case study site and data

The approach presented in this chapter is applied to a Chinese embayment, the Xiangshan Gang (Figure 4.2). This ecosystem has an intensive use of coastal resources with a large aquaculture production and multiple uses of its catchment area. Local farmers indicate a decrease in fish price due to the deterioration of the fish taste following the fish cultivation boom, just as occurred in other Chinese provinces (Zhang et al., 2002). The bay shows several eutrophication symptoms, the most important being: (i) HAB events which originate at the sea boundary and are also driven by pressures exerted in the bay (SOA, 2006; Zhang et al., 2007; ZOFB, 2008); and (ii) sediment anoxic layer under fish cages (Ning and Hu, 2002; Huang et al., 2008b). *Chapter 2* describes other features of the Xiangshan Gang and its catchment.
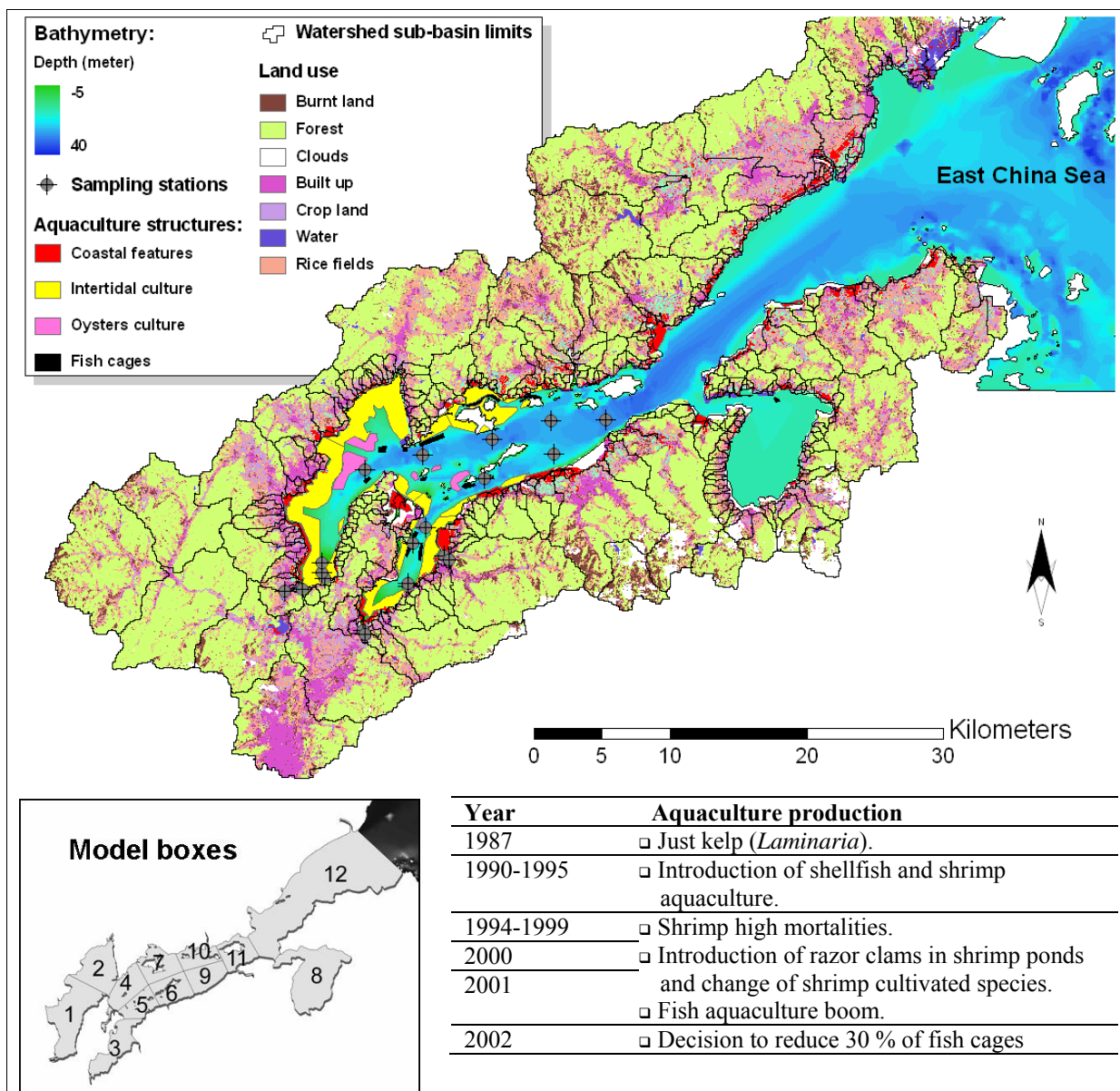
Figure 4.2. Xiangshan Gang characterization: Bathymetry; sampling stations; aquaculture structures and production; watershed sub-basin limits and land use; and model boxes.

The Ningbo municipality, of which the Xiangshan Gang is part, has a strategic plan for the sustainable development of this area, namely to address its environmental problems. Several actions are foreseen to balance its protection and its use in order to take advantage of the ecological and marine resources (Ningbo Municipal People's Government, 2006). Furthermore the motivation at the provincial level is that water quality in Xiangshan Gang should be classified as level I of the Chinese seawater quality standards (Cai and Sun 2007), which corresponds to the best class.

Data description and analysis

The work described herein involved assembling a wide range of data (Ferreira et al. 2008b) and the model outputs described in *Chapter 2*. Table 4.1 synthesises the mixed dataset used

for the development of the case study, including both environmental and socio-economic data (Ferreira et al. 2008b).

Table 4.1. Synthesis of dataset used in the integrated modelling and assessment approach. Data compiled from *Chapter 2* work, otherwise reference is provided.

	Parameters	Dataset
ASSETS		
Zoning	Annual median salinity	Water quality database of Xiangshan Gang 9 stations; Monthly sampling June 2005 - June 2006.
IF	System volume	GIS calculation
	Catchment loads and freshwater flow	SWAT model outputs
	Aquaculture substance loads	Number of fish cages per box; Average fish production per cage; Food waste; Nutrient load per fish produced, based on dry feed conversion rate (Cai and Sun, 2007)
	Sea water salinity and nutrient concentration	East China Sea database; 1 station near Xiangshan Gang; Seasonal sampling 2002.
EC	Chl-a concentration and DO	Water quality database of Xiangshan Gang 9 stations; Monthly sampling from June 2005 to June 2006. Ecosystem model outputs for Chl-a concentration.
	Macroalgae	Local expert knowledge
	HAB	Qualitative and quantitative data from research surveys (ZOFB, 2008)
	SAV	Local expert knowledge
FO	Expected pressure change	Local expert knowledge
ΔDPSIR		
Drivers	Value of the drivers	Aquaculture production survey; Changes of fish production target for scenarios; Model outputs for shellfish production; Unit net profit per kg of aquatic resource produced.
Pressures	Catchment and aquaculture loads	Described for ASSETS - IF
State	Ecological state	Nutrient concentration in the bay from water quality database and ecosystem model outputs; Outputs of the ASSETS EC index; shellfish productivity from production survey and ecosystem model outputs.
	Partial ecosystem value	Same as value of the drivers.
Impact	Value of environmental externalities	Average investment and operational costs per cubic meter of wastewater to treat annually, from projects in China (U.S. Department of Commerce, 2005).
Response	Response cost	Finfish reduction calculated based on changes of net profit (detailed in the Drivers); WWTP investment costs (detailed in the Impact).

IF, Influencing factors; EC, Eutrophic conditions; Chl-a, chlorophyll-a; DO, dissolved oxygen; HAB, harmful algal bloom; SAV, submerged aquatic vegetation; WWTP, wastewater treatment plant.

Scenarios

At present, the high nutrient concentration causes this ecosystem to be poorly classified relatively to the Chinese seawater quality standards (National Standard of People's Republic of China, 1997). Furthermore, there are several eutrophication symptoms, some of which threat aquaculture activities inside the bay (Chen et al., 1992; ZOFB, 2008). The scenarios analysed comprise the settings that Xiangshan Gang stakeholders considered important to be tested in order to improve water quality (Ferreira et al., 2008b): (i) a reduction of fish cages corresponding to 38% less of total fish production (scenario 1), (ii) extend wastewater treatment to the entire population (scenario 2), (iii) simultaneous reduction of fish cages and wastewater treatment plant (WWTP) implementation (scenario 3). *Chapter 2* presents more detail about the scenarios. Table 4.2 synthesises the substance loading used to simulate each scenario.

Table 4.2. Scenario definition.

	Total loads (t d^{-1})		
	DIN	Phosphate	POM
Standard	18.9	5.0	451.7
Scenario 1	16.2	3.9	410.1
Scenario 2	17.5	4.2	413.8
Scenario 3	14.7	3.1	372.1

ASSETS model application

The application of the ASSETS model to the Xiangshan Gang followed the procedure described in Bricker et al. (2003) and applied worldwide (http://www.eutro.org/). Furthermore, this case study includes the updates made to the methodology (Bricker et al., 2008). The ASSETS model and its application are well described in Bricker et al. (1999, 2003, 2007 and 2008), Scavia and Bricker (2006) and Ferreira et al. (2007b). Herein, the application to the study site is briefly explained. Firstly, the ecosystem is divided into salinity zones (tidal freshwater, mixing water, seawater) according to ASSETS thresholds (Bricker et al., 2003). The Xiangshan Gang zoning is carried out by calculating the annual median salinity for each box of the ecosystem model (Figure 4.2), considering data in Table 4.1. For boxes with no sampling stations, the average annual salinity of the connecting boxes is calculated. Secondly, the ASSETS indices - influencing factors (IF), eutrophic condition (EC) and future outlook (FO) - are calculated based on field data for 2005-2006 and local expert knowledge as detailed below. Thirdly, these indices are combined into a single overall score, which is assigned into one of five categories (Bricker at al., 2003): high, good, moderate, poor

or bad. Finally, the ASSETS indices are recalculated for the model simulations following procedure detailed below.

Influencing factors - IF

The IF index calculates the pressure on the system as a combination of the nutrient loading with the system susceptibility to eutrophication (flushing and dilution factors) (Bricker at al., 2008). The simple nutrient mass balance model used in ASSETS (Bricker et al. 2003) is applied to combine human pressure and Xiangshan Gang susceptibility, using the data synthesised in Table 4.1. The outputs are used to classify the influence factors (IF) index according to thresholds defined in Bricker et al. (2003). Nutrient loads include both catchment and aquaculture sources. The IF outputs indicate the relative importance of these sources compared with the inputs from the sea boundary.

Eutrophic condition - EC

The EC index (Bricker at al., 2003; 2008) is calculated by determining for each salinity zone the level of expression of the (i) primary symptoms, chlorophyll-a (chl-a) and macroalgae; and (ii) secondary symptoms, low dissolved oxygen (DO), harmful algae blooms (HAB) and loss of submerged aquatic vegetation (SAV). Data from a water quality database is used to calculate the chl-a and DO symptoms, using the 90th percentile and 10th percentile values, respectively (Table 4.1), as described in Bricker et al. (2003). The remaining symptoms are calculated based on mixed type of information including local knowledge and outputs from local research surveys (Table 4.1).

Future outlook - FO

The FO index is calculated based on envisaged actions by local managers. Given the proactive managers of this ecosystem (Ferreira et al., 2008b and their willingness to improve the rating of the estuary according to the Chinese water quality standards (Cai and Sun, 2007), more improvements are foreseen. The FO outputs indicate whether the eutrophic condition will worsen, improve or remain the same, based on the system susceptibility and the predicted future nutrient loads (Bricker at al., 2003; 2008).

ASSETS application to ecosystem model outputs

The ASSETS application to the model outputs followed the procedure carried out by Nobre et al. (2005). The model simulations are used to computed the IF and EC indices. FO is considered to remain the same for any scenario given the willingness to further improve this ecosystem. For each scenario IF is recalculated according to projected nutrient loads (Table

4.2). For each scenario EC is recalculated based on the simulated chl-a concentration provided by the ecosystem model. A monthly random sample of the model results is used in order to reproduce the field data sampling frequency (Nobre et al., 2005). All the remaining symptoms for calculation of EC are assumed to remain constant. Rationale and limitations of this assumption are further explored in the results and discussion section. The ASSETS outputs for the standard simulation are compared with the data-based application, to verify validity of ASSETS application to the model outputs.

Differential Drivers-Pressure-State-Impact-Response application

A differential version of the Drivers-Pressure-State-Impact-Response (Differential DPSIR or ΔDPSIR) approach (Nobre, 2009) is adopted to evaluate the effectiveness of the proposed management scenarios (i.e. the Response) compared with the standard simulation. For that purpose the changes in the indicators of Drivers, Pressure and State due to the scenario implementation are investigated as well the resulting economic and ecological Impacts (Figure 4.3). The overall objective is to synthesise the model outputs into meaningful information for managers (Figure 4.1). Economic values are presented in Chinese currency (1 USD = 8.06 Yuan; at the time of the study).
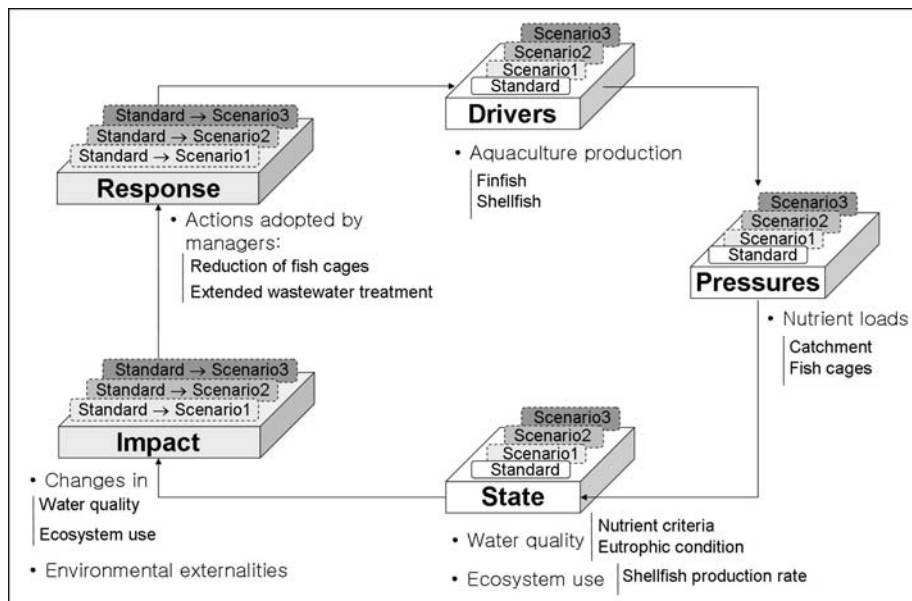
Figure 4.3. Differential DPSIR application to evaluate simulated scenarios.

Drivers

The Drivers are quantified based on the value of the economic activities established in the coastal zone (including the catchment) (Nobre, 2009). For this analysis, aquaculture production represents the Drivers and is assumed that there are no changes in the remaining economic activities. Therefore, the changes in the value of the Drivers ($\Delta VDrivers$) between

the standard simulation and each scenario are quantified based on the change of the net profit of the activities that depend on the ecosystem (*ΔVDriversEcosyst*) while change of the value of the activities on the catchment (*ΔVDriversExternal*) equals to zero. The aquaculture net profit for each scenario and standard simulation is estimated based on: (i) the fish weight production and the simulated weight production of shellfish; and (ii) the net profit per unit produced, which is obtained based on an aquaculture economic survey for finfish, oyster, and clams (de Wit et al., 2008). The shellfish species simulated in the multilayered ecosystem model are *Ostrea plicatula* (Chinese oyster), *Sinonvacula constricta* (razor clam), *Tapes philippinarum* (Manila clam) and *Tegillarca granosa* (muddy clam), as detailed in *Chapter 2*. The finfish production is a forcing function of the model that contributes with dissolved and particulate wastes (as explained in *Chapter 2*). Scenarios 1 and 3 implement the change of fish cages that managers want to test.

Pressures

The significant Pressures to monitor are the discharge of nutrients from the catchment and aquaculture. These are estimated based on the multilayered ecosystem model developed in *Chapter 2*. In order to inform managers about the significance of the loads (from an eutrophication perspective) the outputs of the IF index of the ASSETS model are used. A high score of IF index indicates that control measures can be adopted at the catchment-estuary level, while a low score indicates nutrient reduction actions should be taken at a wider level or are not manageable.

State

The ecological State corresponds to the condition of the aquatic ecosystem resulting from both natural and anthropogenic factors. The State of the ecosystem is analysed considering relevant environmental and ecological criteria:

(i) The nutrient (DIN and phosphate) criteria following Chinese seawater quality standards (National Standard of People's Republic of China, 1997). The simulated annual averages of DIN and phosphate concentration are classified according to the thresholds defined in the standards for these parameters. A monthly random sample of the model results for the day time is used in order to reproduce the field data sampling frequency.

(ii) The eutrophication condition (EC) index of the ASSETS model is used for the classification of the Xiangshan Gang eutrophic state, as detailed previously.

(iii) The shellfish productivity, given as the average physical product (APP) as defined by Jolly and Clonts (1993), is used as a proxy for the ecosystem use. APP calculation consists of dividing the total weight of shellfish harvested by the total weight of seeding.

The calculation of the ecosystem total economic value (TEV) is beyond the scope of this study. Herein, the partial ecosystem value (PEV) is calculated following the simple approach of the ΔDPSIR (Nobre, 2009). The PEV corresponds to the value of the Drivers that depend on the ecosystem (*VDriversEcosyst*) as calculated in the Drivers section.

Impact

The ecological Impact corresponds to the effect of the anthropogenic pressures in the State of the ecosystem and corresponds to the changes in the State at a given time period or between simulation scenarios (Nobre, 2009). The ecological Impact is calculated based on the variation in the (i) nutrients classification according to the Chinese seawater quality standards, (ii) eutrophic condition as determined with the EC index of the ASSETS model, and (iii) shellfish productivity.

The corresponding economic Impact (Nobre, 2009) is calculated based on i) changes of the partial ecosystem value given as the value of the Drivers that depend of the ecosystem (*VDriversEcosyst*, as calculated in the Drivers section), and on ii) the value of environmental externalities (*VExternalities*). The economic Impact component of the Differential DPSIR aims to provide a measure of the direct changes on the economic activities that depend of the ecosystem as well the indirect effects that are not captured in the those activities (environmental externalities). In this analysis, the environmental externalities concerning the shift from the standard simulation to each scenario correspond to the total nutrient load reduction, due to both the fish cage reduction and the WWTP implementation. The avoided costs for treating downstream an equivalent amount of nutrients (Farber et al., 2006) correspond to the *VExternalities*. Data about several WWTP projects in China and respective investments (Table 4.1) provided the investment and operational costs of effluent treatment. Although the WWTP implementation represents a cost for the municipality, thus is accounted as a Response cost, it must also be accounted here as a positive environmental externality. The average investment and operational costs are given expressed per cubic meter (U.S. Department of Commerce, 2005): 4 163 Yuan per cubic meter of total capacity to treat annually and 0.7 Yuan per cubic meter of wastewater to treat, respectively. As such the fish nutrient load reduction is converted into equivalent wastewater flow to treat annually considering the nitrogen (N) and phosphorus (P) removal estimated for the WWTP (Ferreira et al., 2008b): 12.3 mg N L^{-1} and 2.8 mg P L^{-1}. The fish cage nutrient reduction (Table 4.2)

converts into about an equivalent wastewater avoided to treat annually of 81 million m^3, considering the N load, and 66 million m^3, considering the P load. A precautionary approach is adopted, therefore the larger volume of wastewater avoided to treat is chosen. Calculation of the investment cost per annum considered 30-year depreciation for the WWTP facility.

Response

The Response is characterised by the measures that the local stakeholders defined in each scenario. The corresponding costs are the production losses due to reduction of the finfish cages in scenarios 1 and 3, and the WWTP investment and operational costs in scenarios 2 and 3. It is assumed that the cost for the reduction of the fish cages is equivalent to the net profit that would be obtained by the fish farmers if production is maintained at the standard level. Calculation of the fish aquaculture net profit is detailed previously in the Drivers section. The WWTP related costs are calculated based on several WWTP projects in China and respective investments (U.S. Department of Commerce, 2005) as detailed previously in the economic Impact section for calculation of the *VExternalities*.

Overview

The overall economic gain or loss in adopting each scenario (*VManagement*) is calculated by means of a balance between: (i) the direct costs related to ecosystem management (*VResponse*), (ii) the resulting changes in the economic value of the drivers (including only shellfish production given that the finfish cage reduction is an adopted measure and thus is already accounted in the *VResponse*), and (iii) the value of the environmental externalities (*VExternalities*). This synthesis value reflects not only the Impacts caused by the simulated Response actions on the Drivers economic value but also on the indirect value of environmental effects, while accounts for the costs required to implement the Response.

Like any modelling exercise, the application of IEA approaches (the ASSETS and the ΔDPSIR) to interpret modelling outputs presents limitations. The most outstanding on this particular case study regards HAB events. The ecosystem model excludes HAB simulation due to the lack of underlying deterministic knowledge about the complexity of its causes and its chaotic behaviour (Huppert, et al., 2005; Huang et al., 2008a). Consequently, the changes that occur in any of the simulated scenarios regarding HAB events are not predicted. Therefore, and based on the fact that HAB's are occurring at least since 1992 in Xiangshan Gang (Chen et al., 1992) it is assumed that the HAB symptom for ASSETS calculation

remained constant. Likewise, the economic Impacts due to changes in aquaculture closure owing to HAB's are assumed to suffer no changes.

RESULTS AND DISCUSSION

Eutrophication assessment of Xiangshan Gang

Data-based application

The majority of the Xiangshan Gang classifies as seawater zone (276 km^2) corresponding to boxes 4 to 12. The remaining area (84 km^2) of this coastal embayment classifies as mixing water and corresponds to boxes 1 to 3.

Influencing Factors (IF): The influence from aquaculture and catchment loads on the bay's nutrient concentration is moderate high for N and high for P. These IF ratings indicate large nutrient loads compared to the system dilution and flushing potential, which on the other hand points towards a large potential for nutrient reduction from a catchment-bay management perspective.

Eutrophic Condition (EC): Table 4.3 synthesises the EC calculation. For primary symptoms problems are observed concerning the level of expression of chl-a in the mixing and seawater zones. In the mixing zone a high level of expression is obtained given that the chl-a 90th percentile values fall within the range for medium eutrophic conditions and that occur with a high spatial coverage. In the seawater zone a moderate level of expression is obtained given that two out of six stations register medium eutrophic conditions and all the remaining fall below the 5 μg L^{-1} threshold. The frequency of the chl-a problems in both zones is considered periodic, given the seasonal phytoplankton peaks observed in the bay on previous years. For macroalgae no problems are reported since only cultivated biomass is registered in both zones. Concerning secondary symptoms, there are no problems with low DO concentrations given that calculated 10th percentile value for both zones are higher than the threshold adopted as indicative of biological stress (Bricker et al., 2003). SAV symptom is ambiguous to classify because Xiangshan Gang is a highly modified ecosystem, where most of its intertidal and near shore area is converted into aquaculture areas, so no inferences can be done regarding the loss of those habitats as a result of euthrophication process. On the other hand measurements of the sediment below fish cage, carried out on research programmes in 2002, estimate 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).

Table 4.3. Xiangshan Gang eutrophic condition (EC) classification, based on data.

Primary symptoms: Moderate (0.31)

Chl-a	Percentile 90 value	Spatial coverage	Frequency	Level of expression (score)
Mixing	15.5	High	Periodic	High (1)
Seawater	7.5	Moderate	Periodic	Moderate (0.5)
Xiangshan Gang				**High (0.62)**

Macroalgae	Problem status	Spatial coverage	Frequency	Level of expression (score)
Mixing	No problem	N.A.	N.A.	Low (0)
Seawater	No problem	N.A.	N.A.	Low (0)
Xiangshan Gang				**Low (0)**

Secondary symptoms: High (1)

DO	Percentile 10 value	Spatial coverage	Frequency	Score (classification)
Mixing	5.7	N.A.	N.A.	Low (0)
Seawater	6.6	N.A.	N.A.	Low (0)
Xiangshan Gang				**Low (0)**

HAB	Problem status	Duration	Frequency	Score (classification)
Mixing	Observed	Weeks/months	Periodic	High (1)
Seawater	Observed	Weeks/months	Periodic	High (1)
Xiangshan Gang				**High (1)**

SAV	Change	Magnitude of change		Score (classification)
Mixing	Loss	Very low		Low (0.25)
Seawater	Loss	Very low		Low (0.25)
Xiangshan Gang				**Low (0.25)**

Xiangshan Gang Eutrophic Condition (EC): High

Chl-a, chlorophyll *a*; DO, dissolved oxygen; HAB, harmful algal bloom; SAV, submerged aquatic vegetation; N.A., not applicable.

As such and adopting the precautionary principle it is considered, for each salinity zone, SAV loss problems with a coverage equivalent to the fish cage area. Therefore, a SAV loss with a very low magnitude of change is obtained for both the mixing and seawater zones. There is a long record of HAB events in Xiangshan Gang either originated in the bay or coming from the East China Sea (Chen et al., 1992; ZOFB, 2008). Given the examples provided in Table 4.4 this parameter of the EC classifies as high for Xiangshan Gang.

Future Outlook (FO): The local government plans aim to improve water quality (Cai and Sun, 2007). Managers willingness to improve is manifested in the stakeholders meeting described by Ferreira et al. (2008b). The magnitude of change of nutrient pressures tested in the three scenarios corresponds to improve low of conditions in Xiangshan Gang, according to the FO index.

Table 4.4. Example of HAB events in Xiangshan Gang (ZOFB, 2008).

Year	Occurrences	Location / Coverage	Red tide dominant species and effects
2000	Incident 24 May – 15June	200 km^2	*Peridium* sp., *Prorocentrum* sp.; Severe finfish mortalities with economic impact estimated as 10 million Yuan
2003	Mid May and mid June	Inside Xiangshan Gang	*Skeletonema* sp., *Chaetoceros* sp.*;* Occurred near finfish cultivation area with no severe economic loss reposrted.
	Overall 21 occurrences	18 outside, 3 inside extended for more than 30 days	
2004	May 27 to June 3	Inside Xiangshan Gang	*Gymnodinium* sp. and *Chaetoceros* sp.
2005	June	Entrance of Xiangshan Gang / 1600 km^2	*Karenia mikimotoi*; Hemolitic toxicities with razor clam mortalities
	May/June	Outer and inner area of Xiangshan Gang	*Prorocentrum* sp.; Prohibited sale of seafood from affected areas.

Overall ASSETS score: The combination of moderate high influencing factors, high eutrophic conditions and improve low future outlook results into a bad eutrophication assessment score for the Xiangshan Gang condition in 2005-2006.

Assessment of simulated scenarios

The nutrient load decrease estimated in the three scenarios did not suffice to change the IF classification, which remained moderate high and high, for N and P, respectively. These outputs indicate that in all the scenarios there are still management options for reducing a substantial part of the nutrient loading into the bay.

Table 4.5 synthesises the EC calculation for the data and model simulations, with highlight for the chl-a symptom. The chl-a maximums observed and simulated for the mixing and seawater zones led to the same classification for the data-based and standard model applications of ASSETS model: high and moderate level of expression of chl-a symptom for the mixing and seawater zones, respectively. Thus, when comparing the ASSETS application to the data and the standard simulation the same results are obtained for the primary symptoms, the EC and overall ASSETS score (Table 4.5). For the application of the ASSETS to the multilayered ecosystem model outputs is assumed that all the symptoms besides chl-a remain constant, for the following reasons: given that no problems are observed for the macroalgae and DO symptoms for the data-based application, it is unlikely that in scenario simulation where nutrient pressure is reduced these symptoms would increase. For the SAV

symptom the same assumptions are applied as for the data-based ASSETS application. Therefore, the magnitude of SAV change classifies as very low.

Table 4.5. Synthesis of ASSETS application to model outputs and comparison with data-based application.

			Data	Standard	Scenario1	Scenario2	Scenario3
Chl-a	P90	Mixing	15.5	11.1	9.6	8.8	7.4
		Seawater	7.5	7.2	6.3	5.9	5.1
	Spatial coverage	Mixing	High	High	High	High	High
		Seawater	Moderate	Moderate	Low	Low	Low
	Level of expression	Mixing	High (1)	High (1)	High (1)	High (1)	High (1)
		Seawater	Moderate (0.5)	Moderate (0.5)	Low (0.25)	Low (0.25)	Low (0.25)
		Bay	High (0.62)	High (0.62)	Moderate (0.42)	Moderate (0.42)	Moderate (0.42)
Primary symptoms			Moderate (0.31)	Moderate (0.31)	Low (0.21)	Low (0.21)	Low (0.21)
Eutrophic condition (EC)			High	High	Moderate high	Moderate high	Moderate high
Overall ASSETS score			Bad	Bad	Poor	Poor	Poor

P90, 90th percentile; N.A., not applicable.

The nutrient load reduction decreases the probability of HAB events with origin from inside the bay. However, given that this effect cannot be deterministically predicted and that HAB events occur in Xiangshan Gang since at least 1992 (Chen et al., 1992) the precautionary approach is adopted and is assumed that no changes occur in HAB events. A high classification for this symptom based on the past events in Table 4.4 is therefore considered.

The main conclusions of the ASSETS outputs for each scenario are described next. A reduction of seawater zone boxes with chl-a maximum above the 5.0 μg L^{-1} threshold occurred in all scenarios: from 4 boxes (out of a total of 9 boxes) reduced to 2 boxes in scenarios 1 and 2, and to 1 box in scenario 3. Therefore, the spatial coverage of the chl-a symptom reduced to low, which resulted into a low chl-a level of expression according to the ASSETS decision rules (Bricker et al., 2003). In all the boxes of the mixing zone the chl-a maximum falls within the range for medium eutrophic conditions for all scenarios, thus for this zone a high level of expression of chl-a is estimated (Table 4.5). Nevertheless, in scenario 3 the chl-a 90th percentile in 2 boxes, out of the 3 boxes of the mixing zone, is close to the 5.0 μg L^{-1} threshold, which is a sign of improvement. The combination of the scores obtained for the mixing and seawater zones results into the reduction from high to moderate chl-a level of

expression for the Xiangshan Gang for all the scenarios. Therefore, the primary symptoms, which are calculated by combining the chl-a and macroalgae symptoms reduced from moderate to low (Table 4.5). Cascading effects on the EC and overall ASSETS score are obtained (Table 4.5). It is foreseen improvements of the overall euthrophic state due to implementation of any of the scenarios. Nevertheless, that is still a poor score as shown in Table 4.5.

Integrated ecological-economic assessment

Drivers

Table 4.6 synthesises the Drivers quantification. The reduction of fish production in scenarios 1 and 3 corresponds to the measures being simulated, i.e. 38% reduction of total production. The simulated shellfish production decreases in all scenarios as a result of the substance loading reduction with the overall aim to improve water quality, as explained in *Chapter 2*; therefore, the net profit for aquaculture production decreases in all scenarios. Exception is for the razor clam, because according to the data survey its production represents losses for the farmers. As such, a reduction in its production results in economic gains.

Table 4.6. Drivers quantification: fish and shellfish aquaculture production (tonnage and net profit).

Aquaculture production		Finfish	Shellfish			
			Oyster	Razor	Manila	Muddy
Net profit (Yuan kg^{-1})		5.9	1.3	-3.1	2.8	4.2
Standard	Ton	9 370	41 834	1 709	622	784
	10^6 Yuan	55.3	54.4	-5.3	1.7	3.3
Shift to scenario1	Ton	-3 561	-4 205	-168	-56	-161
	10^6 Yuan	-21	-5.5	0.5	-0.2	-0.7
Shift to scenario2	Ton	0	-7 977	-348	-100	-285
	10^6 Yuan	0	-10.4	1.1	-0.3	-1.2
Shift to scenario3	Ton	-3 561	-11 618	-523	-157	-402
	10^6 Yuan	-21.0	-15.1	1.6	-0.4	-1.7

Pressures

The load reduction from standard simulation to each scenario is synthesised in Figure 4.4a. At the management level the catchment-aquaculture sources contribution is classified as moderate high and high for N and P, respectively, according to the IF index of the ASSETS model (Figure 4.4b). This means that despite the nutrient load reduction simulated in the three scenarios there are still other control measures that can be adopted from the catchment-bay perspective in order to reduce the nutrient concentration in the bay.
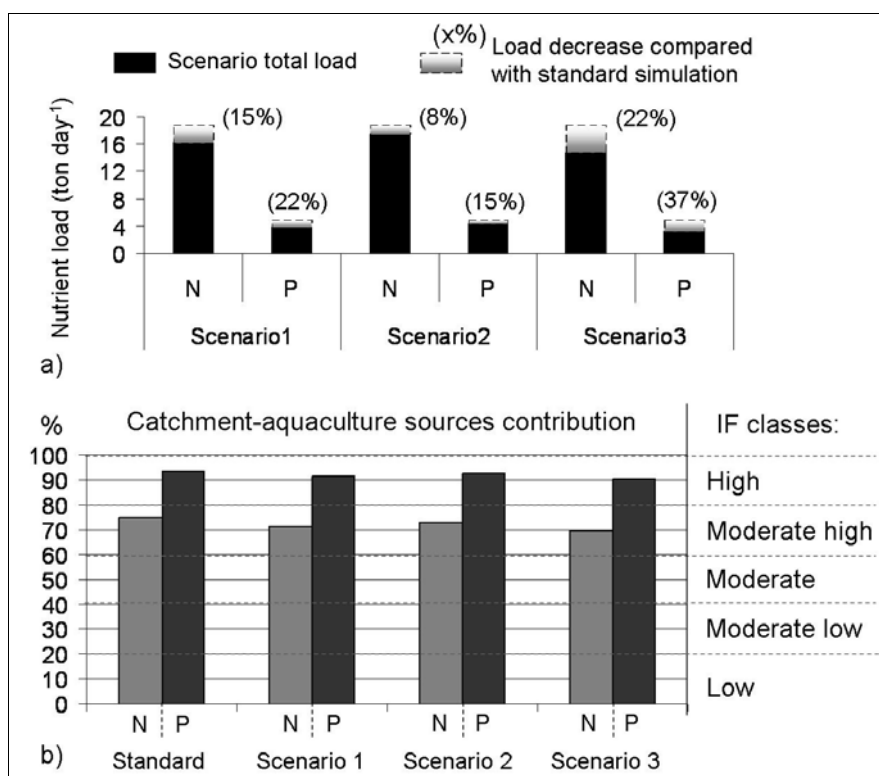
Figure 4.4. Pressure change: a) nutrient load (research level); b) catchment-aquaculture sources contribution using IF index of the ASSETS model (management level).

