UNIVERSIDADE DO ALGARVE

Study of the physical, chemical and biological processes in semi-intensive fishponds: development of a mathematical model as a tool for managing white seabream (*Diplodus sargus*) production

Dalila do Rosário Encarnação Serpa

Doutoramento em Ciências do Mar, da Terra e do Ambiente

Ramo de Ciências do Mar

(Especialidade de Processos de Ecossistemas Marinhos)

2011

UNIVERSIDADE DO ALGARVE

Study of the physical, chemical and biological processes in semi-intensive fishponds: development of a mathematical model as a tool for managing white seabream (*Diplodus sargus*) production

Dalila do Rosário Encarnação Serpa

Doutoramento em Ciências do Mar, da Terra e do Ambiente

Ramo de Ciências do Mar

(Especialidade de Processos de Ecossistemas Marinhos)

Trabalho efectuado sob a orientação de: Professora Doutora Maria Teresa Dinis

Professor Doutor Luís Cancela da Fonseca

Professor Doutor Pedro Duarte

Doutor Miguel Caetano

2011

“The important thing in science is not so much to obtain new facts as to discover new ways of thinking about them”.

William Bragg

Este trabalho é dedicado aos meus pais
e às minhas irmãs

Agradecimentos

Foram muitas as pessoas que contribuíram para a concretização deste trabalho e que me apoiaram ao longo dos últimos anos. A todas elas não posso deixar de expressar os meus sinceros agradecimentos.

Gostaria especialmente de agradecer à Professora Doutora Maria Teresa Dinis por ter aceitado orientar esta tese, por todo o apoio prestado ao longo do trabalho e pela sua constante disponibilidade.

Ao Professor Doutor Pedro Duarte, por toda a ajuda prestada nas várias etapas do trabalho, pela disponibilidade e paciência para responder às minhas dúvidas, pelos ensinamentos sobre modelação ecológica e pelos comentários e sugestões que tanto enriqueceram este trabalho.

Ao Professor Doutor Luís Cancela da Fonseca, pelas sugestões construtivas, pelo encorajamento e por todo o apoio e ajuda prestada durante a realização deste trabalho.

Ao Doutor Miguel Caetano, por ter aceitado orientar esta tese numa fase já adiantada dos trabalhos, por todo o interesse demonstrado, pelo apoio constante, pela amabilidade e simpatia.

À Doutora Manuela Falcão, por me ter iniciado na investigação científica e por me ter ensinado muito do que sei hoje. Pelo seu incentivo e pela generosidade que lhe é característica.

Ao Doutor Pedro Pousão, por me ter acolhido na Estação Piloto de Piscicultura de Olhão e por ter facilitado todo o trabalho experimental, pela sua constante disponibilidade e interesse neste estudo.

À Doutora Narcisa Bandarra, por ter prestado, com toda a disponibilidade, esclarecimentos fundamentais no âmbito da bioquímica dos organismos e, pela sua amabilidade.

Às minhas colegas do Laboratório de Química do IPIMAR de Olhão, Cristina, Jú, Dora e Iolanda, pelo excelente ambiente de trabalho e por todo o apoio prestado. Um grande e especial obrigada à Milú, pela preciosa ajuda no trabalho laboratorial e de campo, pelos bons momentos, pelo incentivo e amizade.

Gostaria também de agradecer, a todos os que na Estação de Piscicultura do IPIMAR colaboraram neste trabalho, em especial ao Hugo, por todo o apoio no trabalho de campo e pelo seu interesse. Um grande obrigado também ao Miguel, Ezequiel, Sara, Ana, Lídia, Marisa, Marco, Ana Marçalo, Luísa, João, Telma, e Isabel por toda a ajuda prestada e pelo bom ambiente de trabalho. Agradeço ainda a todos os meus colegas do IPIMAR de Olhão, em especial à Alexandra Garcia, Pedro Lino, Susana Carvalho, Francisco Leitão, Ana Moura, pelo apoio, incentivo e pelos momentos de boa disposição. Não posso deixar também de agradecer ao Sr. Tibério Simões, Zé Luís Sofia, Luz Abreu e ao Sr. Fernando Catarino por estarem sempre disponíveis a ajudar e pelo excelente ambiente de trabalho.

Aos meus pais por terem sempre apoiado e respeitado as minhas decisões, e pelo incentivo nos momentos mais críticos.

Às minhas irmãs, Catarina e Carmo, é bom saber que há alguém com quem pudemos sempre contar. Ao Pedro, pela amizade, carinho e pelas palavras de encorajamento nos momentos mais complicados. A todos os meus amigos, pelos momentos de boa disposição e descontração, que foram determinantes para conseguir terminar este trabalho.

Gostaria ainda de agradecer às duas Instituições que tornaram possível a realização deste trabalho. Ao Instituto de Investigação das Pescas e do Mar (IPIMAR), pela disponibilização das instalações e equipamentos. À Fundação para a Ciência e Tecnologia, pelo apoio financeiro disponibilizado através da concessão da bolsa de doutoramento (SFRH/BD/27643/2006).

Obrigada…

Abstract

Semi-intensive aquaculture has been recognised as an “environmentally friendly” option. However, the low profitability and competitiveness of these systems compromise their economic viability. The optimization of production is thereby crucial for the sustainability of semi-intensive pond aquaculture, and implies that fish yields are maximized with minimum impacts on the environment. Understanding the physical, chemical and biological processes occurring in fishponds is of outmost importance for defining farming strategies that optimize fish production. This knowledge is even more relevant when dealing with newly cultivated species, as the white seabream (*Diplodus sargus*). Due to the lack of information on the performance of this species in earth ponds, one of the main objectives of the present work was to study the physical, chemical and biological processes in white seabream ponds, over a production cycle. The most relevant results of this experimental work were that: i) the impacts of fish activity on bottom sediments are only noticeable above a fish biomass of 0.5 kg m^{-3} and a feeding rate of 5 kg d^{-1}; ii) pond sediment and water quality was comparable to that of natural systems, suggesting that the assayed farming conditions ensure a good pond environment; and iii) pond water quality was strongly dependent on inflowing water and on benthic nutrient fluxes, emphasizing the relevance of optimum water exchange rates and sediment treatment to an efficient pond management. The other main objective of this work was to develop an ecological model to be used as a tool for managing semi-intensive systems, to improve their economic and environmental performance. The added value of a modeling approach is that, due to their ability to integrate the complexity of fishpond processes, models can be used to simulate the effect of different management scenarios on the pond environment and on the adjacent coastal systems. The model was implemented and tested with the white seabream as a case study, using data collected over the experimental work, together with literature data. Model construction was done in 3 steps: i) implementation of a biogeochemical model; ii) implementation of a fish Dynamic Energy Budget (DEB) model and iii) coupling of the two models. The biogeochemical model developed in this study is a mechanistic model that reproduces the dynamics of organic and inorganic nutrient (nitrogen and phosphorus) forms as well as of oxygen, in the pelagic and benthic compartments of an earth pond. This model not only helped understanding the interactions between pond variables and processes but also how pond structural features and operational parameters affect the water and sediment quality of pond systems. The fish DEB model was able to reproduce the growth of white seabream as well as of gilthead seabream (*Sparus aurata*), a traditionally cultivated species in semi-intensive ponds. This model was used to investigate which biological processes are more likely to influence fish performance and to explain inter-species growth variability. A comparison between the DEB model parameters of the two Sparidae revealed that white seabream lower growth rates are presumably linked to a higher energy demand for body maintenance and a lower feed absorption efficiency. The coupled model was able to reproduce fish pond dynamics, and was further used to simulate different management scenarios, related to stocking densities, water exchange rates and feeding strategies. Scenarios and standard farming conditions were compared in terms of their effects on pond water and sediment quality, as well as on final fish yields and nutrient discharges into the environment. Using the Analytical Hierarchical Process (AHP) methodology, scenarios were ranked in order to evaluate the best management options for optimizing white seabream production. Results revealed that doubling the standard stocking density and improving feed absorption efficiency, may enhance the performance of semi-intensive white seabream production systems. Aside from providing a tool for managing aquaculture systems, this work contains valuable information for defining guidelines on environmental standards (e.g. Maximum Recommended Values) for marine fish farming.

Keywords: Biogeochemical model; Fish Dynamic Energy Budget model; Pond management; Semi-intensive aquaculture; Economic and environmental sustainability; *Diplodus sargus*.

Resumo

No atual contexto de declínio dos recursos marinhos, a aquacultura poderá desempenhar um papel determinante como fonte de proteína alternativa, para responder à crescente procura de produtos alimentares de origem marinha e reduzir a pressão da atividade piscatória. Como consequência da intensificação da produção à escala global, os impactes ambientais da aquacultura têm vindo a aumentar. Neste trabalho foi feita uma extensa revisão bibliográfica sobre os principais impactes ambientais desta atividade e possíveis medidas de mitigação destes impactes. A importância do conceito de capacidade de carga para a sustentabilidade da aquacultura foi também abordada neste trabalho de revisão, e são apresentadas algumas metodologias e ferramentas que podem ser utilizadas para a redução ou antecipação dos impactes da aquacultura, como é o caso dos modelos matemáticos e dos Sistemas de Apoio à Decisão.

Se a piscicultura intensiva é por vezes associada a uma degradação ambiental, aquela que é tipicamente realizada em regime semi-intensivo nos países mediterrânicos, tem sido reconhecida como uma opção "amiga" do ambiente. O menor grau de artificialidade (ou seja, as densidades de carga mais baixas, os caudais reduzidos e a menor quantidade de alimento fornecido) destes sistemas comparativamente com os sistemas intensivos reduzem substancialmente a sua pegada ecológica. Devido à crescente preocupação dos consumidores com a segurança alimentar e o bem-estar das espécies cultivadas, a procura de produtos provenientes da aquacultura semi-intensiva tem vindo a aumentar nos últimos anos. Apesar deste aumento da procura, estes sistemas são muitas vezes caracterizados por uma baixa rentabilidade, que resulta principalmente de uma baixa produtividade associada a elevados custos de produção. Para além da reduzida rentabilidade, a baixa competitividade da piscicultura semi-intensiva face aos baixos preços dos produtos de origem intensiva, compromete fortemente a sua viabilidade económica. Para assegurar o futuro deste tipo de aquacultura, é necessário desenvolver protocolos que permitam

otimizar a produção, ou seja maximizar a produção, mantendo produtos de alta qualidade e minimizando os impactes ambientais desta atividade.

Uma vez que o conhecimento dos processos físicos, químicos e biológicos em tanques de piscicultura é de primordial importância para a definição de estratégias de cultivo que permitam otimizar a produção nestes sistemas, um dos principais objectivos deste estudo foi estudar estes processos ao longo de um ciclo de produção de sargo (*Diplodus sargus*). Os modelos matemáticos, por serem capazes de integrar a dinâmica dos processos que ocorrem nos tanques de cultivo, podem ser utilizados para a simulação de diferentes cenários de gestão bem como para a previsão do impacte desta atividade no meio recetor, constituindo por isso uma ferramenta valiosa para a sustentabilidade da piscicultura semi-intensiva. Neste sentido, o outro grande objetivo deste trabalho consistia no desenvolvimento de um modelo ecológico para tanques de terra de cultivo semi-intensivo de peixes, de modo a maximizar a eficiência económica e a eficiência ambiental destes sistemas. Este modelo foi depois aplicado ao caso concreto do cultivo de sargo (*Diplodus sargus*), para avaliar a viabilidade do cultivo desta espécie em regime semi-intensivo. Uma vez que a diversificação de espécies tem sido referida como uma das estratégias para aumentar a rentabilidade e a competitividade da piscicultura semi-intensiva, o sargo, sendo uma espécie autóctone e de elevado valor comercial nos países mediterrânicos, é considerado como um potencial candidato à aquacultura semi-intensiva em tanques de terra.