State and Impact

The standard model results for average nutrient concentration range from 34.4 μmol L^{-1} for DIN and 1.0 μmol L^{-1} for phosphate in the box that connects with the sea boundary, to 48.4 μmol L^{-1} for DIN and 2.2 μmol L^{-1} for phosphate in one of the inner boxes. The water quality in Xiangshan Gang on average is classified above the limit of Class IV (meaning a poor quality) according to the Chinese seawater quality standards for nutrient concentration parameter. Calculations based on the sampled water quality data also confirm this result. The 90th percentile value of phytoplankton biomass (which is used as a proxy of the maximum values) calculated from model results ranges from 4.4 μg chl-a L^{-1} in Box 11 to 11.1 μg chl-a L^{-1} in Box 3. Considering the results of the ASSETS model application the chlorophyll-a symptom expressed as high in the inner boxes and as moderate in the middle and outer boxes with higher seawater renewal. The shellfish productivity, given as the weight of harvest obtained per weight of seeding varies significantly among species as shown in Figure 4.5. The Chinese oyster is the cultivated species that produces by far the largest harvest per seeding effort, on average per kg of seed produces 20 kg of oyster. The lowest productivity is for cultivation of Manila clam, which generates per kg of seed less than 5 kg harvest output. In general, all species exhibit a marked variability of productivity among the inner and outer

boxes (Figure 4.5). The Chinese oyster productivity, for instance, varies from 14, in Box 1, to 40, in Box 12. An exception is for instance the Manila clam. The highest productivity occurs in Box 3 (ca. 6), which is even slightly higher than in Box 10.
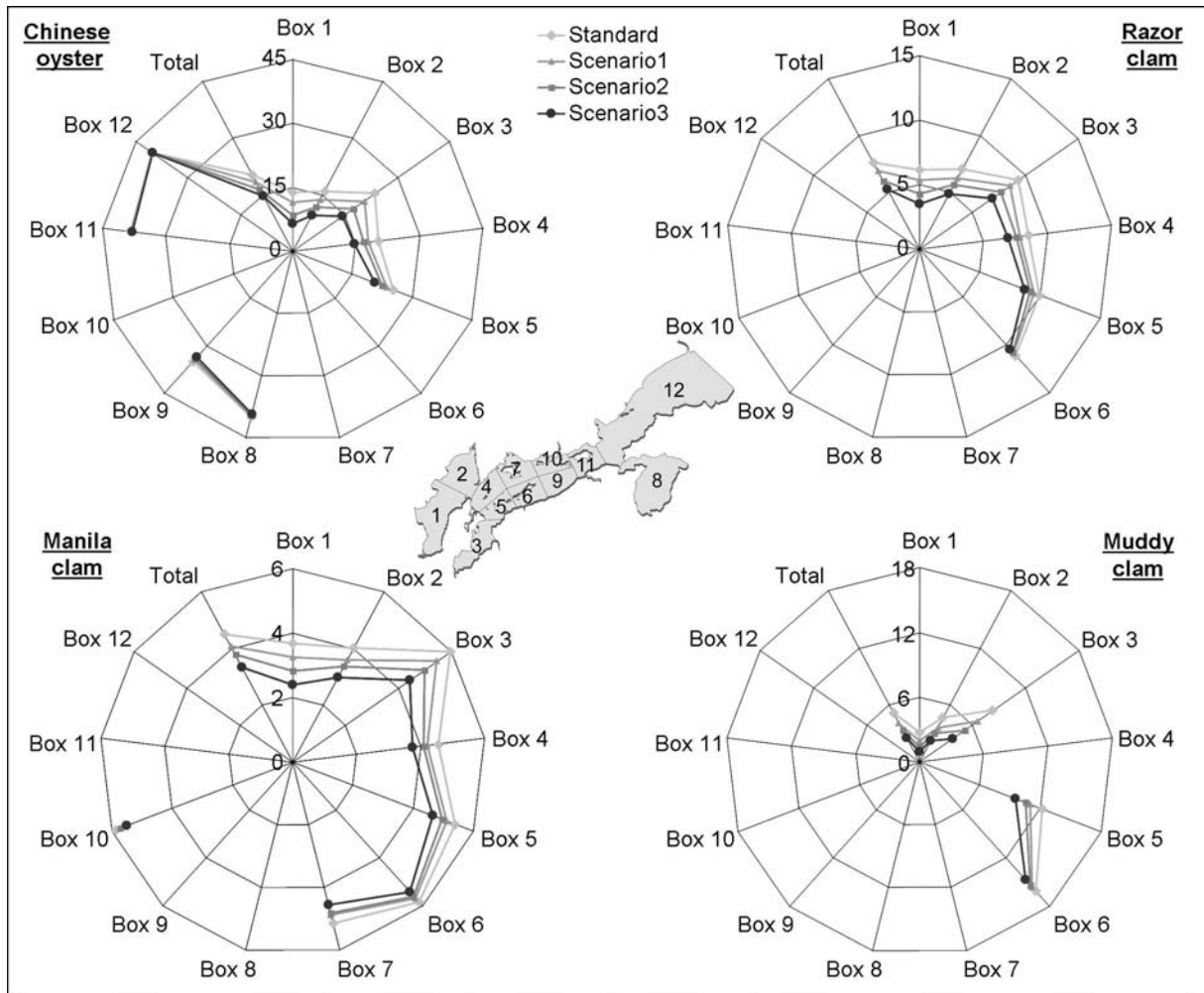
Figure 4.5. Shellfish productivity per box expressed as the average physical product (APP: ratio of total weight of shellfish harvested to total weight of seeding), for Chinese oyster, razor clam, Manila clam and muddy clam.

The change of the ecosystem State predicted in each scenario when compared with the standard simulation corresponds to the ecological Impact and is detailed herein. The State classification per box is shown in Figure 4.6 for the standard simulation and each scenario, thus providing an image of the State evolution regarding the nutrients and chl-a criteria. The simulated actions had a limited Impact on the State classification regarding nutrient concentration in the bay. No changes are estimated for DIN. For phosphate there are improvements with implementation of scenario 3 in boxes 6 and 12, which shifted into Class IV and into Class II/III, respectively, when compared with standard simulation (Figure 4.6). Nevertheless, and more important than to estimate the Impact of nutrient load reduction in the bay nutrient concentration is to examine the eutrophication symptoms (Bricker et al., 2003).

As regards chl-a concentration, the model estimates improvements for boxes 6 and 7 in all scenarios, corresponding to the shift from medium to low phytoplankton levels (Figure 4.6).

	DIN	Phosphate	Chl-a
Legend:	Classification according to Chinese seawater quality standards:		ASSETS classification for chl-a concentration:
	Class IV	Class II/III	Low
	Above Class IV	Class IV	Medium (Eutrophic condition)
		Above Class IV	
Standard			
Scenario 1	No change	No change	
Scenario 2	No change	No change	Same as scenario 1
Scenario 3	No change		
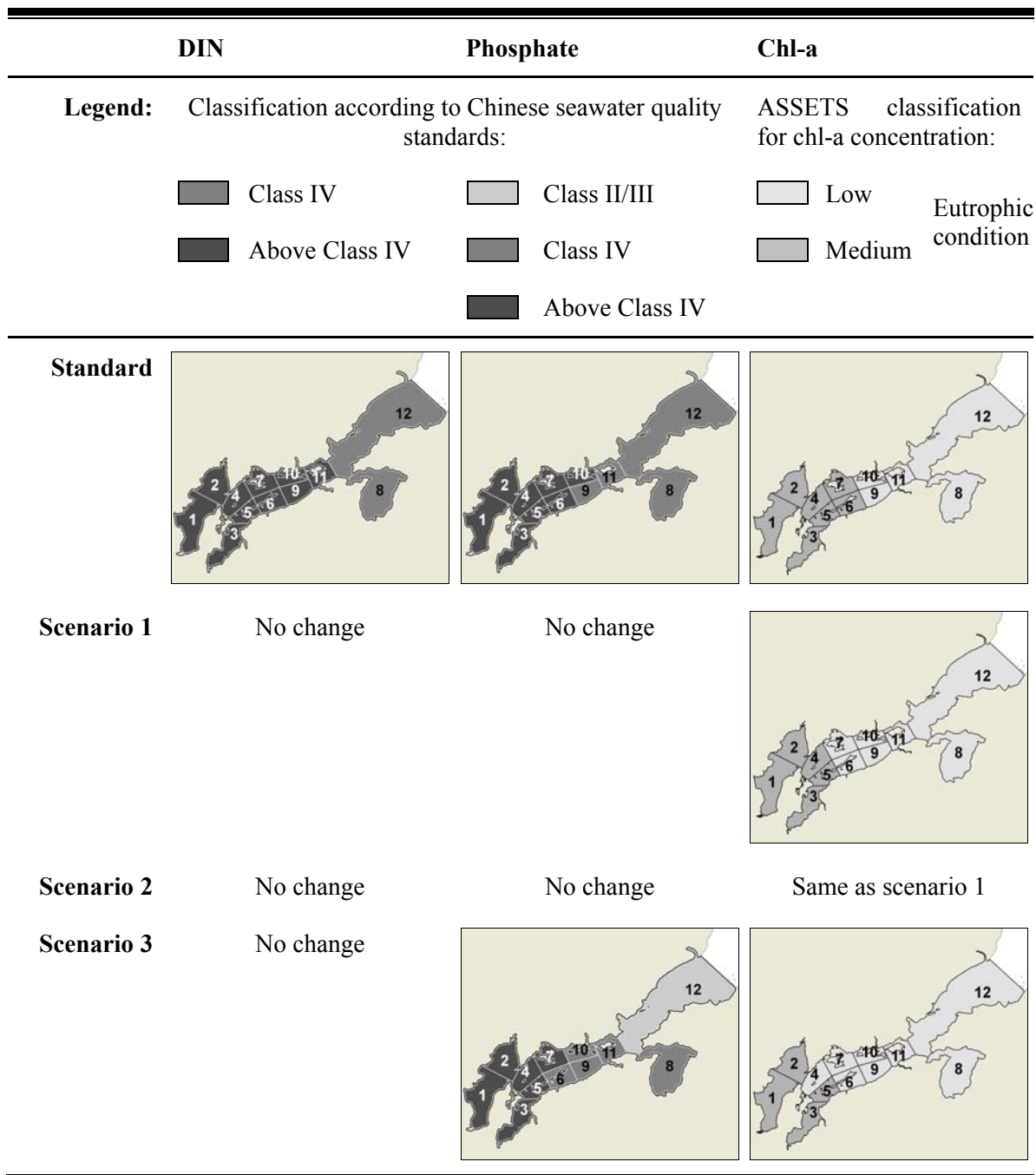
Figure 4.6. Ecosystem State classification of nutrients and chl-a per box for standard simulation and indication of changes as simulated in each scenario.

In Box 4 the same improvement is estimated but only with implementation of scenario 3 (Figure 4.6). It is important to note that in scenario 3, in three out of the four boxes which still exhibit medium chl-a level, the estimated maximum chl-a concentration are very close to the threshold that divide the low from the medium class (5.0 μg L^{-1}): 5.3 μg L^{-1} in Box 1 and 5.1 μg L^{-1} in boxes 2 and 5. The changes estimated for the chl-a concentration lead to changes of the EC index of the ASSETS model, as detailed in the results section for the ASSETS

application to the model outputs. For any of the scenarios the eutrophication condition for Xiangshan Gang is estimated to shift from high to moderate high (Table 4.5). Although this still corresponds to a poor ASSETS score (Table 4.5) it shows improvements. The model estimates a decrease of shellfish productivity with implementation of any of the scenarios (Figure 4.5). This is mainly a consequence of the reduction of substance loading into the bay in order to improve its water quality, which corresponds to the drivers for the shellfish growth (as indicated in *Chapter 2*). As show in Figure 4.5, this effect is in general more significant in the inner boxes (boxes 1 to 3), which are the ones with higher cultivated area, and particularly for oysters and razor clams.

The associated economic Impacts are assessed based on the changes in the aquaculture net profit and the value of environmental externalities related with nutrient loads. The Drivers section details the changes in the aquaculture net profit (Table 4.6). The *VExternalities* (Table 4.7) is calculated as the: (i) avoided cost to treat the nutrient load reduction due to fish cage reduction in scenario 1 (ca. 11 347 million Yuan per year); (ii) avoided wastewater treatment costs due to reduction of wastewater in scenario 2 (ca. 5 945 million Yuan per year); and (iii) the avoided cost due to an equivalent wastewater reduction of 124 million m3 per year in scenario 3 (ca. 17 322 million Yuan per year).

Table 4.7. Value of environmental externalities: avoided costs due to fish cage reduction (in scenarios 1 and 3) and WWTP costs (in scenarios 2 and 3).

	Scenario 1 Fish cage reduction	Scenario 2 Wastewater treatment	Scenario 3 Scn1 + Scn2
Equivalent wastewater reduced million m^3 year^{-1}	81	43	124
Operational cost million Yuan year^{-1}	57	30	87
Investment cost			
million Yuan	338 696	178 359	517 055
million Yuan year^{-1} *	11 290	5 945	17 235
Total avoided cost million Yuan year^{-1}	11 347	5 975	17 322

* Considering 30-year depreciation for the WWTP (wastewater treatment plant).

Table 4.8 synthesizes the estimates of the economic Impacts of the implementation of each scenario. The changes of the value of the drivers that depend on the ecosystem are negative for all scenarios due to the aquaculture production decrease. Given that higher positive

environmental externalities are estimated, then a positive economic Impact is calculated for all the scenarios.

Table 4.8. Economic impacts of the shift from the standard simulation to each scenario.

		Standard simulation to		
		Scn 1	Scn 2	Scn 3
Changes of aquaculture net profit (million Yuan year^{-1})	Shellfish	*-6*	*-11*	*-16*
	Finfish	*-21*	*0*	*-21*
$\Delta VDrivers_{Ecosystem}$ (million Yuan year^{-1})		-27	-11	-37
Avoided costs for nutrient treatment **VExternalities** (million Yuan year^{-1})		11 347	5 975	17 322
VImpact (million Yuan year^{-1})		**11 320**	**5 965**	**17 285**

The current analysis presents only the predictable Impacts of substance load reduction. Other effects are left out such as HAB events due to general knowledge limitations as explained previously. Nuisance and toxic algal blooms can have a significant impact on the local economy. For instance as early as in 1992 Chen et al., (1992) indicates that a red tide in Xiangshan Bay posed great harm on cultured macroalgae and shellfish. Later in June 2005 a large scale red tide in Zhejiang coastal area resulted in economic losses of about 20 million Yuan that also affected Xiangshan Gang (SOA, 2006). On the other hand, the effective monitoring of red tides and emergency response plan reduced the economic loss of millions of Yuans (Ye and Huang, 2003), estimated as about 150 million Yuan near the maritime space of Zhejiang province in (Cai, 2001).

Response

The response cost corresponds in scenario 1 to the decrease of fish production, estimated in the Drivers section as 21 million Yuan per year. In scenario 2 the response cost is given as the implementation and operation cost of the wastewater treatment plant built for Xiangshan Gang population, which is estimated as about 5 975 million Yuan per year (as detailed for calculation of the *VExternalities*). The response cost for scenario 3 corresponds to the sum of actions adopted in scenarios 1 and 2 and is estimated as about 5 996 million Yuan per year.

Overview of the integrated environmental-economic assessment

The ecological and economic ΔDPSIR analysis about simulated scenarios related with aquaculture and nutrient management in Xiangshan Gang is synthesised in Table 4.9. A decrease of aquaculture net profit is estimated in all scenarios (Table 4.6 and Table 4.9).

Firstly, due to the fish cage reduction simulated in scenarios 1 and 3. Secondly, because the imposed reduction of substance loads causes a decrease of shellfish productivity (Figure 4.6). The fish cage reduction and WWTP implementation scenarios decreased the Pressures of nutrients in Xiangshan Gang (Figure 4.4 and Table 4.9). The corresponding ecological Impacts are analysed regarding different criteria. The nutrient concentration inside the bay improved slightly according to Chinese seawater quality standards only for P; for N no changes are estimated. The ASSETS application to the ecosystem model outputs indicates an improvement of the chl-a level of expression that results for any scenario on the improvement of the eutrophic condition from high to moderate high. The corresponding value of the environmental benefits is estimated based on the avoided costs due to the reduction of nutrients from fish cages and of population wastewater into bay (Table 4.7).

Table 4.9. Synthesis of the ecological and economic variables of the differential DPSIR analysis for the shift from standard simulation to each scenario.

			Standard simulation to		
			Scn1	Scn2	Scn3
ΔDrivers	Aquaculture net profit ($VDrivers_{Ecosyst}$)	10^6 Yuan year^{-1}	-27	-11	-37
ΔPressures	Nutrient load	Ton N year^{-1}	-999	-1 316	-2 314
		Ton P year^{-1}	-402	-439	-842
ΔState = Impact	Nutrient classification	No. of boxes that changed N class	No changes[*1]		
		No. of boxes that changed P class	No changes[*2]		Improved[*3]
	ASSETS classification	Chl-a level of expression	High to moderate		
		Overall Eutrophic Condition (EC)	High to moderate high		
	Shellfish productivity	% change	-12 %	-23 %	-34 %
	Economic impact (*VImpact*)	10^6 Yuan year^{-1}	11 320	5 965	17 285
Response	Response cost	10^6 Yuan year^{-1}	21	5 975	5 996
Overall	***VManagement***	10^6 Yuan year^{-1}	11 299	-11	11 289
	Shellfish aquaculture net profit	10^6 Yuan year^{-1}	-6	-11	-16
	Fish cage net profit	10^6 Yuan year^{-1}	-21	0	-21
	WWTP cost	10^6 Yuan year^{-1}	0	-5 975	-5 975
	VExternalities	10^6 Yuan year^{-1}	11 347	5 975	17 322

[*1] 10 out of 12 boxes above class IV and 2 in class IV; [*2] 8 out of 12 boxes above class IV and 4 in class 4; [*3] shift of 2 boxes into class IV and 1 box into class II/III.

The value of environmental externalities surpasses the negative change of the drivers that depend on the ecosystem (Table 4.8); therefore a positive economic Impact is estimated (Table 4.9). The Response cost for the planned action in scenario 1 is 2 orders of magnitude lower than for scenarios 2 and 3. These two scenarios account for the implementation and operation of a WWTP (estimated as about 5 975 million Yuan per year), while in scenario 1 the response cost corresponds to the closure of 38% of finfish aquaculture net profit (estimated as about 21 million Yuan per year). The overall economic balance (*VManagement*) estimates a similar gain for scenarios 1 and 3 and a loss for scenario 2 (Table 4.9). The *VManagement* represents the outcome of the balance between (i) the aquaculture net profit decrease (due to the fish aquaculture reduction and decrease of shellfish productivity), (ii) the response costs (including only the WWTP costs to avoid the double-accounting of the fish aquaculture net profit decrease), and (iii) the quantifiable environmental externalities. In scenario 2 the *VExternalities* cancel out the Response cost leading to a negative overall balance. Therefore, the negative balance corresponds to the reduction of shellfish productivity. In scenarios 1 and 3, the positive balance is mainly due to the avoided costs for treatment of fish effluents.

Herein, and based on the integrated modelling and assessment approach recommendations about improving water quality and minimize aquaculture production decrease are presented. To start with, in a follow up work and based on the current set of models the modelling teams could provide estimates about the reduction targets required to reach the aimed water quality condition. Future actions to improve water quality in Xiangshan Gang should include extended stakeholder meetings to define further nutrient reduction measures; namely, related with agriculture practice, which are not explored in the current modelling exercise. The catchment model of the multilayered modelling system (detailed in *Chapter 2*) could assist the determination of the most effective measures. Alternatively to current options about decreasing fish cages it might be interesting to make a cost-benefit analysis to evaluate other fish cultivation practices; namely, in integrated multi-trophic aquaculture (IMTA) systems in land-based ponds (as discussed in the second part of this chapter for an abalone-seaweed IMTA) or fish cages with a floating bag system (Ayer and Tyedmers, 2009). A benefit of the land ponds is that it allows disconnection from the bay during HAB events. An intermediate measure might be to re-establish massive kelp or other seaweeds cultivation, especially near fish cage areas, in order to reduce loading of dissolved nutrients into the bay. Until late 1980's kelp was the major aquatic resource produced in the bay. The seaweed replacement by other aquatic resources and in particular the fish cultivation boom in the second half of the 1990's is believed to be related with several of the HAB events (Feng et al, 2004). Feng et al. (2004)

illustrates the ecological and economic advantages of restoring the vast kelp cultivation. To minimize the expected effects of decrease of substance loads into the decline of shellfish production, the outputs of the approach suggest the displacement of part of the shellfish production in order to distribute cultivation areas more evenly over the bay. For instance 89% of the total shellfish cultivation area is located in boxes 1 to 5, which correspond to only about 34% of the total Xiangshan area. In general, the shift of part of the shellfish culture from the up- to downstream area of the bay is advisable. For instance, Chinese oyster productivity is almost 3 times higher in Box 12 than in Box 1. As such, a fraction of Box 1 oyster production (which concentrates 22 % of total oyster production) should be distributed over other boxes. This action might require wider zoning efforts for the bay in order to optimize and harmonize space allocation between several coastal uses.

If further management actions for water quality improvement are to be adopted, the wider side effects related with the water quality improvement must be accounted in the economic analysis; such as the development of other coastal uses such as tourism and recreational fisheries, which can diversify the source of income in the region.

The monitoring of HAB events, in particular for determining the origin (if from inside the bay or from outside) and the triggering mechanisms is recommended for the management of the Xiangshan Gang eutrophic condition. Babaran et al. (1998) exemplify how research about initiating and triggering mechanisms that cause HAB's can be applied for managing aquaculture sites subject to these events. Roelke and Buyukates (2001) provide example about establishing preventive management schemes based on an early-warning indicator monitoring. Additionally, for future cost-benefit analysis the appraisal of detailed economic impacts of HAB events on aquaculture production is recommended.

CONCLUSIONS

The synthesis of model outputs using IEA methodologies provides useful insights for managers about what is expectable to change in water quality and ecosystem state as a result of simulation scenarios. In particular the use of the ΔDPSIR enabled an estimate of the ecological-economic impacts of the tested management solutions. The comparison among scenario outputs provided insights for the adoption of future policy and research. The use of ΔDPSIR is enhanced by using together other IEA approaches, more targeted that can provide classification of ecosystem status regarding specific problems. In this case study, the ASSETS screening model played a crucial role to understand the effects of changes of nutrient loading

(from both catchment and aquaculture activities) in the eutrophic condition of bay. There are nevertheless limitations inherent to any modelling exercise given the incomplete representation of reality. For instance, relevant variables for the environmental assessment might be left out given the complexity and current knowledge of processes to be simulated, such as HAB events. In such cases, if possible is recommended to fill modelling gaps based on expert knowledge, rather than overlook the effect (Peirce, 1998). Notwithstanding, the scenario prediction besides providing insights to managers concerning the variables simulated in the ecosystem model, can also assist the planning of the impact assessment evaluations of future coastal management actions. Namely, to identify the relevant variables/indicators for the characterisation and analysis of the: (i) catchment and coastal activities, (ii) relevant pressures, and (iii) ecological features to monitor.

The design of the integrated modelling and assessment approach for a specific case study must be tailored to address the needs of managers and other stakeholders of the ecosystem. This is an essential step to ensure that the relevant local issues are included in the modelling and thus that the overall approach is useful. Actions adopted by managers after the application of an integrated modelling and assessment approach, should be followed by monitoring to: (i) assess the consequent impacts; (ii) verify the modelling predictions; and (iii) contribute to knowledge specially to fulfil modelling gaps.

The case study developed in this chapter lays the groundwork for more complex applications of the integrated environmental modelling and assessment approach elsewhere. The subject analysed herein is highly relevant for the integrated management of coastal zones given the existing challenges to promote sustainable aquaculture development and the management of nutrient loading from coastal activities. In particular, the case study illustrated the usefulness of the integrated environmental modelling and assessment approach to assist the development of an ecosystem approach to aquaculture.

4.2 Farm level assessment: IMTA evaluation using real farm data

Context

The previous part of this chapter analysed the multilayered ecosystem model scenarios to support the development of EAA at the waterbody/catchment level. The analysis of aquaculture at the individual farm level is also important for the development of EAA. Aquaculture has been mostly associated with negative impacts, mostly due to unsustainable cultivation practices (Paez-Osuna et al., 1999; Feng et al., 2004; Xu et al., 2007). However, depending on the aquaculture practice and cultivated species, aquaculture can also generate environmental benefits (Neori et al., 2004; Newell, 2004; Žydelis et al., 2008; Gren et al., 2009). The adoption of best management practices (BMPs) for new and existing farms is important to minimize the potential negative impacts on water quality and ecosystem deterioration, while potentially leading to increasing profit margins, as exemplified by Valderrama and Engle (2002) for shrimp aquaculture.

Summary

This chapter illustrates the application of the ΔDPSIR for the ecological-economic assessment of aquaculture options at the farm level. An abalone farm located in South Africa is used to exploit the detailed dataset about its environmental and economic performance. The case study consists of assessing the ecological-economic effects of the abalone-seaweed IMTA on the farm's performance and the corresponding environmental externalities.

This section corresponds to the manuscript submitted to Aquaculture:

Ecological-economic assessment of aquaculture options: comparison between monoculture and integrated multi-trophic aquaculture, by Nobre, A.M., Robertson-Andersson, D., Neori, A., Sankar, K.

(For consistency with submitted version this chapter is written in American English)

Ecological-economic assessment of aquaculture options: comparison between monoculture and integrated multi-trophic aquaculture

INTRODUCTION

Aquaculture has grown at an average annual rate of 8.8% since 1970, with an increase in the production of seafood (excluding plants) of about 8 fold up to 2004 (FAO, 2006). Sustainability issues related to socially and environmentally irresponsible aquaculture practices reported for certain cultivation systems have generated concerns about the industry, particularly highly industrialized and intensified monoculture farms (Paez-Osuna et al., 1999; GESAMP, 2001; Islam, 2005; Xu et al., 2007; Allsopp et al., 2008). Because of their impact on the environment and of their negative feedbacks on the aquaculture operations, the expansion of aquaculture has been limited (GESAMP, 2001; Islam, 2005; Gibbs, 2009). The broader public is generally unaware of the benefits that aquaculture can generate to the environment (Newell, 2004; Lindahl et al., 2005; Ferreira et al., 2007a; Rice, 2008; Žydelis et al., 2008) and to society (promotion of poverty reduction through employment, higher income and food security – FAO, 2005; Msuya, 2006; Troell et al., 2006; Kaliba et al., 2007; Robertson-Andersson et al., 2008). Given the importance of food security on the one hand (Ahmed and Lorica, 2002), and given the negative ecological-economic impacts of poorly conceived aquaculture practices on the other hand (Islam, 2005), an integrated planning and management of aquaculture is required (GESAMP, 2001). Furthermore, external benefits of socially and environmentally responsible (sustainable) aquaculture can have direct economic value, since consumers have been showing increased awareness of, and preference for, sustainable seafood harvesting (FAO, 2006). The main technological approaches that have been developed to meet environmental concerns (Refstie et al., 2001; Neori et al., 2004; Gutierrez-Wing and Malone, 2006) include: (i) improved feed and water management, (ii) water recirculating systems, (iii) bacterial biofilters and (iv) extractive species (filter feeders, detritivores and macroalgae).

More recently, the integration of fed species and extractive species in the modern form of polyculture called integrated multi-trophic aquaculture (IMTA, also known as 'partitioned aquaculture' and 'aquaponics'), has been developed to ease environmental concerns because it addresses issues of both productivity and nutrient loading into the environment (Neori et al., 2004; FAO, 2006; WGEIM, 2006; Abreu et al., 2009; Buschmann et al., 2009; Troell et al., 2009). IMTA has been gaining recognition as a sustainable approach to aquaculture because

of its combination of environmental, economic and social advantages (Whitmarsh et al., 2006; Ridler et al., 2007; Allsopp et al., 2008). A key component of IMTA is the use of macroalgae: while taking up dissolved inorganic nutrients (nitrogen and phosphorus), the produced algal biomass is a renewable protein-enriched feed to other cultivated species, and a product on its own (Chopin et al., 2001). Abalone farming is an aquaculture industry that can particularly benefit from the implementation of IMTA with marine macroalgae (seaweeds), which are the natural abalone food. South Africa, the third largest abalone producer in the world (Gordon and Cook, 2004), has begun implementing IMTA with the seaweed *Ulva lactuca* L. and the abalone *Haliotis midae* L. (Robertson-Andersson et al., 2008). This move has largely emerged for the following reasons:

(i) Demand for natural stocks of South African kelp as feed for abalone is approaching the maximum sustainable yield of the concession areas (Troell et al., 2006) and insufficient access by some farms to wild kelp beds (Bolton, 2006; Smit et al., 2007; Hwang et al., 2009).

(ii) Diets of mixed algal species accelerate abalone growth rates relative to single-species diets (Naidoo et al., 2006; Dlaza et al., 2008).

(iii) Cultivation of seaweeds in the farm's abalone effluent allows water recirculation and reduces nutrient discharge into the environment (Robertson-Andersson, 2007).

(iv) A land based seaweed facility allows the abalone farm to disconnect itself from the sea for extended periods by water recirculation through seaweed ponds during red tides and oil spills (Robertson-Andersson, 2007).

Aquaculture, like other uses of marine resources where the environmental and the socio-economic systems are intertwined, require for its sustainable development information about the ecological and economic impacts of different practices. This implies communication between the scientific, management and policy-making communities, and the integration among disciplines using mutually understandable concepts (GESAMP, 2001). The Drivers-Pressure-State-Impact-Response (DPSIR) approach is a potential analytical framework for determining the impacts of aquaculture options. This approach has been applied to assist in the evaluation of environmental impacts and of ecosystem management approaches (Stanners et al., 2008). In particular, the DPSIR has been widely used to report about quantification of the impacts of human activities on coastal activities (Borja et al., 2006; Elliott, 2002; IMPRESS 2003; Nobre, 2009). The DPSIR is a conceptual framework for integrated environmental assessment that provides (i) a systematic view of the socio-economic and environmental interactions and (ii) a reporting framework to policy-makers and public (Bowen and Riley, 2003; Ledoux and Turner, 2002; Nobre, 2009). The application of the

DPSIR is based on the use of indicators (Stanners et al., 2008). It facilitates the structuring of data following the causal chain D-P-S-I-R: Drivers are the anthropogenic activities generating Pressures that perturb the State of the ecosystem, thus causing an Impact on the ecosystem, which calls for management and policy-making Responses to improve the State of the ecosystem (Borja et al., 2006; IMPRESS 2003). A recent version of the DPSIR, the Differential DPSIR (ΔDPSIR), establishes an explicit link between the ecological and the economic systems and screens the evolution of ecological and economic variables over time or between simulated scenarios (Nobre, 2009). The ΔDPSIR approach provides a tool for the assessment of changes in environmental quality and consequent effects on the economic system, including on the value of anthropogenic activities and of the ecosystem (Nobre, 2009).

The aim of the work presented herein is to couple ecological and economic information to support resource managers in the assessment of the ecological and economic impacts of aquaculture operations. This paper uses the integration of seaweed production in the abalone industry in the form of IMTA as a case study, and the ΔDPSIR framework (Nobre, 2009) as an approach to evaluate the ecological and economic impacts. The objectives are to:

(i) Assess the environmental and economic impacts to the main stakeholders of the shift from abalone monoculture to IMTA with seaweeds using data from a farm located in South Africa (Roberston-Andersson, 2007; Roberston-Andersson et al., 2008; Sankar, 2009).

(ii) Carry out a mass balance analysis to manage nutrient limitation due to seaweed expansion in the South African farm. Includes data analyzes from an Israeli IMTA farm with abalone, fish and seaweeds (Neori and Shpigel, 2006) to provide guidance on possible solutions for the sustainable management of the nutrient limitation that occurs when expanding the seaweed production.

METHODOLOGY

General approach

The ΔDPSIR methodology (Nobre, 2009) is applied to evaluate aquaculture options. The ΔDPSIR includes quantification of ecological and economic variables. The ecological assessment includes quantification of indicators of Pressure, State, and Impact. The economic assessment consists in a cost-benefit analysis to evaluate a given Response from an environmental and economic perspective; includes quantification of the value of the divers, of the ecosystem, of the impact, of the response and the economic value of management (Nobre,

2009). The ecological and economic variables are used to quantify the Drivers, Pressures and ecosystem State in two or more time snapshots (or scenarios); these values are then used to calculate (or predict) the relevant overall Impacts that result of the management Response over the time interval (or between two scenarios).

The ΔDPSIR components are defined as follows (Nobre, 2009):

 (i) Drivers - the anthropogenic activities that may have an environmental effect at a given moment in time; it is a socio-economic component of the ΔDPSIR.

(ii) Pressures – direct positive and negative (e.g., biofiltration or sewage effluents, respectively) influence of the Drivers on the environment.

(iii) State - the condition of the ecosystem at a given moment in time. It has both ecological and economic dimensions and is influenced by both anthropogenic Pressures and natural factors. The ecological dimension of State can include water quality and habitat biodiversity quantified using existing classification tools such as ASSETS eutrophication model (Bricker et al., 2003) and benthic diversity index (Pinto et al., 2009). The economic dimension can be provided by the values of environmental goods and services, quantified by the total economic value (TEV) of an ecosystem (Turner et al., 2003). Calculation of an objective and complete TEV, however, is a complex exercise with limitations (Chee, 2004; Emerton and Bos, 2004; Kumar and Kumar, 2008).

(iv) Impact - the environmental effect of the Pressures, i.e. changes in the State of the ecosystem between two points in time or between two scenarios. An environmental Impact can be either positive (e.g. restoration of a habitat) or negative (e.g., eutrophication). The associated economic Impact includes direct gains/losses (e.g., related to tourism, transportation and fisheries) as well as indirect gains/losses of non-use value of ecosystems (e.g., related to value of mangroves in reproduction of marine animals). In the ΔDPSIR framework the value of the economic Impacts is determined by one of two possible approaches (Nobre, 2009): i) if the TEV was calculated in the State component of the analysis, the economic Impact is given as the difference in TEV between two points in time or between two scenarios; otherwise (ii) where TEV is not computed, the economic Impact can be calculated based on changes in the profit of the Drivers that depend on changes in the State of the ecosystem and on the value of environmental externalities, which can be calculated based on replacement, restoration and avoided costs (Emerton and Bos 2004; Ledoux and Turner, 2002) associated with the quantified environmental Impacts.

(v) Response - management actions and policies such as i) measures taken to improve the State of the water body (a new wastewater treatment plant), ii) a waste discharge permit that increases pollution of a receiving water body or, iii) change of aquaculture practices that can improve the State of the receiving coastal waters. The economic dimension of Response is quantified by calculating the cost of the measures and actions identified.

Figure 4.7 schematizes the DPSIR application to evaluate aquaculture options.
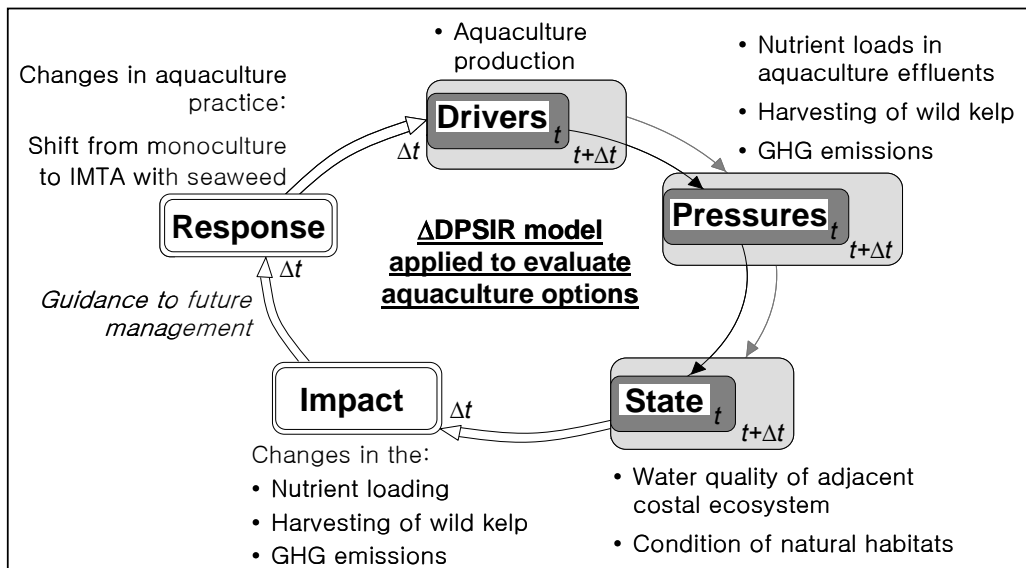
Figure 4.7. Application of the differential DPSIR to evaluate the seaweed role in IMTA.

Case study site and data

This study was conducted with data from an abalone farm located in the Western Cape, South Africa, the Irvine and Johnston (I & J), Cape Cultured Abalone Pty, Ltd. The farm started operating in 1994 a flow-through abalone monoculture using seawater pumped from the sea. In 2007 this farm installed pilot scale seaweed culture ponds, through which effluent from the abalone culture facility was recirculated. Seaweed from these ponds supplied 10% of the abalone seaweed requirements. An expansion of the seaweed ponds planned for 2009 will supply 30% of the abalone seaweed requirements. Data on the performance of this farm were taken from Roberston-Andersson (2007), Roberston-Andersson et al. (2008) and Sankar (2009). The I & J farm was analyzed according to the three operation settings: the flow-through abalone monoculture (setting 1); and the two scales of an integrated abalone/seaweed recirculating system (settings 2 and 3) (Table 4.10). In both settings 2 and 3, half of the abalone production was still cultivated using a monoculture flow-through system (Table 4.10).

Table 4.10. Settings adopted for the application of the differential DPSIR to the I & J farm.

Production (ton year[-1])	Setting 1 Monoculture	Setting 2 Monoculture + IMTA		Setting 3 Monoculture + IMTA	
Abalone	240	120	120	120	120
Seaweed		120		360	
ΔDPSIR analysis	Shift from: setting1 to setting2; Setting1 to setting3				

Seawater was pumped from the sea into top header tanks at a rate of 1 200 m^3 hr^{-1}. From there it was gravity fed un-filtered to the abalone tanks. In setting 1, effluent water was discharged to the sea. In setting 2 the effluent water from half the farm was channeled via a conveyor filter, which removed about 85% of the water-borne faeces, to four seaweed paddle ponds (with area of 140 m^2 and volume of 108 m^3 per pond). Water from the seaweed ponds was collected in a sump tank, from where half was pumped back into the header tank and half discharged.

As guidance for the sustainable management of the nutrient limitation that occurs in I & J when expanding the seaweed production another IMTA was analyzed - the IMTA farm Seaor Marine Ltd., located on the Israeli Mediterranean coast, 35 km north of Tel-Aviv (Neori et al., 2004). The Seaor Marine farm balances fish and abalone nutrient excretion with seaweed nutrient uptake and abalone seaweed consumption (Neori and Shpigel, 2006; Neori et al., 1998; Shpigel and Neori, 1996; Shpigel et al., 1996). The data for the nutrient budget of this farm consists on (i) N removal rate by seaweeds (4 g N m^{-2} d^{-1}), N uptake efficiency (85%) and total production area (3.5 ha); (ii) fish N excretion rate (182.5 g N per kg of fish produced); (ii) abalone N excretion rate (126.3 g N per kg of abalone produced) (Neori et al., 1998, 2004).

Differential Drivers-Pressure-State-Impact-Response application to the case study

We have assessed the economic and environmental cost/benefits to the main stakeholders of the shift from the monoculture setting 1 to the IMTA settings 2 and 3 (Table 4.10). ΔDPSIR analysis examined the ecological and economic effects of the integration of seaweed production in the I & J abalone farm (Response) by a quantification of the Drivers and

Pressures at the different operation settings, and the resulting Impacts. The application of the ΔDPSIR is generally schematized in Figure 4.7 and detailed below.

All monetary values in this paper were expressed in U.S. dollar (USD). Currency conversion used the average exchange rate for 2007 from IMF (International Monetary Fund) data (1 USD = 7.24 Rand and 1 Euro = 1.312 USD). Furthermore adjustments were made to equalize purchasing power between USA and South Africa using the purchasing power parity (PPP) for 2007 (1 USD = 4.273 Rand) obtained from IMF database.

Drivers

The Drivers of this case study are aquaculture production of abalone and seaweed (Table 4.10). These Drivers were quantified by profits of the I & J farm, with cost and revenue data extracted from Robertson-Andersson (2007), Robertson-Andersson et al. (2008) and Sankar (2009). Abalone sales were 240 ton year^{-1} with revenue of 378 Rand kg^{-1} (converts to 88.46 USD kg^{-1}, using PPP) live abalone in all operation settings. The costs for setting 1 were calculated based on an analysis of the farm running costs (Table 4.11). For settings 2 and 3 the costs were calculated based on setting 1 running costs and on the additional costs or savings associated with the shift to the abalone/seaweed recirculation system.

Table 4.11. Running costs of I & J farm when producing abalone in monoculture using a flow-through system (setting 1) (compiled from Robertson-Andersson (2007)).

Setting 1 - Total running costs	
6 740 thousand USD per annum	
Running cost breakdown	**%**
Labor	31.3
Sales related costs	21.5
Kelp feed	10.6
Repairs and maintenance	7.2
Electricity	6.8
Artificial feed	5.6
Research and development	3.2
Security	2.5
Technology	2.1
Insurance	4.2
Miscellaneous (e.g. generator, tractor)	5.0

Pressures

The Pressure exerted on coastal ecosystem by aquaculture of abalone, fish and seaweed can be assessed by a range of indicators as synthesized in Table 4.12.

Table 4.12. General indicators of Pressure exerted on the coastal ecosystem by aquaculture of abalone, seaweed and fish.

Abalone	Fish	Seaweed
N and P nutrient discharge in the effluents		
Oxygen concentrations in the effluents		
pH in the effluents		
Turbidity in the effluents		
BOD in the effluents		
Temperature in the effluents		
GHG* emission due to electricity consumption in farm operations (aeration, pumping, agitation, wastewater treatment)		
GHG emission due to electricity consumption in preparation of artificial feed and additives		
-		Net CO$_2$ uptake
Harvest of natural kelp beds as abalone feed	Harvest of fish to prepare fish feed	-

*Greenhouse gas.

For the case study the Pressure indicators considered important were: (i) nutrient discharge; (ii) harvesting of natural kelp for feed and (iii) emission of greenhouse gases (GHG). Approximate values for Pressures on the coastal ecosystem were estimated using the procedure detailed below. More detailed studies could include life-cycle assessment (LCA) (Ayer and Tyedmers, 2009).

Pressure I: Nutrient discharge in the effluent was calculated as the product of wastewater flow and the yearly average nutrient content at the systems' outlets, from the abalone tanks in setting 1, and from the seaweed ponds in setting 2 (Table 4.13). A simple nutrient mass balance model (described below) predicted effluent discharge in setting 3.

Table 4.13. Water quality and water flow in the monoculture (setting 1) and IMTA system (setting 2) for the I & J farm (compiled and combined from Robertson-Andersson (2007), Robertson-Andersson et al. (2008), Sankar (2009)).

	Parameter	Units	Setting 1	Setting 2
	Water pumped into the system/ wastewater flow	$m^3 h^{-1}$	2 772	1 386
	Recirculation	%	0	50
Yearly average nutrient concentration	Incoming seawater	$\mu mol\ N\ L^{-1}$	7.66	7.66
		$\mu mol\ P\ L^{-1}$	0.64	0.64
	Abalone tank outflow	$\mu mol\ N\ L^{-1}$	16.61	8.15
		$\mu mol\ P\ L^{-1}$	3.20	3.57
	Seaweed pond outflow	$\mu mol\ N\ L^{-1}$		3.82
		$\mu mol\ P\ L^{-1}$		3.41

Pressure II: The use of the cultivated seaweed as feed for the abalone in the IMTA settings reduces harvesting of the natural kelp beds and as such contributes to protecting the ecological functions provided by these ecosystems (Troell et al., 2006). The reduced harvesting due to the shift from monoculture to IMTA, represents a decrease in harvest Pressure. This calculation considered: (i) the seaweed production on the farm (120 and 360 ton year^{-1} in settings 2 and 3, respectively); and (ii) the average kelp bed density in the concession areas of the South African coast (5.43 kg m^{-2}, Robertson-Andersson (2007)).

Pressure III: The change in GHG emissions was determined by a simple mass balance of sources and sinks in the different operation settings. These included seaweed CO_2 uptake and the difference in GHG emissions between monoculture and IMTA operations. Further research should include LCA of carbon inventories in both monoculture and IMTA activities. CO_2 uptake by the cultivated seaweed was based on the seaweed yield, converted to net primary production (NPP) using the following conversion ratios: (i) 0.133 dry to fresh mass (Robertson-Andersson, 2007); (ii) 0.25 carbon in dry mass (Alongi, 1998); (iii) 0.8 NPP to gross primary production (GPP) (Sundbäck et al., 2004) and (iv) 3.66 CO_2 to C. The GHG emission was estimated from the electricity consumption data provided by the farm and the relative CO_2 emission (0.95 kg kWh^{-1}) reported from the utility that provides 95% of the South Africa's electricity (ESKOM).

State

Absolute quantification of the State of the costal ecosystem that receives the effluent had to be neglected, due to lack of synoptic data.

Impact

This case study considers the ecological Impacts that the seaweeds caused on the farm environmental performance instead of considering the Impacts on the State of the ecosystem resulting from the adoption of the IMTA. The changes in farm nutrient discharges, GHG emissions and area of the kelp natural beds not harvested were used to quantify the environmental externalities that result from the shift from abalone monoculture to the abalone-seaweed IMTA.

The corresponding economic Impact of IMTA was based on difference in the aquaculture profit between the monoculture and the IMTA setting as calculated in the Drivers section (it is assumed that there are no changes in the remaining activities that depend on the ecosystem) plus the value of the environmental externalities. The value of the environmental externalities was calculated based on avoided or additional costs due to: (i) nutrient treatment, (ii) kelp bed restoration and (iii) GHG offset. The calculation of these costs/benefits is as follows:

Externality I: IMTA with seaweeds reduces nutrient discharge. Avoided treatment costs were used to quantify the economic benefits. The total avoided treatment costs for the I & J farm when implementing the IMTA with seaweeds were calculated based on the estimated net nutrient removal compared with the monoculture setting and on the unit value of nutrient removal costs. However, effluent treatment costs vary widely with the characteristics of the effluent, regulations and technology. For instance in Crab et al. (2007) cost of treatment by frequently used biofilters in aquaculture ranged from 0.26 to 1.50 USD per kg of fish produced. Instead, the nutrient trading system established in the Chesapeake Bay watershed in Virginia (SWCB, 2006) was used to determine the value of the external benefits per unit of nutrient removal: 24.38 USD and 11.11 USD per kg of N and P removed, respectively.

 Externality II: The value of the benefits generated as a result of the avoided kelp harvest was calculated based on estimates of avoided harvested area due to seaweed production in the I & J IMTA farm and on the unit avoided costs for kelp bed restoration. Calculation of the unit avoided cost was based on the restoration costs in San Clemente Kelp Mitigation Project (Seaman, 2007; R. Grove (Southern California Edison) *personal communication*, 2008). For a total restoration area of 60.75 ha, a cost of 20.7 million USD (at 2007 prices) was estimated, which converts to an average of 34.09 USD m^{-2}.

Externality III: The cost/benefit associated with the change in the GHG emission was estimated using the voluntary carbon market system and the estimated GHG emissions of the I & J farm. The average applicable offset rate (14.27 Euro per ton CO_2, which converts to about 18.72 USD per ton CO_2) was calculated based on data of 90 providers (Carbon Catalog, 2008).

Response

The Response is a socio-economic component of the ΔDPSIR analysis (Nobre, 2009) and in this case study equals the cost of the measures adopted by the farm managers to set up settings 2 and 3. Estimates were based on Robertson-Andersson (2007) data about seaweed pond investment costs (Table 4.14). Calculation of the Response cost per annum considered 10-year depreciation for the concrete structure and 5-year for other components.

Table 4.14. I & J seaweed pond investment costs (compiled from Robertson-Andersson (2007)).

Paddle pond costs	USD
Concrete	9 361
Other components (e.g. electric motor)	4 645
Paddle wheels (shared by two ponds)	8 031
Total investment	**USD**
4 ponds	78 858
12 ponds	236 574

A Nutrient mass balance model for the recirculating system

A simple two compartment abalone-seaweed mass balance model was developed to predict the nutrient discharge for the projected expansion of the seaweed ponds in setting 3 (Figure 4.8). The nutrient sources include the nutrients from seawater (F_{sea}), the net nutrient production in the abalone tanks ($F_{abalone}$) and the seaweed fertilization ($F_{fertilizer}$). The nutrient sinks include seaweed nutrient uptake (F_{algae}) and nutrient discharge to the sea ($F_{effluent}$).
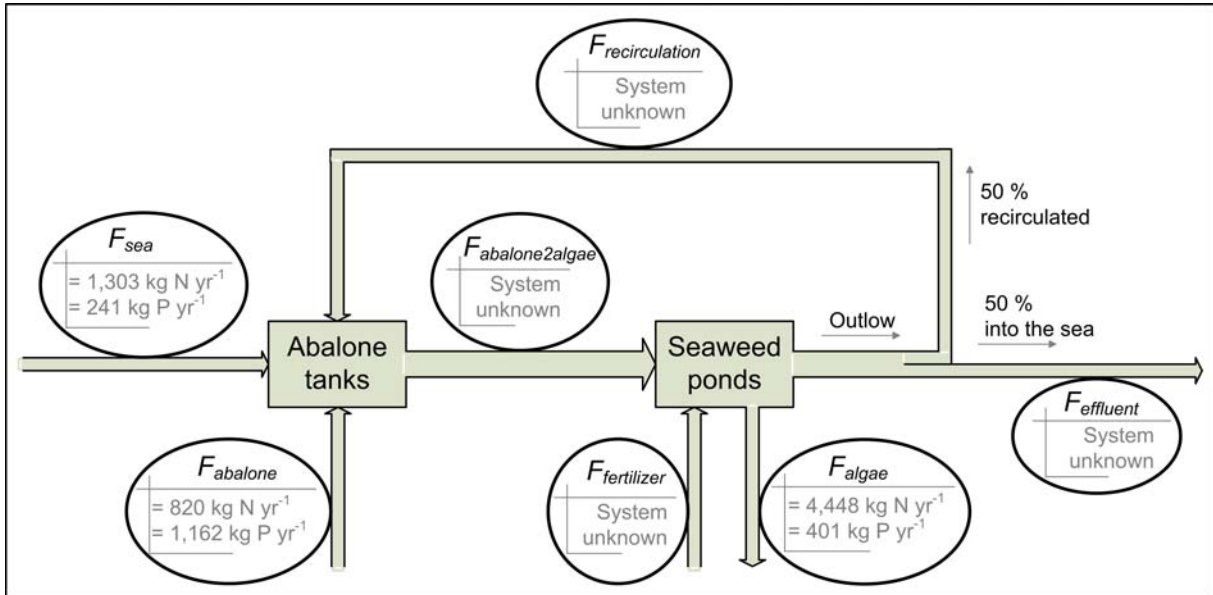
Figure 4.8. Nutrient mass balance model for setting 3 (recirculating IMTA system with 12 seaweed ponds to be implemented in the I & J farm).

The nutrient balance of the farm is assumed to be at steady state, where sources equal sinks for the entire farm (Eq. 4.2.1), the abalone tanks (Eq. 4.2.2) and the seaweed ponds (Eq. 4.2.3):

$$F_{sea} + F_{abalone} + F_{fertilizer} = F_{algae} + F_{effluent} \qquad \text{Eq. 4.2.1}$$

$$F_{sea} + F_{recirculation} + F_{abalone} = F_{abalone2algae} \qquad \text{Eq. 4.2.2}$$

$$F_{fertilizer} + F_{abalone2algae} = F_{recirculation} + F_{algae} + F_{effluent} \qquad \text{Eq. 4.2.3}$$

Where, F_{sea} (kg year^{-1}) is the nutrient mass flow from seawater (kg year^{-1}) and is calculated by the seawater nutrient concentration data and seawater pumped per annum (Table 4.13); $F_{abalone}$ (kg year^{-1}) is the overall nutrient mass produced in the abalone tanks and is calculated from the balance of nutrient flow into and out of the tanks (Table 4.13); $F_{fertilizer}$ (kg year^{-1}) is the nutrient mass required to subsidize the seaweed growth, in addition to the nutrient supply by the seawater and the abalone; F_{algae} (kg year^{-1}) corresponds to the algal nutrient uptake and is a product of the total seaweed pond cultivation area (4 and 12 ponds, of 140 m^2 each, in settings 2 and 3, respectively) by the nutrient uptake rate (r_{uptake} estimated for setting 2 as 7.3 g m^{-2} d^{-1} for N and as 0.7 g N m^{-2} d^{-1} for P); $F_{effluent}$ (kg year^{-1}) is the nutrient mass discharge to the sea; $F_{recirculation}$ (kg year^{-1}) is the nutrient mass in seaweed effluents that re-enters into the system and in this case study (50% recirculation) it is equal to the $F_{effluent}$; $F_{abalone2algae}$ (kg year^{-1}) is the nutrient mass outflow from the abalone tanks to the seaweed ponds.

Furthermore, $F_{fertilizer}$ can be defined as the nutrients required for the seaweed maximal yield minus the other nutrient sources for the seaweed ponds:

$$F_{fertilizer} = F_{algae}/e_{uptake} - F_{abalone2algae} \qquad \text{Eq. 4.2.4}$$

Where, e_{uptake} is the seaweed nutrient removal efficiency (%) that corresponds to the proportion of nutrients removed relative to the available nutrients. e_{uptake} was estimated for setting 2 as 53% for N and 5% for P.

For setting 3 it is assumed that values for F_{sea}, $F_{abalone}$, r_{uptake} and e_{uptake} are the same as in setting 2.

Considering the above assumptions and Eq. 4.2.2, Eq. 4.2.3 and Eq. 4.2.4 the model may be defined as a system of four equations with the following four unknowns (Figure 4.8):

$$F_{fertilizer} = F_{algae}/e_{uptake} - (F_{sea} + F_{recirculation} + F_{abalone}) \qquad \text{Eq. 4.2.5}$$

$$F_{abalone2algae} = F_{sea} + F_{recirculation} + F_{abalone} \qquad \text{Eq. 4.2.6}$$

$$F_{recirculation} = (F_{fertilizer} + F_{abalone2algae} - F_{algae})/2 \qquad \text{Eq. 4.2.7}$$

$$F_{effluent} = F_{recirculation} \qquad \text{Eq. 4.2.8}$$

The solution to the system can be defined as:

$$F_{fertilizer} = F_{algae}*[(1+ e_{uptake})/2*e_{uptake}] - F_{sea} - F_{abalone} \qquad \text{Eq. 4.2.9}$$

$$F_{abalone2algae} = F_{sea} + (F_{algae}/2) * (1/e_{uptake} -1) + F_{abalone} \qquad \text{Eq. 4.2.10}$$

$$F_{recirculation} = F_{algae}/2 * (1/ e_{uptake} -1) \qquad \text{Eq. 4.2.11}$$

$$F_{effluent} = F_{algae}/2 * (1/ e_{uptake} -1) \qquad \text{Eq. 4.2.12}$$

Here, Eq. 4.2.9 calculates the quantity of fertilizer required for the planned seaweed production and Eq. 4.2.12, the farm's nutrient discharge.

RESULTS AND DISCUSSION

The main issues considered for the analysis of the seaweed role in the abalone IMTA system as well the indicators used to quantify each of the DPSIR components are listed in Table 4.15 and quantified in Table 4.16.

Table 4.15. Synthesis of the differential DPSIR application to the I & J farm.

		Setting 1 to setting 2	Setting 1 to setting 3
Issues		Assess the role of seaweed in IMTA: Shift of abalone monoculture in flow-through system to polyculture combining abalone and seaweed in 50% recirculating system.	
		4 seaweed ponds (feed 10% of the farm)	12 seaweed ponds (feed 30% of the farm)
Drivers		Abalone aquaculture production (quantified using the profit).	
Pressures		Nutrients in aquaculture effluent. Harvesting of natural kelp as feed for abalone. Greenhouse gas (GHG) balance	
State		The State of the adjacent coastal ecosystem was not analyzed.	
Impact	**Ecol.**	Change in nutrient discharge. Change in harvesting from natural kelp bed. Change in GHG emissions	
	Econ.	Cost/benefits associated with nutrient treatment. Cost/benefits associated with kelp bed restoration. Cost/benefit associated with GHG offset.	
Response		Implementation of the ponds (cost of building and operation of the seaweed ponds).	
Time period		Shift between the monoculture and IMTA settings.	

Table 4.16. Quantification of the ecological and economic variables of the differential DPSIR for the I & j farm.

		Setting1	Setting2	Setting3	**Setting 1 to 2**	**Setting 1 to 3**
Drivers	Profit (10^3 USD year^{-1})	14 491	14 695	15 212	**204**	**721**
Pressure	N discharge (ton year^{-1})	11.3	6.3	7.6	**-5.0**	**-3.7**
	P discharge (ton year^{-1})	4.8	3.7	6.3	**-1.1**	**1.4**
	Kelp harvest (ha year^{-1})	14.4	12.2	7.8	**-2.2**	**-6.6**
	GHG (10^3ton CO_2 year^{-1})	11.6	11.2	11.3	**-0.35**	**-0.29**
Impact	Environ. externalities (10^3 USD year^{-1}):				**894**	**2 339**
	Changes in (10^3 USD year^{-1}): N discharge				(121.9)	(90.3)
	P discharge				(12.5)	(-16.0)
	Kelp harvest				(753.4)	(2 260.1)
	GHG				(6.5)	(5.0)
	Total impact *[1] (10^3 USD year^{-1})				**1 098**	**3 060**
Response	Implementation cost (10^3 USD year^{-1})				**12**	**36**
Net value of cost/benefits*[2] (10^3 USD year^{-1})					**1 086**	**3 024**

GHG – Greenhouse gas, CO_2. *[1] Total impact is given by the sum of change in profit with the value of externalities. *[2] Net value of cost/benefits is given by total impact minus the response implementation cost.

Drivers

The estimated profit was higher in the IMTA farms than in the monoculture farm (Table 4.16). IMTA reduced farm running costs relative to the abalone monoculture (Table 4.17). The items that contributed most to this result were: (i) faster abalone growth to market size when fed a mixed diet of kelp and cultivated seaweed (4, 3.8 and 3.3 years in settings 1, 2 and 3, respectively) (Naidoo et al., 2006); (ii) reduced kelp feed by 120 ton in setting 2 and 360 ton in setting 3; (iii) energy savings due to a lower pump head in recirculation (5 m), relative to a head of 15 m in pumping water from the sea to the monoculture (Robertson-Andersson, 2007). The shift from a monoculture to IMTA increases employment for the seaweed operation by 1 manager with 2 workers and 1 manager with 4 workers in settings 2 and 3 respectively; this adds to labor costs (Table 4.17), but constitutes a social benefit.

Table 4.17. Additional costs associated with the seaweed ponds and savings that result from the shifting of monoculture (setting 1) to the IMTA (settings 2 and 3) in the I & J farm.

	Setting 1 to setting 2	Setting 1 to setting 3
Costs (x10³ USD per annum):		
Labor for seaweed ponds	33	43
Savings (x10³ USD per annum):		
Abalone faster growth caused by mixed diet	168	590
Energy reduction	13	9
Kelp feed costs	55	165

Pressures

Nutrient discharge

Significant decreases in N (-44%) and P (-23%) discharges are estimated upon shifting the farm from setting 1 to setting 2 (Table 4.16). The reduction in N discharge is the result of seaweed uptake and decreased N accumulation in the abalone tanks (Table 4.18). The reduction of P discharge is mainly explained by a 50% reduction in water discharge to the sea that counteracts a small increase in the P concentration at the outlet of the recirculating system (Table 4.18).

Table 4.18. Nutrient sources and sinks for the I & J farm in (i) a flow-through 120 ton abalone monoculture system and (ii) a 120 ton abalone and seaweed (four ponds) IMTA system.

Source of nutrient flow	N (kg year⁻¹)		P (kg year⁻¹)	
	(i)	(ii)	(i)	(ii)
	Flow-through	Recirculating	Flow-through	Recirculating
From sea	2 606	1 303	481	241
Abalone tank	3 045	820	1 925	1 162
Fertilizer	-	10	-	14
Seaweed uptake	-	1 483	-	134
Recirculated	-	650	-	1 282
Out to sea	5 651	650	2 407	1 282

The nutrient mass balance model predicts a decrease in N discharges (-33%) but an increase in P discharges (+30%) upon shifting from setting 1 to setting 3 (Table 4.16). The increase of the P discharge is due to the estimated high fertilizer that is required for the production of 360 ton of seaweeds (Table 4.19).

Table 4.19. Nutrient source and sink predictions for I & J farm: (i) for the projected 120 ton abalone farm combined with twelve seaweed ponds (360 ton) in a recirculating system; and (ii) for a sensitivity analysis for the nutrient removal efficiency, where is tested e_{uptake} values from the literature (75% for N and 12.5% for P) instead of using values from setting 2 (53% for N and 5% for P).

Source of nutrient flow	Setting 3 recirculating system		Sensitivity analysis	
(kg year^{-1})	N	P	N	P
From sea *[1]	1 303	241	1 303	241
Produced in abalone tank*[1]	820	1 162	820	1 162
Expected seaweed uptake*[2]	4 448	401	4 448	401
Required fertilization*[3]	4 274	2 846	3 066	406
Recirculated *[3]	1 949	3 847	741	1 407
Out to sea *[3]	1 949	3 847	741	1 407

*[1] Assumed the same as in setting 2. *[2] Based on nutrient uptake rate as estimated in setting 2 and area of production. *[3] Model outputs.

Harvesting of natural kelp beds

The on-farm grown seaweed production of 120 ton for setting 2 corresponded to an estimated decrease in the harvesting of natural kelp beds of approximately 2.2 ha year^{-1} compared with the abalone monoculture setting 1. This reduction represents a cut of 3% in the total kelp harvest from South African natural kelp beds, which was about 4 050 ton year[-1] in 2003 (Troell et al., 2006). The expansion of the seaweed production to 360 ton (setting 3) represents a tripling in the estimated benefits to kelp beds relative to setting 2.

CO_2 balance

The CO_2 emission balance (ton CO_2 year^{-1}) indicates a net reduction relative to setting 1 of 345 ton in setting 2 and of 268 ton in setting 3, mainly thanks to the reduction in pump head height. Electricity saving of 350 MWh was estimated in setting 2 and of 245 MWh in setting 3. The CO_2 uptake through seaweed net primary production was about 12 ton CO_2 in setting 2 and 35 ton CO_2 in setting 3.

State and Impact

The State of the ecosystem for each operation setting was not quantified, due to a lack of synoptic data. However, given the general decrease in the Pressures (nutrient discharge and kelp harvest), positive Impacts and improvements in that State can be expected. Still, preliminary investigations on the ecological effects of abalone farm discharge did not find significant impacts, arguably due to dispersion of the effluents by high wave energy along the studied area (Samsukal, 2004). We have therefore applied the precautionary principle (Rio Declaration Principle 15 established at the 1992 United Nations Conference on Environment

and Development) and considered changes in nutrient discharge as an environmental externality with associated costs/benefits.

The increased P discharge in setting 3 Pressure compared with setting 1 (+30%) represented the only negative environmental impact (Table 4.16), largely due to the addition of fertilizer to sustain the necessary seaweed production. Remaining Pressure indicators showed that shifting from monoculture to an IMTA system translated into the following beneficial impacts: (i) reduced in N discharge (-44% in setting 2 and -33% in setting 3); (ii) reduced P discharge in setting 2 (-23%); (iii) reduced use of natural kelp beds (-3% and -8.9% of total kelp harvesting in South Africa in setting 2 and 3 respectively); and (iv) reduced GHG emissions (-3% in setting 2 and -2.3% in setting 3).

The quantified environmental externalities corresponded to an overall economic benefit to the environment of about 0.9 million and 2.3 million USD year^{-1} upon shifting the farm practice from abalone monoculture (setting 1) to the IMTA settings 2 and 3, respectively (Table 4.16). The economic value of the environmental externalities included the following items (Table 4.16):

(i) Avoided costs for N treatment - reduction in N discharge of 5 001 and 3 702 kg year^{-1} in settings 2 and 3 respectively, multiplied by 24.38 USD per kg of N removal, which corresponded to benefits of 121.9 thousand and 90.3 thousand USD year^{-1}, respectively.

(ii) Avoided and added costs for P treatment - P discharge reduction of 1 124 kg year^{-1} and an increase of 1 440 kg year^{-1} in settings 2 and 3 respectively, multiplied by 11.11 USD per kg of P removal, which corresponded to a benefit of 12.5 thousand USD year^{-1} and a cost of 16.0 thousand USD year^{-1}, respectively.

(iii) Avoided costs concerning kelp bed restoration in settings 2 and 3 - decreased kelp harvesting in concession areas of 22 099 m^2 year^{-1} in setting 2 and of 66 298 m^2 year^{-1} in setting 3, multiplied by the average kelp restoration cost of 34.09 USD m^{-2}, which corresponded to 753.4 thousand and 2 260.1 thousand USD year^{-1} respectively.

(iv) Avoided costs concerning changes in GHG emissions - emission reductions of 345 ton CO_2 year^{-1} and 268 ton CO_2 year^{-1} in setting 2 and setting 3 respectively, multiplied by the average CO_2 offset rate of 18.72 USD per ton CO_2, which corresponded to benefits of 6.5 thousand and 5.0 thousand USD year^{-1}, respectively.

The overall economic Impact associated with the shift from monoculture to IMTA is 1.1 million and 3.1 million USD year^{-1} in settings 2 and 3 respectively (Table 4.16). These positive values are a result of the benefits generated by the seaweeds directly to the farms

(increased profits, Table 4.16) and indirectly to the environment and the public (value of the externalities, Table 4.16).

Response

Shifting the farm from monoculture to IMTA involves financial costs (i.e. seaweed pond construction) of about 12 and 36 thousand USD year^{-1} in settings 2 and 3, respectively (Table 4.16). It is interesting to note that this investment is recovered in less than one year, given that the increase of profits obtained when shifting from monoculture to IMTA settings (0.20 and 0.72 million USD year^{-1} to settings 2 and 3, respectively, Table 4.16), is significantly higher than the total investment cost, estimated as only 79 thousand USD and as 237 thousand USD in settings 2 and 3 respectively (Table 4.14).

Managing nutrient limitation due to seaweed expansion

Nutrient mass balance in I & J, Cape Cultured Abalone Pty, Ltd. farm

The seaweed nutrient requirements are met in full by inputs from the sea and from abalone production in setting 2 (99.7% of N and 99.5% of P), but only partially in setting 3 (48.8% of N and 64.9% of P), as shown in Figure 4.9. Expansion of the seaweed production to 360 ton (setting 3) is thus nutrient limited and requires an external source of fertilizer.
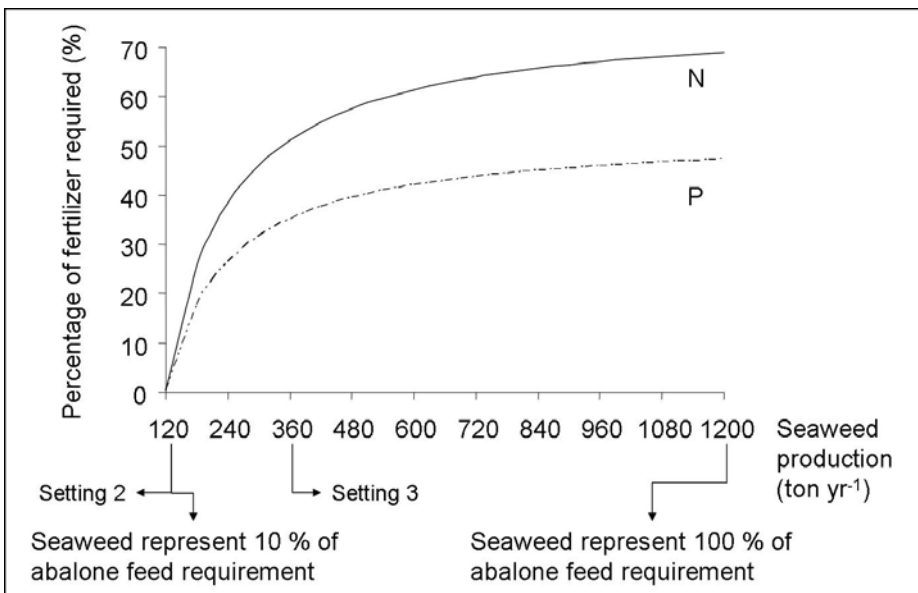
Figure 4.9. Nutrient mass balance model estimates of % of fertilizer required for seaweed production in the I & J farm as a function of target yield.