O ensaio experimental destinado a estudar os processos físicos, químicos e biológicos em tanques de piscicultura, teve a duração de aproximadamente dois anos, e foi realizado nos tanques de terra da Estação Piloto de Piscicultura do IPIMAR, localizada no Parque Natural da Ria Formosa, em Olhão. Durante o ensaio, foram recolhidas amostras de material particulado, sedimento e água num tanque de produção de sargo e num tanque controle (sem peixes). A condição do sargo foi também avaliada ao longo do ensaio, tendo-se realizado amostragens

biológicas regulares. No que diz respeito ao material orgânico particulado, os resultados revelaram que a taxa de deposição das partículas aumentou linearmente com o tempo no tanque de cultivo. A contínua deposição de material particulado, proveniente da água de entrada, do alimento não digerido, da excreção fecal e do fitoplâncton senescente, conduziu a um enriquecimento orgânico dos sedimentos, sobretudo a partir do segundo ano de produção. Como consequência da mineralização da matéria orgânica acumulada nos sedimentos, houve um aumento das concentrações de nutrientes na água intersticial, principalmente durante os períodos de temperatura mais elevada, que estimulou a produção microfitobentónica no tanque de produção. Apesar dos impactes da atividade piscícola nos sedimentos de fundo só se tornarem evidentes a partir de uma biomassa de 0.5 kg m^{-3} e de uma quantidade diária de alimento fornecido superior a 5 kg, a qualidade dos sedimentos no tanque de cultivo foi comparável à laguna adjacente (Ria Formosa). Dado que a taxa de sobrevivência dos sargos no final do ensaio foi bastante elevada (94%), os resultados sugerem que as condições de cultivo experimentadas não causam constrangimentos nos tanques de produção, podendo servir como base para o setor produtivo. Outro indício da sustentabilidade do protocolo de cultivo seguido neste trabalho é a boa qualidade da água no tanque de cultivo, pois a sua composição química foi semelhante à da água de entrada e à do tanque de controle. Uma vez que a qualidade da água nos sistemas de cultivo semi-intensivo é frequentemente determinada pela qualidade dos sedimentos e da água de abastecimento, este trabalho experimental tinha também como propósito, estimar o contributo destas fontes de nutrientes para a disponibilidade de azoto e fósforo na água de um tanque de produção de sargo. O transporte sedimento-água de nutrientes, foi estimado de acordo com duas metodologias diferentes. Enquanto os fluxos difusivos foram calculados de acordo com a 1ª Lei de Fick, através dos gradientes de concentração entre a água intersticial e a água sobrenadante medidos no campo, os fluxos biológicos foram determinados em experiências de incubação, realizadas em laboratório. Os resultados mostraram que os fluxos difusivos de amónia e fosfato foram mais elevados no segundo ano do ciclo de produção, como consequência do

enriquecimento orgânico dos sedimentos do tanque de cultivo. A atividade da macrofauna bentónica contribuiu de forma substancial para o transporte sedimento-água de nutrientes, uma vez que os fluxos biológicos foram cerca de uma ordem de magnitude superior aos fluxos difusivos. Com base na informação disponível sobre as principais fontes e sumidouros de nutrientes num tanque de cultivo de peixes, foi feito um balanço de massas, para avaliar o contributo relativo dos sedimentos e da água de entrada para a disponibilidade de nutrientes dissolvidos na coluna de água. De acordo com os resultados deste balanço, os “inputs” diários de azoto de fósforo devido à água de entrada, foram notoriamente superiores aos dos sedimentos. Por outro lado, grande parte dos nutrientes dissolvidos é perdida através dos efluentes. Estes resultados evidenciam portanto a relevância da otimização das taxas de renovação de água e do tratamento dos sedimentos para uma eficiente gestão dos tanques.

Após a recolha da informação relativa aos processos físicos, químicos e biológicos num tanque de cultivo de sargo, iniciou-se o desenvolvimento do modelo matemático. A construção do modelo foi feita em três etapas: i) desenvolvimento de um modelo biogeoquímico; ii) desenvolvimento de um modelo de crescimento dos peixes; e iii) acoplamento dos 2 modelos. O modelo biogeoquímico desenvolvido neste estudo foi capaz de reproduzir a dinâmica dos elementos com maior probabilidade de afetar a produção piscícola e de causar impactes ambientais indesejáveis no meio recetor, como é o caso do azoto, do fósforo e do oxigénio. Uma das particularidades deste modelo, relativamente aos modelos existentes para tanques de terra, é a sua capacidade de simular não só as concentrações das formas inorgânicas de nutrientes, mas também as formas orgânicas, as quais têm sido referidas como de extrema relevância para a dinâmica dos nutrientes em ecossistemas aquáticos. Neste modelo, o compartimento pelágico foi acoplado ao compartimento bentónico, devido à importância das trocas entre o sedimento e a coluna de água em tanques de piscicultura pouco profundos (1.5 m). O modelo biogeoquímico foi calibrado usando as séries de dados recolhidas no tanque controle durante o ensaio experimental em tanques de terra, uma vez que nesta fase do trabalho não foram contempladas as interações entre os peixes e o ambiente. Esta estratégia permitiu reduzir a complexidade do modelo e assim identificar mais facilmente as interações entre as diferentes variáveis e processos na coluna de água e no sedimento, de modo a facilitar a calibração do modelo acoplado. A variabilidade dos compostos na coluna de água e na água intersticial deste tanque foi razoavelmente bem prevista pelo modelo. Nos sedimentos, os compostos orgânicos de fósforo foram simulados com grande exatidão, mas os conteúdos em carbono e azoto orgânicos foram mais imprecisos. A análise de sensibilidade realizada ao modelo permitiu compreender as interações entre as diferentes variáveis e os processos de um tanque de terra. Por exemplo, as formas inorgânicas de azoto e fósforo na água intersticial, foram particularmente sensíveis aos parâmetros do modelo relacionados com os processos aeróbios, o que explica a inter-variabilidade entre estes compostos e o oxigénio dissolvido na água intersticial. Os efeitos das características estruturais dos tanques e de alguns parâmetros operacionais (como por exemplo, a taxa de renovação da água), ao nível da qualidade da água e do sedimento, foram também testados no modelo, tendo-se recolhido informação importante para o dimensionamento de novas unidades de piscicultura e para a gestão das que estão em actividade. Para além de constituir uma base de conhecimento da biogeoquímica de tanques de terra, o presente trabalho permitiu identificar os processos que necessitam de um estudo mais aprofundado, como é o caso da ressuspensão do sedimento, da produtividade primária e da bioturbação, de forma a melhorar o desempenho do modelo e a assegurar a sua aplicação a outros sistemas aquáticos.

Um modelo biológico capaz de reproduzir o crescimento das espécies cultivadas constitui uma ferramenta útil para a gestão de tanques de piscicultura, uma vez que permite estudar a influência das condições de cultivo e de diversos parâmetros ambientais e fisiológicos, na produção piscícola. Neste trabalho foi desenvolvido um modelo de crescimento de peixes baseado na teoria DEB (“Dynamic Energy Budget”), por esta assentar em princípios fisiológicos comuns a todas as espécies e pelo facto da diversidade inter-específica ser traduzida através de um número relativamente reduzido de parâmetros, o que facilita a aplicação do modelo a outros sistemas de monocultivo, mas também a sistemas de policultivo. Outra das vantagens deste modelo de crescimento, reside no facto de este puder ser facilmente calibrado, o que é particularmente relevante quando existe pouca informação para estimar os parâmetros da espécie em estudo. Correndo o modelo para uma população de *n* peixes, cada um com um conjunto específico de parâmetros atribuído aleatoriamente, é possível selecionar o conjunto de parâmetros que permite o melhor ajuste entre os valores de peso e comprimento previstos e observados, e depois correr o modelo para simular um peixe médio. Esta estratégia de modelação foi adotada por a fisiologia do sargo estar relativamente pouco estudada, e por isso existirem algumas incertezas relativamente aos parâmetros desta espécie. Neste trabalho, para além do crescimento do sargo, o modelo foi utilizado para simular o crescimento da dourada (*Sparus aurata*), uma espécie tradicionalmente cultivada nos países mediterrânicos. Uma vez que o modelo conseguiu simular bastante bem o crescimento destas duas espécies de Sparídeos, foi possível averiguar quais os processos biológicos mais prováveis de afetar o crescimento piscícola. De acordo com os resultados do modelo, o crescimento é sobretudo determinado pela ingestão e pela eficiência de absorção do alimento, bem como pela fração de energia alocada para o metabolismo e para o crescimento do indivíduo. Uma vez que o sargo tem sido referido como uma espécie de crescimento lento comparativamente à dourada, o modelo foi utilizado para investigar as diferenças entre o crescimento destas duas espécies. A comparação entre os parâmetros específicos de cada espécie, revelou que o crescimento mais lento do sargo resulta de um maior gasto energético no metabolismo basal e de uma menor eficiência na absorção do alimento. De facto, um incremento de apenas 12% na eficiência de absorção do alimento resultou num aumento de cerca de 120g no peso do peixe no final do ciclo de produção, o que poderá ser suficiente para garantir a viabilidade do cultivo de sargo em sistemas semi-intensivos. Considerando que esta espécie omnívora, é presentemente alimentada com uma ração otimizada para a dourada, uma espécie tipicamente carnívora, os resultados do modelo sugerem que o desenvolvimento de rações específicas para o sargo pode contribuir para uma maximização da produção.

Após calibração do modelo biogeoquímico e do modelo biológico, estes foram acoplados. O acoplamento consistiu basicamente em utilizar os “outputs” do modelo biológico como “inputs” para o modelo biogeoquímico e vice-versa. No geral, o modelo acoplado conseguiu reproduzir a variabilidade dos compostos na coluna de água e nos sedimentos de um tanque de piscicultura, assim como o crescimento do sargo ao longo de um ciclo de produção. O balanço de massas construído com base nos resultados do modelo permitiu avaliar a eficiência da alimentação dos peixes. De acordo com os resultados deste balanço, mais de metade do alimento fornecido não é ingerido pelos peixes, o que justifica a baixa taxa de conversão alimentar (3.7) obtida para esta espécie. Este tipo de balanços permite também quantificar as fontes e sumidouros de nutrientes dissolvidos e sob a forma particulada, podendo ser utilizado para definir formas práticas de melhorar a qualidade da água e dos sedimentos dos tanques de cultivo, ao longo do ciclo produtivo. Devido à capacidade do modelo de reproduzir a dinâmica dos tanques de piscicultura, este foi utilizado para testar diferentes cenários de gestão: i) aumento da densidade de carga; ii) aumento/diminuição das taxas de renovação da água; iii) aumento/diminuição da quantidade de ração fornecida; iv) diminuição do conteúdo em fósforo da ração; v) aumento da eficiência de absorção do alimento vi) diminuição da velocidade de decaimento da ração. Os diferentes cenários e a simulação padrão foram comparados relativamente aos seus efeitos na qualidade da água e do sedimento no tanque de cultivo, na produção final de peixe e nas descargas de nutrientes para o meio recetor. Utilizando o método de Avaliação Multicritério (AHP – “Analytical Hierarchical Process”), foi atribuída uma pontuação a cada cenário por forma a identificar as medidas de gestão que permitem otimizar o cultivo de sargo. Os resultados no presente estudo revelaram que a duplicação da densidade piscícola e um aumento na eficiência

de absorção do alimento poderão melhorar o desempenho dos sistemas semi-intensivos de produção de sargo e assegurar a viabilidade do cultivo desta espécie. Para além do modelo constituir uma importante ferramenta de gestão para tanques de aquacultura, este trabalho contém ainda informação relevante para a definição de parâmetros de qualidade ambiental (como por exemplo, Valores Máximos Recomendados) para a piscicultura marinha realizada em tanques de terra.

Palavras-Chave: Modelo biogeoquímico; Modelo de crescimento de peixes; Gestão de tanques de piscicultura; Aquacultura semi-intensiva; Sustentabilidade económica e ambiental; *Diplodus sargus*.

Table of Contents

Agradecimentos i

Abstract v

Keywords vi

Resumo vii

Palavras-Chave xiv

Table of Contents xv

List of Figures xxiii

List of Tables xxxi

Chapter 1. General Introduction 1

1 Aquaculture 2

1.1 Basic concepts 2

1.2 Global production 2

1.3 Environmental impacts 5

1.4 Semi-intensive pond aquaculture 5

1.4.1 Advantages and disadvantages 5

1.4.2 Economical and environmental sustainability 6

2 Physical, chemical and biological processes in fishponds 10

3 Ecological modelling 11

4 Thesis aims and structure 13

Chapter 2. Impacts of Aquaculture and Mitigation Measures 17

Abstract 18

1 Introduction 18

2 Aquaculture Environmental Impacts 21

2.1 Organic matter enrichment 25
2.2 Nutrient enrichment 27
2.3 Chemical contamination 29
2.4 Spread of parasites and diseases 30
2.5 Habitat destruction and modification 31
2.6 Introduction of new species and new genetic varieties 32
2.7 Harvest of wild stocks as feed or seed/broodstock to aquaculture operations 34
2.8 Socio-economic impacts 35
3 The concept of carrying capacity in aquaculture 39
3.1 CC categories and definitions 40
3.2 CC and limiting factors 41
3.3 Methods for determining CC 44
4 Aquaculture and environmental management towards sustainable development 49
4.1 Mitigation measures 50
4.1.1 Interference in biogeochemical processes 51
4.1.2 Interference with the life cycles of wild species 53
4.1.3 Impacts of introduction of new species or genetic varieties 55
4.1.4 Degradation of genetic diversity 55
4.1.5 Modification and/or destruction of habitats 56
4.2 Ecoaquaculture 56
4.3 Drivers, pressures, states, impacts and responses (DPSIR) 61
4.4 Decision support systems 62
5 Concluding Remarks 64
Chapter 3. Physical, chemical and biological processes in semi-intensive fishponds 65

Chapter 3.1 Geochemical changes in white seabream (*Diplodus sargus*) earth ponds during a production cycle 67

Abstract 68

1 Introduction 69

2 Materials and Methods 70

2.1 Experimental system 70

2.2 Field sampling 71

2.2.1 Sediment-traps settled particles 72

2.2.2 Sediment cores 72

2.3 Analytical methodology 72

2.3.1 Porewater samples 73

2.3.2 Solid fraction 73

2.4 Fish sampling 74

2.5 Data analysis 74

3 Results 75

3.1 Particulate material 75

3.2 Sediment 77

3.3 Fish data 78

4 Discussion 80

Acknowledgments 82

Chapter 3.2 Contribution of benthic nutrient mineralization and inflowing water to the nutrient availability in semi-intensive fish earth ponds: consequences for pond water quality 83

Abstract 84

1 Introduction 85

2 Materials and Methods 87

2.1 Description of the system 87

2.2 Field sampling 87

2.3 Laboratory experiments 89

2.4 Analytical procedures 89

2.5 Calculations 90

2.5.1 Diffusive fluxes 90

2.5.2 Biologically-mediated fluxes 91

2.5.3 Nutrient inflow rates 91

2.6 Data analysis 92

3 Results 92

3.1 Field experiments 92

3.1.1 Overlying and inflowing water 92

3.1.2 Porewater 92

3.1.3 Diffusive fluxes 95

3.2 Laboratory experiments 95

3.2.1 Biologically-mediated fluxes 95

4 Discussion 98

4.1 Nutrient budgets 101

5 Conclusions 106

Acknowledgments 106

Chapter 4. Modelling biogeochemical processes in semi-intensive fish earth ponds: model development and calibration 107

Abstract 108

1 Introduction 109

2 Methodology 111

2.1 Description of the system 111

2.2 Model description 111

2.2.1 Pelagic module 111

2.2.2 Benthic module 113

2.3 Model equations 114

2.4 Model implementation 114

2.5 Model calibration 115

2.6 Sensitivity analysis 115

2.7 Statistical analysis 117

3 Results 118

3.1 Model calibration 118

3.1.1 Water column variables 118

3.1.2 Porewater variables 119

3.1.3 Sediment variables 119

3.2 Sensitivity analysis 124

3.2.1 Water column variables 125

3.2.2 Porewater variables 126

3.2.3 Sediment variables 130

4 Discussion 131

5 Conclusions 140

Ackowledgments 140

Appendix A 141

Appendix B 145

Appendix C 149

Appendix D 157

Chapter 5. Modelling the growth of white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) in semi-intensive earth ponds using the Dynamic Energy Budget approach **163**

Abstract 164

1 Introduction 165

2 Species information 167

3 Methodology 168

3.1 The DEB theory 168

3.2 Model development 169

3.2.1 Model equations 170

3.2.1.1 Ingestion 170

3.2.1.2 Assimilation 172

3.2.1.3 Reserves utilization 172

3.2.1.4 Growth 173

3.2.1.5 Temperature limitation 174

3.2.2 Parameter estimation 174

3.2.2.1 Maximum surface area-specific ingestion rate ($\{\dot{P}_{Xm}\}$) 174

3.2.2.2 Absorption efficiency (κ_X) 177

3.2.2.3 Fraction of reserves allocated for soma (κ), volume-specific costs for growth ($[E_G]$) and maximum energy storage density ($[E_m]$) 178

3.2.2.4 Volume-specific costs for maintenance ($[\dot{p}_M]$) 178

3.2.2.5 Shape coefficient (δ_m) 180

3.2.2.6 Arrhenius temperature (T_A) 180

3.2.3 Model forcing and calibration 181

3.2.4 Model implementation 183
3.2.5 Sensitivity Analysis 183
3.2.5.1 IBM model 183
3.2.5.2 "State variable" model........ 184
4 Results 184
5 Discussion........ 196
6 Conclusions 201
Acknowledgments........ 201
Chapter 6. Mathematical modelling as a tool for managing semi-intensive production systems 203
Abstract 204
1 Introduction 205
2 Methodology........ 208
2.1 Description of the system........ 208
2.2 Model development........ 209
2.3 Model forcing and calibration 214
2.4 Model implementation 215
2.5 Model performance 215
2.6 Scenario analysis 216
2.6.1 Scenario classification 217
3 Results 219
3.1 Model performance 220
3.1.1 Water column variables........ 220
3.1.2 Porewater variables........ 222
3.1.3 Sediment variables........ 225

3.1.4 Biological variables 227

3.2 Scenario analysis 228

3.2.1 Water quality 229

3.2.2 Sediment quality 233

3.2.3 Fish biomass 236

3.2.4 Environmental impacts 237

3.2.5 Scenario classification 237

4 Discussion 239

4.1 Model performance 239

4.2 Nutrient budgets 244

4.3 Scenario analysis 248

5 Conclusions 252

Acknowledgments 253

Chapter 7. General conclusions and Future Perspectives 255

1 General Conclusions 256

2 Future Perspectives 260

Chapter 8. References 263

List of Figures

Figure 1.1 – Total aquaculture (except aquatic plants) and capture fisheries production versus human consumption, from 2000 to 2009. Source: FAO (2011a). 4

Figure 1.2 – Global aquaculture production by trophic level, in 2009 (Adapted from Primavera (2006)). 4

Figure 2.1 – The parabolic relationship between stock and yield in bivalve culture. Carrying capacity increases with stock up to a point above which individual growth is severely compromised due to food limitation (see text). 43

Figure 2.2 – Areas cultivated in Sungo Bay since 1999 with kelps (*Laminaria japonica*), oysters (*Crassostrea gigas*) and scallops (*Chlamys farreri*), including part of a two dimensional model grid (upper left corner), for which the spatial step is 500 m (refer text). 47

Figure 2.3 – Diagram showing the interactions and feedbacks among different carrying capacity categories towards an accepted aquaculture scenario, and the tools used for physical, production and ecologic CC and for the scenario selection. Continuous lines show direct influences of CC over the accepted scenario or other CC category. Physical CC limits production CC. Dashed lines showing feedbacks from production and ecological CC to economic and social CC (see text). 50

Figure 3.1.1 – Location of the Aquaculture Research Station (ARS). 71

Figure 3.1.2 – Monthly ration (kg) supplied to white seabream during the production cycle. 72

Figure 3.1.3 – Particle-settling rates (g m^{-2} d^{-1}) in Pf (♦) and Pnf (o), during the sampling period. 75

Figure 3.1.4 – Total organic carbon (TOC), total organic phosphorus (TOP), phosphorus bound-Fe (P-Fe) and chlorophyll *a* (Chl *a*) concentrations in superficial sediments (0-2 cm) of Pf (♦) and Pnf (o), during the sampling period. 78

Figure 3.1.5 – Correspondence analysis ordination plot of physical (Temp – temperature, in ºC), chemical (NH_4^+ - ammonium; NO_3^- - nitrates; NO_2^- - nitrites; $Si(OH)_4$ – silicates; HPO_4^{2-} - phosphates; DON – dissolved organic nitrogen; DOP – dissolved organic phosphorus, in µM) and biological (Chl *a* – chlorophyll *a*; Phaeop – phaeopigments, in µg g^{-1}) parameters in porewater and superficial sediments of Pf (♦) and Pnf (O) during the sampling period. Parameters annual range in Pf. 79

Figure 3.2.1 – Location of the Aquaculture Research Center. 88

Figure 3.2.2 – Overlying water ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), phosphate (HPO_4^{2-}), dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) concentrations (µM) in the fishpond (Pf, ♦), control pond (Pnf, ○) and inflowing water (Iw, ∗), throughout the trial. 93

Figure 3.2.3 – Ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-) and phosphate (HPO_4^{2-}) molecular diffusive fluxes (mean ± *standard deviation*, nmol cm^{-2} d^{-1}) in the control pond (Pnf, ***a***) and in the fishpond (Pf, ***b***), throughout the trial. 96

Figure 3.2.4 – Dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) molecular diffusive fluxes (mean ± *standard deviation*, nmol cm^{-2} d^{-1}) in the control pond (Pnf, ***a***) and fishpond (Pf, ***b***), throughout the trial. 97

Figure 3.2.5 – Average daily contributions (µM d^{-1}) of different nutrient sources (benthic fluxes, inflowing water, fish excretion and dissolution of uneaten food) and sinks (phytoplankton uptake, microphytobenthos uptake and outflowing water), for dissolved nitrogen (N) and phosphorus (P) availability in pond water, Pf (*top*) and Pnf (*bottom*), during the first//second year of the trial. 105

Figure 4.1 – Conceptual model for nitrogen (N) and phosphorus (P) transformations in the water column and sediments of fish earth ponds. POP: particulate organic phosphorus; DOP: dissolved organic phosphorus; HPO_4^{2-}: phosphate; Pads: inorganic phosphorus adsorbed to

sediments; PON: particulate organic nitrogen; DON: dissolved organic nitrogen; NH_4^+: ammonium; NO_x: oxidized nitrogen forms. Adapted from Worsfold et al. (2008). 113

Figure 4.2 – Average daily water temperature (ºC) and wind speed (m s^{-1}) in an earth pond without fish, from June 2003 to March 2005. 117

Figure 4.3 – Predicted (line) and observed (diamonds) values of total particulate matter (TPM, mg L^{-1}) and particulate organic matter (POM, mg L^{-1}) in the water column of an earth pond without fish. 119

Figure 4.4 – Predicted (blue line) and observed (red diamonds and line) ammonium ($NH_4^+{}_w$), oxidized nitrogen forms (NO_{xw}), dissolved organic nitrogen (DON_w), phosphate ($HPO_4^{2-}{}_w$), dissolved organic phosphorus (DOP_w) and dissolved oxygen (DO_w) concentrations in the water column of an earth pond without fish. 120

Figure 4.5 – Predicted (line) and observed ± standard deviation (diamonds) ammonium ($NH_4^+{}_s$), oxidized nitrogen forms (NO_{xs}) and dissolved organic nitrogen (DON_s) porewater concentrations in an earth pond without fish. 122

Figure 4.6 – Predicted (line) and observed ± standard deviation (diamonds) phosphate ($HPO_4^{2-}{}_s$) and dissolved organic phosphorus (DOP_s) porewater concentrations in an earth pond without fish. 123