Nutrient mass balance in Seaor Marine Ltd. farm

The same methodology for nutrient analysis was applied to an Israel IMTA abalone farm, which includes fish and seaweeds with the following results:

A three-species IMTA farm, such as the Seaor Marine Ltd. farm, with fish, abalone and seaweeds is an efficient and economically profitable solution to the seaweed fertilizing issue. The effluents from fed-fish culture supply the nutrients necessary for high yields of protein-rich seaweeds (Neori et al., 2004). The entire seaweed N requirements of 60 ton N in the budget for Seaor Marine Ltd. farm (Table 4.20) are met by the abalone and fish excretions (72 ton N). The seaweeds remove 71% of the total N input. From an economic perspective, either the integration of fish production in the IMTA, or merely obtaining fish effluent from a separate fish monoculture farm, would generate economic benefits to both the abalone-seaweed, and the fish operations in the form of seaweed fertilizer and avoided fish effluent treatment. Such practice is even more advantageous where polluter pays taxes are applicable.

Table 4.20. Nutrient budget in Seaor Marine Ltd IMTA farm combining fish, seaweed and abalone (compiled from Neori et al. (1998, 2004)).

	Seabream	Abalone	Seaweed
Cultivation area (ha)	1.00	1.85	3.50
Production (ton year^{-1})	265.0	185.0	2 215.0
N release (ton year^{-1})	48.4.0	23.4	-
N uptake (ton year^{-1})	-	-	* 51.1

* Considering the 85% N uptake efficiency the 51.1 ton N removal by seaweed corresponds to a requirement of 60.1 ton year^{-1}.

Insights from the nutrient mass balance model

A sensitivity analysis of the nutrient mass balance model indicates that the nutrient discharge depends considerably on the efficiency of nutrient uptake by the seaweed. The simulations for setting 3 used values calculated from setting 2, with nutrient removal efficiencies (e_{uptake}) of 53% for N and 5% for P. Replacing these efficiencies with conservative literature estimates that apply to this farm, 75% for N and of 12.5% for P (Neori et al., 2000; Schuenhoff et al., 2003), significantly reduces the calculated discharge to the sea (Table 4.19). N uptake efficiency by seaweeds in a given location depends on the daily N load per unit area by a saturation curve (Cohen and Neori, 1991; Neori et al., 2003). Practically, the N load depends on the ratio of N excretion by the animals (abalone and fish) and seaweed pond area; a lower ratio leads to a higher N uptake efficiency, but with a lower uptake rate per unit area.

DISCUSSION

The ecological and socio-economic ΔDPSIR analysis of the abalone and seaweed IMTA used variations in key Pressure indicators upon shifting a farm from abalone monoculture to IMTA

with seaweeds. In general, the incorporation of seaweeds decreased the Pressures caused by the abalone production. The environmental benefits included reduction in nitrogen discharge into the sea, reduction in harvest of natural kelp harvesting and reduction in CO_2 emissions. Depending on the upscaling setting, phosphorus discharge could increase due to fertilization of the seaweeds; this could, however, be neutralized by integrating the production of other organism such as fish in the IMTA. The overall economic gain, thanks to adopting the IMTA design compared with an abalone monoculture, is valued at 1.1 million or 3.0 million USD year^{-1}, depending on the scale of the seaweed facility. These values represent the outcome of the balance between the farm direct benefits expressed as increased profit, the implementation costs of the seaweed ponds and the quantifiable environmental externalities that arise due to the shift from abalone monoculture to IMTA with seaweeds. The external environmental benefits contribute about 80% of the economic gains upon shifting to the IMTA, which means that the increase in profitability to the farms brings even larger benefits to the environment and the public. From both ecological and economic perspectives, the benefits associated with the shift from monoculture to the IMTA increases with an increase in seaweed production. However, the expansion of the on-farm grown seaweeds should be carefully designed in order to efficiently address the resulting nutrient limitations. The balanced three-species IMTA farm in Israel provides an example on how to manage nutrient limitation. In that case the on-farm grown seaweeds receive all the additional nutrients from the fish effluents.

Social relevance

Aquaculture can accrue social benefits in employment, income and food security, particularly important to poor, rural coastal communities worldwide (Ahmed and Lorica, 2002; Katranidis et al., 2003; Kaliba et al., 2007). The South African abalone farm case study exemplifies the positive impact an aquaculture industry can have on local communities. The I & J farm employs 5.5% of the men and 1.5% of the women from the local communities of Blompark, Groeneweldskerma and Masakhane (CSS, 2005, Robertson-Andersson, 2007). These communities are characterized by high unemployment (85.7%), with more than 50% of the labor force being unskilled and semi skilled, using criteria as defined by Lewis (2001) (Robertson-Andersson, 2007). This is particularly relevant where unemployment is not only an economic issue but also a critical socio-political issue (Kingdon and Knight, 2003; Evett, 2006): four in every ten adults of working age in South Africa are unemployed or have no access to or means of earning an income (Evett, 2006). According to Lewis (2001), the overall unemployment in South Africa in 2000 was above 36% and 50% in unskilled and semi-skilled workers respectively. The direct permanent employment in the South African

abalone industry has a large local impact in previously disadvantaged coastal communities, where any increase in employment is valuable. A more detailed analysis is still required to determine the full cascade of social impacts that the IMTA approach can have. In particular, and in order to complement the present ecological-economic assessment, future analysis should focus on the social cost/benefits of the IMTA settings compared with the monoculture production.

CONCLUSION

The application of the ΔDPSIR to the present case study indicates that the shift from abalone monoculture to IMTA with seaweeds increases the farm profitability and brings even larger benefits to the environment and the public, through reduced Pressures on the adjacent coastal ecosystem and increased employment. As the cost of energy increases and where pollution taxes are adopted, the economic incentives for the implementation of IMTA farms, compared with monoculture abalone farms, are likely to mount. From a social and environmental perspective, the three-species IMTA with fish, abalone and seaweeds produces more value and resources for human consumption while still managing the waste produced. This outcome should be considered by industry and regulators involved with the current expansion in abalone culture worldwide. The present ΔDPSIR analysis can help owners and regulatory officials in balancing the design of the farm with respect to nutrient mass balance towards reduced negative environmental externalities. As kelp is reaching limits of sustainable harvesting, particularly in kelp concession areas with high abalone farm concentrations, and with the forceful socio-economic incentives quantified in the present paper, it can be expected that two- and three-species IMTA farms will become the industry norm, rather than the exception.

The estimates of the economic value of the environmental externalities obtained by the ΔDPSIR analysis provide the aquaculture industry, the coastal zone and resource managers an indication of the benefits to farms and society by implementation of ecologically balanced IMTA farms, relative to monoculture systems. More such analyses should be undertaken on other aquaculture practices and for other species of fish, shrimp, shellfish and macroalgae. Likewise, the ΔDPSIR could be applied to compare the ecological and economic impacts of fisheries vs. aquaculture. Those studies should include a detailed quantification of aquaculture industry impacts on the entire cascade of employment and income of local communities.

Chapter 5. Ecological-economic dynamic modelling

Context

One of the missing links in ecosystem modelling is economics. Integration with economics for scenario testing is important to help define the focus of management measures. Dynamic ecological-economic modelling is required to simulate the feedbacks between the ecological and economic systems. Insights provided by the outcomes of such modelling tools are important for coastal management. For instance, with limited resources, is important to prioritize actions that bring larger benefits to the public and at the same time allow the development of private activities.

Summary

This chapter presents the MARKET model, which dynamically couples the ecological and economic components of aquaculture production. The model is herein applied to simulate shellfish production in a Chinese bay under different assumptions for price and income growth rates and the maximum area available for shellfish cultivation.

This chapter corresponds to the published manuscript:

(For consistency with published version this chapter is written in American English)

A dynamic ecological-economic modeling approach for aquaculture management

INTRODUCTION

Global consumption of finfish and shellfish as food has doubled since 1973. Evidence suggests that the large increase in the aquatic resources production in recent decades has resulted from the enormous growth in seafood demand in the developing countries (Delgado et al., 2003). China is the largest aquaculture producer in the world, with an average annual growth rate from 1980 to 2004 of 15 % (Gíslason et al., 2006), and the only nation where farmed production exceeds wild catch (Sanchez et al., 2007). In 2006, 68 % of total aquatic production in China was from aquaculture (FAO, 2009). The development of aquaculture in China has had a positive impact in terms of its contribution to nutrition, employment, and improvement in socio-economic status of both rural and urban communities (FAO, 2004). About 4.3 million rural workers are directly employed in aquaculture with an annual per capita net income of 8 667 Yuan (which converts to 1 075 USD considering the exchange rate at the time of study, 1 USD = 8.06 Yuan) (FAO, 2005). Given the significance of aquaculture in China, changes in mariculture production due to changes in economic inputs or biophysical variability have a wider socio-economic impact on communities.

Just like any other food-producing sector in the world, aquaculture relies on renewable and non-renewable resources. Sustainable development and management of aquaculture thus requires an appropriate understanding of the conflicts and interactions between the resource use and its users. Such understanding contributes to improve governance in resource use, which is an important prerequisite of the sector's sustainability and one of the objectives of building an ecosystem approach to aquaculture (EAA) (Soto et al., 2008). Aquaculture is considered as the "solution" for bridging the supply and demand gap of aquatic food globally. There is however concern about the negative environmental impacts that some aquaculture practices can exert on coastal resources and ecosystems (Tovar et al., 2000; Xu et al., 2007).

The carrying capacity of the coastal ecosystem can represent a limit to the increase in aquaculture production. Depending on culture practices, this might be related to space limitations, availability of food resources or on the environmental capacity to assimilate aquaculture generated wastes (Sequeira et al., 2008). Apart from ecological limitations there are also economic cost limitations to production, illustrated through an analysis of the marginal cost in relation to marginal revenue (Gravelle and Rees, 1993). An economic

analysis of aquaculture production must be based on realistic production cost and income projections that account for these economic limitations.

The focus of aquaculture management is often on maximizing the output and not the profit, which is not only economically inefficient, but carries unnecessary ecological risks. If the goals of sustainable aquaculture development are to be achieved, then there is need to understand both ecological and economic limitations. Aquaculture operations depend directly on the availability and quality of the marine resources and environment. If the marine ecosystem is overexploited the negative impacts will be felt in aquaculture farming operations and by all other downstream activities dependent on aquatic resources farming. This is particularly important for a country such as China that accounts for 68 % of the world aquatic production, and where some of the marine ecosystems have a high percentage of reclaimed areas for aquaculture, e.g., 77 % of the coastal usable area of Xiamen is occupied by aquaculture activities (Xue, 2005).

To ensure sustainable aquaculture production, it is crucial to understand the ecological and economic limits beyond which mariculture becomes less efficient. Dynamic modeling can provide a tool that facilitates the understanding of the complex feedbacks between ecological and economic aspects of aquaculture production. Resource managers and policymakers have come to understand that the sustainability of ecological and economic systems is tightly coupled (GESAMP, 2001). However, the complexity of the interactions may make informed resource decision-making extremely difficult, particularly given the dynamic nature of ecosystems and the difference in the scale of analysis of ecological and economic systems.

The integration between ecological and economic models is currently a developing discipline (Drechsler et al., 2007). Several conflicts were identified (Bockstael et al., 1995; Drechsler and Watzold, 2007) that explain the decoupling of these two disciplines, namely: (i) the scales of analysis; (ii) the communication/understanding between ecology and economics; and (iii) the implicit assumptions of each one.

In recent years there was an increase in the development of integrated ecological-economic models (Drechsler et al., 2007). According to Bulte and van Kooten (1999), Armstrong (2007) and Drechsler et al. (2007) these models tend to be less complex than the biological/ecological models alone. Jin et al. (2003) categorize ecological-economic models into 3 groups: (i) bioeconomic model approach; (ii) integration of complex environmental and economic models; and (iii) linear models, for instance the coupling of linear economic input-output model with a food web model.

This paper aims to develop a dynamic environmental and economic model as a tool for mariculture management and for EAA, and to illustrate a coupling approach. The main objectives are to:

1. Develop a conceptual model of the ecological-economic interactions in mariculture;

2. Implement a dynamic ecological-economic model in order to simulate (i) the socio-economic component of shellfish aquaculture production, (ii) its effects on the estuarine and coastal ecosystems, and (iii) feedbacks of the environmental system on the socio-economic system;

3. Simulate a set of scenarios to compare the model outputs with expected trends and to test its capability to simulate management scenarios.

METHODOLOGY

Conceptual approach

The Modeling Approach to Resource economics decision-maKing in EcoaquaculTure (MARKET) (Figure 5.1), illustrates the major interactions which should be considered in mariculture between ecological and economic systems.
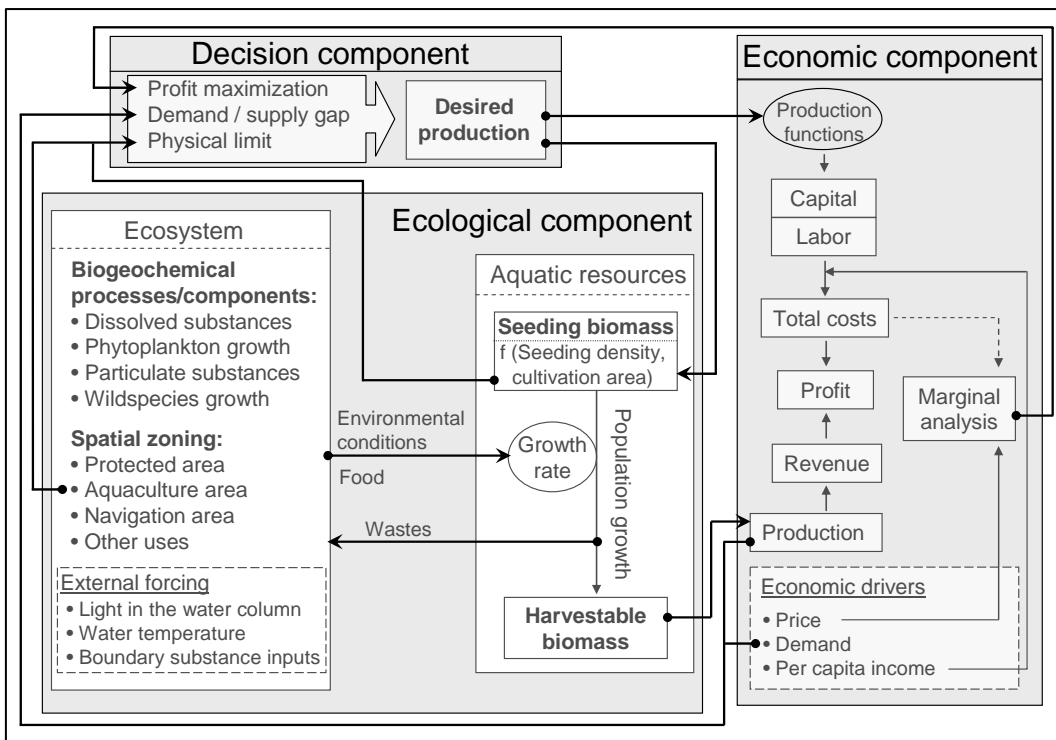
Figure 5.1. MARKET conceptual model: ecological-economic interactions in mariculture.

The MARKET model includes three components (Figure 5.1): (i) the ecological component, which includes the relevant ecosystem biogeochemistry and the growth of aquatic resources;

(ii) the economic component, which invests capital and labor for the production of the aquatic resources; and (iii) the decision component, which determines the desired production for the next production cycle. The three components interact as follows (Figure 5.1):

At the beginning of a production cycle, the ecological component is used to determine the seeding biomass corresponding to the desired production for that cycle and to allocate the required cultivation space. The ecosystem water quality and environmental conditions are used to calculate the scope for growth of the cultivated species. In parallel, the aquatic resource production affects the biogeochemistry of the ecosystem, either through waste generation and/or uptake of particulate and dissolved substances, depending on species and culture practice. The adult individuals are subsequently harvested and transferred to the economic component at the end of the production cycle, and the harvested biomass is used by this module to calculate the revenue generated. Concurrently, in the economic component the production inputs, such as labor and capital required to produce the desirable yield (as calculated in the decision component), are determined and used to calculate the production cost. In addition, the economic component determines the marginal cost and marginal revenue in order to inform the decision component about profitability. The decision component then determines the changes in the desired production for the next cycle based on the following criteria: (i) profit maximization, based on the comparison of marginal cost and marginal revenue; (ii) the gap between demand and supply, based on the comparison of the local demand against shellfish production, in order to monitor if the market can absorb an increase in production or if there is already a surplus; and (iii) physical limit, in order to ensure that the cultivation area does not exceed the maximum available area for aquaculture, as defined by ecosystem managers.

Ecological and economic limits

The ecosystem carrying capacity and economic production capacity can be limited by the following factors:

1. Space limitation, which is defined by stakeholders with respect to allocation of ecosystem area to cultivation and other uses.

2. Food limitation (in the case of extensive aquaculture), which is a function of available ecosystem resources, cultivation densities and practices. It affects the growth rate of aquatic resources.

3. Aquaculture waste limitation, which causes an effect on environmental conditions such as dissolved oxygen, thereby causing a feedback on the growth rate of aquatic resources. These effects depend on the cultivation practice and on the assimilation capacity of the ecosystem.

4. Cost limitations related to the amount of inputs that can be used.

5. Diminishing returns to scale, such that each additional unit of variable input yields less and less additional output (production).

6. Profit maximization, whereby the profit maximizing firms will increase production as long as their profits will continue to rise. Profits will start to decrease beyond the output level where marginal cost equals marginal revenue.

Case study: site and data description

The MARKET model was applied to simulate shellfish production in Xiangshan Gang, a coastal embayment located in Zhejiang Province, in the East China Sea (Figure 5.2) in the vicinity of the largely industrialized centre of Ningbo City.

Zhejiang Province is known for its valuable marine resources, although it is less dependent on the primary sector than China in general (Table 5.1). Considering the total value of all marine and inland fish farming and the direct employment it generates (Table 5.1) this industry creates almost 20 direct fish farming jobs per 1 million Yuan (124 000 USD) of value in fish farming.
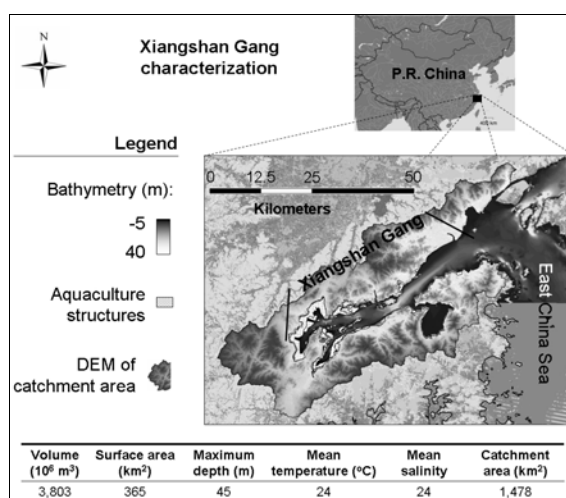
Figure 5.2. Xiangshan Gang map and physical data.

In Zhejiang, total aquatic outputs declined by 2 % from 2004 to 2005, while secondary and tertiary sectors continued to grow rapidly (Information Center of General Office of Zhejiang Provincial Government, 2006). A synthesis of the case study socio-economic indicators is provided in Table 5.1.

Table 5.1. Case study socio-economic indicators. Compiled from FAO (2005) and NBSC (2007).

	China	Zhejiang Province	Ningbo City
Population, million inhabitants	1 300	47	6
Urban per capita annual disposable income, Yuan (USD)	10 397 (1 290)	10 156 (1 260)	26 598 (3 300)
Primary sector share of economy, %	15 %	7 %	7 %
Fish production, million ton	47	4.9	0.9
Total fisheries value, Yuan billion (USD billion)	332 (41.2)	14.0 (1.7)	n/a
Related industry value, Yuan billion (USD billion)	126 (15.6)	3.0 (0.4)	n/a
Related services value, Yuan thousand (USD thousand)	119 400 (14 814)	300 (37)	n/a
Marine farming value, Yuan billion (USD billion)	73 (9.1)	n/a	n/a
Inland farming value, Yuan billion (USD billion)	143 (17.7)	n/a	n/a
Total fisheries employment, million jobs	7.0	n/a	n/a
Fish farming employment, million jobs	4.3	n/a	n/a

Note: Conversion to USD is shown between 'brackets' after values in Yuan considering the exchange rate at the time of study: 1 USD = 8.06 Yuan.

The Xiangshan Gang covers an area of 365 km2 and an annual shellfish production of about 38 000 ton (Sequeira et al., 2008). Figure 5.2 provides further details about the characteristics of the bay. An ecosystem model developed for the Xiangshan Gang was used in order to simulate the shellfish production and the biogeochemistry of the system (Ferreira et al., 2008b; Sequeira et al., 2008). Data on the ecosystem and shellfish cultivation were obtained from Ferreira et al. (2008b) and Sequeira et al. (2008).

Economic data used in this study are from various sources and include: (i) data on the reference production, cost and net profit obtained in a local survey on the economics of aquaculture (de Wit et al., 2008); (ii) the sensitivity (elasticity) of demand to price and income obtained from demand functions analysis, while the capital and labor elasticities are obtained from a production function analysis (Musango et al., 2007); (iii) other data such as production and price growth rates are from various issues of the China Statistical Yearbooks (NBSC, 2007) while the interest rate was taken from International Monetary Fund (IMF) statistics.

Model implementation

The MARKET model was implemented for shellfish production in Xiangshan Gang using a visual modeling platform (PowerSim™). Table 5.2 and Table 5.3 specify the model parameters and the initial conditions of the state variables.

Table 5.2. MARKET model parameters.

Parameter	Symbol	Value	Unit	Comment
Simulation setup				
Simulation timestep	ts	0.01	year	
Ecological timestep	ts_{ecol}	0.01	year	
Economic timestep	ts_{econ}	1	year	
Simulation period	$SimP$	50	year	
Ecological system				
Cultivation cycle	tp	1	year	
Seeding period	sp	0.25	year	0.00-0.25 year every year
Seeding density	n_{seed}	45	ind m^{-2}	Sequeira et al. (2008)
Weight class:	$s:$			
Weight class 1	s_1	5	g ind^{-1}	0 to 10 g ind^{-1}
Weight class 2	s_2	15	g ind^{-1}	10 to 20 g ind^{-1}
Weight class 1	s_3	20	g ind^{-1}	20 to 30 g g ind^{-1}
Mortality rate	μ	0.46	year^{-1}	Sequeira et al. (2008)
Maximum cultivation area	$MaxA$	302 950 000	m^2	83 % of estuary area
Ecosystem model seed weight	w	1.5	g ind^{-1}	Sequeira et al. (2008)
Economic system				
Price elasticity of demand	e_d	-0.07	(-)	Ferreira et al. (2008b)
Income elasticity of demand	e_y	0.87	(-)	Ferreira et al. (2008b)
Per capita income growth rate	r_y	0.1	year^{-1}	NBSC (2007)
Price growth rate	r_p	0.02	year^{-1}	NBSC (2007)
Demand growth rate	r_d	0.0856	year^{-1}	$r_d=e_y*r_y+e_d*r_p$
Elasticity of labor	α_L	0.44	(-)	Musango et al. (2007)
Elasticity of capital	α_K	0.53	(-)	Musango et al. (2007)
Depreciation fraction	d_f	0.1	(-)	$d_f=ts_{econ}/d_p$
Depreciation period	d_p	10	year	Assumption
Interest rate	r	0.06	year^{-1}	IMF
Maintenance Fraction	m_f	0.16	year^{-1}	Assumption

A key feature for implementation of the integrated ecological-economic model was to accommodate the different resolutions at which the ecological and the economic systems are studied, which are hours to days, and annual quarters to years, respectively. The scaling issue was addressed by using two different timesteps for each model, 0.01 year (3.65 days) for the ecological model and 1 year for the economic model (Table 5.2). The ecological model runs every timestep while the economic and decision models run only with a periodicity corresponding to its timestep, i.e. every 100 timesteps of the simulation. The simulation

period considered is 50 years and the shellfish production cycle (*tp* in year) is one year (Table 5.2). The seeding occurs during the first 91 days of the year (Table 5.2) and the harvest accumulates until the last timestep of each year (0.99 year), at which the harvestable biomass is communicated to the economic model. The decision and economic models operate at the last timestep of each year (0.99 year).

Table 5.3. Initial value of MARKET model variables.

State variable	Symbol	Initial value	Unit	Comment
Cultivation area	A	23 083 092	m^2	Sequeira et al. (2008)
Local demand	LD	37 222 000	kg	Assumed equal to initial HSY
Price	P	12.5	Yuan kg^{-1}	de Wit et al. (2008)
Shellfish production	Q	37 222 000	kg	de Wit et al. (2008)
Labor	L	128211	Man-Day (MD)	de Wit et al. (2008)
Capital	K	37 030 726	Yuan	de Wit et al. (2008)
Unit labor cost	UVC_L	7.38	Yuan MD^{-1}	de Wit et al. (2008)
Unit cost of other variable inputs	$UVCo$	0.19	Yuan kg^{-1}	de Wit et al. (2008)

The implementation of each simulation block of the MARKET model (Figure 5.1) is explained below.

Ecological component

The implementation of the ecological component of the MARKET model followed a three stage approach:

Stage 1 – Decoupled ecosystem modeling. This stage comprehends simulation of Xiangshan Gang biogeochemistry and shellfish growth using an ecosystem model, which was decoupled from the MARKET model.

Stage 2 – Simplification of main interactions between ecosystem model and shellfish production. In this stage the ecosystem model was used to determine the shellfish growth rate as function of cultivated area and thus of seeding biomass (given that seeding density is a constant).

Stage 3 – Integration in the MARKET model of the main interactions with the ecosystem model. In this stage a population model was used to simulate the harvestable available biomass (to be used as an input in the economic model at the end of the production cycle)

based on the seeding input (obtained from the decision model output at the beginning of each production cycle) and on the shellfish growth rate (obtained from stage 2).

Stage 1 - Decoupled ecosystem model

An ecosystem model, developed with the widely used EcoWin2000 modeling platform (Ferreira, 1995; Nunes et al., 2003; Nobre et al., 2005; Sequeira et al., 2008), was applied to simulate the key biogeochemical features of Xiangshan Gang as well as shellfish aquaculture (Ferreira et al., 2008b; Sequeira et al., 2008). The spatial domain of the model was divided into 24 compartments (12 horizontal x 2 vertical layers). The catchment loads (dissolved nutrients and particulate matter) and fish cage wastes were simulated as a forcing function (Ferreira et al., 2008b). The transport of substances was simulated using an offline data series of water fluxes between boxes and across the sea boundaries, provided by a detailed hydrodynamic model (Ferreira et al., 2008b). In each box the main state variables simulated were dissolved inorganic nutrients (nitrogen and phosphorus), suspended particulate matter, phytoplankton biomass, shellfish individual scope for growth and population dynamics, following the approach described for instance in Ferreira et al. (2008a).

For the simulation of feedbacks between the economic and environmental components, both the economic and the decision models should be coupled with the ecosystem model, although in the current implementation of the MARKET model simulations were made in decoupled mode.

Stage 2 - Simplification of main interactions between ecosystem model and shellfish production

In order to implement the ecological component of the MARKET model the main interactions between the ecosystem model and the aquatic resources production were simplified. It was considered that these are represented by (i) the seeding biomass (i.e. the cultivation area assuming that the seeding density is a constant) and (ii) the resulting growth of the bivalves.

The decoupled ecosystem model of the bay (Sequeira et al., 2008) was run in order to determine the shellfish growth rate as a function of the cultivated area. Several cultivation areas were used to run the ecosystem model using the same setup for the remaining initial state variables, parameters and boundary conditions. Therefore, the simulation accommodates the potential food availability constraints due to an increase in the number of filter feeders. It was found that the growth rate is inversely proportional to the cultivated area (Eq. 5.1).

$$G = -2.3 \times 10^{-8} \cdot A + 20.71 \qquad \text{Eq. 5.1}$$

Where, G is the annual growth rate (year^{-1}) and A is the cultivation area (m^2).

The disruption of shellfish production due to food availability, which potentially could occur as a result of an increase of cultivated area, is never reached, even when the maximum cultivated area (considered to be 83 % percent of the bay area) is attained.

Stage 3 - Integration in the MARKET model of the main interactions with the ecosystem model:

In the current implementation of the MARKET model, shellfish growth provides a proxy for the ecosystem feedbacks. The ecological component was implemented by means of a population model (Ferreira et al., 2007a), which was used to simulate the growth of the cultivated seed up to a harvestable size (Eq. 5.2).

$$dN(s,t)/dt = -d[N(s,t) * g(t)]/ds - \mu * N(s,t) \qquad \text{Eq. 5.2}$$

Where, s is weight class (in g ind^{-1}, defined in Table 5.2), t is time (in year), N is number of individuals (in ind) of weight class s, g is scope for growth (in g ind^{-1} year^{-1}), and μ is mortality rate (in year^{-1}, defined in Table 5.2).

Every year at the end of the production cycle the new cultivation area for the next year (Eq. 5.3) is calculated as a function of previous cultivated area and rate of change in production (r_{cq}, in year^{-1}, obtained from the decision component):

$$dA/dt = A * r_{cq} \qquad \text{Eq. 5.3}$$

At the start of each seeding period (sp, in year, defined in Table 5.2) the total seeding of individuals in Class 1 (N_1, Eq. 5.4) is determined based on cultivation area (A from Eq. 5.3) and seeding density (n_{seed}, ind m^{-2}, defined in Table 5.2):

$$N_1 = A * n_{seed} \qquad \text{Eq. 5.4}$$

Scope for growth (g, Eq. 5.5) is calculated as a proxy of the population growth (G from Eq. 5.1), and thus is a function of cultivated area.

$$g = G * w$$ Eq. 5.5

Where, w (in g ind^{-1}, defined in Table 5.2) is the average individual seed weight used in the ecosystem model.

At the end of the year the individuals accumulated in the harvestable classes (N_2+N_3, as calculated from Eq. 5.2) are converted into the harvestable biomass (HB, in kg, Eq. 5.6):

$$HB = (N_2 * s_2 + N_3 * s_3) * \beta$$ Eq. 5.6

Where, β is the conversion from g to kg.

Current implementation of the ecological model assumes that decisions to change production are implemented through changes in the cultivation biomass. On the other hand, the changes in the cultivation biomass affect the growth of shellfish (due to food availability) and consequently the harvestable biomass. At this stage of development, the ecosystem feedbacks are implicitly included in the MARKET model through the shellfish growth. Future developments of the model will include explicit integration of the economic and decision systems into the ecosystem model in order to monitor shellfish biodeposition as well as the role of filter-feeders on phytoplankton uptake. Phytoplankton removal equates to the reduction of coastal eutrophication symptoms, providing an additional ecosystem service.

Economic component

In each simulation year, the decision model calculates the desired production rate, communicates it to the economic model and thus drives the change in the production inputs (Figure 5.1). The economic component of the MARKET model is divided into sub-models that simulate: (i) the harvest of the available biomass determined by the ecological model, (ii) the production inputs (labor and capital), (iii) the corresponding production cost, (iv) the generated revenue and net profit of the bivalve production for a given year, and (vi) the marginal cost and marginal revenue in order to provide information required by the decision model. The implementation of the economic model also includes simulation of the exogenous functions that drive the aquatic resource production, namely: (i) price, (ii) household income, and (iii) local demand. Both the economic drivers and sub-models are further detailed below.

Economic drivers:

The economic drivers are implemented following standard economic theory. A rise in income is expected to positively influence the demand for fish and aquatic products and an increase in price is expected to negatively influence the demand for aquatic species and aquatic products

(Jolly and Clonts, 1993). In the model the changes in demand (r_d, in year^{-1}) are determined by changes in the income and prices, as defined in Table 5.2. Both the price elasticity of demand (e_d, Table 5.2) and income elasticity of demand (e_y, Table 5.2) were obtained from a national level demand function analysis (Ferreira et al., 2008b). This model assumes that the changes of the local demand follow the changes of the national demand, as information to derive local level demand functions was not available. The local demand (*LD*, in kg) forcing function (Eq. 5.7) is initialized considering the local consumption data as the initial local demand (Table 5.3).

$$dLD/dt = r_d * LD \qquad\qquad \text{Eq. 5.7}$$

The local farmers are assumed to be price takers, whereby the aquatic product prices are determined by the global market. The changes in the domestic price reflect the Chinese inflation rate for the period 1995-2006. The yearly average including outliers is 2.8 %, while when excluded, the average is 1.5 % (NBSC, 2007). A constant price growth rate (r_p, in year^{-1}) of 2 % per year was therefore assumed based on the averaged inflation data. The price (*P*, in Yuan kg^{-1}) forcing function is given by Eq. 5.8:

$$dP/dt = r_p * P \qquad\qquad \text{Eq. 5.8}$$

In addition to price and demand the economic model is also forced by the annual growth of the per capita income (r_y, in year^{-1}). The per capita income growth rate is used to calculate the changes in the demand (r_d), as defined in Table 5.2, and is also used to force the changes of the unit labor cost as defined in Eq. 5.23. A constant per capita income growth rate of 10 % per year was assumed based on the real per capita income growth data (NBSC, 2007).