Figure 4.7 – Predicted (line) and observed ± standard deviation (diamonds) organic carbon (POC_s), nitrogen (PON_s) and phosphorus (POP_s) content in the sediments of an earth pond without fish. 125

Figure 4.8 – Predicted dissolved oxygen porewater (DO_s) concentrations in an earth pond without fish. 135

Figure 4.9 – Average daily nitrogen fluxes (μM N d^{-1}), for the 2-year simulation period, in an earth pond without fish. 137

Figure 4.10 – Average daily phosphorus fluxes (μM P d^{-1}), for the 2-year simulation period, in an earth pond without fish. 138

Figure 5.1 – Energy flow through an organism in a standard DEB model. Arrows indicate rates and squares are state variables. 169

Figure 5.2 – Ingestion rate (J d^{-1}) - volumetric length (cm^2) relationships for white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) at the reference temperature (20ºC = 293.15 K). The slope of the linear regression between these two variables corresponds to maximum surface area-specific ingestion rate {$\dot{P}_{Xm}$} for these species. 177

Figure 5.3 – Structural volume ($V^{1/3}$) – total length (L, cm) relationships for juveniles (green dots) and adult (red dots) white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) individuals. The slope of the linear regression between these two variables corresponds to the shape coefficient (δ_m) at the different life stages. 180

Figure 5.4 – Arrhenius plot (ln rate *versus* T^{-1}) for the oxygen consumption data of white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*). The slope of the linear regression corresponds to the Arrhenius temperature (T_A). 181

Figure 5.5 – Daily time series of food input (kg d^{-1}) and water temperature (ºC) registered during the white seabream (top graph) production cycle (from 18th June 2006 to 31th March 2005) and the gilthead seabream (bottom graph) production cycle (2nd May 2005 to 22nd March 2007). 182

Figure 5.6 – Aquaculture Research Center. 183

Figure 5.7 – Predicted (blue diamonds) total length and wet weight values for 10000 white seabreams (*Diplodus sargus*) from Simulations1. The green line and whiskers represent the average predicted values ± 1 sd while red dots and whiskers correspond to average measured values ± sd. 186

Figure 5.8 – Predicted (blue diamonds) total length and wet weight values for 10000 gilthead seabreams (*Sparus aurata*) from Simulations1. The green line and whiskers represent the average predicted values ± 1 sd while red dots and whiskers correspond to average measured values ± sd....187

Figure 5.9 – Predicted (blue diamonds) total length and wet weight values for 10000 white seabreams (*Diplodus sargus*) from Simulations2. The green line and whiskers represent the average predicted values ± 1 sd while red dots and whiskers correspond to average measured values ± sd....188

Figure 5.10 – Predicted (blue diamonds) total length and wet weight values for 10000 gilthead seabreams (*Sparus aurata*) from Simulations2. The green line and whiskers represent the average predicted values ± 1 sd while red dots and whiskers correspond to average measured values ± sd....189

Figure 5.11 – Principal Component Analysis (PCA) showing the contribution of DEB parameters to white seabream (top graph) and gilthead seabream (bottom graph) growth in Simulations1....190

Figure 5.12 – Principal Component Analysis (PCA) showing the contribution of DEB parameters to white seabream (top graph) and gilthead seabream (bottom graph) growth in Simulations2....191

Figure 5.13 – Average ± sd measured (diamonds) and predicted values (lines) of wet weight (g) and total length (cm) for white seabream (top graph) and gilthead seabream (bottom graph) in Simulations1....193

Figure 5.14 – Average ± sd measured (diamonds) and predicted values (lines) of wet weight (g) and total length (cm) for white seabream (top graph) and gilthead seabream (bottom graph) in Simulations2....194

Figure 6.1 – Coupling of the biogeochemical and DEB models. POM_w – Particulate Organic Matter in the water column; PON_s and POP_s – Particulate Organic Nitrogen and Phosphorus

in sediments; DON_s and DOP_s – Dissolved Organic Nitrogen and Phosphorus in porewater; DIN_s and DIP_s – Dissolved Inorganic Nitrogen and Phosphorus in porewater; DO_w and DO_s – Dissolved Oxygen in the water column and in porewater; TDN_w and TDP_w – Total Dissolved Nitrogen and Phosphorus in the water column. 210

Figure 6.2 – Predicted (line) and observed (diamonds) values of total particulate matter (TPM_w, mg L^{-1}) and particulate organic matter (POM_w, mg L^{-1}) in the water column of a white seabream production pond. 222

Figure 6.3 – Predicted (blue line) and observed (red diamonds and line) ammonium ($NH_4^+{}_w$), oxidized nitrogen forms (NO_{xw}), dissolved organic nitrogen (DON_w), phosphate ($HPO_4^{2-}{}_w$), dissolved organic phosphorus (DOP_w) and dissolved oxygen (DO_w) concentrations in the water column of a white seabream production pond. 223

Figure 6.4 – Predicted (line) and observed ± standard deviation (diamonds) ammonium ($NH_4^+{}_s$), oxidized nitrogen forms (NO_{xs}) and dissolved organic nitrogen (DON_s) porewater concentrations in a white seabream production pond. 224

Figure 6.5 – Predicted (line) and observed ± standard deviation (diamonds) phosphate ($HPO_4^{2-}{}_s$) and dissolved organic phosphorus (DOP_s) porewater concentrations in a white seabream production pond. 225

Figure 6.6 – Predicted (line) and observed ± standard deviation (diamonds) organic carbon (POC_s), nitrogen (PON_s) and phosphorus (POP_s) content in the sediments of a white seabream production pond. 226

Figure 6.7 – Average ± standard deviation measured and predicted values of white seabream wet weight (respectively, diamonds and solid line) and total length (respectively, circles and dashed line) over a production cycle.. 228

Figure 6.8 – Final fish biomass (kg) in a white seabream production pond, for the standard simulation and for each of the management scenarios analysed in the present work......... 236

Figure 6.9 – Predicted dissolved oxygen (DO_s) porewater concentrations in a white seabream production pond...243

Figure 6.10 – Average nitrogen fluxes (µM d^{-1}) in a semi-intensive white seabream production pond. Abbreviattions: PON_{iw} - particulate organic nitrogen in inflowing water; TDN_{iw} - total dissolved nitrogen in inflowing water; PON_{ow} - particulate organic nitrogen in outflowing water; TDN_{ow} - total dissolved nitrogen in outflowing water; Min. - mineralization; Nitrif. – nitrification and Denit. - denitrification. 244

Figure 6.11 – Average phosphorus fluxes (µM d^{-1}) for a semi-intensive white seabream production pond. Abbreviattions: POP_{iw} - particulate organic phosphorus in inflowing water; TDP_{iw} - total dissolved phosphorus in inflowing water; POP_{ow} - particulate organic phosphorus in outflowing water; TDP_{ow} - total dissolved phosphorus in outflowing water; Min. - mineralization; Ads. - adsorption and Desorp. - desorption. 245

List of Tables

Table 2.1 – Amounts (kg per ton of product) of Total Suspended Solids (TSS), Biochemical Oxygen Demand (BOD), Particulate Organic Matter (POM), Nitrogen (N) and Phosphorus (P) discharged from different aquaculture units. 23

Table 2.2 – Top ten aquaculture producer countries in 2006 and its respective aquaculture revenues. 36

Table 2.3 – Drivers, Pressures, States, Impacts and Responses for a hypothetical aquaculture development. 62

Table 3.1.1 - Deposition rates of particulate organic carbon (POC), particulate organic nitrogen (PON) and particulate organic phosphorus (POP) in the fish production pond (Pf) and in the non-fish production pond (Pnf), during the experiment. 76

Table 3.1.2 – Total length (cm), body weight (g) and condition factor (k) of white seabream throughout the production cycle. 80

Table 3.2.1 – Porewater ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), phosphate (HPO_4^{2-}), dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) concentrations (mean ± *standard deviation*) in the upper sediment layers (2 cm) of the fishpond (Pf) and control pond (Pnf), during the experiment. 94

Table 3.2.2 – Ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), phosphate (HPO_4^{2-}), dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) biologically-mediated fluxes (F_{bio}) and diffusive fluxes (Js) for the fishpond (Pf) and control pond (Pnf) sediments (mean ± *standard deviation*) in incubation chambers at two different temperatures (T). 97

Table 3.2.3 – Brief review of diffusive fluxes (nmol cm^{-2} d^{-1}) in different types of ecosystems. 101

Table 3.2.4 – Description of the main sources and sinks of dissolved N and P in the fishpond: source data (annual averages) and brief explanation of the calculations. Values for the control pond are given between brackets. 104

Table 4.1 – EcoDynamo objects implemented for earth ponds and respective state variables.. 116

Table 4.2 – Results of model II regressions for water column variables. 121

Table 4.3 – Results of model II regressions for porewater variables. 123

Table 4.4 – Results of model II regressions for sediment variables. 124

Table 4.5 – Sensitivity analysis for water column state variables. Results are expressed as the % of variation relative to the average value in the standard simulation. 126

Table 4.6 – Sensitivity analysis for porewater state variables. Results are expressed as the % of variation relative to the average value in the standard simulation. 128

Table 4.7 – Sensitivity analysis for sediment state variables. Results are expressed as the % of variation relative to the average value in the standard simulation. 130

Table 5.1 – DEB parameters for white seabream. 175

Table 5.2 – DEB parameters for gilthead seabream. 176

Table 5.3 – White seabream oxygen consumption rates (mg O_2 g^{-1} d-1, mean ± sd) at different temperatures (17 to 25 ºC) and corresponding volume-specific costs for maintenance ($[\dot{p}_M]$, J cm^{-3} d^{-1}). 179

Table 5.4 – Resting oxygen consumption rates (mg O_2 g^{-1} d^{-1}, mean ± sd) for gilthead seabream (Guinea and Fernandéz 1997) and corresponding volume-specific costs for maintenance ($[\dot{p}_M]$, J cm^{-3} d^{-1}). 179

Table 5.5 – Calibrated DEB parameters for white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) in Simulations1. Sensitivity analysis results as % of variation in white seabream mean total length and wet weight after changing each parameter by ±25%. 195

Table 5.6 – Calibrated DEB parameters for white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) in Simulations2. Sensitivity analysis results as % of variation in white seabream mean total length and wet weight after changing each parameter by ±25%. 196

Table 6.1 – Diet composition, relatively to food dry weight. 212

Table 6.2 – Juvenile and adult white seabream (*Diplodus sargus*) elemental composition, relatively to fish dry weight. 213

Table 6.3 – Scenarios and management options for semi-intensive pond aquaculture. Current rearing conditions (standard simulation): Stocking density = 1.2 kg m^{-3}; Water exchange rate = 25 to 100 m^3 h^{-1}; Feeding rate = 0.8 to 1.2% of fish body weight; Phosphorus content in feeds = 1.2%; Feed absorption efficiency = 72%; Pellets sinking velocity = 0.035 m s^{-1}. 217

Table 6.4 – Results of model II regressions for water column variables. 221

Table 6.5 – Results of model II regressions for porewater variables. 225

Table 6.6 – Results of model II regressions for sediment variables. 227

Table 6.7 – Results of model II regressions for fish growth variables. 228

Table 6.8 – Water quality assessment for the standard simulation and for each of the management scenarios considered in the present work. Quality status: BLUE – High; GREEN – Good; YELLOW – Moderate; ORANGE – Poor; and RED – Bad. Adapted from: Austoni et al. (2004). 230

Table 6.9 – Sediment quality assessment (Austoni et al. 2004) for the standard simulation and for each of the management scenarios considered in the present work. Quality status: BLUE – High; GREEN – Good; YELLOW – Moderate; ORANGE – Poor; and RED – Bad. 234

Table 6.10 – Total nitrogen (TN) and phosphorus (TP) discharges (kg) during a white seabream production cycle. Values in bold correspond to the best scenarios. 238

Table 6.11 – Analytical Hierarchical Approach for each scenario analysed in the present work, considering equal weights (A_1) and different weights (A_2 – higher weight for the production indicator) for the indicators. Values in bold correspond to the best scenarios. 239

Chapter 1

General Introduction

1 Aquaculture

1.1 Basic concepts

A simple definition for aquaculture is “the farming of aquatic species, either plants or animals, in all types of water environments (fresh, brackish and marine environments), including natural or manmade systems” (Pillay and Kutty 2005; FAO 2010; NOAA 2010). According to the degree of artificiality, aquaculture systems are commonly referred as extensive, semi-intensive and intensive systems (Funge-Smith and Philips 2001). As the different levels of intensification represent a continuum, no specific definition can be given for each system (Funge-Smith and Philips 2001). Extensive systems deeply resemble natural systems, whereas intensive systems are highly artificial (Funge-Smith and Philips 2001). Aquaculture units may also cultivate one or more species, being classified respectively as, monoculture or polyculture systems. More recently, traditional polyculture systems have evolved into integrated systems, which may be differentiated into integrated multi-trophic aquaculture systems (IMTA), also known as ‘partioned aquaculture’ or ‘aquaponics’, that combine species from different trophic levels (Krom et al. 1995; Buschmann et al. 1996; Brummet 1999; Alongi et al. 2000; Choo 2001; Funge-Smith and Philips 2001; Neori et al. 2004; Primavera 2006; Abreu et al. 2009; Troell et al. 2009; Bosma and Verdegem 2011), and into systems combining aquaculture with other productive activity, such as agriculture or livestock (Funge-Smith and Philips 2001; Jamu and Piedrahita 2001; Neori et al. 2004; World Bank 2006; Bosma and Verdegem 2011).

1.2 Global Production

As a consequence of the continuous decrease in marine biodiversity, caused by anthropogenic activities such as environmental pollution, habitat destruction, fishing, and by global climate change (Lotze et al. 2006; Halpern et al. 2008; FAO 2010; Merino et al. 2010), a collapse of currently fished taxa is expected to happen by 2048 (Worm et al. 2006), if trends are not reversed. In this context of declining marine resources, aquaculture can have a major role in providing alternative protein sources to meet the increasing worldwide demand for seafood and to ease fishing pressure on marine stocks (FAO 2010; Bosma and Verdegem 2011; Grigorakis and Rigos 2011).

According to FAO estimates (FAO 2011a), aquaculture and capture fisheries supplied the world with 144 million tonnes of animal products, in 2009 (Figure 1.1), from which 118 million tonnes (82%) were used for human consumption (Figure 1.1). Assuming that most of aquaculture production (55 million tonnes in 2009) is directly used as human food, one may estimate that aquaculture provides almost half (47%) of the aquatic animal products that are eaten today. If trends from the last decade are maintained, i.e. aquaculture growing at ≈6% per year and fisheries decreasing ≈1% per year (Figure 1.1), the former industry will overcome fisheries production in a decade. The majority of cultivated animal species belong to the lower end of the food chain, e.g. shellfish, herbivorous and omnivorous fish (Figure 1.2). However, production of species higher in the food chain, such as shrimp, salmon, and marine finfish, is now growing (Figure 1.2) in response to the increasing market demand in developed countries (FAO 2007, 2010). Although animal species dominate aquaculture production (76%), aquatic plants have an important contribution (24%) to the global industry (Figure 1.2).

The commercialisation of seafood products, is estimated to have yielded 105.3 thousands of millions of US$, in 2009, and an additional 4.8 thousands of millions of US$ were generated from aquatic plants trade (FAO 2011a). Nevertheless, the revenues from the entire sector are expected to be significantly higher, because the value of aquaculture hatchery and nursery production and that of the breeding of ornamental is not usually included in the estimations (FAO 2010).

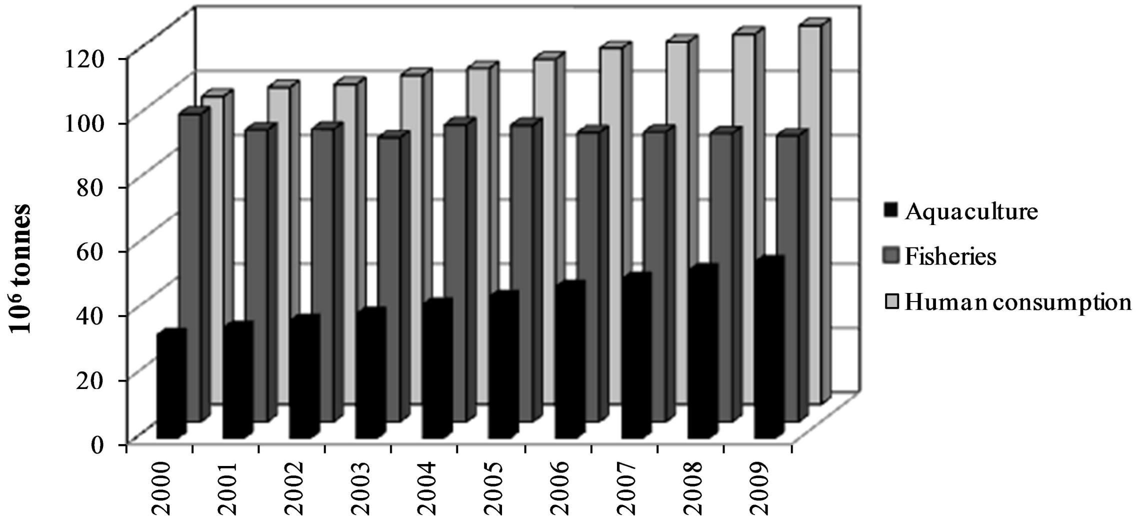


Figure 1.1 – Total aquaculture (except aquatic plants) and capture fisheries production versus human consumption, from 2000 to 2009. Source: FAO (2011a).

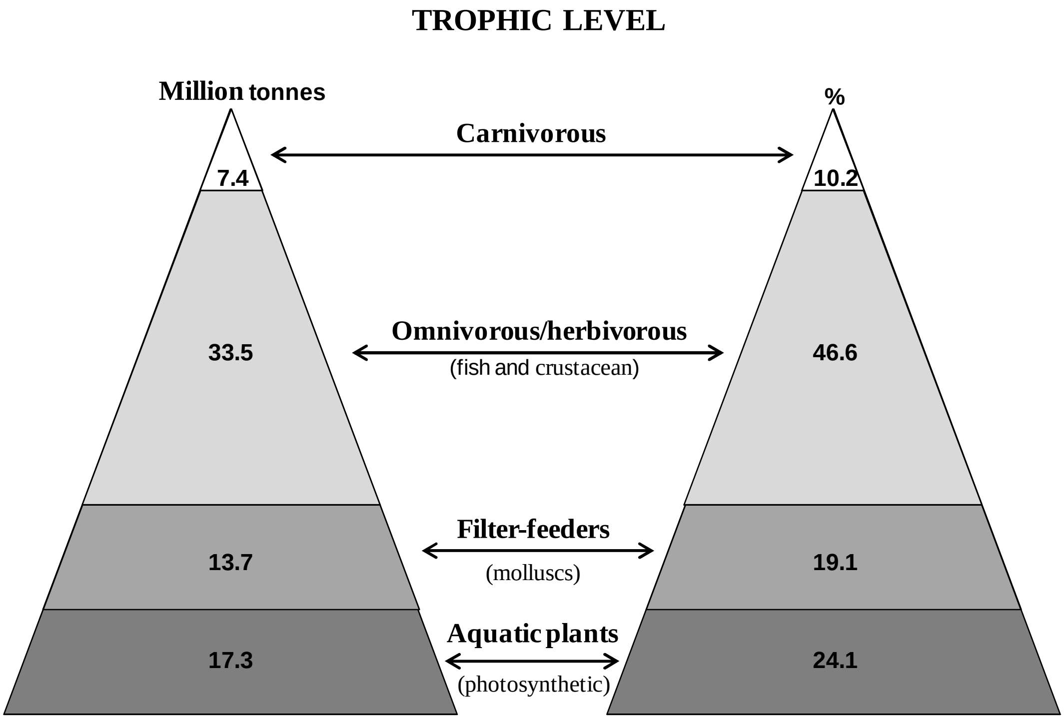


Figure 1.2 – Global aquaculture production by trophic level, in 2009 (Adapted from Primavera (2006)).

1.3 Environmental impacts

Although responsible aquaculture can provide significant socio-economic benefits, such as: development of the economies of low profit-food deficit countries, increase of rural development, alleviation of poverty and hunger, and promotion of gender opportunities (Black 2001; World Bank 2006; FAO 2010), uncontrolled and irresponsible aquaculture operations can cause a wide range of negative impacts. Some of these impacts are related to: i) organic and nutrient pollution (Wu 1995; Kelly et al. 1996; Deb 1998; Naylor et al. 2000; Tovar et al. 2000a, b; Pearson and Black 2001; Páez-Osuna 2001a, b; Read and Fernandes 2003; Gyllenhammar and Håkanson 2005; Bosma and Verdegem 2011; Grigorakis and Rigos 2011); ii) chemical contamination (Choo 2001; Read and Fernandes 2003; Islam et al. 2004; Grigorakis and Rigos 2011); iii) spread of parasites and diseases (Naylor et al. 2000; Nash 2005); iv) habitat destruction and modification (Wu 1995; Deb 1998; Naylor et al. 2000; Black 2001; Páez-Osuna 2001b; Ruiz et al. 2001; Pérez et al. 2008); v) introduction of exotic species and new genetic varieties (Black 2001; Naylor et al. 2005; Grigorakis and Rigos 2011) and vi) depletion of wild stocks as feeds or seed to aquaculture operations (Deb 1998; Choo 2001; Kaiser 2001; Páez-Osuna 2001b; Grigorakis and Rigos 2011). Aside from damages to aquatic environments, social conflicts and economic breakdowns may also occur when the ecosystem functioning is radically altered and the resources that support other human activities are affected (Boyd and Clay 1998; FAO 2007; Allsopp et al. 2008; Grigorakis and Rigos 2011).

1.4 Semi-intensive pond aquaculture

1.4.1 Advantages and disadvantages

Aquaculture is carried out in a wide variety of systems, and ponds are the most common production systems on a worldwide basis (Culberson and Piedrahita 1996). Due to their manageability in terms of waste, nutrient recycling and feed conversion, land-based aquaculture systems are more promising for the sustainability of marine aquaculture than open-water systems (Neori et al. 2004). Semi-intensive pond systems traditionally used in Mediterranean countries, have been particularly recognized as an environmentally-friendly option (Boyd and Tucker 1998; SEACASE 2009). The lower degree of artificiality (i.e. lower stocking densities, water use and feed inputs) of these systems comparatively to intensive systems, substantially reduces their ecological footprint (Troell 1997; Boyd and Tucker 1998; Kautsky et al. 2000; Banas et al. 2008; Bosma and Verdegem 2011), i.e. the quantity of environmental goods and services consumed (e.g. food, space, water) in the generation of goods and processing of associated wastes (e.g. feces, ammonia excretion) (Rees and Wackernagel 1994). Aside from ecological benefits, semi-intensive aquaculture systems promote the development of rural and coastal areas and create employment opportunities (WorldBank 2006; FAO 2007; SEACASE 2009; Bosma and Verdegem 2011; Grigorakis and Rigos 2011).

Despite its environmental and socio-economical advantages, semi-intensive fish farms are usually characterized by a low profitability, as a consequence of high production costs (high labour and land costs) and low productivity (SEACASE 2009; Bosma and Verdegem 2011). A low profitability associated with the increasing market competition with low-price products from intensive aquaculture, strongly limits the economic viability of this activity (SEACASE 2009). Thus, to make this environmentally-friendly aquaculture more competitive it is necessary to increase revenues.

1.4.2 Economical and environmental sustainability

One of the strategies that have been proposed to increase the competitiveness of semi-intensive aquaculture is the association of its products with certification or ecolabelling schemes that trace the origin, quality, safety, and the environmental conditions prevailing during the production cycle (Boyd 2003; SEACASE 2009; Allsopp et al. 2008; FAO 2010; Bosma and Verdegem 2011). The increasing consumer awareness on animal health/welfare and environmental/biodiversity protection (FAO 2007; Verbeke et al. 2007; Allsopp et al. 2008; Bosma and Verdegem 2011), will most likely increase the demand for semi-intensive aquaculture products (SEACASE 2009; Bosma and Verdegem 2011), and the sector would largely benefit from certification.