Production sub-model:

The shellfish production (*Q*, in kg) for a given year (Eq. 5.9), is based on the desired production determined for that year and is limited by the harvestable biomass simulated in the ecological system (*HB*, in kg, Eq. 5.6). Thus, herein we assume that the harvest shellfish yield equals to the shellfish production.

$$Q = Min(DQ, HB) \qquad\qquad \text{Eq. 5.9}$$

Where, *DQ* (in kg), is the desired production determined for that year, which was calculated in the previous year as the desired production for the next cycle, following Eq. 5.32, in the decision system.

Production inputs sub-model:

This sub-model examines the capital and labor input levels resulting from the changes in the desired production:

$$dL / dt = R_L \qquad\qquad\qquad \text{Eq. 5.10}$$

$$dK / dt = R_K \qquad\qquad\qquad \text{Eq. 5.11}$$

Where, L (in Man-Day) is the labor used for the production and is calculated based on the required changes in labor inputs (R_L, in Man-Days year^{-1}); K (in Yuan) represents the assets used in production and is calculated based on the required changes in the value of capital (R_K, in Yuan year^{-1}).

The changes in both labor (R_L, Eq. 5.12) and capital (R_K, Eq. 5.13) are determined as a function of the desired change in production (R_{CQ}, in kg year^{-1}, calculated in the decision model, Eq. 5.31) and respectively on the marginal productivity of labor (MP_L, in kg Man-Days^{-1}) and on the marginal productivity of capital (MP_K, in kg Yuan^{-1}):

$$R_L = R_{CQ} / MP_L \qquad\qquad\qquad \text{Eq. 5.12}$$

$$R_K = R_{CQ} / MP_K \qquad\qquad\qquad \text{Eq. 5.13}$$

Where, MP_L and MP_K are determined following Eq. 5.14 and Eq. 5.15, respectively, as defined in Yunhua et al. (1998).

$$MP_L = \alpha_L * Q / L \qquad\qquad\qquad \text{Eq. 5.14}$$

$$MP_K = \alpha_K * Q / K \qquad\qquad\qquad \text{Eq. 5.15}$$

Where, α_L and α_K (dimensionless, Table 5.2) are the elasticity of labor and capital, respectively, and were determined based on the production function (Musango et al., 2007) defined in Eq. 5.16:

$$\ln Q = 0.44 * \ln L + 0.53 * \ln K + 1.16 \qquad\qquad\qquad \text{Eq. 5.16}$$

Production cost sub-model:

The production cost sub-model determines the total cost of shellfish production (TC_Q, in Yuan, Eq. 5.17) as the sum of the fixed cost (FC, in Yuan) and the variable cost (VC, in Yuan):

$$TC_Q = FC + VC \qquad\qquad \text{Eq. 5.17}$$

Where, FC and VC are calculated following Eq. 5.18 and Eq. 5.21, respectively.

$$FC = DK + IKL \qquad\qquad \text{Eq. 5.18}$$

Where, FC is given by the depreciation of capital (DK, in Yuan) and by the interest on capital loan (IKL, in Yuan). DK and IKL are given by Eq. 5.19 and Eq. 5.20, respectively.

$$DK = d_f * K \qquad\qquad \text{Eq. 5.19}$$

Where, d_f (dimensionless) represents the depreciation fraction (Table 5.2).

$$IKL = r * K \qquad\qquad \text{Eq. 5.20}$$

Where, r (year^{-1}) is the interest rate (Table 5.2).

The variable cost includes the labor cost (VC_L), the maintenance cost (VC_M) and other variable costs (VC_O), all are expressed in Yuan:

$$VC = VC_L + VC_M + VC_O \qquad\qquad \text{Eq. 5.21}$$

The labor cost is calculated based on the labor and on the unit labor cost (UVC_L, in Yuan Man-Day^{-1}):

$$VC_L = L * UVC_L \qquad\qquad \text{Eq. 5.22}$$

The unit labor cost changes as a function of the per capita income growth rate (r_y, in year^{-1}, defined in Table 5.2):

$$dUVC_L / dt = r_y * UVC_L \qquad\qquad \text{Eq. 5.23}$$

The maintenance cost is determined as a fraction (m_f, defined in Table 5.2) of the capital (K) as defined in a local economic survey (de Wit et al., 2008) and following Eq. 5.24:

$$VC_M = m_f * K \qquad\qquad \text{Eq. 5.24}$$

The other variable costs include costs of feeding, seeding and interest on loan among others. This variable is calculated based on the shellfish production (Q, in kg, Eq. 5.9) and on the unit cost of other variables (UVC_O, in Yuan kg^{-1}):

$$VC_O = Q * UVC_O \qquad\qquad \text{Eq. 5.25}$$

The unit cost of other variables changes as a function of the price growth rate (r_p, in year^{-1}, defined in Table 5.2):

$$dUVC_O / dt = r_p * UVC_O \qquad\qquad \text{Eq. 5.26}$$

Net profit sub-model:

The dynamics of net profit (NP, in Yuan, Eq. 5.27) are determined by the revenue (derived from the dynamics of production output and price) and the total cost incurred (which includes fixed and variable costs):

$$NP = (Q * P) - (FC + VC) \qquad\qquad \text{Eq. 5.27}$$

Marginal cost and revenue sub-model:

For each economic timestep the marginal cost (MC, in Yuan kg^{-1}) is determined as the increase in total cost that results of producing an additional unit of shellfish:

$$MC = \Delta TC / \Delta Q \qquad\qquad \text{Eq. 5.28}$$

For calculation of MC we consider an output increment of one kg of shellfish ($\Delta Q = 1$ kg). Thus, for every one unit of additional Q, Eq. 5.28 reduces to:

$$MC = TC_{Q+1} - TC_Q \qquad\qquad \text{Eq. 5.29}$$

Where, TC_{Q+1} (in Yuan) is the total cost to produce $Q+1$, and TC_Q (Eq. 5.17) the total cost as calculated previously for Q. TC_{Q+1} is calculated using the production cost sub-model (Eq. 5.17 to Eq. 5.26) to compute the cost of the inputs (labor and capital) needed to produce $Q+1$. On the other hand, the required labor and capital to produce the additional output are determined by multiplying one kg of shellfish by the inverse of marginal productivity of labor

($\frac{1}{MP_L}$ which expresses as Man-Days kg^{-1}) and the inverse of marginal productivity of capital

($\frac{1}{MP_K}$ which expresses as Yuan kg^{-1}), respectively.

Assuming that the shellfish farmers are price takers, the marginal revenue (*MR*, in Yuan kg^{-1}) was equated to the price of shellfish (*P*, in Yuan kg^{-1}, Eq. 5.8).

$$MR = P \qquad\qquad\qquad\qquad\qquad\qquad \text{Eq. 5.30}$$

Both marginal cost (*MC*) and marginal revenue (*MR*) are used by the decision model for calculation of the profit maximization criteria.

Decision component

The decision component is the engine of the MARKET model. This simulation block determines the production in the following year, therefore driving both the ecological and economic components. In the MARKET model it is assumed that the farmers' decision is based on (i) the profit maximization, (ii) the gap between demand and supply, and (iii) the available area for aquaculture activities, i.e. the physical limits. Each of the three criteria is further detailed below:

i) Profit maximization: the local farmers are assumed to be perfectly rational and that their interest in aquaculture production is to maximize individual profit. Therefore, they will aim to increase production only up to an output level whereby marginal cost equals marginal revenue. In this analysis, the farm managers are assumed to have knowledge on the cost and demand functions facing the shellfish production and about other actors in the system. Although none of these conditions are likely to be met in reality, these provide a baseline economic decision-making rule to maximize profit in order to test the application of the MARKET model. Both marginal cost and marginal revenue values are provided by the economic model (Eq. 5.29 and Eq. 5.30). If the marginal revenue is greater than the marginal cost (*MR* > *MC*) the decision model defines an increase in desired production for the next period and the inverse occurs when the marginal revenue is less than the marginal cost (*MR* < *MC*). If the marginal revenue equals marginal cost (*MR* = *MC*) then the model decides to maintain the desired production for the next period at current production level.

ii) Demand / supply gap: is calculated as the difference between the local demand and the shellfish production, both given by the economic model (Eq. 5.7 and Eq. 5.9). It indicates whether the demand is met by production (if $Q \geq LD$), or if the market can absorb an increase in production (if $Q < LD$).

iii) Physical limit: the farmers can expand up to a maximum available area for aquaculture (*A* = *MaxA*). In the model the maximum cultivation area is a parameter of the ecological component (Table 5.2). This area should be defined by ecosystem managers based on a zoning policy decision or simply based on the physical limits of the ecosystem.

The decision on whether to increase, decrease or maintain production is simulated based on the decision rules shown in Figure 5.3. If all the three criteria are favorable to increase production (*MR* > *MC* AND *LD* > *Q* AND *A* < *MaxA*), the desired production increases at a percentage of current year production. If the current profitability is negative (*MR* < *MC*) then the decision model defines a decrease in the desired production which is proportional to the current year production. If none of the previous conditions are met and if the maximum profitability is achieved (*MR* = *MC*), or demand is met (*Q* ≥ *LD*) or the maximum cultivation area is attained (*A* ≥ *MaxA*) then the decision model maintains the current year production.
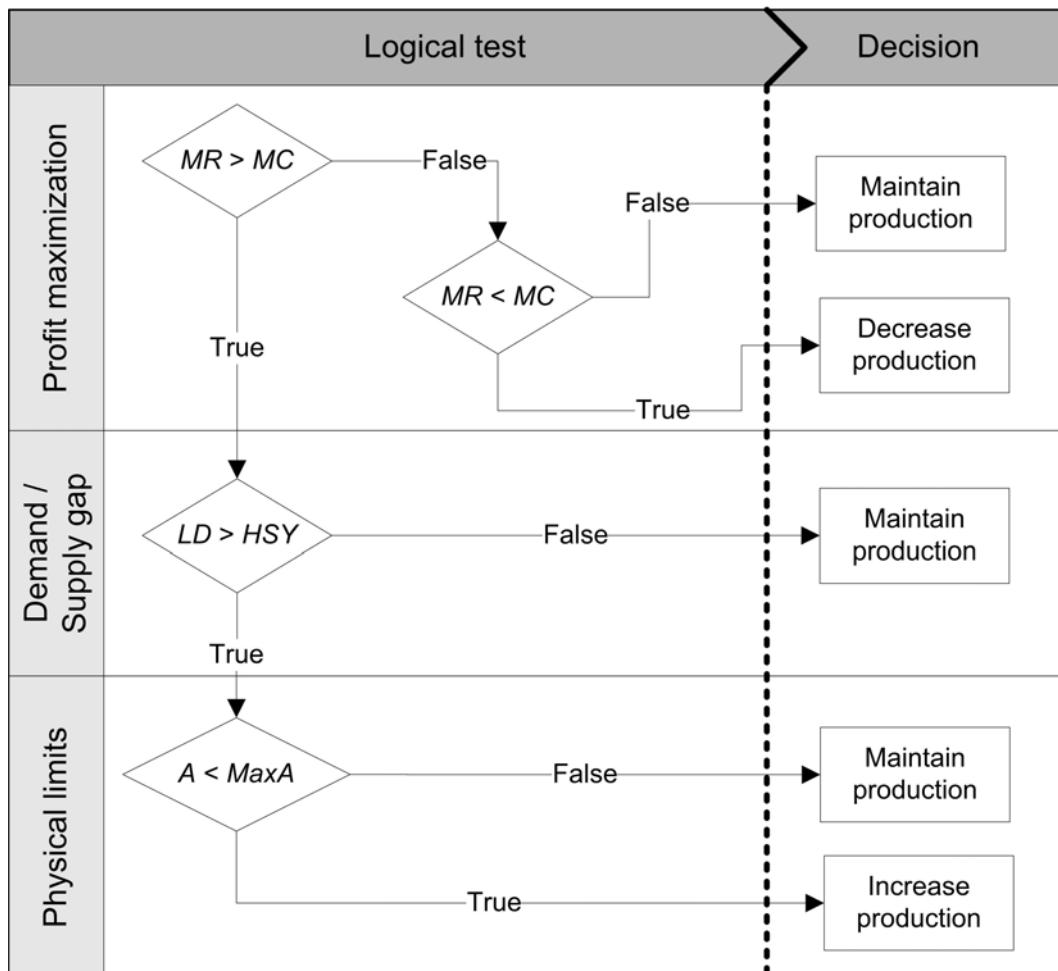
MR: Marginal revenue; MC: Marginal cost; LD: Local demand; HSY: Harvested shellfish yield; A: Cultivated area; MaxA: Maximum cultivation area.

Figure 5.3. Decision model implementation: logical test for decision about increase, decrease or maintaining current production.

The change in the quantity that aquaculture managers want to produce in the next cycle, i.e. the desired change in production (R_{CQ}, in kg year^{-1}), is calculated as a fraction of current year production by means of Eq. 5.31:

$$R_{CQ} = Q * r_{cq}$$
<div align="right">Eq. 5.31</div>

Where, Q (in kg) represents the current year production and is calculated in the economic model (Eq. 5.9); r_{cq} (in year^{-1}), is the annual change rate in production and is conditioned by the decision whether to increase, decrease or maintain production (according to Figure 5.3 and as explained above). Depending on the decision taken r_{cq} is given as:

 (i) If decision is to increase production, then the rate of change in production is 10 % per year of current production ($r_{cq} = 0.1$ year^{-1});

(ii) If decision is to decrease production, then the rate of change in production is -30 % per year of current production ($r_{cq} = -0.3$ year^{-1});

 (iii) If decision is to maintain production, then the rate of change in production is 0 % per year of current production ($r_{cq} = 0.0$ year^{-1}).

Further research is needed to understand how this decision is normally taken in the real world in order to improve the definition of the rate of change in production.

The desired production for the next cycle (DQ, in kg) is then given by current production (Q) and by the desired change in production for the next cycle ($R_{CQ}*tp$):

$$DQ = Q + R_{CQ} * tp$$
<div align="right">Eq. 5.32</div>

Where, tp (in year) is the shellfish production cycle period (defined in Table 5.2).

Model assessment and scenario definition

At this stage of development and given the deterministic nature of the MARKET model, it cannot incorporate the randomness involved in decisions by individual farmers. In addition, it does not integrate the complex dynamics that govern for instance a policy change that decides a shift from shellfish to finfish or macroalgal production. In order to validate the MARKET model at that level, a very specific dataset would be required: a data series of both economic production and environmental factors for a given ecosystem where the main changes in aquaculture production are only constrained by the ecological and economic factors in a perfectly rational way.

The applicability of the model was thus assessed by comparing the general trends of simulation results with the expected outcomes according to standard economic theory for consumption and production and according to ecological economics theory: It is expected that shellfish is a normal good, meaning that rising income will lead to rising demand and vice-versa. It is also expected that a rising demand will lead to an expansion in farming activities up to a level that is both economically profitable and sustained by the ecosystem. In order to support the comparison with expected results a set of scenarios was defined (Table 5.4) aimed to test the model response to changes in price and income growth rates, and maximum cultivation area. Another reason to run these scenarios was to demonstrate the capabilities of the MARKET model to simulate relevant management scenarios. For instance scenario 3 exemplifies a management decision to set a lower maximum cultivation area as compared to the standard scenario. Scenario 4 develops this by introducing a compensation measure to farmers whereby the reduction of the maximum cultivation area is followed by a price increase.

Table 5.4. Scenarios analyzed in the MARKET model.

Scenario	Price growth rate (% per year): r_p	Income growth rate (% per year): r_v	Maximum cultivation area (% of bay area): *MaxA*
Standard	2 %	10 %	83 % of bay
Scenario 1	1 %	Standard	Standard
Scenario 2	Standard	5 %	Standard
Scenario 3	Standard	Standard	42 % of bay
Scenario 4	3 %	Standard	42 % of bay

RESULTS

The standard simulation results indicate that the production is limited by the maximum cultivation area in the 27th year (Figure 5.4b). Afterwards, the economic limitation to production (marginal cost equals marginal revenue) is experienced after 10 years in the 37th year (Figure 5.4c). These two limitations in production are visible in the net profit curve shown in Figure 5.4d.
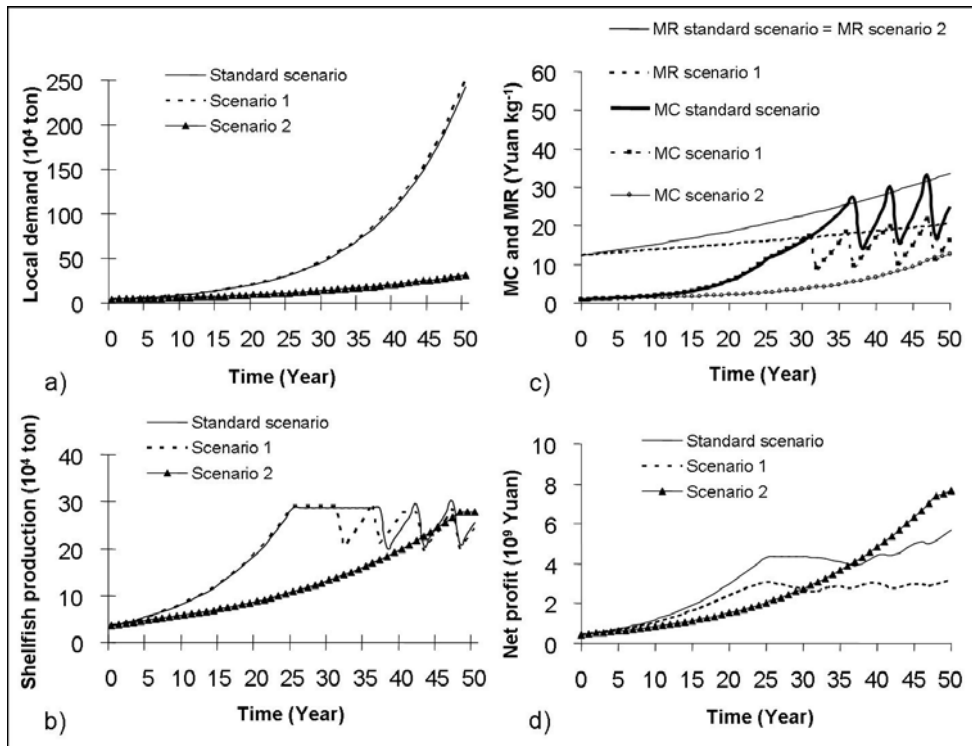
Figure 5.4. Simulation results for standard scenario, scenario 1 and scenario 2 for: a) local demand (*LD*), b) shellfish production (*Q*), c) marginal cost and revenue (*MC* and *MR*) and d) net profit (*NP*).

In scenario 1 the reduction of half the price growth rate (r_p = 1 % per year, Table 5.4) is tested. The economic limit to production (marginal cost equals marginal revenue) in this scenario is reached sooner than in the standard and other simulations (Figure 5.4c). The net profit also decreases (Figure 5.4d). This is because the price is a major determinant in the profitability of the aquatic operations. Therefore, with other variables growing at the rate of the standard simulation, the profitability decreases.

In scenario 2, a decrease of the per capita income growth rate to half the standard simulation (r_y = 5 % per year, Table 5.4) is tested, while the values of price growth rate and cultivation area are the same as in the standard simulation (Table 5.4). The income growth rate does influence the demand: with a lower income growth rate, the demand in scenario 2 is lower than in the standard scenario (Figure 5.4a) and the exploitation rate is therefore lower (Figure 5.4b). As a result of the reduced harvest, there is less pressure on the aquatic resources. Although the demand is lower than in the standard simulation, in the long run the shellfish production in scenario 2 presents higher profits than in the standard simulation: the marginal cost is less than marginal revenue in the entire simulation and from 40th year, the net profit in scenario 2 diverges beyond the standard simulation (Figure 5.4d). This outcome is further explored in the discussion section.

In scenario 3, a decrease in the maximum cultivation area (*MaxA* = 42 % of total bay area, Table 5.4) was tested. This can simulate for instance a management decision of allocating

more area of the bay for other purposes such as tourism or navigation. Up to the point where the physical limit to production is achieved, which occurs at the 18th year, all the variables (including net profit) for standard scenario and scenario 3 coincide (Figure 5.5), given that the only difference between these two scenarios is the maximum cultivation area. From the 18th year, the limitation in the production area reduces the amount of harvestable biomass in scenario 3 compared with standard scenario (Figure 5.5b). This further leads to reduced profits in scenario 3 compared with standard (Figure 5.5d). However, it is interesting to note that due to the lower production over time (from 18th year) the marginal cost increases at a lower rate causing a decrease in profitability ($MC = MR$) only at the 47th year, whereas in the standard scenario marginal cost equals marginal revenue in the 37th year.
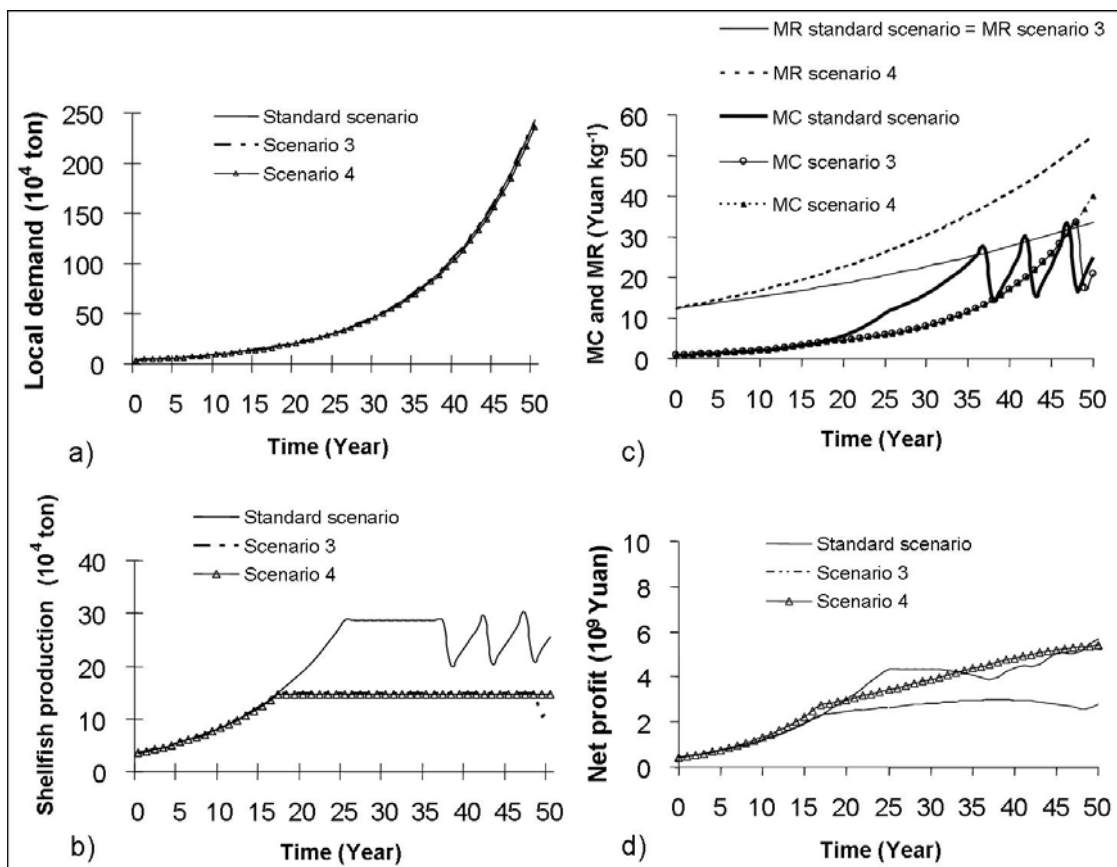
Figure 5.5. Simulation results for standard scenarios, scenario 3 and scenario 4 for: a) local demand (*LD*), b) shellfish production (*Q*), c) marginal cost and revenue (*MC* and *MR*) and d) net profit (*NP*).

Scenario 4 combines the reduction of maximum cultivation area (also simulated in scenario 3) with an increase in the price growth rate (r_p = 3 % per year, Table 5.4). This scenario can exemplify a policy measure to compensate for the limitation on the aquaculture expansion potential. The outputs for this scenario show that from the 18th year the shellfish production is less than the amount simulated in the standard scenario (Figure 5.5b), however, given the

increase in price growth rate, the profits are sustainable in the long run: the marginal cost is less than the marginal revenue in the entire simulation (Figure 5.5c) and the net profit is in the same range as the net profit for the standard simulation (Figure 5.5d). The shellfish production for scenario 4 and scenario 3 are also similar except with a slight difference for scenario 3 in the 47th year. This is because at that point, the marginal cost for scenario 3 equals marginal revenue, which implies a decision to decrease production. This occurrence is mainly explained by the lower price growth rate for scenario 3 than for scenario 4.

DISCUSSION

A comparison of the model results for all the simulations, as discussed below, indicates that the MARKET model followed the expected trends regarding the standard economic theory for consumption and production. Likewise the interrelationship between net profit, physical space and food limitation was modeled successfully, according to ecological economics theory.

Since the income growth rate in scenario 2 (r_y = 5 % per year) is half than for other scenarios (r_y = 10 % per year), the local demand in scenario 2 is significantly lower (Figure 5.4a and Figure 5.5a). On the other hand, given that the model assumes price as inelastic, the proportional change in local demand due to changes in price growth rate is lower: scenario 1, where the price growth rate is lowest (r_p = 1 % per year, Table 5.4), when compared to scenarios that consider an equal income growth rate of 10 % per year (standard scenario, scenario 3 and scenario 4, Table 5.4) shows a slightly higher local demand (Figure 5.4a and Figure 5.5a).

In the scenario with a lower demand (scenario 2) the harvested shellfish was reduced (Figure 5.4b). In the long run, production was limited by the maximum cultivation area in all the scenarios (Figure 5.4b and Figure 5.5b). This outcome indicates that the current annual rates for shellfish demand are not sustainable over extended periods of time in this ecosystem. From the ecosystem perspective this restriction was only caused by the physical limitation given that the ecosystem model results indicate that the food available suffices to yield the production up to the maximum cultivation area of 83 % of the Xiangshan total area. Nevertheless, this occurs with a slower scope for growth as described in Eq. 5.1.

Following the physical limitation, the standard scenario, scenario 1 and scenario 3 experienced an economic limitation to production (reached when marginal cost equals marginal revenue, shown in Figure 5.4c and Figure 5.5c), while scenario 2 and scenario 4 did

not. The explaining variables were a combination of price, production level and factors affecting the production cost: The comparison of scenario 1 (r_p = 1 % per year) with the standard scenario (r_p = 2 % per year), and of scenario 3 (r_p = 2 % per year) with scenario 4 (r_p = 3 % per year) highlighted the impact that a lower price growth rate has on economic limitation to production: in scenario 1 it is reached sooner than in the standard scenario and in scenario 3 it is reached at 47th year while in scenario 4 it is never reached (Figure 5.4c and Figure 5.5c, respectively). The comparison of scenario 3 with the standard scenario indicated that the lower production level in scenario 3 caused the marginal cost to equalize with the marginal revenue later than in the standard simulation (Figure 5.5c). In scenario 2, where the only difference from the standard scenario is a lower income growth rate and consequent lower demand, the economic limitation to production ($MC = MR$) was not reached, while it did occur in the standard scenario (Figure 5.4c). The main explanation is the lower production level (caused by the lower demand) together with the effect of the lower income in the cost of labor for the shellfish production (as unit labor cost changes as a function of the per capita income growth rate in the model).

An interesting outcome of scenario 2 was that although the lower income resulted in a lower demand, it also caused a decrease in production cost which resulted in a net profit dynamics that in the long run exceeded the net profit of the standard scenario (Figure 5.4d). This scenario raises the issue that a lower demand does not always imply a corresponding decrease in net profit. This is a topic for further research in the context of economic policy mitigation plans: MARKET or other similar models can support a more in-depth analysis, e.g., to determine where to target public intervention. In this case, if any public intervention took place, it should focus on the promotion of social security (due to the lower income), while private fish farmers were protected from the lower demand. In the remaining scenarios, the net profit dynamics followed the expected results: the decrease in price caused a decrease in profits and vice-versa, as shown by comparison of scenarios that differ only in price (the standard scenario with scenario 1 in Figure 5.4d, and scenario 3 with scenario 4 in Figure 5.5d); the reduction of the production level due to the reduction of the cultivation area also lead to a decrease in profits (as tested in scenario 3 compared with standard scenario, Figure 5.5d). For all the simulations performed within our case study, the profits of shellfish production were assured.

CONCLUSIONS

The MARKET model allows for an integrated dynamic analysis of (i) the demand for mariculture products, (ii) economic production and cost limiting factors, (iii) the biological growth of aquatic resources, (iv) interactions with the environmental conditions and (iv) the spatial limitations of culture in coastal ecosystems. Our approach can contribute to mariculture management and for implementation of an ecosystem approach to aquaculture (EAA).

Simulation of shellfish production in a Chinese embayment was chosen as a case study illustrating the implementation of the MARKET model. A key feature of the model implementation was to incorporate the different time scales at which the ecological and economic systems function. In this study, we have used several management scenarios to show that the model reproduces the expected trends and provides further insights. In all the scenarios, production in the long run does not meet increasing demand. In this case study the physical limitation of the bay was the first limiting factor for all the scenarios, that is, space is expected to impose limitations on production before it becomes less profitable to expand production. Overall, the MARKET model can help to understand the succession of the limiting factors in mariculture industry and whether the production can meet the demand for aquatic resources.

The MARKET model can be widely applied, provided that case-specific information exists on shellfish demand, price, income, production functions, physical area available for cultivation, and environmental conditions that have an effect on the growth of aquatic resources and are affected by its production. It is recommended that future MARKET model developments include: (i) an improvement of the decision model, in particular for decisions by farmers about changes of production level, (ii) explicit dynamic coupling with an ecosystem model, and (iii) implementation for other aquaculture species and culture practices, especially those that normally raise more concerns related with environmental management, such as finfish monoculture.

Chapter 6. **Integration of ecosystem-based tools**

Context

The preceding *Chapters 2 to 5* present different methodologies for (i) integrated simulation of coastal ecosystems, (ii) ecological-economic assessment of the effectiveness of response actions, and (iii) dynamic ecological-economic modelling of aquaculture production in coastal ecosystems.

Summary

This chapter presents the integrated application of these and a wider spectrum of ecosystem-based tools for coastal ecosystem research and management, such as geographic information systems, remote sensing and economic valuation methods. This chapter illustrates the application of such a set of tools to support coastal management, using a coastal lagoon located in southwest Europe as a case study.

This chapter corresponds to the published manuscript:

Nobre, A.M and Ferreira, J.G., 2009. Integration of ecosystem-based tools to support coastal zone management. Journal of Coastal Research, SI 56: 1676-1680.

(For consistency with published version this chapter is written in American English)

Integration of ecosystem-based tools to support coastal zone management

INTRODUCTION

Coastal zones exhibit complex interactions at different levels: (i) they are under influence of a great variety of pressures at the interface between land and sea, (ii) they are subject to feedback effects between natural and human systems (Turner et al. 2003), (iii) they exhibit complex relationships between the physical and biological processes – in particular estuaries are characterized by complex ecological feedbacks (Bergamasco et al., 2003). Coastal zones are highly productive and provide significant direct and indirect socio-economic benefits, e.g. food, biodiversity, nutrient cycling, climate regulation, recreation, culture and amenity (MA, 2005). As a result coastal zones concentrate 40% of the world population and 61% of world's total GNP (MA, 2005). However, their misuse is causing degradation and consequently decreases of the services that these coastal ecosystems deliver (MA, 2005). The Millennium Assessment (MA, 2005) also indicates impact on human health: of the annual cost due to coastal water pollution (16 billion USD) a large proportion is related to human health.

To address coastal zone problems, ecosystem-based management (EBM) and integrated coastal management (ICM) are required (Browman and Stergiou, 2005; Murawski et al., 2008). ICM is a well established approach (GESAMP, 1996; Cicin-Sain and Knecht, 1998) defined as a dynamic process for the management of the use, development and protection of the coastal zone (Murawski et al., 2008). It consists of an integrated approach from different perspectives (GESAMP, 1996). EBM is an emerging scientific consensus (Murawski et al., 2008), defined as the use of the best available knowledge about the ecosystem to manage marine resources (Fluharty, 2005). The integration of (i) science with management and (ii) natural with social sciences, is critical for effective governance of coastal zones (Cheong, 2008).

The role of science is to provide the insights and information required to support managers and decision makers (GESAMP, 1996; Browman and Stergiou, 2005). This implies the use of scientific applications that enable (i) the understanding of biogeochemical processes, (ii) interaction of ecological and socio-economic components, and (iii) synthesis and communication of complex outputs to managers. The integration of tools, such as geographic information systems (GIS), ecological modeling of catchment and coastal systems, economic

valuation methods and integrated environmental assessment (IEA), can empower coastal managers with a scientific framework for sound decision-making.

The objective of this paper is to review the most used tools for coastal ecosystem research and how they can empower coastal managers for (i) performance evaluation of previously adopted responses and (ii) definition of policies. We provide examples, where possible, of the application of these tools for management of Ria Formosa, a coastal lagoon in the South of Portugal.

GENERAL APPROACH

Integration of ecosystem-based tools

Integrated approaches for environmental management including of coastal ecosystems, have in common (i) the integration of the environmental and socio-economic systems, and (ii) the communication between the scientific, management and local communities (Harris, 2002; Greiner, 2004; Chang et al., 2008; Tompkins et al., 2008).

Figure 6.1 synthesizes most common tools used for integrated coastal research and management and the links that are normally established among them. These tools can be used isolated or combined. The inclusion of "System monitoring" in the diagram, highlights the fact that all the tools require data to be applied. The components of the integrated approach (depicted in Figure 6.1) are detailed herein in separate sections. For each tool the relevance for coastal ecosystem management was described and illustrated using the same ecosystem. Whenever possible the integration among tools is exemplified.
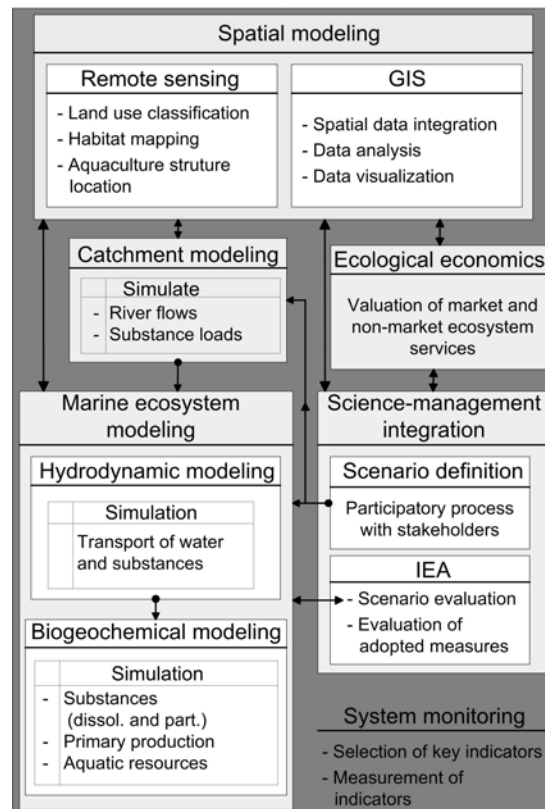
Figure 6.1. Integration of tools for coastal ecosystem management.

Case study

Ria Formosa was chosen as a case study due to the considerable interaction between the ecological and the socio-economic systems of this coastal zone: (i) on the one hand this ecosystem has an environmental importance recognized by several international conventions and directives (e.g. RAMSAR, Birds and Habitats EU Directives) and is classified as a Natural Park by the Portuguese legislation, on the other hand (ii) Ria Formosa and its catchment support several economic activities that represent the main source of employment and income in the region. The main economic activities include extensive bivalve aquaculture, tourism, agriculture and livestock, manufacturing industry, fish aquaculture and salt production.

REMOTE SENSING

Understanding the upstream processes that exert a pressure on coastal zones is a very important component. Remote sensing (RS) can provide valuable information, namely for: land use mapping, altimetry, drainage network and other watershed data required for hydrological modeling (Pandey et al., 2005). RS can also be particularly useful for mapping habitats within coastal systems, e.g. wetlands and mangroves (Green et al., 1999) as well as to monitor key surface water quality variables (Chen et al., 2004). Green et al. (1999) and Chen et al. (2004) provide detailed guidance about the use of RS for ICM. The major strength of RS is that it allows (i) spatially extensive surveys, (ii) monitoring of past situations and (iii) multitemporal sensing of e.g., habitat coverage and condition (Lillesand and Kiefer, 2000). Such information forms the basis for evaluation of ecosystem services, resource conservation status or pressure evolution over time.

In the Ria Formosa, RS was used to classify the catchment land cover (Figure 6.2). The enhanced nearest neighbor algorithm was used for supervised classification of a Landsat-7 TM scene (30m resolution). Statistical validation of the supervised classification was carried out by computing a confusion matrix (Lillesand and Kiefer, 2000) using surveyed test zones. The Khat statistic, which provides an indication of classification performance (Lillesand and Kiefer, 2000), indicates that classification obtained is 84% better than one resulting from chance.
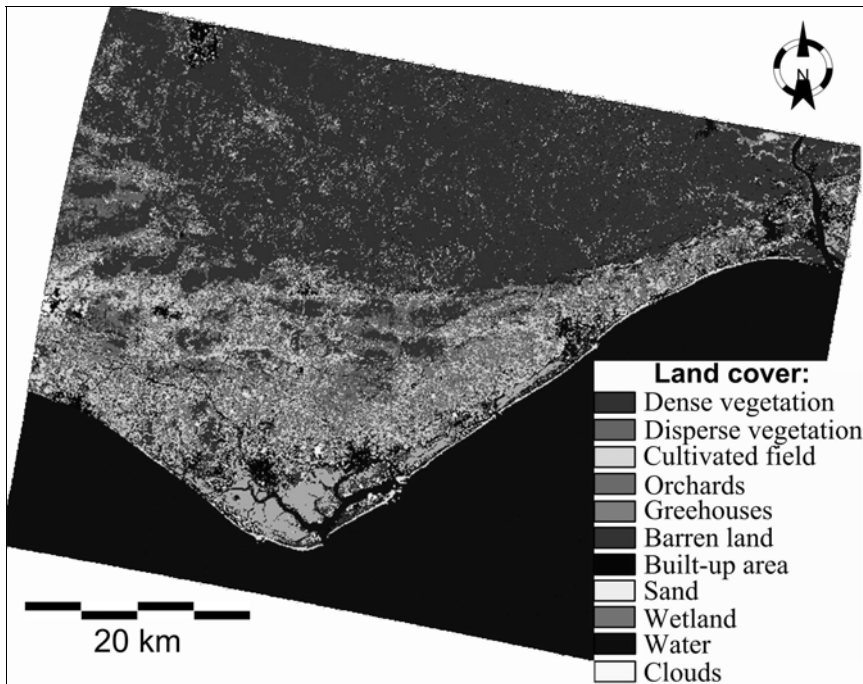
Figure 6.2. Ria Formosa land cover classification results.

GEOGRAPHIC INFORMATION SYSTEMS

GIS can be used for spatial data integration (e.g. bathymetry, sampling stations, habitat area, catchment land use), data analysis (e.g. calculation of waterbody volume and area, thematic mapping such as interpolation of sampling station data, zoning) and data visualization (e.g., of the generated thematic maps). These capabilities make it a useful tool for ICM (Douven et al., 2003; Tolvanen and Kalliola, 2008) either as a data generator (if used to extract data for other tools, e.g. setup ecological models) or as an 'end in itself' (if used for communicating information to managers). GIS can integrate with other applications as e.g., ecological models, by offline coupling, whereby the model receives some of its input data from the GIS, or using a tighter integration whereby both the model and GIS share a common interface and communicate directly (Fedra, 1996). Sardá et al. (2005), illustrates the integration of data into GIS, and its use for data processing and visualization targeted to managers. The use of embedded GIS basic functions into Decision Support Systems (DSS) can empower managers by enabling to manipulate, display and analyze spatial data and models (Fedra, 1996). Another example of GIS use for ICM is to support marine spatial planning for the implementation of relevant legislation (Gilliland and Laffoley, 2008; Maes, 2008). Examples are provided by (i) Cheong (2008) for the delineation of Exclusive Economic Zones required by the Law of the Sea Convention of 1982, (ii) Ferreira et al. (2006) for the division of

168

transitional and coastal waters into waterbodies as determined by the Water Framework Directive (WFD, 2000/60/CE) and (iii) Boyes et al. (2007) for zoning based on legislation applicable within the Irish Sea.

For Ria Formosa there are several examples of the use of GIS for ICM, namely (i) zoning of Ria Formosa for the application of WFD as described by Ferreira et al. (2006) and (ii) identification of conflicting uses by the Natural Park authority (ICN, 2005). Existing spatially distributed data (either produced by research institutes, universities or local managers) could be compiled for the development of a DSS to support local managers to implement existing and develop future plans.

CATCHMENT MODELING

Integrated land use catchment modeling emerged as a requirement from policy makers and managers to understand the feedback between changing land use and changing environmental conditions (Veldkamp and Verburg, 2004). Several studies were developed to understand the effects of land use policy on the environmental and socio-economic systems (Veldkamp and Verburg, 2004; Macleod et al., 2007). Furthermore, information about catchment pressures is of paramount importance to simulate the downstream coastal ecosystems (Neal et al., 2003). In particular, estimates of substance loads entering from the catchment are required to simulate the biogeochemical conditions of the coastal water bodies. Depending on objectives and available data simpler or more complex approaches can be used: direct estimation techniques, simple export coefficient methods or more complex catchment models (McGuckin et al., 1999; Letcher et al., 2002; Endreny and Wood, 2003; Pandey et al., 2005; Wade et al., 2005). The advantage of catchment models is that they allow for scenario simulation of catchment land use. This can be integrated with coastal ecosystem models to determine the impact of the catchment loads.

In the case of Ria Formosa, the runoff is concentrated in the winter months (ca. 71%). Loads entering into this coastal system have been calculated based on river water quality and flow data together with waste water discharge data (MAOT, 2000; Ferreira et al., 2003). However, it is desirable to apply catchment models to determine daily nutrient and sediment loads as well to test relevant management scenarios and respective impacts.

COASTAL ECOSYSTEM MODELING

For the simulation of estuarine and coastal ecosystems there are a large number of models of varying complexity, regarding spatial and temporal scales, components of the ecosystem and

processes included (Fulton et al., 2003). Model development normally depends on the research objectives. Recently the use of ecosystem modeling to assist ICM became an emerging requirement (Fulton et al., 2003; Nobre et al., 2005). In particular modeling can be useful to overcome data limitations and to simulate scenarios. For instance ecological modeling can play an important role for the implementation of the WFD (de Jonge, 2007). The development of ecological models usually implies integration with some of the other tools in review, at least for the model setup and forcing with boundary conditions (Fedra, 1996; Neal et al., 2003).

In the Ria Formosa several models have been applied at different levels, namely a detailed hydrodynamic model and an ecosystem box model that simulate transport, nutrient cycling, primary production and secondary production (bivalves) (Nobre et al. 2005). The ecosystem model was run to simulate different scenarios relevant for eutrophication management.

ECONOMIC VALUATION

Ecosystem valuation aims to estimate the total and marginal value of the ecosystem services (both the market and the non-market components). There are several difficulties in placing an economic value on natural assets and specially of calculating an absolute economic value of ecosystems (Costanza et al., 1997; Ledoux and Turner, 2002). Nevertheless, it is of crucial importance that an effort is made to calculate the changes caused on human welfare due to the changes that affect ecosystem functioning (Costanza et al., 1997). Valuation can be regarded as a policy tool in the sense that it enables an accounting of ecosystem goods and services, together with the market services, in decision-making and management of coastal systems (Barbier et al., 1996; Costanza et al., 1997; Ledoux and Turner, 2002). There is a variety of economic valuation methods broadly categorized either as revealed preference methods (such as hedonic pricing, travel cost or replacement cost) or as stated preference methods (such as contingent valuation and choice experiment), each with advantages and limitations depending on the application. Ledoux and Turner (2002) and Birol et al. (2006) provide a review of the application of such methods for water resources management.

Given the ecological importance of the Ria Formosa and the benefits it generates, it would be appropriate to conduct such a valuation exercise. Considering the economic activities that depend on this ecosystem (aquaculture, fisheries, tourism and salt production) an average benefit of 338 million Euros year^{-1} (2000 prices) is estimated. This value corresponds to the average net profit generated by these activities for the period between 1980 and 1999 (Nobre, 2009). Updated and more detailed studies are required to capture other direct and indirect use

values. Particularly important is to estimate the values associated with the wetland area (ca. 17 % of Ria Formosa Natural Park area) given the range of benefits this type of ecosystem provides, i.e. food resources, flood water retention, groundwater recharge/discharge and nutrient abatement (Acharya, 2000). Detailed guidelines to carry out such studies are provided by Barbier et al. (1996). In order to estimate an approximate range, the wetlands potential value was evaluated using values provided by Ghermandi et al. (2008), ca. 100 to 10 000 USD (2003) ha-1 yr-1, based on an extensive review of economic value estimates of wetlands worldwide. The estimated value of wetlands in Ria Formosa ranges between 0.30 and 29.54 million Euro yr-1 (2000 prices) (USD conversion to Euros was based on the Consumer Price Index rate from the Bureau of Labor Statistics and currency conversion from the IMF).

ASSESSMENT METHODOLOGIES

Integrated Environmental Assessment (IEA) methodologies can be broadly defined as interdisciplinary approaches targeted to guide decision-makers and managers about environmental problems, and in more general terms for natural resources management (Toth and Hizsnyik, 1998). IEA methodologies are by themselves integrative tools (Cheong, 2008) that promote the interaction of ecological and socio-economic disciplines or simply the synthesis of complex information to managers. The Drivers-Pressure-State-Impact-Response (DPSIR) is one such tool that has been widely applied to synthesize natural and socio-economic sciences for marine policy formulation (Cheong, 2008) and for ICM (Ledoux and Turner, 2002). For the application of assessment approaches the selection of key indicators is critical (Håkanson and Blenckner, 2008). Borja et al. (2008) reviews existing methodologies to assess ecosystem ecological status in order to address legislation adopted worldwide for management of ecological quality or integrity.

Ferreira et al. (2003) exemplifies the use of an IEA methodology to inform managers about eutrophication status in Ria Formosa. The work carried out concluded that there is a moderate low eutrophic condition, for which the main symptom identified is periodic blooms of macroalgae in some locations of Ria Formosa (Ferreira et al., 2003). Further research investigated the effects of nutrient loading scenarios on the eutrophic state of Ria Formosa by coupling the eutrophication assessment methodology with the ecosystem ecological model (Nobre et al. 2005). The eutrophication assessment methodology used was the USA National Estuarine Eutrophication Assessment (NEEA) method and its successor the ASSETS screening model (Bricker et al., 2003).

The case study presented by Nobre (2009) exemplifies how an IEA approach could support the strategic management of Ria Formosa natural resources from both ecological and socio-economic perspectives: The comparison of drivers, pressures and ecosystem state in two different periods (1980/85 and 1995/99) indicates that although there was a significant management response (namely the construction of waste water treatment plants), the negative economic impacts represented 80% to 220% of the response cost (Nobre, 2009). The decrease of the economic benefits was mainly due to the decrease of bivalve production, which is believed to be related to the appearance of a parasite (Campos and Cachola, 2006). Aquaculture production in Ria Formosa presently accounts for 47% of the Portuguese mariculture products and it is estimated that bivalve aquaculture alone is responsible for the direct employment of 4 500 people (ICN, 2005) or up to 10 000 according to unofficial estimates (Campos and Cachola, 2006). *Ruditapes decussatus* is the local clam species and its production in Ria Formosa is highly significant (ca. 90% of Portuguese production, in 2001). This species is highly priced (Matias et al., 2008), however, it is being displaced by the Manila clam *Ruditapes philippinarum* (Campos and Cachola, 2006). Notwithstanding the incentives for conservation of local clam, the stipulated activities in Ria Formosa Natural Park Management Plan preview for bivalve related management an amount that represents 1.9% of planned total budget (ICN, 2005). Results and information synthesized herein, suggest that is advisable to invest in the proper management of bivalve aquaculture and natural beds with a special emphasis on the seeding procurement or development of local hatcheries, which might have a positive effect on (i) mitigating disease introduction (Nobre, 2009), (ii) limiting the introduction of alien species (Campos and Cachola, 2006) and (iii) on *Ruditapes decussatus* seed availability (Matias et al., 2008).

It is advisable that the relevant authorities should define a set of indicators to monitor effectiveness of the goals established in the several management plans that exist for this ecosystem, the most important being: (i) Management Plan of Coastal Zone between Vilamoura e Vila Real de Santo António approved through Resolution No. 103/2005 of 27 June 2005, focus on the strip of land 500m wide from the seawater baseline and on the marine area limited by the 30m bathymetric line, (ii) Ria Formosa Natural Park Management Plan approved through Regulatory Decree No. 2/91 of 24 January 1991 and currently is under revision, focus on Ria Formosa lagoon ecosystem, and (iii) Hidrographic basin plans of the Algarve streams approved through Regulatory Decree No. 12/2002 of 9 March 2002, focus on the drainage basin of several streams encompassing Ria Formosa catchment area.

CONCLUDING REMARKS

This paper describes a range of tools that can be used to provide coastal managers with scientifically based information for performance evaluation of previously adopted responses as well as future management policies. In order to capitalize on the use of these tools and their integration a tighter iterative collaboration at the ecosystem level between managers and scientists is required, whereby the former should provide the latter with specific management objectives or goals for conservation of a given ecosystem and the services it delivers (Rosenberg and McLeod, 2005). This approach asks scientists for: (i) suggestions about how to achieve those objectives within budget and timeframe constraints, and (ii) monitoring tools to assess the performance of policies adopted. Scientists engaged in this process should focus on addressing the management needs and communicating the information in an understandable and accessible away (Tribbia and Moser, 2008). Nevertheless, there are always uncertainties associated with scientific knowledge and predictions. These should be acknowledged, particularly with respect to accuracy, but without holding the ecosystem-based management process.

Chapter 7. General discussion

This chapter presents a general discussion on the work developed in this thesis. This discussion consolidates the outcomes of the work presented in *chapters 2 to 6*. The first part discusses the methodological developments, both individually and in terms of how they complement each other. The second part discusses the use of several study sites and the main conclusions for each. The third part presents final conclusions.

7.1 Integrated ecological-economic modelling and assessment approach

The integrated ecological-economic modelling and assessment approach consists of complementary approaches developed to assess the coastal ecosystem at different scales and translate scientific-based information into meaningful knowledge for managers. Some of the developments contribute novel methodologies for integrated coastal zone management (ICZM) and, in particular, support an ecosystem approach to aquaculture. The main methodologies include:

- A multilayered ecosystem model: simulates the cumulative impacts of multiple uses of coastal zones. This approach combines the simulation of the biogeochemistry of a coastal ecosystem with the simulation of its main forcing functions, such as catchment loading and aquaculture activities;

- A coupled ecological-economic assessment methodology – the ΔDPSIR approach (http://www.salum.net/ddpsir/): informs managers and decision-makers about the ecological and economic impacts of previously adopted ICZM programmes as well as about future response scenarios. The key feature of the ΔDPSIR is to provide an explicit link between ecological and economic information related to the use and management of a coastal ecosystem within a specific timeframe;

- A Modelling Approach to Resource economics decision-maKing in EcoaquaculTure – the MARKET model: provides understanding of the ecological and economic limits beyond which mariculture becomes less efficient. The key feature of the MARKET model is that it dynamically simulates the ecological and economic interactions.

Overall, the multilayered ecosystem model can provide valuable insights to ICZM, for instance, as regards management scenarios that account for the cumulative impacts of multiple uses of coastal zones. This approach can be particularly useful if managers are engaged in the process, in which case it requires the explanation of the model capabilities and limitations to managers and of the management requirements to the modelling team. Scenario testing can help managers design the most effective measures for attaining their goals. After implementing a set of measures, for instance, in the context of an ICZM programme,

managers need to be able to assess the outcomes of the initiative in order to follow an adaptative management approach. The ΔDPSIR provides a framework to accomplish this evaluation. One of the case studies of this thesis illustrates the application of the ΔDPSIR approach and exemplifies how this methodology can support the strategic management of natural resources in a coastal lagoon from both ecological and economic perspectives. The approach is further extended in another case study in which the ΔDPSIR methodology is applied to evaluate scenarios simulated with the multilayered ecosystem model. The application of this combined modelling and assessment approach explicitly links the ecological and economic information about the aquatic resource use and management options simulated for the coastal ecosystem. Overall, the ΔDPSIR application is tested using different datasets and scales of analysis: (i) to analyse past management of a coastal lagoon, based on data; (ii) to evaluate impacts of management scenarios on a coastal bay, based on model outputs; (iii) and to assess the performance of an individual aquaculture farm. Finally, the MARKET model explicitly couples ecological and economic interactions for aquaculture production. A key feature of the coupled ecological-economic model implementation is the incorporation of the different time scales at which the ecological and economic systems function. The MARKET model further develops the multilayered ecosystem model and the ΔDPSIR approach, both of which cannot dynamically simulate the feedbacks between the ecological and economic systems.

In this work, the described methodologies are applied to address a current management challenge. The focus of the work is on dealing with the challenges of sustainable mariculture development; mainly due to its socio-economic importance and complex interactions with the environmental system. The multilayered ecosystem model is applied to test scenarios designed to improve water quality and manage aquaculture. The model outputs are analysed using the ΔDPSIR approach to assess the ecological-economic impacts of the scenarios on aquaculture production at the waterbody/watershed level. Additionally, the ΔDPSIR approach is used to evaluate the ecological-economic effects of different aquaculture practice options at the individual farm level, which is other important scale of analysis for the development of an ecosystem approach to aquaculture. Finally, the MARKET model is applied to dynamically simulate the interactions between the ecological and economic systems to understand the ecological and economic limits beyond which mariculture becomes less efficient.

The integrated ecological-economic modelling and assessment approach for management of coastal ecosystems presents several limitations, which include:

❑ Limitations inherent to ecological/ecosystem modelling:

- Ecological modelling, and thus scenario simulation, is limited to the variables for which there exists comprehensive knowledge;

- For known processes, there are complex interactions that cannot be accounted for in an ecosystem model;

- The degree of model complexity, regarding spatial/temporal resolution, is limited to a level that allows a manageable treatment of results, which implies that only averaged values are obtained.

❑ Limitations inherent to integrated assessment:

- Integrated assessments, for instance the DPSIR, do not generate neutral knowledge (Svarstad et al., 2008); rather, the results depend on the analyst's point of view. For that reason, when performing such assessments, it is important to engage all stakeholders.

❑ Limitations inherent to integrating the natural and socio-economic system:

- Ecosystem valuation has a number of limitations, not only methodological but also moral (Hampicke, 1999; Emerton and Bos, 2004). For a number of reasons, is difficult to compute an objective and holistic total economic value of a given ecosystem (Nijkamp and van den Bergh 1997);

- The stochastic nature of decisions by individual farmers and the complex dynamics that govern, for instance, a policy change, are difficult to incorporate in a dynamic model. As such, the MARKET model allows for scenario testing under restricted assumptions.

When working with modelling and assessment methodologies to support coastal management, it is important to identify the above-mentioned limitations in order to avoid the misconception that science can address all coastal problems and questions made by managers. However, all these methodologies have useful applications, as presented throughout this thesis. The ecosystem-based management approach takes into account the balance between scientific limitations and capabilities to address management needs. It endorses the use of the best available knowledge about the ecosystem to manage coastal resources and maintain its services; thus promoting an adaptative understanding about ecosystem processes to respond to uncertainties (Murawski, 2007).

7.2   Concluding remarks about the study sites

This section presents the consolidated conclusions about each study site:

Xiangshan Gang exemplifies a South East Asian coastal ecosystem characterised by (i) multiple human pressures, in particular large aquaculture production areas, and (ii) ongoing management efforts to improve water quality in order to diversify the uses, such as promoting tourism. Major outcomes of the research include:

- The assessment of the trophic condition of the bay results in a poor estimated score for the implementation of any scenario. The improvement of water quality will require broader actions than those tested in the modelling exercise. There is still a high to moderate high proportion of anthropogenic nutrient sources that can be reduced;

- Harmful algal bloom events are the most relevant eutrophication symptom in Xiangshan Gang. Management of this complex phenomenon requires further research and monitoring, including a systematic analysis about the origin of the occurrences, triggering mechanisms and detailed economic impacts;

- Further actions to decrease pressure on the coastal ecosystem should also include land use change of the catchment area. The multilayered ecosystem model can assist managers in testing the effects of different land cover and agriculture practice on the bay water quality and aquaculture production;

- A solution with potential ecological and economic gains is to re-establish kelp or other seaweed cultivation in order to reduce dissolved nutrient concentration in the bay;

- In the simulated scenarios, the reduction of emissions from wastewater and fish cages causes a reduction in shellfish production of about 8% to 47%, depending on the scenario;

❑ The MARKET model indicates that production in the long run does not meet increasing demand. If aquaculture reduction is not an option, because substance loading, which provides food for shellfish, will decrease as a result of further water quality improvement plans, an ecosystem approach is required to optimise growth conditions:

   o Displace the shellfish culture to areas of the bay with best growth conditions. For instance, model outputs estimate that Chinese oyster productivity is almost 3 times higher in the downstream area of the bay than in the inner part;

   o Integrate shellfish production near fish cage areas, in order to sustain shellfish food resources even if, at the ecosystem level, the substances might be reduced;

   o Where integrated aquaculture in the embayment is not desirable or possible, in-land integrated multi-trophic aquaculture (IMTA) systems might be an option.

❑ Integration of the insights provided by the model outputs with spatial zoning tools might assist in optimizing the location of competing and synergistic activities;

❑ Model improvements should include better simulation of the bay hydrodynamics, detailed data series for the sea boundary inputs, extended spatial coverage of the water sampling network inside the bay, detailed data series of fish cage inputs and full mapping of aquaculture structures and practices;

❑ In the long run and from a broad society perspective, the costs incurred to take some of these actions might be paid back by the avoided costs of restoring ecosystems and improved food security.

Ria Formosa exemplifies western shallow coastal systems, with a conservation value protected by several international conventions/directives and a coastal community characterised by low population density with a high degree of socio-economic interaction with the lagoon. The most important management issues identified in the Ria Formosa for the period analysed (1985 to 1995), were seasonal variations of the local human population and a decrease in clam stocks. The major outcomes of the research include:

- A comparison of drivers, pressures and ecosystem state in two different periods (1980/85 and 1995/99) indicates that although there was a significant management response (such as the construction of waste water treatment plants), the ecosystem state worsened in terms of abnormal clam mortalities due to a parasite and benthic eutrophication symptoms in specific problematic areas. The corresponding negative economic impacts represent 80% to 220% of the response cost;

- The value of economic activities dependent on the lagoon suffered a significant reduction (ca. -60%). The decrease of the economic benefits was mainly due the decrease of bivalve production, a consequence of the abnormal clam mortalities. The social consequences are also relevant given that bivalve aquaculture production is responsible for the direct employment of about 4 500 to 10 000 people;

- The local clam species (*Ruditapes decussatus*) is highly priced, and its production in Ria Formosa is significant compared with total national production. However, it is being displaced by the Manila clam (*Ruditapes philippinarum*);

- Evaluation of these events indicates that future management policies should focus on conservation of the local clam, a step that should result in positive impacts to both the local socio-economy and biodiversity;

- Notwithstanding, the activities stipulated in the Ria Formosa Natural Park Management Plan include an amount for bivalve related management that represents 1.9% of planned total budget;

- Future actions should invest in the proper management of bivalve aquaculture and natural beds with a special emphasis on the seeding procurement or development of local hatcheries, which might have a positive effect on (i) mitigating disease introduction, (ii) limiting the introduction of alien species, and (iii) on local clam seed availability (Matias et al., 2008);

❑ The relevant authorities should also define a set of indicators to monitor the effectiveness of the goals established in the various management plans that exist for this ecosystem.

The Irvine and Johnston (I & J), Cape Cultured Abalone Pty, Ltd farm illustrates the shift of an abalone monoculture in a flow-through system into an abalone-seaweed IMTA with recirculation. This case study is relevant not only for other Southern African abalone farms, which together are the largest abalone producers outside Asia (783 ton per year), but also for land-based farms located elsewhere and for other species, such as fish or shrimp. The major outcomes of the ΔDPSIR application include:

❑ The comparison of monoculture with both IMTA settings indicates an overall economic gain of between 1.1 and 3.0 million U.S. dollar per annum. This range of values reflects the effects of adopting IMTA on (i) economic value of drivers, i.e. farm's profit, (ii) value of environmental externalities, and (iii) implementation costs;

❑ The environmental benefits include reduction in nitrogen discharge into the sea, reduction in the harvest of natural kelp and reduction in CO_2 emissions. Alone, these represent about 80 % of the estimated overall gains;

❑ The ΔDPSIR analysis suggests that the value of the benefits to the public by adopting the IMTA designs were larger than the gains in the farm's profitability;

❑ The benefits associated with shifting from a monoculture to the IMTA increase with an increase in seaweed production. However, the resulting nutrient limitation should be addressed;

❑ One solution is the three-species IMTA with fish, abalone and seaweeds. This system produces more value and resources for human consumption while still managing the waste produced;

❑ The abalone-seaweed case provided a convenient IMTA system, given that while seaweeds act as nutrient biofilters, they are also the natural abalone feed. For other species, the generated algal biomass can be converted into other products, such as energy, fertilizers or pharmaceuticals; a similar study might also be conducted using other extractive species, such as filter-feeders.

7.3   Conclusions

Research about the ecological and economic assessment of coastal ecosystems is important because (i) of the importance of and high demand for coastal zones, (ii) the symptoms of overuse and misuse of these ecosystems, and (iii) the need for methodologies to evaluate the outcomes of coastal management initiatives and to support coastal planning.

The specific problem addressed in this work is the assessment of changes of the ecosystem state and their interactions with the anthropogenic system. This thesis provides a methodology to assess the impacts of management responses and multiple coastal zone uses on the ecosystem state and generated benefits. The study focuses on the challenges of sustainable aquaculture research and management. Despite the limitations described above, the integrated ecological-economic approach for management of coastal ecosystems contributes new knowledge for addressing the following research needs:

□   Simulation of the cumulative impacts of multiple uses of coastal zones;

□   Management-oriented assessment of ecological and economic impacts on coastal ecosystems;

□   Dynamic simulation of ecological-economic interactions of mariculture production.

The results obtained for the different case studies illustrate this method's application for assessing the ecological and economic impacts of management responses and scenarios simulated to test management actions. The outcomes of the approach were synthesised into information for managers. The integrated approach was applied to analyse aquaculture production at both the ecosystem and farm levels. The outcomes illustrated the usefulness of this approach for assisting the development of an ecosystem approach to aquaculture, as advocated by FAO (FAO, 2007; Soto et al., 2008). Furthermore, the simulation of the feedbacks between the ecological and economic systems supported the dynamic analysis of (i) the demand for aquaculture products, (ii) economic production and cost-limiting factors, (iii) the growth of aquatic resources, (iv) interactions with environmental conditions, and (iv) the spatial limitations of culture in coastal ecosystems.

Future applications must include the interaction and communication with stakeholders of the ecosystem, preferably at earlier stages. Such procedures will contribute to the definition of evaluation criteria in the development of management programmes. Additionally, they will ensure that the relevant variables for managers and resource users are included in the modelling frameworks. Early interaction should be followed up with iterative communication between researchers, stakeholders with a management role and users of the goods and services of an ecosystem.

The methodology developed in this thesis can be further applied to address new coastal management issues, such as coastal vulnerability to natural catastrophes. It can also support implementation of current legislation and policies, such as the EU ICZM recommendation or the development of River Basin Management Plans following the requirements of the EU Water Framework Directive. On the other hand, it can be used to address recurring issues, such as the assessment of the outcomes of past or on-going coastal management plans.

References

Abreu, M. H., Varela, D. A., Henríquez, L., Villarroel, A., Yarish, C., Sousa-Pinto, I. and Buschmann, A. H., 2009. Traditional vs. Integrated Multi-Trophic Aquaculture of *Gracilaria chilensis* C. J. Bird, J. McLachlan & E. C. Oliveira: Productivity and physiological performance. Aquaculture, 293(3-4): 211-220.

Acharya, G., 2000. Approaches to valuing the hidden hydrological services of wetland ecosystems. Ecological Economics, 35:63-74.

Ahmed, M., Lorica, M.H., 2002. Improving developing country food security through aquaculture development - lessons from Asia. Food Policy, 27(2):125-141.

Allen, B.P., Loomis, J.B., 2006. Deriving values for the ecological support function of wildlife: an indirect valuation approach. Ecological Economics, 56:49–57.

Allsopp, M., Johnston, P., Santillo, D., 2008. Challenging the aquaculture industry on sustainability. Greenpeace International, Netherlands, 24 pp.

Alongi, D.M., 1998. Coastal ecosystem processes. CRC Press, Boca Raton, 419 pp.

Anderson, D.M., Hoagland, P., Kaoru, Y., White, A.W., 2000. Estimated annual economic impacts from harmful algal blooms (HABs) in the United States. Woods Hole Oceanographic Institute Technical Report, WHOI-2000-11, 101 pp.

Anker, H.T., Nellemann, V., Sverdrup-Jensen, S., 2004. Coastal zone management in Denmark: ways and means for further integration. Ocean & Coastal Management, 47(9-10): 495-513.

Arhonditsis, G.B., Stow, C.A., Paerl, H.W., Valdes-Weaver, L.M., Steinberg, L.J., Reckhow, K.H., 2007. Delineation of the role of nutrient dynamics and hydrologic forcing on phytoplankton patterns along a freshwater-marine continuum. Ecological Modelling, 208: 230-246.

Armstrong, C.W., 2007. A note on the ecological-economic modelling of marine reserves in fisheries. Ecological Economics, 62: 242-250.

Artioli, Y., Bendoricchio, G., Palmeri, L., 2005. Defining and modelling the coastal zone affected by the Po river (Italy). Ecological Modelling, 184(1): 55-68.

Astron, 2005. Seas the Opportunity: A Strategy for the Long Term Sustainability of Scotlands Coasts and Seas. Scottish Executive, Edinburgh. 47 pp.

Ayer, N.W., Tyedmers, P.H., 2009. Assessing alternative aquaculture technologies: life cycle assessment of salmonid culture systems in Canada. Journal of Cleaner Production, 17(3):362-373.

Azevedo, C., 1989. Fine structure of *Perkinsus atlanticus* n. sp. (Apicomplexa, Perkinsea) parasite of the clam *Ruditapes decussatus* from Portugal. Journal of Parasitology 75:627–635.

Baan, P.J.A., van Buuren, J.T., 2003. Testing of indicators for the marine and coastal environment in Europe. Part 3: Present state and development of indicators for eutrophication,

hazardous substances, oil and ecological quality. Technical report No 86. European Environment Agency, 65 pp.

Babaran, R.P., Espinosa, R.A., Abalos, T.U., 1998. Initiating and triggering mechanisms causing harmful algal blooms. Journal of Shellfish Research, 17(5): 1623-1626.

Balaguer, P., Sardá, R., Ruiz, M., Diedrich, A., Vizoso, G., Tintoré, J., 2008. A proposal for boundary delimitation for integrated coastal zone management initiatives. Ocean & Coastal Management, 51(12): 806-814.

Bald, J., Sinquin, A., Borja, A., Caill-Milly, N., Duclercq, B., Dang, C., de Montaudouin, X., 2009. A system dynamics model for the management of the Manila clam, *Ruditapes philippinarum* (Adams and Reeve, 1850) in the Bay of Arcachon (France). Ecological Modelling, In Press.

Barbier, E.B., Acreman, M.C., Knowler, D., 1996. Economic valuation of wetlands: a guide for policy makers and planners. Ramsar Convention Bureau, Gland, Switzerland.

Barragán, J.M., 2003. Coastal zone management in Spain (1975-2000). Journal of Coastal Research, 19: 314-325.

Batjes, N.H., 2002. Soil parameter estimates for the soil types of the world for use in global and regional modelling (version 2.1; July 2002). ISRIC report 2002/02c, International Food Policy Research Institute (IFPRI) e International Soil Reference and Information Centre (ISRIC), Wageningen.