Other strategy that can increase both the profitability and competitiveness of semi-intensive fish farms is species diversification (SEACASE 2009). Autochthones species with high market value and high flesh quality, like the sole (*Solea senegalensis*), the sharpsnout seabream (*Diplodus puntazzo*) and the white seabream (*Diplodus sargus*), are already being tested to be cultivated in Mediterranean semi-intensive earth pond systems (Sá et al. 2006, 2007; SEACASE 2009; Ferreira et al. 2010; Grigorakis and Rigos 2011), since the traditionally-cultivated species, e.g. gilthead seabream (*Sparus aurata*) and seabass (*Dicentrarchus labrax*), can no longer compete with prices of intensive aquaculture products (SEACASE 2009; Barazi-Yeroulanos 2010).

Some authors have also suggested an increase in stocking densities as a way to increase the productivity of semi-intensive fish farms (SEACASE 2009), however higher biomasses often lead to a deterioration of the pond environment (Lin and Yi 2003; Viadero Jr. 2005) and an increase in waste emissions (Bergheim and Brinker 2003; Lin and Yi 2003; Viadero Jr. 2005). To make this a viable option, it must be ensured that the assimilative capacity of the fishpond (Boyd 2003) as well as of the receiving environment, are not exceeded (Wu 1995; Naylor et al. 2000; Gyllenhammar and Håkanson 2005; Gatlin et al. 2007). In fact, for semi-intensive fish farming to be sustainable there should be an improvement in its environmental efficiency as well as on the economic efficiency (Boyd 2003; SEACASE 2009; Chávez-Crooker and Obreque-Contreras 2010).

As one of the major environmental constraints of semi-intensive fish farming is related to organic and nutrient loadings to coastal waters (Boyd and Tucker 1998; Boyd 2003; Jegatheesan et al. 2007), mainly resulting from feed waste products; the sustainability of these systems, as for aquaculture in general, strongly depends on feed management (Black 2001; Choo 2001; World Bank 2006; Bosma and Verdegem 2011; Grigorakis and Rigos 2011). Optimization of feeding rates (i.e. the amount of daily feed supplied as a function of fish biomass) is essential to avoid situations of over or under-feeding that may compromise the viability of farming units, and to reduce effluent discharges (Boyd 2003). Improvements in feed pellet technology, namely the development of species-specific feeds and the increase in pellet stability or reduction of its sinking rates, may improve feed conversion rates (FCRs) for cultivated species (i.e. the amount, in kg, of fish biomass produced per kg of feed supplied), by maximizing the amount of feed ingested and minimising organic/nutrient loadings (Choo 2001; World Bank 2006). Another feed-related environmentally friendly option is the replacement of fish meal and oils of formulated feeds by vegetable protein sources such as soybean, corn meal and rice bran (Black 2001; Kaushik et al. 2004; World Bank 2006; Allsopp et al. 2008; Dias 2009; FAO 2010). For some herbivorous and omnivorous fish it has been possible to completely replace animal protein by vegetable one without impacts on fish growth and yields (Tacon et al. 2006). However, for carnivorous species, ecofeeds are still to be perfect according to fish requirements (Tacon et al. 2006; Drakeford and Pascoe 2008; Dias et al. 2009; FAO 2010). Aside from reducing aquaculture dependence on fisheries resources (Black 2001; Kaushik et al. 2004; World Bank 2006), ecofeeds of vegetable-origin may reduce nutrient loadings (Ferreira et al. 2010), contributing for the minimization of the ecological footprint of semi-intensive fish farming.

Nevertheless, in order for this type of ecofeeds to be viable they have to be produced through sustainable agriculture (Allsopp et al. 2008; FAO 2010).

Biological methods, such as polyculture (SEACASE 2009; Bosma and Verdegem 2011) or integrated multi-trophic aquaculture (IMTA) systems (Neori et al. 2004; Primavera 2006; World Bank 2006; FAO 2007; Chávez-Crooker and Obreque-Contreras 2010; Nobre et al. 2010), may also be a sustainable option for reducing the environmental impacts of semi-intensive fish farming while increasing its economic efficiency (Brummet 1999; Whitmarsh et al. 2006; Allsopp et al. 2008; Bunting and Shpigel 2009; Troell et al. 2009; Nobre et al. 2010). IMTA systems are particularly environmentally benign because organic waste products from fed fish are recycled by the extractive species cultivated in the same units. Seaweed and shellfish based-integrated systems are particularly promising due to the high market demand for these products (Neori et al. 2000; Neori et al. 2004; Zhou et al. 2006; Abreu et al. 2009).

In semi-intensive fishponds, sustainability is also dependent on an efficient water management. If, by on one hand, water exchange rates define pond water quality, which is crucial for fish welfare and growth, on the other it determines the amount of effluents discharged into the environment (Hopkins et al. 1993; Brambilla et al. 2007; d'Orbcastel et al. 2008). The reduction of water exchange rates is often referred as a best management practice (BMP) for semi-intensive farms, because it minimizes the risk for sudden changes in water quality parameters and minimizes the adverse effects of effluents discharge (Páez-Osuna 2001a, b; Boyd 2003; Primavera 2006). However, the quality of pond and outlet waters depends on the quality of inlet water (Bosma and Verdegem 2011). For instance, for aquaculture units located on coastal eutrophic systems, water exchange can increase the risk for eutrophication within fishponds

(Boyd and Tucker 1998; Deb 1998; Páez Osuna 2001b; Boyd 2003). On the other hand, in oligotrophic systems, this risk decreases due to the lower nutrient concentrations in source water (Maldonado et al. 2005). In fact, in the latter systems, water exchange may be an effective strategy for reducing the concentrations of potentially toxic compounds like ammonia and nitrites, allowing substantially higher stocking densities (Burford and Lorenzen 2004).

2 Physical, chemical and biological processes in fishponds

The ecology of semi-intensive fishponds consists of a variety of interdependent physical, chemical and biological processes (Boyd and Tucker 1998). The knowledge of these processes is of outmost importance for defining farming strategies (protocols, monitoring systems and techniques) that optimize fish production, i.e. maximize production while minimizing environmental impacts (Boyd and Tucker 1998; Hargreaves 1998; Hargreaves and Tucker 2003). Although processes occurring in semi-intensive fishponds are similar to other aquatic marine systems (Boyd and Tucker 1998; Hargreaves 1998; Alongi et al. 1999; Burford et al. 2003; Boyd et al. 2006; Nhan et al. 2006; Serpa et al. 2007a, b; Rodriguez-Gallego et al. 2008), the shallowness of fishponds and the high inputs of allochtonous material, induce changes in natural processes (Culberson and Piedrahita 1996; Rise and Roos 1997; Boyd and Tucker 1998; Hargreaves 1998; Jamu and Piedrahita 2001; Mischke and Zimba 2004; Torres-Beristain et al. 2006; Serpa et al. 2007b; Yokoyama et al. 2009; Bosma and Verdegem 2011; Joyni et al. 2011). The spatial and temporal variability of the physical, chemical and biological processes occurring in fishponds strongly depends on i) farming conditions, such as stocking densities (Rowland et al. 1995; Tovar et al. 2000a; Ingram 2008; van de Nieuwegiessen et al. 2009) and feeding practices (Tovar et al. 2000a; Paspatis et al. 2000; Başçınar et al. 2007; Booth et al. 2008; Piedcausa et al. 2010; Bosma and Verdegem 2011), ii) cultivated species biology and feeding behaviour (Chakrabarty and Das 2007; Piedcausa et al. 2010; Bosma and Verdegem 2011) and; iii) site-specific environmental parameters, e.g. climate and sediment characteristics (Culberson and Piedrahita 1996; Hargreaves 1998; Serpa et al. 2007b; Bosma and Verdegem 2011). Therefore, the investigation of fishpond dynamics in a specific aquaculture system is crucial for the optimization of semi-intensive fish farming (Bosma and Verdegem 2011).

3 Ecological modelling

Even though an adaptive approach, i.e. the monitoring of variables and processes over time, should be followed to avoid the deterioration of pond environment and to minimize the impacts of effluent discharges on receiving waters (Crawford 2003; Gibbs 2009), a modelling approach has been widely used in pond management (Piedrahita et al., 1984; Culberson and Piedrahita 1996; Montoya et al. 2000; Lefebvre et al. 2001; Li and Yakupitiyage 2003; Jiménez-Montealegre et al. 2002a; Burford and Lorenzen 2004), since models are able to integrate the dynamics of physical, chemical and biological processes occurring in these systems (Piedrahita et al. 1984; Silvert and Cromey 2001). By being able to predict pond dynamics, models have been used for designing or improving monitoring strategies (Crawford 2003) as well as to simulate the effect of different management scenarios (e.g. stocking densities, water exchange and feeding rates) on the pond environment and on coastal systems (Piedrahita 1991; Burford and Lorenzen 2004). An ecological model is also a valuable tool for preventing episodic events of fish mortality by improving our understanding of the complex feedbacks between cultivated species and environmental variables, such as temperature (Via et al. 1998; Das et al. 2005; Del Toro-Silva et al. 2008; Dalvi et al. 2009), ammonia concentrations (Biswas et al. 2006; Remen et al. 2008) and oxygen availability (Del Toro-Silva et al. 2008; Remen et al. 2008; Bosma and Verdegem 2011). Moreover, models can be used to predict the impact of fish farming in the surrounding environment (Silvert and Cromey 2001; Rodriguez-Gallego et al. 2008; Piedcausa et al. 2010; Tsagaraki et al. 2010).

Despite its numerous advantages, models have limitations (Gibbs 2009) that restrict their ability to reproduce the overall variability of real systems. For this reason, even in the presence of validated models for fishpond systems an adaptive approach should be followed (Gibbs 2009). The confrontation of model results and observations allows model improvements over time, as more knowledge is accumulated about the ecosystem under study (Serpa and Duarte 2008). In addition, uncertainties associated with model parameters and results may help define sampling strategies and experiments to fill the gaps (Serpa and Duarte 2008).

Ecological models developed for fishpond systems may include different components, like transport, thermodynamic, biogeochemical and biological sub-models. Transport sub-models are used for simulating the interactions between pond hydrodynamics and sediment transport/resuspension (Peterson et al. 2000), whereas thermodynamic sub-models are used for temperature calculations (Culberson and Piedrahita 1996; Lamoreaux et al. 2005). The biogeochemical sub-model reproduces the cycles of elements that are most likely to affect biological activity, such as nitrogen (Kochba et al. 1994; Hargreaves 1997; Lefebvre et al. 2001; Jiménez-Montealegre et al. 2002a; Burford and Lorenzen 2004), phosphorus (Montoya et al. 2000; Lefebvre et al. 2001) and oxygen (Meyer and Brune 1982; Culberson and Piedrahita 1996), and often include a pelagic and a benthic compartment to reproduce water-sediment interactions. The biological sub-model simulates the growth (Cuenco et al. 1985a; Yi 1998; Hernández et al. 2003; Zhou et al. 2005; Libralato and Solidoro 2008; Moss et al. 2009; Pecquerie et al. 2011), as well as the production and biological interactions between individuals (Cuenco et al. 1985b, c) or cultivated species (Gazi et al. 2009). This module also predicts the effects of environmental (e.g. temperature), physiological (e.g. assimilation and excretion rates) and husbandry factors (e.g. stocking rates, feeding rates) on fish performance (Cuenco et al. 1985b, c), allowing fish farmers to adjust management strategies that maximize the growth rates of cultivated fish (Alunno-Bruscia et al. 2009; van der Veer et al. 2009). Biological sub-models are usually coupled to a biogeochemical sub-model to reproduce the feedbacks between cultivated species and the environment (Piedrahita et al. 1984; Jiménez-Montealegre et al. 2002a; Li and Yakupitiyage 2003; Piedcausa et al. 2010).