Belfiore, S., 2000. Recent developments in coastal management in the European Union. Ocean & Coastal Management, 43(1):123-135.

Belfiore, S., Barbière, J., Bowen, R., Cicin-Sain, B., Ehler, C., Mageau, C., McDougall, D., Siron, R., 2006. A handbook for measuring the progress and outcomes of integrated coastal and ocean management. Heileman, S. (Eds). IOC Manuals and Guides, 46; ICAM Dossier, 2. Paris, UNESCO. 224 pp.

Bergamasco, A., De Nat, L., Flindt, M.R., Amos, C.L., 2003. Interactions and feedbacks among phytobenthos, hydrodynamics, nutrient cycling and sediment transport in estuarine ecosystems. Continental Shelf Research, 23:1715-41.

Best, M., Massey, A.,Prior, A., 2007. Developing a saltmarsh classification tool for the European Water Framework Directive. Marine Pollution Bulletin, 55(1-6):205-214.

Bidone, E.D., Lacerda, L.D., 2004. The use of DPSIR framework to evaluate sustainability in coastal areas. Case study: Guanabara Bay basin, Rio de Janeiro, Brazil. Regional Environmental Change, 4(1):5-16.

Billé, R., 2007. A dual-level framework for evaluating integrated coastal management beyond labels. Ocean & Coastal Management, 50(10):796-807.

Birol, E., Karousakis, K., Koundouri, P., 2006. Using economic valuation techniques to inform water resources management: A survey and critical appraisal of available techniques and an application. Science of The Total Environment, 365:105-122.

Bockstael, N., Costanza, R., Strand, I., Boynton, W., Bell, K., Wainger, L., 1995. Ecological economic modeling and valuation of ecosystems. Ecological Economics, 14:143-159.

Boissonnas, J., Connolly, N., Mantoura, F., D'Ozouville, L., 2002. Integrating Marine Science in Europe. European Science Foundation Marine Board, 148 pp.

Bolton, J.J., 2006. Do we have the vision to integrate our marine aquaculture? South African Journal of Science, 102:57-58.

Bondad-Reantaso, M.G., Subasinghe, R.P., Arthur, J.R., Ogawa, K., Chinabut, S., Adlard, R., Tan, Z., Shariff, M., 2005. Disease and health management in Asian aquaculture. Veterinary and parasitology, 132:249–272.

Borja, A., 2006. The new European Marine Strategy Directive: Difficulties, opportunities, and challenges. Marine Pollution Bulletin, 52(3):239-242.

Borja, A., Galparsoro, I., Solaun, O., Muxika, I., Tello, E.M., Uriarte, A., Valencia, V., 2006. The European Water Framework Directive and the DPSIR, a methodological approach to assess the risk of failing to achieve good ecological status. Estuarine, Coastal and Shelf Science, 66(1-2):84-96.

Borja, A., Bricker, S.B., Dauer, D.M., Demetriades, N.T., Ferreira, J.G., Forbes, A.T., Hutchings, P., Jia, X., Kenchington, R., Marques, J.C. and Zhu, C., 2008. Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. Marine Pollution Bulletin, 56:1519-1537.

Bowen, R.E., Riley, C., 2003. Socio-economic indicators and integrated coastal management. Ocean & Coastal Management, 46:299-312.

Bower, B.T., Turner, R.K., 1998. Characterising and analysing benefits from integrated coastal management (ICM). Ocean and Coastal Management, 38:41–66.

Boyes, S.J., Elliott, M., Thomson, S.M., Atkins, S., Gilliland, P., 2007. A proposed multiple-use zoning scheme for the Irish Sea: An interpretation of current legislation through the use of GIS-based zoning approaches and effectiveness for the protection of nature conservation interests. Marine Policy, 31:287-298.

Brady, M., 2003. The relative cost-efficiency of arable nitrogen management in Sweden. Ecological Economics 47:53–70.

Breton, F., 2006. Report on the use of the ICZM indicators from the WG-ID. A contribution to the ICZM evaluation. Version 1."indicators and data" working group (WG-ID) of the European ICZM expert group. 63 pp.

Bricker, S.B., Clement, C.G., Pirhalla, D.E., Orlando, S.P. and Farrow, D.R.G., 1999. National Estuarine Eutrophication Assessment. Effects of Nutrient Enrichment in the Nation's Estuaries. NOAA - National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science, Silver Spring.

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

Bricker, S., Longstaff, B., Dennison, W., Jones, A., Boicourt, K., Wicks, C., Woerner, J., 2007. Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change, National Estuarine Eutrophication Assessment Update. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Centers for Coastal Ocean Science, Silver Spring, MD.

Bricker, S.B., Longstaff, B., Dennison, W., Jones, A., Boicourt, K., Wicks, C., Woerner, J., 2008. Effects of nutrient enrichment in the nation's estuaries: A decade of change. Harmful Algae, 8(1):21-32.

Brock, T.D., 1981. Calculating solar radiation for ecological studies. Ecological Modelling, 14:1-9.

Browman, H.I., Stergiou, K.I. (Eds.), 2005. Politics and socio-economics of ecosystem-based management of marine resources. Marine Ecology Progress Series, 300:241-296.

Bulte, E.H., van Kooten, G.C., 1999. Metapopulation dynamics and stochastic bioeconomic modeling. Ecological Economics, 30:293-299.

Burbridge, P.R., 1997. A generic framework for measuring success in integrated coastal management. Ocean & Coastal Management, 37(2): 175-189.

Burks, B.D., Minnis, M.M., 1994. Onsite Wastewater Treatment Systems. Hogarth House, Madison.

Buschmann, A. H., Cabello, F., Young, K., Carvajal, J., Varela, D. A. and Henríquez, L., 2009. Salmon aquaculture and coastal ecosystem health in Chile: Analysis of regulations, environmental impacts and bioremediation systems. Ocean & Coastal Management, 52(5): 243-249.

Cai, Y., 2001. Xiangshan Harbor Red Tide Emergency Monitoring and Disaster Precaution Technique. In 2nd International Conference on Harmful Algae Mitigation and Mitigation (HAMM): November 12-16, 2001, Qingdao, China.

Cai, H., Sun, Y., 2007. Management of Marine Cage Aquaculture. Environmental Carrying Capacity Method Based on Dry Feed Conversion Rate. Environmental Science and Pollution Research, 14:463-469.

Campos, C., Cachola, R.A., 2006. The introduction of the Japanese Carpet Shell in coastal lagoon systems of the Algarve (south Portugal): a food safety concern. Internet Journal of Food Safety, 8:1-2.

Cao, W., Wong, M.H., 2007. Current status of coastal zone issues and management in China: A review. Environment International, 33: 985-992.

Carbon Catalog, http://www.carboncatalog.org/, data searched on 21st November 2008

CGIAR, Consultative Group on International Agricultural Research. 2005. CGIAR-CSI SRTM 90m DEM Digital Elevation Database. Consortium for Spatial Information, Consultative Group on International Agricultural Research. Available online at: http://srtm.csi.cgiar.org/.

Chang, Y.C., Hong, F.W., Lee, M.T., 2008. A system dynamic based DSS for sustainable coral reef management in Kenting coastal zone, Taiwan. Ecological Modelling, 211:153-168.

Chee, Y.E. 2004. An ecological perspective on the valuation of ecosystem services. Biological Conservation, 120:549-565.

Chen, Q., Zhang, Y., Ekroos, A., Hallikainen, M., 2004. The role of remote sensing technology in the EU water framework directive (WFD). Environmental Science & Policy, 7:267-276.

Chen, Y.Q., Rong, P., Wu, X.J., 1992. Environment of the East China Sea Area and Its CoastaI Economic Development. Science and Technology Review, 9:30-32.

Cheong, S.M., 2008. A new direction in coastal management. Marine Policy, 32:1090-1093.

Chopin, T., Buschmann, A.H., Halling, C., Troell, M., Kautsky, N., Neori, A., Kraemer, G.P., Zertuche-Gonzalez, J.A., Yarish, C., Neefus, C., 2001. Integrating seaweeds into marine aquaculture systems: a key toward sustainability. Journal of Phycology, 37:975-986.

Cicin-Sain, B., Knecht, R.W., 1998. Integrated Coastal and Ocean Management: Concepts and Practices. Island Press, Washington, DC.

Cicin-Sain, B., Belfiore, S., 2005. Linking marine protected areas to integrated coastal and ocean management: a review of theory and practice. Ocean and Coastal Management, 48:847–868.

Clark, J.R., 1996. Coastal zone management handbook. Lewis/CRC Publishers, Boca Raton, Florida. 694 pp.

Cohen, I., Neori, A., 1991. *Ulva lactuca* biofilters for marine fishponds effluents. Botanica Marina, 34:475–82.

Cordah Ltd, 2001. Indicators to monitor the progress of integrated coastal zone management: A review if worldwide practice. Scottish Executive Central Research Unit. 54 pp.

Costanza, R., D'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., Van Den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature, 387:253-260.

Crab, R., Avnimelech, Y., Defoirdt, T., Bossier, P., Verstraete, W., 2007. Nitrogen removal techniques in aquaculture for a sustainable production. Aquaculture, 270:1-14.

Costanza, R., Farley, J., 2007. Ecological economics of coastal disasters: Introduction to the special issue. Ecological Economics 63: 249-253.

Crawford, C., 2003. Environmental management of marine aquaculture in Tasmania, Australia. Aquaculture, 226:129–138.

Cromey, C.J., Nickell, T.D., Treasurer, J., Black, K.D., Inall, M., 2009. Modelling the impact of cod (*Gadus morhua* L.) farming in the marine environment - CODMOD. Aquaculture, 289:42-53.

CSS (Central Statistical Service). 2005. October Household Survey 2001: Central Statistical Service, Pretoria.

de Jonge, V.N., 2007. Toward the application of ecological concepts in EU coastal water management. Marine Pollution Bulletin, 55:407-414.

de Wit, M.P., Jiao, H., Luo, Q., Musango, J.K., Yan, X., Ye, C., Zhang, X., Zhang, Z., Zhu, M., 2008. Notes on production, costs and revenue in selected Chinese aquaculture systems:

Results from an economic survey. CSIR Report: CSIR/NRE/RBSD/ER/2008/0024/C, Stellenbosch, South Africa.

Deboudt, P., Dauvin, J.C., Lozachmeur, O., 2008. Recent developments in coastal zone management in France: The transition towards integrated coastal zone management (1973-2007). Ocean & Coastal Management, 51(3):212-228.

DEFRA (Department for Environment Food and Rural Affairs), 2008. A strategy for promoting an integrated approach to the management of coastal areas in England. 48 pp.

Delft Hydraulics, 2006. Delft3D-FLOW, a simulation program for hydrodynamic flows and transports in2 and 3 dimensions. User manual. Version 3.11.

Delgado, C.L., Wada, N., Rosegrant, M.W., Meijer, S., Ahmed, M., 2003. Outlook for fish 2020: Meeting global demand. International Food Policy Research Institute (IFPRI) 2020 Vision Food Policy Report.

Dempster, T., Sanchez-Jerez, P., 2008. Aquaculture and coastal space management in Europe: An ecological perspective. In: M. Holmer, K. Black, C. M. Duarte, N. Marbà, I. Karakassis (eds.), Aquaculture in the ecosystem. Springer. 326 pp.

Dennison, W.C., 2008. Environmental problem solving in coastal ecosystems: A paradigm shift to sustainability. Estuarine, Coastal and Shelf Science, 77(2):185-196.

Dlaza, T.S., Maneveldt, G.W., Viljoen, C., 2008. Growth of post-weaning abalone *Haliotis midae* fed commercially available formulated feeds supplemented with fresh wild seaweed. African Journal of Marine Science, 30(1):199–203.

DOENI (Department of the Environment in Northern Ireland), 2006. An Integrated Coastal Zone Management Strategy for Northern Ireland 2006 - 2026. 86 pp.

Douven, W.J.A.M., Buurman, J.J.G., Kiswara, W., 2003. Spatial information for coastal zone management: the example of the Banten Bay seagrass ecosystem, Indonesia. Ocean & Coastal Management, 46:615-634.

Dowd, M., 2005. A bio-physical coastal ecosystem model for assessing environmental effects of marine bivalve aquaculture. Ecological Modelling, 183:323-346.

Drechsler, M., Grimm, V., Mysiak, J., Watzold, F., 2007. Differences and similarities between ecological and economic models for biodiversity conservation. Ecological Economics, 62:232-241.

Drechsler, M., Watzold, F., 2007. Ecological-economic modelling for the sustainable use and conservation of biodiversity. Ecological Economics, 62:03-206.

Driml, S.M., 1997. Bringing ecological economics out of the wilderness. Ecological Economics, 23:145–153.

Duarte, C.M., Marbá, N., Holmer, M., 2007a. Rapid domesticationof marine species. Science, 316: 382-383.

Duarte, P., Azevedo, B., Ribeiro, C., Pereira, A., Falcão, M., Serpa, D., Bandeira, R., Reia, J., 2007b. Management oriented mathematical modelling of Ria Formosa (South Portugal). Transitional Waters Monographs, 1:13–51.

Ducrotoy, J.P., Elliott, M., 2006. Recent developments in estuarine ecology and management. Marine Pollution Bulletin, 53(1-4):1-4.

Ducrotoy, J.P., Elliott, M., 2008. The science and management of the North Sea and the Baltic Sea: Natural history, present threats and future challenges. Marine Pollution Bulletin, 57(1-5):8-21.

Dumbauld, B.R., Ruesink, J.L., Rumrill, S.S., 2009. The ecological role of bivalve shellfish aquaculture in the estuarine environment: A review with application to oyster and clam culture in West Coast (USA) estuaries. Aquaculture, 290:196-223.

Economopoulos, A.P., 1993. Assessment of Sources of Air, Water, and Land Pollution: A Guide to Rapid Source Inventory Techniques and Their Use in Formulating Environmental Control Strategies. World Health Organization, Geneva.

Ehler, C.N., Cicin-Sain, B., Knecht, R., South, R., Weiher, R., 1997. Guidelines to assist policy makers and managers of coastal areas in the integration of coastal management programs and national climate-change action plans. Ocean & Coastal Management, 37:7-27.

Elliott, M. 2002. The role of the DPSIR approach and conceptual models in marine environmental management: an example for offshore wind power. Marine Pollution Bulletin, 44(6):iii-vii.

Elliott, M., Burdon, D., Hemingway, K.L., Apitz, S.E., 2007. Estuarine, coastal and marine ecosystem restoration: Confusing management and science—a revision of concepts. Estuarine Coastal and Shelf Science, 74(3):349–366.

Elofsson, K., 2003. Cost-effective reductions of stochastic agricultural loads to the Baltic Sea. Ecological Economics, 47:13–31.

Emerton, L., Bos, E., 2004. Value: Counting Ecosystems as an Economic Part of Water Infrastructure. IUCN, Gland, Switzerland and Cambridge, UK. 88 pp.

Endreny, T.A., Wood, E.F., 2003. Watershed Weighting of Export Coefficients to Map Critical Phosphorous Loading Areas. Journal of the American Water Resources Association, 39:165–181.

Enemark, J., 2005. The Wadden Sea protection and management scheme--towards an integrated coastal management approach? Ocean & Coastal Management, 48: 996-1015.

Eom, Y.S., Larson, D.M., 2006. Improving environmental valuation estimates through consistent use of revealed and stated preference information. Journal of Environmental Economics and Management, 52:501-516.

EPA (U.S. Environmental Protection Agency), 2008. Handbook for Developing Watershed TMDLs. U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds.. Draft December 2008. Avalaible at: http://www.epa.gov/OWOW/TMDL/, last accessed on 29 September.

Eremina, T.R., Stetsko, E.V., 2003. Legal provision for integrated coastal zone management. Textbook published by Russian State Hydrometeorological University (RSHU), based on UNESCO/CSI related project on Sustainable Coastal Development in the White Sea - Barents

Sea Region. Available at: http://www.unesco.org/csi/act/russia/legalpro1.htm, last accessed on July 2009.

Evett, K. 2006. Our Future for life: sustainable job creation. Highbury Monarch Communications Pty. Cape Town. South Africa. 242pp.

Fabbri, K.P., 1998. A methodology for supporting decision making in integrated coastal zone management. Ocean & Coastal Management, 39(1-2): 51-62.

FAO (Food and Agriculture Organization), 2009. Fishery statistical collections, Global aquaculture production 1950-2007. Available online: http://www.fao.org/fishery/statistics/global-aquaculture-production/en, searched on 6th May 2009.

FAO, 2004. The State Of World Fisheries and Aquaculture (SOFIA) 2004. FAO Fisheries Department, Rome. 153 pp.

FAO. 2005. National Aquaculture Sector Overview - China. Text by Shuping, C., FAO Fisheries and Aquaculture Department. Rome.

FAO. 2006. State of World Aquaculture, FAO Fisheries Technical Paper No. 500. FAO Fisheries and Aquaculture Department, Rome, 134 pp.

FAO, 2007. The state of world fisheries and aquaculture, 2006. FAO Fisheries and Aquaculture Department. 180 pp.

FAO, 2009. The State Of World Fisheries and Aquaculture (SOFIA) 2008. FAO Fisheries Department, Rome. 196 pp.

Farber, S., Costanza, R., Childers, D. L., Erickson, J., Gross, K., Grove, M., Hopkinson, C. S., Kahn, J., Pincetl, S., Troy, A., Warren, P., Wilson, M., 2006. Linking Ecology and Economics for Ecosystem Management. Bioscience, 56(2):121-133.

Fedra, K., 1996. Distributed models and embedded GIS: Integration strategies and case studies. In: Goodchild, M.F., Steyaert, L.T., Parks, B.O., Johnston, C., Maidment, D., Crane, M. and Glendinning, S. (eds.). GIS and environmental modeling: Progress and research issues. John Wiley & Sons, Inc., pp. 413-417.

Feng, Y., Hou, L., Ping, N., Ling, T., Kyo, C., 2004. Development of mariculture and its impacts in Chinese coastal waters. Reviews in Fish Biology and Fisheries, 14:1-10

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

Ferreira, J. G., Duarte, P., Ball, B., 1997. Trophic capacity of Carlingford Lough for oyster culture - analysis by ecological modelling. Aquatic ecology, 31(4): 361-378.

Ferreira, J.G., Simas T., Nobre A., Silva M.C., Shifferegger K., Lencart-Silva J., 2003. Identification of sensitive areas and vulnerable zones in transitional and coastal Portuguese systems. INAG, Portuguese Water Institute and IMAR, Institute of Marine Research, Portugal, 131 pp.

Ferreira, J.G., Andersson, H.C., Corner, R., Groom, S., Hawkins, A.J.S., Hutson, R., Lan, D., Nauen, C., Nobre, A.M., Smits, J., Stigebrandt, A., Telfer, T., de Wit, M., Yan, X., Zhang,

X.L., Zhu, M.Y., 2005. SPEAR, Sustainable options for people, catchment and aquatic resources. IMAR—Institute of Marine Research, Portugal, 71 pp. Available at http://www.biaoqiang.org/

Ferreira, J.G., Nobre, A.M., Simas, T.C., Silva, M.C., Newton, A., Bricker, S.B., Wolff, W.J., Stacey, P.E., Sequeira, A, 2006. A methodology for defining homogeneous water bodies in estuaries - Application to the transitional systems of the EU Water Framework Directive. Estuarine Coastal Shelf Science, 66:468-482.

Ferreira, J.G., Hawkins, A.J.S., Bricker, S.B., 2007a. Management of productivity, environmental effects and profitability of shellfish aquaculture - the Farm Aquaculture Resource Management (FARM) model. Aquaculture, 264:160-174.

Ferreira, J.G., Bricker, S.B., Simas, T.C., 2007b. Application and sensitivity testing of a eutrophication assessment method on coastal systems in the United States and European Union. Journal of Environmental Management, 84(4):433-445.

Ferreira, J.G., Hawkins, A.J.S., Monteiro, P., Moore, H., Service, M., Pascoe, P.L., Ramos, L., Sequeira, A., 2008a. Integrated assessment of ecosystem-scale carrying capacity in shellfish growing areas. Aquaculture, 275(1-4):138-151.

Ferreira, J.G., Andersson, H.C., Corner, R.A., Desmit, X., Fang, Q., de Goede, E.D., Groom, S.B, Gu, H., Gustafsson, B.G., Hawkins, A.J.S., Hutson, R., Jiao, H., Lan, D., Lencart-Silva, J., Li, R., Liu, X., Luo, Q., Musango, J.K., Nobre, A.M., Nunes, J.P., Pascoe, P.L., Smits, J.G.C., Stigebrandt, A., Telfer, T.C., de Wit, M.P., Yan, X., Zhang, X.L., Zhang, Z., Zhu, M.Y., Zhu, C.B., Bricker, S.B., Xiao, Y., Xu, S., Nauen, C., Scalet, M., 2008b. SPEAR. Sustainable Options for People, Catchment and Aquatic Resources. IMAR-Institute of Marine Research. 184 pp.

Ferreira, J.G., A. Sequeira, A.J.S. Hawkins, A. Newton, T. Nickell, R. Pastres, J. Forte, A. Bodoy, S.B. Bricker, 2009. Analysis of coastal and offshore aquaculture: application of the FARM model to multiple systems and shellfish species. Aquaculture, 292, 129-138.

Field, B.C., 1997. Environmental economics, an introduction, 2nd edn. McGraw-Hill, Singapore, 496 pp.

Fluharty, D., 2005. Evolving ecosystem approaches to management of fisheries in the USA. Marine Ecology Progress Series, 300:248–253.

Forst, M.F., 2009. The convergence of Integrated Coastal Zone Management and the ecosystems approach. Ocean & Coastal Management, 52(6):294-306.

Franco, A.R., Ferreira, J.G., Nobre, A.M., 2006. Development of a growth model for penaeid shrimp. Aquaculture, 259:268-277.

Fujiwara, T., Sanford, L.P., Nakatsuji, K., Sugiyama, Y., 1997. Anti-cyclonic circulation driven by the estuarine circulation in a gulf type ROFI. Journal of Marine Systems, 12:1-4.

Fulton, E.A., Smith, D.M., Johnson, C.R., 2003. Effect of complexity on marine ecosystem models. Marine Ecology Progress Series, 253:1-16.

Gangnery, A., Bacher, C., Buestel, D., 2004. Modelling oyster population dynamics in a Mediterranean coastal lagoon (Thau, France): sensitivity of marketable production to environmental conditions. Aquaculture, 230:323-347.

GESAMP (IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). 1996. The contributions of science to coastal zone management. Reports and Studies, 61: 66p.

GESAMP (IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). 2001a. Planning and management for sustainable coastal aquaculture development. GESAMP Reports and Studies, 68: 90pp.

Ghermandi, A., van den Bergh, J.C.J.M., Brander, L.M., de Groot, H.L.F., Nunes, P.A.L.D., 2008. The economic value of wetland conservation and creation: a meta-analysis. FEEM (Fondazione Eni Enrico Mattei) working paper, 25pp.

Gibbs, M.T., 2004. Interactions between bivalve shellfish farms and fishery resources. Aquaculture, 240:267–296.

Gibbs, M.T. 2009. Implementation barriers to establishing a sustainable coastal aquaculture sector. Marine Policy, 33(1):83-89.

Gibson, J., 2003. Integrated Coastal Zone Management Law in the European Union. Coastal Management, 31(2):127-136.

Gilliland, P.M., Laffoley, D., 2008. Key elements and steps in the process of developing ecosystem-based marine spatial planning. Marine Policy, 32:787-796.

Gíslason, A., Shen, B.L.Y., Halldórsson, V., 2006. Glitnir seafood industry report - China. Glitnir Seafood Team. 50 pp.

Goggin, C.L., Barker, S.C., 1993. Phylogenetic position of the genus Perkinsus (Protista, Apicomplexa) based on small subunit ribosomal RNA. Molecular and Biochemical Parasitology, 60:65-70.

Gordon, R., Cook, P.A., 2004. World abalone fisheries and aquaculture update: supply and market dynamics. Journal of Shellfisheries Research, 23:935-939.

Grant, J., Bacher, C., Cranford, P. J., Guyondet, T., Carreau, M., 2008. A spatially explicit ecosystem model of seston depletion in dense mussel culture. Journal of Marine Systems, 73: 155-168.

Gravelle, H., Rees, R., 1993. Microeconomics. 2nd edition. Longman, London & New York. 752 pp.

Green, E.P., Mumby, P.J., Edwards, A.J., Clark, C.D., 1999. The Remote Sensing Handbook for Tropical Coastal Management, UNESCO Publishing.

Greiner, R., 2004. Systems framework for regional-scale integrated modelling and assessment. Mathematics and Computers in Simulation, 64:41-51.

Gren, I.M., Folmer, H., 2003. Cooperation with respect to cleaning of an international water body with stochastic environmental damage: the case of the Baltic Sea. Ecological Economics, 47:33–42.

Gren, I.M., Lindahl, O., Lindqvist, M., 2009. Values of mussel farming for combating eutrophication: An application to the Baltic Sea. Ecological Engineering, 35(5):935-945.

GRID-Geneva: Global Resource Information Database, Geneva office. 2004. FAO – UNESCO soil map of the world; 130 soil types (grid version). United Nations Environment Programme, Division of Early Warning and Assessment, Genebra. Available online at: www.grid.unep.ch/

Grizzetti, B., Bouraoui, F., Granlund, K., Rekolainen, S., Bidoglio, G., 2003. Modelling diffuse emission and retention of nutrients in the Vantaanjoki watershed (Finland) using the SWAT model. Ecological Modelling, 169:25–38.

Grumbine, E.R., 1994. What is ecosystem management? Conservation Biology, 8(1): 27-38.

GTOS (Global Terrestrial Observing System), 2005. Coastal GTOS - Strategic design and phase 1 implementation plan. Global Terrestrial Observing System, Food and Agriculture Organization of the United Nations Rome. 108 pp.

Gutierrez-Wing, M.T., Malone, R.F., 2006. Biological filters in aquaculture: Trends and research directions for freshwater and marine applications. Aquacultural Engineering, 34(3):163-171.

Håkanson, L., Blenckner, T., 2008. A review on operational bioindicators for sustainable coastal management-Criteria, motives and relationships. Ocean & Coastal Management, 51:43-72.

Hale, L.Z., 2000. Achieving Integration in Coastal Management: The Challenge of Linking National and Local Levels of Government. Korea Observer, 30(4):671-690.

Halpern, B.S., McLeod, K.L., Rosenberg, A.A., Crowder, L.B., 2008. Managing for cumulative impacts in ecosystem-based management through ocean zoning. Ocean & Coastal Management, 51(3):203-211.

Hampicke, U., 1999. The limits to economic valuation of biodiversity. International Journal of Social Economics, 26: 158-173.

Hardman-Mountford, N.J., Allen, J.I., Frost, M.T., Hawkins, S.J., Kendall, M.A., Mieszkowska, N., Richardson, K.A., Somerfield, P.J., 2005. Diagnostic monitoring of a changing environment: An alternative UK perspective. Marine Pollution Bulletin, 50:1463-1471.

Harris, G., 2002. Integrated assessment and modelling: an essential way of doing science. Environmental Modelling & Software, 17(3):201-207.

Hatton MacDonald, D., Connor, J., Morrison, M., 2004. Economic instruments for managing water quality in New Zealand. Final report for the NZ Ministry for the environment. S/03/1393. CSIRO Land and Water, 73 pp.

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.

HELCOM, 2007. The HELCOM Baltic Sea Action Plan. HELCOM Ministerial Meeting. Krakow, Poland, 15 November 2007. 101 pp

Hershman, M.J., Good, J.W., Bernd-Cohen, T., Goodwin, R.F., Lee, V., Pogue, P., 1999. The Effectiveness of Coastal Zone Management in the United States. Coastal Management, 27(2): 113-138.

Hoffmann, J., 2009. Indicators for an ICZM. Experience with a problem-oriented approach. Journal of Coastal Conservation, DOI 10.1007/s11852-009-0052-0.

Hofmann, J., Behrendt, H., Gilbert, A., Janssen, R., Kannen, A., Kappenberg, J., Lenhart, H., Lise, W., Nunneri, C., Windhorst, W., 2005. Catchment-coastal zone interaction based upon scenario and model analysis: Elbe and the German Bight case study. Regional Environmental Change, 5:54-81.

Huang S., Lou, H., Xie, Y., Hu, J., 2003. Hydrodynamic Environment and its effects in the Xiangshan Bay. International Conference on Estuaries and Coasts, November 9-11, 2003, Hangzhou, China.

Huang, D.W., Wang, H.L., Feng, J.F., Zhu, Z.W., 2008a. Modelling algal densities in harmful algal blooms (HAB) with stochastic dynamics. Applied Mathematical Modelling, 32:1318-1326.

Huang, X.Q., Wang, J.H., Jiang, X.S., 2008b. Marine environmental capacity and pollution control in Xiangshan Gang. China Ocean Press. 348pp.

Humphrey, S., Burbridge, P., Blatch, C., 2000. US lessons for coastal management in the European Union. Marine Policy, 24(4):275-286.

Huppert, A., Blasius, B., Olinky, R., Stone, L., 2005. A model for seasonal phytoplankton blooms. Journal of Theoretical Biology, 236:276-290.

Hur, H.B., Jacobs, G.A., Teague, W.J., 1999. Monthly Variations of Water Masses in the Yellow and East China Seas. Journal of Oceanography, 55:171-184.

Hwang, E.K., Baek, J.M., Park, C.S., 2009. The mass cultivation of *Ecklonia stolonifera* Okamura as a summer feed for the abalone industry in Korea. Journal of Applied Phycology. DOI 10.1007/s10811-009-9402-4.

ICN (National Nature Conservation Institute), 2005. Plano de Ordenamento do Parque Natural da Ria Formosa. Estudos de caracterização, diagnóstico e programa de execução (In Portuguese).

Imperial, M.T., Hennessey, T. M., 1996. An ecosystem-based approach to managing estuaries: An assessment of the National Estuary program. Coastal Management, 24:115-139.

IMPRESS (working group 2.1. for the identification of pressures and assessment of impacts). 2003. Analysis of Pressures and Impacts. Guidance document No 3. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Office for Official Publications of the European Communities, Luxembourg. 157 pp.

Information Center of General Office of Zhejiang Provincial Government, 2006. The Economic Development of Zhejiang Province. The People's Government of Zheijang Province.

Islam, Md.S., Tanaka, M., 2004. Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. Marine Pollution Bulletin, 48:624–649.

Islam, M., 2005. Nitrogen and phosphorus budget in coastal and marine cage aquaculture and impacts of effluent loading on ecosystem: review and analysis towards model development. Marine Pollution Bulletin, 50(1):48-61.

Isobe A., Fujiware, E., Chang, P., Sugimatsu, K., Shimizu, M., Matsuno, T., Manda, A., 2004. Intrusion of Less Saline Shelf Water into the Kuroshio Subsurface Layer in the East China Sea. Journal of Oceanography, 60:853-863.

James, I.D., 2002. Modelling pollution dispersion, the ecosystem and water quality in coastal waters: a review. Environmental Modelling & Software, 17:363-385.

Jin, D., Hoagland, P., Dalton, T.M., 2003. Linking economic and ecological models for a marine ecosystem. Ecological Economics, 46:367-385.

Jolly, C.M., Clonts, H.A., 1993. Economics of aquaculture. Haworth Press, USA. 319 pp.

Jørgensen, S.E., Bendoricchio, G., 2001. Fundamentals of ecological modelling, 3rd edition. Developments in Environmental Modelling 21. Elsevier, Netherlands. 530 pp.Kaliba, A., Ngugi, C., Mackambo, J., Osewe, K., Senkondo, E., Mnembuka, B., Amisah, S., 2007. Potential effect of aquaculture promotion on poverty reduction in Sub-Saharan Africa. Aquaculture International, 15:445-459.

Kalnay, E., Kanamitsu, M., Kistler, R., Collins, W., Deaven, D., Gandin, L., Iredell, M., Saha, S., White, G., Woollen, J., Zhu, Y., Chelliah, M., Ebisuzaki, W., Higgins, W., Janowiak, J., Mo, K.C., Ropelewski, C., Wang, J., Leetmaa, A., Reynolds, R., Jenne, R., Joseph, D., 1996. The NCEP/NCAR 40-year reanalysis project, Bull. Amer. Meteor. Soc., 77:437-470.

Katranidis, S., Nitsi, E., Vakrou, A., 2003. Social acceptability of aquaculture development in coastal areas: the case of two Greek islands. Coastal Management, 31:37-53.

Kay, R., Eliot, I., Panizza, V., Donaldson, B., 1997. Reforming coastal management in Western Australia. Ocean & Coastal Management, 35(1):1-29.

Kearney, J., Berkes, F., Charles, A., Pinkerton, E., Wiber, M., 2007. The Role of Participatory Governance and Community-Based Management in Integrated Coastal and Ocean Management in Canada. Coastal Management, 35(1):79-104.

Kingdon, G.G., Knight, J., 2003. Well-being poverty versus income poverty and capabilities poverty. Global Poverty Reduction Group, Centre for the Study of African Economies, University of Oxford. 32 pp.

Knecht, R.W., Cicin-Sain, B., Fisk, G.W., 1996. Perceptions of the performance of state coastal zone management programs in the United States. Coastal Management, 24(2):141-163.

Knecht, R.W., Archer, J., 1993. "Integration" in the US coastal zone management program, Ocean & Coastal Management, 21:183–200.

Knecht, R.W., Cicin-Sain, B., Fisk, G.W., 1997. Perceptions of the performance of state coastal zone management programs in the United States. II. Regional and state comparisons. Coastal Management, 25(3):325-343.

Krishnamurthy, R.R., Kannen, A., Ramanathan, A.L., Tinti, S., Glavovic, B.C., Green, D.R., Han, Z., Agardy, T.S. (Eds.), 2008. Integrated Coastal Zone Management - The Global Challenge. Research Publishing, Singapore, Chennai.

Kumar, M., Kumar, P., 2008. Valuation of the ecosystem services: A psycho-cultural perspective. Ecological Economics, 64:808-819.

La Rosa, T., Mirto, S., Mazzola, A., Maugeri, T.L., 2004. Benthic microbial indicators of fish farm impact in a coastal area of the Tyrrhenian Sea. Aquaculture, 230:153–167.