So far, fishpond models have been specifically used for researching nitrogen (Kochba et al. 1994; Hargreaves 1997; Jiménez-Montealegre et al. 2002a; Burford and Lorenzen 2004) phosphorus (Montoya et al. 2000) or oxygen dynamics (Meyer and Brune 1982; Culberson and Piedrahita 1996), while less effort has been made to develop more complete predictive models of pond dynamics (Piedrahita et al. 1984; Lefebvre et al. 2001; Li and Yakupitiyage 2003; Mukherjee et al. 2008).

4 Thesis aims and structure

The low profitability and competitiveness of semi-intensive fishpond aquaculture in Mediterranean countries strongly compromises the economic viability of production systems. The key for the sustainability of these systems seems to rely on the development of new methodologies and approaches that improve their economic as well as environmental efficiency. The challenge is to find the best solutions for each system, because there are many variables involved. For example, as these systems strongly rely on natural resources, the quality of source water or the assimilative capacity of receiving waters may prevent the application of a specific approach to one fish farm but not to another.

One of the main objectives of the present work was to study the physical, chemical and biological processes in semi-intensive fishponds, in order to understand the functioning of these systems. The other main objective was to develop an ecological model to be used as a tool for managing semi-intensive systems, to improve their economic and environmental performance. In order to accomplish the main objectives of this work, specific objectives were defined:

i) Monitoring water quality, sediment quality and species growth over a production cycle in semi-intensive ponds.

ii) Develop a comprehensive model to reproduce the biogeochemical cycles that are more likely to affect biological activity and cause negative environmental impacts, namely those of nitrogen, phosphorus and oxygen.

iii) Develop a biological model to simulate the growth of finfish in a dynamic environment, using the Dynamic Energy Budget (DEB) Theory.

iv) Couple the previous models to reproduce the interactions between cultivated species and the environment.

The fish species used both as an object of study and as a case study to implement and test the above mentioned model, was the white seabream (*Diplodus sargus*) - a new species in semi-intensive Mediterranean aquaculture (Cejas et al. 2004; Sá et al. 2006, 2007). As there was no other available information regarding the production of this species in earth ponds, the results obtained in the experimental work designed to study fishpond processes, together with literature data were used to implement the above mentioned models. After model calibration/validation, different management scenarios (e.g. stocking densities, water exchange rates, feeding rates) were tested for their effects on the pond environment, final fish production and nutrient discharges, in order to define the best management options for optimizing white seabream production in semi-intensive systems, i.e. to maximize production while minimizing environmental impacts.

This thesis is composed by 7 Chapters, which correspond to papers that have been published in peer-reviewed journals, or that are either submitted or in preparation. The content of each chapter is briefly described in the next paragraphs:

Chapter 1 – General Introduction

Chapter 2 – Impacts of Aquaculture and Mitigation Measures

In this chapter, an extensive review on the environmental impacts of aquaculture and on possible mitigation measures, approaches and tools to reduce or anticipate these impacts was carried out for a global perspective on how to manage aquaculture towards sustainability.

Chapter 3 – Physical, chemical and biological processes in semi-intensive fishponds

As adaptive approaches as well as modelling approaches are required for an efficient pond management, particularly when a new species is being cultivated, a case study was developed to study the physical, chemical and biological processes over a white seabream production cycle. The first part of this chapter deals with the effects of fish farming on bottom sediments, whereas processes affecting pond water are addressed in its second part. Data collected over this experimental work were used for model parameterization and calibration.

Chapter 4 – Modelling biogeochemical processes in semi-intensive fish earth ponds: model development and calibration

This chapter describes the development of a biogeochemical model – one of the specific objectives of the present thesis – integrating physical, chemical and biological processes in earth pond systems. This model was first implemented for a pond without fish and later coupled with the fish growth model (Chapters 5 and 6).

Chapter 5 – Modelling the growth of white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) in semi-intensive earth ponds using the Dynamic Energy Budget approach

In this chapter, a Dynamic Energy Budget (DEB) growth model was implemented for two species: the newly cultivated white seabream; and a traditionally cultivated species - the gilthead seabream. The model was used to investigate which factors (environmental, physiological or husbandry factors) are more likely to affect fish growth performance, and to explain the growth differences between these two species, towards a better understanding of the factors that should be manipulated to improve white seabream growth in culture, since low growth rates have been reported for this species (Cejas et al. 2004; Sá et al. 2006).

Chapter 6 – Mathematical modelling as a tool for managing semi-intensive production systems

This chapter describes the coupling of the biogeochemical and fish growth models developed in Chapters 4 and 5, respectively. Different management scenarios were tested with this model and then scored using the Analytic Hierarchical Process (AHP) methodology, to evaluate the best management options for white seabream production in semi-intensive systems.

Chapter 7 – General Conclusions and Future Perspectives

The major results and conclusions from the previous chapters are synthesized in this chapter. Ideas for future improvements and applications of the model developed in this thesis are presented.

Chapter 2

Impacts of Aquaculture and Mitigation Measures

Serpa D., Duarte P. (2008). Impacts of aquaculture and Mitigation measures. In: Russo, R. (eds.) Dynamic Biochemistry, Process Biotechnology and Molecular Biology 2 (Special issue 1), Global Science Books, 1 – 20.

Abstract

The role of aquaculture in world food production is increasing very fast, contributing with more than 40% for the total production of aquatic organisms. The general approach in modern aquaculture resembles much that of industrial agriculture and husbandry, with large energy subsidies and the usage of many chemicals in, predominantly, monoculture systems, with a large ecological footprint. Despite the large body of regulation available worldwide, there are important ecologic, economic and social impacts in many countries as a result of aquaculture. In some cases, the anticipation of these impacts by local populations represents a negative feedback for aquaculture development. In the present work, a review of those impacts is presented, followed by a discussion of the carrying capacity concept, then by presenting some approaches and methods that may help planning aquaculture developments including the Drives Pressures States Impacts Responses framework, modelling and Decision Support Systems. The analysis of a large number of works suggests that aquaculture management should be participated by local stakeholders and viewed within the context of other management approaches, such as Integrated Coastal Zone Management. This may allow for a better ecosystemic integration of aquaculture with other activities in line with Ecological Engineering concepts. Likely, there should be more investment in low-trophic level species to reduce aquaculture ecological footprint.

1 Introduction

The contribution of aquaculture to global production of aquatic organisms increased from ca. 32%, in 2000, to 42%, in 2006, according to the FAO Fishery Statistical Collections (FAO 2008). Following the same source, total aquaculture production increased over 18 times for the period 1997–2006, from 3,584,160 to 66,728,941 tonnes. Considering the mentioned growth and that intensive aquaculture developed over the last years (Muir 2005), it is expected that its environmental impacts have also increased. These impacts as well as the sustainability of aquaculture were discussed in previous works (e.g. GESAMP 2001; SECRU 2002; Read and Fernandes 2003; GESAMP 2008). However, there are still several issues to clarify about how to guarantee aquaculture sustainability, giving the vagueness of the concept and the lack of a general paradigm to handle this problem that, together with social awareness, creates some negative-feedbacks to aquaculture development in regions where perceived costs outweigh the perceived benefits by local stakeholders (Gibbs 2009). Therefore, the main purpose of this work is to synthesize information and concepts that may be useful in defining a paradigm towards aquaculture sustainability.

This work is structured as follows: Much of what is known about environmental impacts of aquaculture is synthesised in section 2. In the following section, the carrying capacity concept and its application to aquaculture are discussed, together with methods for its quantification. In a section about aquaculture and environmental management, some approaches and tools that may help to manage aquaculture towards sustainability are presented. Finally, some general conclusions are attempted.

Aquaculture industry seems to be following the same steps as agriculture: from traditional polyculture systems with low energy subsidies, to intensive monocultures with high energy inputs and biotechnological innovation. These high energy inputs are in the form of trophic energy, such as sun light and fish food, and auxiliary energy, such as renewable and non-renewable energy sources, to maintain production operations. In many countries, there is a strong investment in the production of carnivore species that comprise more than 30% of world aquaculture production in monetary terms (Primavera 2006), implying a relatively small efficiency in the conversion of primary production, though several steps of the food web, and a large ecological footprint. Furthermore, high intensive aquaculture systems require more pharmaceuticals and other chemicals to protect organisms from disease.

Perhaps one of the main problems in aquaculture, as well as in other human activities, is the apparent difficulty of people to think holistically, especially in industrialized societies, where compartmentalization is frequently equated with efficiency. Due to this limitation, local developments are planned without much consideration about integration with other activities, leading to the production of wastes that may represent an environmental problem when, if otherwise planned, could serve as raw materials for another activity. If some sort of integrated management is applied, such as Integrated Coastal Zone Management, with considerations about the spatial distribution of different activities, to guarantee proper access to resources by all stakeholders but, without consideration of material and energy fluxes related to different activities, there may still be sustainability problems. Therefore, traditional Chinese aquaculture-agriculture-husbandry-waste treatment systems may serve as a good example of empirical yet, holistic approaches, to be incorporated in modern developments but in tight interaction with scientific methods, well in line with the principles of Ecological Engineering (e.g. Yan and Ma, 1991; Mitsch 1997).

Aquaculture may be important to alleviate poverty by generating food, employment and wealth if a more equitable distribution of its benefits is assumed. Its environmental impacts should be assessed at a larger scale than the farm scale, due to the cumulative effects of several farm operations in the same area and their combination with other human activities (GESAMP 2001). However, if aquaculture development is planned in isolation from other activities, if it implies changes in resource ownership, preventing local people from having access to resources, and it leads to concentration of wealth in a few people, its environment, economic and social effects are unsustainable.

2 Aquaculture Environmental Impacts

Aquaculture units can generate considerable amounts of wastes/effluents containing a variety of substances such as, particulate material (mainly resulting from uneaten feed and fecal material), dissolved metabolites (from excretion via gills and kidneys), and various forms of chemicals (e.g. therapeutants, fertilizers, heavy metals), with undesirable environmental consequences (Wu 1995; Kelly et al. 1996; Deb 1998; Tovar et al. 2000a, 2000b; Pearson and Black 2001; Páez-Osuna 2001a, 2001b; Read and Fernandes 2003). The environmental impact resulting from particulate and dissolved organic and inorganic material (Table 2.1) is particularly important because these compounds are directly discharged into the environment affecting both the water column and the sediment compartment (Dalsgaard and Krause-Jensen 2006; Holmer et al. 2007). The magnitude of these impacts depends mainly on farm location, species, culture type, stocking densities, food digestibility, and on other husbandry factors such as feeding practices and disease status (Wu 1995).

The meteorological (e.g. wind patterns), hydrographical (e.g. bathymetry, currents, tidal regime, wave action, sedimentation rates) and geomorphological characteristics of aquaculture sites (Kempf et al. 2002; Nordvarg and Håkanson 2002; Kalantzi and Karakassis 2006; Rodriguez-Gallego et al. 2008), strongly influence the fate of any type of waste released into the water column. For instance, high-energy environments, well swept by bottom currents, are usually less affected by the impacts of waste material than low-energy environments, most likely due to the contribution of hydrodynamics to the dissipation and dispersion of exogenous material (Klaoudatos et al. 2006). Furthermore, re-suspension periodically re-exposes superficial sediments and waste products to oxygen, enhancing organic matter decomposition (Burdige 2006). Conversely, in shallow waters or in restricted exchange environments (e.g. semi-enclosed estuaries, bays or fjords) with weak bottom currents, there is a higher risk of particulate organic matter and nutrients to increase locally (Wallin and Håkanson 1991), causing not only the degradation of water quality but also severe negative impacts on benthic assemblages.