Lal, P., 2003. Economic valuation of mangroves and decision-making in the Pacific. Ocean & Coastal Management, 46(9-10):823-844.

LDEO, Lamont-Doherty Earth Observatory. 2008. IRI/LDEO Climate Data Library. The Earth Institute, University of Columbia. Available online at: http://iridl.ldeo.columbia.edu/.

Ledoux, L., Turner, R.K., 2002. Valuing ocean and coastal resources: a review of practical examples and issues for further action. Ocean & Coastal Management, 45:583-616.

Ledoux, L., Beaumont, N., Cave, R., Turner, R.K., 2005. Scenarios for integrated river catchment and coastal zone management. Regional Environmental Change, 5:82-96.

Lenihan, H.S., Micheli, F., Shelton, S.W., Peterson, C.H., 1999, The influence of multiple environmental stressors on susceptibility to parasites: an experimental determination with oysters. Limnology and Oceanography, 44:910-924.

Letcher, R.A., Jakeman, A.J., Calfas, M., Linforth, S., Baginska, B., Lawrence, I., 2002. A comparison of catchment water quality models and direct estimation techniques. Environmental Modelling & Software, 17:77-85.

Lewis III, R.R., Clark, P.A., Fehring, W.K., Greening, H.S., Johansson, R.O., Paul, R.T., 1999. The Rehabilitation of the Tampa Bay Estuary, Florida, USA, as an Example of Successful Integrated Coastal Management. Marine Pollution Bulletin, 37(8-12):468-473.

Lewis, J., 2001. Policies to Promote Growth and Employment in South Africa, Discussion Paper 16, Southern Africa Department, World Bank, Washington, D.C. 103 pp.

Lillesand, T.M., Kiefer, R.W., 2000. Remote sensing and image interpretation. 4th edition. John Wiley & Sons, Inc, New York.

Lindahl, O., Hart, R., Hernroth, B., Kollberg, S., Loo, L.O., Olrog, L., Rehnstam-Holm, A.S., Svensson, J., Svensson, S., Syversen, U., 2005. Improving Marine Water Quality by Mussel Farming: A Profitable Solution for Swedish Society. AMBIO: A Journal of the Human Environment, 34:131-138.

Lipton, D.W., Hicks, R., 2003. The cost of stress: Low dissolved oxygen and recreational striped bass (Morone saxatilis) fishing in the Patuxent River. Estuaries, 26:310–315.

Liu, C.L., Chen, H.T., Ren, H.B., Zhang, G.S., 2003. Nutrient elements in wet deposition (precipitation) from the Yellow Sea and the East China Sea regions. Marine Environmental Science (China), 22:26-30.

Liu, Y., Gupta, H., Springer, E., Wagener, T., 2008. Linking science with environmental decision making: Experiences from an integrated modeling approach to supporting sustainable water resources management. Environmental Modelling & Software, 23:846-858.

Liu, D., Keesing, J.K., Xing, Q., Shi, P., 2009. World's largest macroalgal bloom caused by expansion of seaweed aquaculture in China. Marine Pollution Bulletin, 58(6): 888-895.

Long, H., Zhou, Y., Yu, Y.F., Fu, G.J., 2008. Analyses on harmful algal blooms in Zhejiang coastal waters from 2001 to 2007. Marine Environmental Science 27, S1-S4.

Luiten, H., 1999. A legislative view on science and predictive models. Environmental Pollution, 100:5-11.

MA (Millennium Ecosystem Assessment). 2005. Global Assessment reports. Volume 1: Current state and trends. Chapter 19: Coastal systems. Island Press, Washington, DC.

Macleod, C.J.A., Scholefield, D., Haygarth, P.M., 2007. Integration for sustainable catchment management. Science of The Total Environment, 373:591-602.

Maes, F., 2008. The international legal framework for marine spatial planning. Marine Policy, 32:797-810.

MAOT (Ministério do Ambiente e do Ordenamento do Território) (2000) Plano de Bacia Hidrogra´fica das Ribeiras do Algarve - Caracterização Geral da Bacia Hidrográfica, 1ª Fase - Análise e Diagnóstico da Situação de Referência, Volume III. Ministério do Ambiente e do Ordenamento do Território: Instituto da Água e Direcção Regional do Ambiente do Algarve (in Portuguese).

Marinov, D., Galbiati, L., Giordani, G., Viaroli, P., Norro, A., Bencivelli, S., Zaldivar, J.M., 2007. An integrated modelling approach for the management of clam farming in coastal lagoons. Aquaculture, 269:306-320.

Martinez-Cordero, F., Leung, P., 2004. Sustainable aquaculture and producer performance: measurement of environmentally adjusted productivity and efficiency of a sample of shrimp farms in Mexico. Aquaculture, 241:249–268.

Matias, D., Joaquim, S., Leitão, A., Massapina, C., 2008. Effect of geographic origin, temperature and timing of broodstock collection on conditioning, spawning success and larval viability of *Ruditapes decussatus* (Linné, 1758). Aquaculture International, doi: 10.1007/s10499-008-9197-3.

McAllister, T.L., Overton, M.F., Brill, J., 1996. Cumulative impact of marinas on estuarine water quality. Environmental Management, 20:385-396.

McCreary, S., Gamman, J., Brooks, B., Whitman, L., Bryson, R., Fuller, B., McInerny, A., Glazer, R., 2001. Applying a Mediated Negotiation Framework to Integrated Coastal Zone Management. Coastal Management, 29(3):183-216.

McFadden, L., 2007. Governing Coastal Spaces: The Case of Disappearing Science in Integrated Coastal Zone Management. Coastal Management, 35(4):429-443.

McGuckin, S.O., Jordan, C., Smith, R.V., 1999. Deriving phosphorus export coefficients for corine land cover types. Water Science and Technology, 39:47-53.

McKindsey, C.W., Thetmeyer, H., Landry, T., Silvert, W., 2006. Review of recent carrying capacity models for bivalve culture and recommendations for research and management. Aquaculture, 261:451-462.

McNeil, T. C., Rousseau, F. R. and Hildebrand, L. P., 2006. Community-Based Environmental Management in Atlantic Canada: The Impacts and Spheres of Influence of the Atlantic Coastal Action Program. Environmental Monitoring and Assessment, 113(1): 367-383.

McNie, E.C., 2007. Reconciling the supply of scientific information with user demands: an analysis of the problem and review of the literature. Environmental Science & Policy, 10(1):17-38.

Moll, A., Radach, G., 2003. Review of three-dimensional ecological modelling related to the North Sea shelf system. Progress in Oceanography, 57:175-217.

Msuya, F.E., 2006. The impact of seaweed farming on the social and economic structure of seaweed farming communities in Zanzibar, Tanzania. In A.T. Critchley, M. Ohno and D.B Largo (Eds), World Seaweed Resources, Version: 1.0. 27 pp.

Murawski, S.A., 2007. Ten myths concerning ecosystem approaches to marine resource management. Marine Policy 31, 681-690.

Murawski, S., Cyr, N., Davidson, M., Hart, Z., NOAA, Balgos, M., Wowk, K., Cicin-Sain, B., Global Forum, 2008. Policy brief on achieving EBM and ICM by 2010 and progress indicators. 4th Global Conference on Oceans, Coast, and Islands: Advancing Ecosystem Management and Integrated Coastal and Ocean Management in the Context of Climate Change. April 7-11, 2008, Hanoi, Vietnam. 70p.

Musango, J.K., de Wit, M.P., Lombard, J.P., Gu, H., Luo, Q., Ye, C., Zhang, X., Zhang, Z., Zhu, M., 2007. Estimating production functions in selected Chinese Aquaculture production systems. CSIR Report: CSIR/NRE/RBSD/ER/2007/0169/C, Stellenbosch, South Africa.

Naidoo, K., Maneveldt, G., Ruck, K., Bolton, J.J., 2006. A comparison of various seaweed-based diets and formulated feed on growth rate of abalone in a land-based aquaculture system. Journal of Applied Phycology, 18:437-443.

NAP (Australian National Action Plan for Salinity and Water Quality), 2000. Monitoring and evaluation: resource condition indicators. Estuarine, Coastal and marine habitat integrity. Natural Resource Management (NRM) Fact sheets, 18 pp.

National Standard of People's Republic of China, 1997. Sea water quality standard, GB3097-1997.

NBSC, National Bureau of Statistics of China, 2007. China Statistical Yearly Data. Available online: http://www.stats.gov.cn/english/statisticaldata/yearlydata/, searched on 4th December 2007.

Neal, C., Leeks, G.J.L., Millward, G.E., Harris, J.R.W., Huthnance, J.M., Rees, J.G., 2003. Land-ocean interaction: processes, functioning and environmental management from a UK perspective: an introduction. The Science of the Total Environment, 314-316:3-11.

Neitsch, S.L., Arnold, J.G., Kiniry, J.R., Williams, J.R., Kiniry, K.W., 2002. Soil and Water Assessment Tool theoretical documentation. TWRI report TR-191, Texas Water Resources Institute, College Station.

Neori, A., Shpigel, M., 2006. An integrated system for farming fish, seaweed and abalone (a solicited contribution). In: CAB International Aquaculture Compendium.Wallingford, UK.

Neori, A., Ragg, L.C., Shpigel, M., 1998. The integrated culture of seaweed, abalone, fish and clams in modular intensive land-based systems: II. Performance and nitrogen partitioning within an abalone (*Haliotis tuberculata*) and macroalgae culture system. Aquacultural Engineering, 17(4):215-239.

Neori, A., Shpigel, M., Ben-Ezra, D., 2000. A sustainable integrated system for culture of fish, seaweed and abalone. Aquaculture, 186(3-4):279-291.

Neori, A., Msuya, F.E., Shauli, L., Schuenhoff, A., Kopel, F., Shpigel, M., 2003. A novel three-stage seaweed (*Ulva lactuca*) biofilter design for integrated mariculture. Journal of Applied Phycology, 15:543-553.

Neori, A., Chopin, T., Troell, M., Buschmann, A.H., Kraemer, G.P., Halling, C., Shpigel, M., Yarish, C., 2004. Integrated aquaculture: rationale, evolution and state of the art emphasizing seaweed biofiltration in modern mariculture. Aquaculture, 231:361-391.

Newell, R.I.E., 2004. Ecosystem influences of natural cultivated populations of suspension-feeding bivalve molluscs: a review. Journal of Shellfisheries Research, 23:51-61.

Nijkamp, P., van den Bergh, J.C.J.M., 1997. New advances in economic modelling and evaluation of environmental issues. European Journal of Operational Research, 99:180-196.

Nikolaidis, N.P., Heng, H., Semagin, R., Clausen, J.C., 1998, Non-linear response of a mixed land use watershed to nitrogen loading. Agriculture, Ecosystems & Environment, 67:251-265.

Nikolaidis, N.P., Karageorgis, A.P., Kapsimalis, V., Drakopoulou, P., Skoulikidis, N., Behrendt, H., Levkov, Z., 2009. Management of nutrient emissions of Axios River catchment: Their effect in the coastal zone of Thermaikos Gulf, Greece. Ecological Modelling, 220:383-396.

Ning, X.R., Hu, X.G., 2002. Aquaculture ecology and carrying capacity assessment in Xiangshan Harbour. Beijing: Ocean Press, pp92-93.

Ningbo Municipal People's Government, 2006. The Eleventh Five-Year Plan. Chapter VI. Accelerating the Advancement of Urbanization, and Promoting harmonious Progress of Different Areas. Available at http://english.ningbo.gov.cn/col/col450/index.html, last access at October 2008.

Nobre, A.M., 2009. An ecological and economic assessment methodology for coastal ecosystem management. Environmental Management, 44(1):185-204.

Nobre, A.M, Ferreira, J.G., 2009. Integration of ecosystem-based tools to support coastal zone management. Journal of Coastal Research, SI 56:1676-1680.

Nobre A.M., Ferreira, J.G., Newton, A., Simas, T., Icely, J.D., Neves, R., 2005. Management of coastal eutrophication: Integration of field data, ecosystem-scale simulations and screening models. Journal of Marine Systems, 56:375-390.

Nobre, A.M., Musango, J.K., de Wit, M.P., Ferreira, J.G., 2009. A dynamic ecological-economic modeling approach for aquaculture management. Ecological Economics. DOI: 10.1016/j.ecolecon.2009.06.019.

NOFB, Ningbo Ocean and Fisheries Bureau. 2007. Distribution plan of fish cages in Xiangshan Bay. 25pp.

NRMMC (National Resource Management Ministerial Council), 2006. National cooperative approach to Integrated Coastal Zone Management. Framework and implementation plan. NRMMC, Department of the Environment and Heritage, Canberra, ACT. 57 pp.

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.

Nunes, P.A.L.D., van den Bergh, J.C.J.M., 2001. Economic valuation of biodiversity: sense or nonsense? Ecological Economics, 39:203-222.

Nunes, P.A.L.D, van den Bergh, J.C.J.M., 2004. Can People Value Protection against Invasive Marine Species? Evidence from a Joint TC–CV Survey in the Netherlands. Environmental & Resource Economics, 28:517-532.

Nunneri, C., Hofmann, J., 2005. A participatory approach for Integrated River Basin Management in the Elbe catchment. Estuarine, Coastal and Shelf Science, 62:521-537.

NZME (New Zealand Ministry for the Environment), 1998. Environmental performance indicators. Summary of proposed indicators for the marine environment. Ref. ME296, 32 pp.

Oglethorpe, D.R., Miliadou, D., 2001. Economic valuation of the non-use attributes of a wetland: a case-study for Lake Kerkini. Journal of Environmental Planning and Management, 43:755-767.

Olsen, S., Tobey, J., Kerr, M., 1997. A common framework for learning from ICM experience. Ocean & Coastal Management, 37(2):155-174.

Olsen, S.B., 2003. Frameworks and indicators for assessing progress in integrated coastal management initiatives. Ocean & Coastal Management, 46(3-4):347-361.

Ortiz-Lozano, L., Granados-Barba, A., Solís-Weiss, V., García-Salgado, M.A., 2005. Environmental evaluation and development problems of the Mexican Coastal Zone. Ocean and Coastal Management, 48:161–176.

Paez-Osuna, F., Guerrero-Galvan, S.R., Ruiz-Fernandez, A.C., 1999. Discharge of Nutrients from Shrimp Farming to Coastal Waters of the Gulf of California. Marine Pollution Bulletin, 38(7):585-592.

Pandey, V.K., Panda, S.N., Sudhakar, S., 2005. Modelling of an Agricultural Watershed using Remote Sensing and a Geographic Information System. Biosystems Engineering, 90:331-347.

Peirce, M., 1998. Computer-based models in Integrated Environmental Assessment. Technical report no. 14. European Environment Agency. 63 pp.

Pickaver, A.H., Gilbert, C., Breton, F., 2004. An indicator set to measure the progress in the implementation of integrated coastal zone management in Europe. Ocean & Coastal Management, 47(9-10):449-462.

Pickaver, A., 2003. Integrated Coastal Zone Management in the Baltic States - State of the Art Report. EUCC - The Coastal Union. 93 pp.

Pinto, R., Patrício, J., Baeta, A., Fath, B.D., Neto, J.M., Marques, J.C., 2009. Review and evaluation of estuarine biotic indices to assess benthic condition. Ecological Indicators, 9:1-25.

Plus, M., Chapelle, A., Lazure, P., Auby, I., Levavasseur, G., Verlaque, M., Belsher, T., Deslous-Paoli, J.-M., Zaldivar, J.-M., Murray, C. N., 2003. Modelling of oxygen and nitrogen cycling as a function of macrophyte community in the Thau lagoon. Continental Shelf Research, 23:1877-1898.

Raick C., Soetaert, K., Grégoire, M., 2006. Model complexity and performance: How far can we simplify? Progress in Oceanography, 70:27-57.

Raillard, O., Ménesguen, A., 1994. An ecosystem box model for estimating the carrying capacity of a macrotidal shellfish system. Marine Ecology Progress Series, 115:117-130.

Rantajärvi, E., Olsonen, R., Hällfors, S., Leppänen, J. M., Raateoja, M., 1998. Effect of sampling frequency on detection of natural variability in phytoplankton: unattended high-frequency measurements on board ferries in the Baltic Sea. ICES Journal of Marine Science, 55:697-704.

Refstie, S., Storebakken, T., Baeverfjord, G., Roem, A.J., 2001. Long-term protein and lipid growth of Atlantic salmon (Salmo salar) fed diets with partial replacement of fish meal by soy protein products at medium or high lipid level. Aquaculture, 193(1-2):91-106.

Rice, M.A., 2008. Environmental effects of shellfish aquaculture in the Northeast. Northeastern Regional Aquaculture Center Publication, 105:1-6.

Ridler, N., Wowchuk, M., Robinson, B., Barrington, K., Chopin, T., Robinson, S., Page, F., Reid, G., Szemerda, M., Sewuster, J. and Boyne-Travis, S., 2007. Integrated Multi-Trophic Aquaculture (IMTA): A potential strategic choice for farmers. Aquaculture Economics & Management, 11(1): 99-110.

Robertson-Andersson, D.V., 2007. Biological and economical feasibility studies of using *Ulva lactuca* (Chlorophyta) in recirculating systems in abalone farming. Doctoral dissertation, University of Cape Town, South Africa.

Robertson-Andersson, D.V., Potgieter, M., Hansen, J., Bolton, J.J., Troell, M., Anderson, R.J., Halling, C., Probyn, T., 2008. Integrated seaweed cultivation on an abalone farm in South Africa. Journal of Applied Phycology, 20(5):579-595.

Roebeling, P.C., Bohnet, I., Smith, M., Westcott, D., Kroon, F., Hartcher, M., Hodgen, M., Vleeshouwer, J., 2005. Landscapes Toolkit for triple-bottom-line assessment of land use scenarios in Great Barrier Reef catchments. MODSIM 2005 International Congress on

Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2005, 711-717.

Roelke, D., Buyukates, Y., 2001. The Diversity of Harmful Algal Bloom-Triggering Mechanisms and the Complexity of Bloom Initiation. Human and Ecological Risk Assessment: An International Journal, 7(5):1347-1362.

Rogers, S.I., Greenaway, B., 2005. A UK perspective on the development of marine ecosystem indicators. Marine Pollution Bulletin, 50:9–19.

Romstad, E., 2003. Team approaches in reducing nonpoint source pollution. Ecological Economics, 47:71-78.

Rosenberg, A.A., McLeod, K.L., 2005. Implementing ecosystem-based approaches to management for the conservation of ecosystem services. Marine Ecology Progress Series, 300:270–275.

RSS, Remote Sensing Systems, 2008. Special Sensor Microwave/Imager (SSM/I) data products. Remote Sensing Systems, Santa Rosa. Available online at: http://www.ssmi.com/.

Rupprecht Consult, IOI (International Ocean Institute), 2006. Evaluation of Integrated Coastal Zone Management (ICZM) in Europe - Final Report. European Commission.

Salomons, W., Turner, K., 2005. Catchment-coastal region research. Regional Environmental Change, 5:50-53.

Samsukal, P., 2004. A preliminary study of effluent water quality of land-based abalone farms in South Africa. Masters thesis. The Norwegian College of Fisheries Science, University of Tromsø Norway.

Sanchez, J., Xinping, W., Han, A., 2007. China, Peoples Republic of Fishery products. Global Agriculture Information Network, Report no. CH7094, USDA Foreign Agricultural Service.

Sankar, K., 2009. Effects of formulated feed fortification with kelp (*Ecklonia maxima* (Osbeck) Papenfuss) on the South African abalone (*Haliotis midae*) in terms of growth, water quality and cultivation system. Masters thesis. University of Cape Town, South Africa.

Sardá, R., Avila, C., Mora, J., 2005. A methodological approach to be used in integrated coastal zone management processes: the case of the Catalan Coast (Catalonia, Spain). Estuarine, Coastal and Shelf Science, 62(3):427-439.

Scavia, D., Bricker, S.B., 2006. Coastal eutrophication assessment in the United States. Biogeochemistry, 79:187-208.

Scheren, P.A.G.M., Kroeze, C., Janssen, F.J.J.G., Hordijk, L., Ptasinski, K.J., 2004. Integrated water pollution assessment of the Ebrie Lagoon, Ivory Coast, West Africa. Journal of Marine Systems, 44(1-2):1-17.

Schuenhoff, A., Shpigel, M., Lupatsch, I., Ashkenazi, A., Msuya, F.E., Neori, A., 2003. A semi-recirculating, integrated system for the culture of fish and seaweed. Aquaculture, 221:167-181.

Seaman, W. 2007. Artificial habitats and the restoration of degraded marine ecosystems and fisheries. Hydrobiologia, 580:143-155.

SEPA (Scottish Environment Protection Agency), 2005. Marine fish farm manual: regulation and monitoring of marine cage fish farming in Scotland—a manual of procedures.

Sequeira A., Ferreira, J.G., Hawkins, A.J.S., Nobre, A., Lourenço, P., Zhang, X.L., Yan, X., Nickell, T., 2008. Trade-offs between shellfish aquaculture and benthic biodiversity: A modelling approach for sustainable management. Aquaculture, 274:313-328.

Shipman, B., Stojanovic, T., 2007. Facts, Fictions, and Failures of Integrated Coastal Zone Management in Europe. Coastal Management, 35(2):375-398.

Shpigel, M., Neori, A., Marshall, A., 1996. The suitability of several introduced species of abalone (gastropoda: haliotidae) for land-based culture with pond grown seaweed in Israel. The Israeli Journal of Aquaculture - Bamidgeh, 48:192-200.

Shpigel, M., Neori, A., 1996. Abalone and seaweeds intensive cultivation in integrated land-based mariculture system: I. Proposed design and cost analyses. Aquacultural Engineering, 15:313-326.

Simas, T.C., Ferreira, J.G., 2007. Nutrient enrichment and the role of salt marshes in the Tagus estuary (Portugal). Estuarine, Coastal and Shelf Science, 75, 393-407.

Simpson, J.H., 1997. Physical processes in the ROFI regime. Journal of Marine Systems, 12:1-4.

Slocombe, D., 1993. Environmental planning, ecosystem science, and ecosystem approaches for integrating environment and development. Environmental Management, 17(3):289-303.

Slocombe, D.S., 1998. Lessons from experience with ecosystem-based management. Landscape and Urban Planning, 40(1-3):31-39.

Smit A.J., Robertson-Andersson, D.V., Peall, S., Bolton, J.J., 2007. Dimethylsulfoniopropionate (DMSP) accumulation in abalone *Haliotis midae* (Mollusca: Prosobranchia) after consumption of various diets, and consequences for aquaculture. Aquaculture, 269:377-389.

Smith, H.D., Potts, J., 2005. Managing Britain's marine and coastal environment, towards a sustainable future. Routledge and the National Maritime Museum. 309 pp.

SNPRCN (Serviço Nacional de Parques, Reservas e Conservação da Natureza) (1986) Plano de Ordenamento do Parque Natural da Ria Formosa (in Portuguese).

SOA (State Oceanic Administration China). 2006. Annual report of marine environment quality in China. 75 pp.

Söderqvist, T., Eggert, H., Olsson, B., Soutukorva, A., 2004. Economic valuation for sustainable development in the Swedish coastal zone. Ambio, 34:169–175.

Sohma, A., Sekiguchi, Y., Kuwae, T., Nakamura, Y., 2008. A benthic-pelagic coupled ecosystem model to estimate the hypoxic estuary including tidal flat - Model description and validation of seasonal/daily dynamics. Ecological Modelling, 215:10-39.

Soto, D., Aguilar-Manjarrez, J., Hishamunda, N. (Eds.), 2008. Building an ecosystem approach to aquaculture. FAO/Universitat de les Illes Balears Expert Workshop. 7–11 May

2007, Palma de Mallorca, Spain. FAO Fisheries and Aquaculture Proceedings. No. 14. Rome, FAO. 2008. 221 pp.

Stanners, D., Bosch, P., Dom, A., Gabrielsen, P., Gee, D., Martin, J., Rickard, L.; Weber, J. L., 2008. Frameworks for environmental assessment and indicators at the EEA. In: T. Hak, B. Moldan, A. L. Dahl (eds.), Sustainability Indicators: A Scientific Assessment. Island Press. 127-144 pp.

Stephenson, K., Aultman, S., Metcalfe, T., Miller, A., 2009. An evaluation of nutrient trading options in Virginia: A role for agriculture? In Southern Agricultural Economics Association Annual Meeting, Atlanta, Georgia, January 31-February 3, 2009.

Stojanovic, T., Ballinger, R.C., Lalwani, C.S., 2004. Successful integrated coastal management: measuring it with research and contributing to wise practice. Ocean & Coastal Management, 47(5-6):273-298.

Sundbäck, K., Linares, F., Larson, F., Wulff, A., Engelsen, A., 2004. Benthic nitrogen fluxes along a depth gradient in a microtidal fjord: The role of denitrification and microphytobenthos. Limnology and Oceanography, 49(4):1095-1107.

Svarstad, H., Petersen, L.K., Rothman, D., Siepel, H., WStzold, F., 2008. Discursive biases of the environmental research framework DPSIR. Land Use Policy, 25(1):116-125.

SWCB (State Water Control Board). 2006. General Virginia pollutant discharge elimination system (VPDES) Watershed permit regulation for total nitrogen and phosphorus discharges and nutrient trading in Chesapeake Bay Watershed in Virginia. 9 VAC 25-820-10 et seq. Effective: November 1, 2006. Virginia Department of Environmental Quality. 19pp.

Taveira Pinto, F., Paskoff, R., 2004. The practice of coastal zone management in Portugal. Journal of Coastal Conservation, 10(1):147-158.

TBNEP (Tillamook Bay National Estuary Project), 1999. Restoring the balance—Comprehensive Conservation and Management Plan for Tillamook Bay, Oregon. Garibaldi, Oregon: Tillamook County Performance Partnership under Cooperative Agreement CE-980127-01 with the U.S. Environmental Protection Agency, 382 pp.

Tol, R.S.J., Klein, R.J.T., Jansen, H.M.A., Verbruggen, H., 1996. Some economic considerations on the importance of proactive integrated coastal zone management. Ocean and Coastal Management, 32:39-55.

Tol, R., Vellinga, P., 1998. The European Forum on Integrated Environmental Assessment. Environmental Modeling and Assessment, 3(3):181-191.

Tolvanen, H., Kalliola, R., 2008. A structured approach to geographical information in coastal research and management. Ocean & Coastal Management, 51:485-494.

Tompkins, E.L., Few, R., Brown, K., 2008. Scenario-based stakeholder engagement: Incorporating stakeholders preferences into coastal planning for climate change. Journal of Environmental Management, 88:1580-1592.

Toth, F., Hizsnyik, E., 1998. Integrated environmental assessment methods: Evolution and applications. Environmental Modeling and Assessment, 3:193-207.

Tovar, A., Moreno, C., Manuel-Vez, M.P., Garcia-Vargas, M., 2000. Environmental impacts of intensive aquaculture in marine waters. Water Resources, 34(1):334-342.

Tribbia, J., Moser, S.C., 2008. More than information: what coastal managers need to plan for climate change. Environmental Science & Policy, 11:315-328.

Troell, M., Robertson-Andersson, D., Anderson, R.J., Bolton, J.J., Maneveldt, G., Halling, C., Probyn, T., 2006. Abalone farming in South Africa: An overview with perspectives on kelp resources, abalone feed, potential for on-farm seaweed production and socio-economic importance. Aquaculture, 257:266-281.

Troell, M., Joyce, A., Chopin, T., Neori, A., Buschmann, A. H. and Fang, J. G., 2009. Ecological engineering in aquaculture - Potential for integrated multi-trophic aquaculture (IMTA) in marine offshore systems. Aquaculture, In Press, doi:10.1016/j.aquaculture.2009.09.010.

Turner, R.K., 2000. Integrating natural and socio-economic science in coastal management. Journal of Marine Systems, 25(3-4):447-460.

Turner, R.K., van den Bergh, J.C.J.M., Söderqvist, T., Barendregt, A., van der Straaten, J., Maltby, E., van Ierland, E.C., 2000. Ecological-economic analysis of wetlands: scientific integration for management and policy. Ecological Economics, 35:7–23.

Turner, R.K., Paavola, J., Cooper, P., Farber, S., Jessamy, V., Georgiou, S., 2003. Valuing nature: lessons learned and future research directions. Ecological Economics, 46:493-510.

U.S. Department of Commerce, 2005. Water supply and wastewater treatment market in China. U.S. Department of Commerce, International Trade Administration. Washington, D.C., 105 pp. Available at http://www.ita.doc.gov, last access at October 2008.

Valderrama, D., Engle, C.R., 2002. Economics of better management practices (BMP) for semi-intensive shrimp farms in Honduras and shrimp cooperatives in Nicaragua. Report prepared under the World Bank, NACA, WWF and FAO consortium program on shrimp farming and the environment. Work in progress for public discussion. Published by the Consortium. 53pp.

van Alphen, J., 1995. The Voordelta integrated policy plan: administrative aspects of coastal zone management in the Netherlands. Ocean & Coastal Management, 26(2): 133-150.

Van Kouwen, F., Dieperink, C., Schot, P., Wassen, M., 2008. Applicability of Decision Support Systems for Integrated Coastal Zone Management. Coastal Management, 36(1):19-34

Veldkamp, A., Verburg, P.H., 2004. Modelling land use change and environmental impact. Journal of Environmental Management, 72:1-3.

Veloso-Gomes, F., Taveira-Pinto, F., Cassar, M., 2003. Portuguese coastal zones and the new coastal management plans. Journal of Coastal Conservation, 9(1):25-34.

Vidal, M., Duarte, C. M., Sbnchez, M.C., 1999. Coastal Eutrophication Research in Europe: Progress and Imbalances. Marine Pollution Bulletin, 38(10):851-854.

Volk, M., Hirschfeld, J., Dehnhardt, A., Schmidt, G., Bohn, C., Liersch, S., Gassman, P. W., 2008. Integrated ecological-economic modelling of water pollution abatement management options in the Upper Ems River Basin. Ecological Economics, 66:66-76.

Wade, A.J., Whitehead, P.G., Jarvie, H.P., Neal, C., Prior, H., Johnes, P.J., 2004. Nutrient monitoring, simulation and management within a major lowland UK river system: the Kennet. Mathematics and Computers in Simulation, 64:307-17.

WAG (Welsh Assembly Government), 2007. Making the Most of Wales' Coast – the Integrated Coastal Zone Management Strategy for Wales. 32 pp.

Wang, F.B., 2002. The water pollution introduced by aquaculture using net-cage and countermeasures controlling of it. Marine Sciences (China), 26:24-26.

Weise, A.M., Cromey, C.J., Callier, M.D., Archambault, P., Chamberlain, J., McKindsey, C.W., 2009. Shellfish-DEPOMOD: Modelling the biodeposition from suspended shellfish aquaculture and assessing benthic effects. Aquaculture, 288:239-253.

Wentz, F.J., Spencer, R.W., 1998. SSM/I Rain Retrievals within a Unified All-Weather Ocean Algorithm. Journal of the Atmospheric Sciences, 55:1613-1627.

Westmacott, S., 2001. Developing decision support systems for integrated coastal management in the tropics: Is the ICM decision-making environment too complex for the development of a useable and useful DSS? Journal of Environmental Management, 62:55-74.

WGEIM (ICES Working Group on Environmental Interactions of Mariculture), 2006. Report of the working group on environmental interactions of mariculture (WGEIM). ICES WGEIM Report 2006, ICES Mariculture Committee, ICES CM 2006/MCC:03, Ref. ACME. 201 pp.

Whalen, J., Uljee, I., White, R., Neis, B., Engelen, G., 2004. An individual based model of the lobster fishery in St. John Bay, Newfoundland, Canada, Research Institute for Knowledge Systems, Maastricht. 87pp.

Whitall, D., Bricker, S., Ferreira, J., Nobre, A., Simas, T., Silva, M., 2007. Assessment of Eutrophication in Estuaries: Pressure-State-Response and Nitrogen Source Apportionment. Environmental Management, 40:678-690.

Whitmarsh, D.J., Cook, E.J., Black, K.D., 2006. Searching for sustainability in aquaculture: An investigation into the economic prospects for an integrated salmon-mussel production system. Marine Policy, 30:293-298.

WL|Delft-Hydraulics (1996). Delft3D-FLOW User Manual Version 3.05. Delft, Netherlands, WL|Delft Hydraulics, 614 pp.

Xu, Z., Lin, X., Lin, Q., Yang, Y., Wang, Y., 2007. Nitrogen, phosphorus, and energy waste outputs of four marine cage-cultured fish fed with trash fish. Aquaculture, 263:130-141.

Xue, G., 2005. China and international fisheries law and policy. Publications on Ocean Development, 50. Martinus Nijhoff Publishers, Leiden, Boston, 326 pp.

Ye, S.F., Huang, X.Q., 2003. HABs in East China Sea：Surveillance and monitoring. Marine environmental Science, 22(2):10-14.

Yuan, D., Lin, B., Falconer, R.A., Tao, J., 2007. Development of an integrated model for assessing the impact of diffuse and point source pollution on coastal waters. Environmental Modelling & Software 22(6):871-879.

Yunhua, L., Beng, C.S., Wenzhi, L., 1998. Education, experience and productivity of labor in China's township and village enterprises: the case of Jiangsu province. China Economic Review, 9(1):47-58.

Zhang, C.L., Liu, J.F., Li, Y.C., Chen, Z., 2002. Analysing the present condition and countermeasure of cultured large yellow croaker Pseudosciaena crocea in Fujian Province. Journal of the Shanghai Fisheries University, 11(1):77-83.

Zhang, L.L., Jiang, X.S., Cai, Y.H., Li, Z.E., 2007. Comprehensive assessment of the situation of water quality at the red tide monitoring area of Xiangshan Harbor in recent four years. Transaction of Oceanology and Limnology, 4:98-103.

ZOFB, Zhejiang Ocean and Fisheries Bureau. 2008. Bulletin of Marine Environment in Zhejiang Province in 2007. pp40.

Žydelis, R., Esler, D., Kirk, M., Boyd, W.S., 2008. Effects of off-bottom shellfish aquaculture on winter habitat use by molluscivorous sea ducks. Aquatic Conservation: Marine and Freshwater Ecosystems. DOI: 10.1002/aqc.977.

Zylicz, T., 2003. Instruments for water management at the drainage basin scale. Ecological Economics, 47:43–51.