Effluents from intensive production systems, with a large feed input, typically have greater negative impacts than effluents from semi-intensive or extensive systems with little or no feed addition (Kautsky et al. 2000; Páez-Osuna 2001a; Banas et al. 2008). However, the economic viability of these systems, relying mostly on natural food, is usually compromised by their limited capacity to control environmental and husbandry factors (e.g. nutrition, predators and disease agents), and by their low productivity. To turn aquaculture into a more productive activity with improved profit margins, farmers worldwide have been intensifying production (World Bank 2006). As stocking densities increase, the systems increasingly require higher water volumes, use of feeds and chemicals, which substantially increase organic and inorganic loadings. For example, the ecological footprint of semi-intensive tilapia production systems is relatively low (approximately equal to the farm area) compared to intensive systems that require an area up to 10000 times higher than the farm area (Folke et al. 1998). The higher the degree of artificiality, more likely is the occurrence of environmental damages because recycling processes and their respective feedback mechanisms vaguely resemble natural systems (Kautsky et al. 2000; Banas et al. 2008).

Table 2.1 – Amounts (kg per ton of product) of Total Suspended Solids (TSS), Biochemical Oxygen Demand (BOD), Particulate Organic Matter (POM), Nitrogen (N) and Phosphorus (P) discharged from different aquaculture units.

Species	Culture method	TSS	BOD	POM	N	P	Reference
		kg per ton of product					
Finfish	Marine cage farming				61- 132	2.2 - 95	Enell and Ackefors (1991), Islam (2005)
Seabreams	Marine cage farming	7038 - 9105	235	843 - 1009	190	28	Jambrina (1995), Barbato et al. (1996), Tovar et al. (2000b)
Octopus	Marine cage farming				111	37	Mazón et al. (2007)
Salmonids	Freshwater cage farming	474 - 4015	285 - 990		71	11	Beveridge et al. (1991), Kelly et al. (1996)
Catfish	Freshwater systems				9.2	0.57	Schwartz and Boyd (1994)
Rainbow trout	Freshwater systems	640	129 - 551			22	Holby and Hall (1991), Boaventura et al. (1997)
Shrimp	Semi-intensive earth ponds	715 - 9105	235	257 - 918	29 – 48	2.6 - 4.6	Páez-Osuna et al. (1997), Biao and Kaijin (2007), Casillas-Hérnandez et al. (2007)

Species cultured in intensive systems, usually high-trophic level species, have a higher ecological footprint than those producing low-trophic level species, as omnivorous or herbivorous fish (e.g. catfish, tilapia) (Table 2.1). Carnivore species require high-proteic manufactured feeds, releasing substantial amounts of wastes that are not easily assimilated by the environment (Karakassis et al. 2000; Choo 2001; Páez-Osuna 2001a; Pearson and Black 2001; King and Pushchak 2008). For instance, a study carried out by Folke et al. (1998) revealed that Atlantic salmon marine cage farming requires an ecosystem area 40000 to 50000 times higher than the farm area. However, as feed technology improves and higher feed conversion rates (FCR) are attained, the footprint of intensive carnivore production is likely to decrease (Black 2001). An additional factor contributing to the high ecological footprint of carnivorous aquaculture is the use of the so-called “trash fish” (i.e. fish unfit to human consumption) for the production of pelleted diets, which consumes a large quantity of natural resources (Black 2001).

The most environmentally benign production systems are probably those cultivating species from the base of the food web, like seaweeds or filter-feeders (Crawford et al. 2003). However, even these systems may have a relevant ecological footprint, depending on the location, farm dimension and stocking densities (Folke et al. 1998; Black 2001; World Bank 2006). For instance, large amounts of biodeposits (e.g. bivalves’ feces and pseudo-feces) may induce changes on benthic processes and benthic communities (Buschmann et al. 1996; Kaiser 2001; SECRU 2002; Watson-Capps and Mann 2005), with consequences for the entire ecosystem.

Aquaculture systems combining species from different trophic levels (e.g. fish-shellfish or fish-seaweeds polyculture) or integrated with other activities like agriculture or waste treatment may significantly lower the environmental impacts of aquaculture because nutrients and organic matter are recycled within the system (Buschmann et al. 1996; World Bank 2006).

2.1 Organic matter enrichment

The immediate effects of particulate organic matter released from aquaculture operations include the stimulation of phytoplankton and bacterial development, which reduces the penetration of light into the water column, subsequently affecting benthic flora (Páez-Osuna 2001a; Ruiz et al. 2001; Watson-Capps and Mann 2005; Pérez et al. 2008). However, in oligotrophic systems such as the Mediterranean Sea, aquaculture impacts on the water column are minimal, presenting only localized or no effects on most water quality parameters (Maldonado et al. 2005). These findings are generally attributed to fast dilution (Pitta et al. 2006) and high nutrient recycling rates within the food web (Machias et al. 2004). Particulate organic loading also contributes to long term changes in the benthic environment (Gowen and Bradbury 1987; Wu 1995; Karakassis et al. 1998; Holmer et al. 2005; Klaoudatos et al. 2006).

On reaching the bottom, biodeposits may be incorporated into the sediment or re-suspended by bottom currents (Jones et al. 2001) that disperse them further away from the discharge point. With the continuous deposition of organic matter, microbial activity is enhanced and sediments become reduced due to an increase in oxygen consumption (Giles et al. 2006; Belias et al. 2007; Holmer and Frederiksen 2007). When the oxygen demand caused by the input of organic matter exceeds the oxygen mixing rate from overlying waters, sediments become anoxic and anaerobic processes dominate (SECRU 2002; Holmer and Frederiksen 2007). Microbiological processes such as denitrification, nitrate, manganese, iron and sulphate reductions, and methanogenesis prevail (Pearson and Black 2001), whilst aerobic respiration and nitrification processes are inhibited by sulphide (Deb 1998). The outcome of these reactions is the production of toxic gases (e.g. ammonia, methane and hydrogen sulphide) and the development of hypoxia in the water column (SECRU 2002).

Changes in the physical and chemical characteristics of sediments generally have strong adverse impacts on the structure of benthic communities (Naylor et al. 2000; Pearson and Black 2001; Kelly and Elberizon 2001; Páez-Osuna 2001a; Nordvarg and Håkanson 2002; Edgar et al. 2005; Watson-Capps and Mann 2005; Klaoudatos et al. 2006; Rodriguez-Gallego et al. 2008). Although initially the diversity and biomass of benthic fauna increases, mostly due to the expansion of opportunistic species (e.g. small annelid and nematode worms) and the immigration of other species, the continuous organic matter input will promote anoxia of the deeper sediment layers leading to the elimination of larger and deeper burrowing long-lived forms and subsequently to a decrease in biodiversity (Kelly and Elberizon 2001; Pearson and Black 2001; Edgar et al. 2005; Felsing et al. 2005; Klaoudatos et al. 2006). The increasing sediment oxygen demand will eventually bring anoxia into the lower levels of the water column, originating the appearance of an azoic zone (Tovar et al. 2000b; Ruiz et al. 2001; Kelly and Elberizon 2001; Pearson and Black 2001; Read and Fernandes 2003; Edgar et al. 2005; Gyllenhammar and Håkanson 2005; Watson-Capps and Mann 2005).

The impacts of aquaculture on benthic primary producers, particularly on seagrass communities, have been widely reported (Ruiz et al. 2001; Pérez et al. 2008). The combined effects of light attenuation, mainly due to the shade effect of aquaculture structures and high concentrations of suspended solids, with the accumulation of organic wastes on bottom sediments, significantly reduces the density of seagrass meadows, such as *Posidonia oceanica* (Cancemi et al. 2003; Pérez et al. 2008). Bottom sediment enrichment may also increase epiphytic growth and herbivore pressure, limiting the seagrasses photosynthetic activity (Ruiz et al. 2001). Moreover, the decomposition of organic matter increases porewater nutrient availability and sulphide concentrations in the root zone, which negatively affects seagrasses health and survival (Pérez et al. 2008).

Changes on the benthic compartment may affect trophic relations and energy transfer along the aquatic food webs (Wu 1995; Deb 1998; Karakassis et al. 2000; Tovar et al. 2000b; Kelly and Elberizon 2001; Pearson and Black 2001; Read and Fernandes 2003; Felsing et al. 2005; Gyllenhammar and Håkanson 2005; King and Pushchak 2008). For instance, studies carried out in marine cage farms revealed that the organic wastes released from aquaculture operations constitute an additional food source for wild fish living in the vicinity of the culture site, making fish to congregate locally (Pearson and Black 2001; Machias et al. 2004; Gyllenhammar and Håkanson 2005). The reduction of the fishing pressure and the refuge/protection provided by aquaculture structures (Pearson and Black 2001; Machias et al. 2004) may additionally contribute for wild fish assemblages. Although the magnitude of these bottom environmental impacts depends on several factors such as, culture type, stocking densities and cultivated species (Wu 1995; Kempf et al. 2002; Kalantzi and Karakassis 2006), in general, the major negative effects are found in the farm area and in its immediate vicinity, decreasing with greater distance from farming operations (Karakassis et al. 1998; Pearson and Black 2001; Kaiser 2001; Cromey et al. 2002; Felsing et al. 2005).

2.2 Nutrient enrichment

Inputs of inorganic compounds (e.g. ammonia, nitrates, nitrites and phosphates) through organic matter breakdown, animal excretion and pond fertilization may also have potentially hazardous effects on the surrounding environment (Wu 1995; Buschmann et al. 1996; Deb 1998; Tovar et al. 2000b, 2000b; Páez-Osuna 2001a; Pearson and Black 2001; Read and Fernandes 2003; Biao and Kaijin 2007; Pérez et al. 2008; Rodriguez-Gallego et al. 2008). Most of the undesirable ecological consequences related to the excessive nutrient availability from aquaculture discharges (Table 2.1) are related to eutrophication, and include, for example, hypernutrification and the depletion of dissolved oxygen that cause the deterioration of water quality (Tovar et al. 2000b; Read and Fernandes 2003). Nutrient loadings also contribute to the pool of plant nutrients in aquatic systems, stimulating the growth of primary producers (Read and Fernandes 2003; Biao and Kaijin 2007) and even changing the structure and composition of these key communities (SECRU 2002).

Should nutrient enrichment coincide with certain physical conditions, and other, poorly understood factors, there may be a growth of toxic phytoplankton species, leading to the formation of Harmful Algal Blooms, HAB (Biao and Kaijin 2007; King and Pushchak 2008). For example, reports of HAB of *Chattonella marina*, presumably, caused by effluent discharges from shrimp farms were documented alongshore the north of the Yellow Sea in 1993 and 1995 (Biao and Kaijin 2007). Toxic phytoplankton blooms may produce different types of toxins (e.g. DSP - diarrheic shellfish poisoning, PSP - paralytic shellfish poisoning, and ASD - amnesiac shellfish disease) that often cause shellfish poisoning and the mortality of benthic fauna and wild/farmed fish, thereby threatening the economic viability of aquaculture activities (Pearson and Black 2001; Read and Fernandes 2003; Gyllenhammar and Håkanson 2005).

Although the potential for eutrophication appears unlikely to marine cage farming due to the dilution effect of seawater (Wu 1995; Pearson and Black 2001), the possibility of localized eutrophication in areas of poor flushing cannot be excluded (Wu 1995; Pearson and Black 2001). In terms of restricted exchange areas, such as coastal lagoons and estuaries, excessive nutrient availability may affect the ecosystem productivity (OAERRE 2001) and in some cases, negatively affect the aquaculture activity itself (Deb 1998; Páez-Osuna 2001b).

2.3 Chemical contamination

The overuse and misuse of chemicals in aquaculture operations is also a reason for apprehension due to the pollution and contamination effects that it may have on the aquatic environment. Chemicals used in aquaculture operations may be categorised as: 1) feed additives (e.g. vitamins, pigments, minerals, and hormones), 2) disinfectants (e.g. bleach, malachite green) and pesticides (e.g. molluscicides and piscicides), 3) liming materials, 4) metals (e.g. antifoulants) and 5) veterinary medicines, including antibiotics, anaesthetics, parasiticides, and vaccines (Read and Fernandes 2003) used to control external and internal parasites or microbial infections (Costello et al. 2001). Other biological products, such as, organic matter decomposers (e.g. bacteria and enzyme preparations) are also used (Gräslund and Bengtsson 2001).

The application of these chemicals is mainly dependent on the culture system. For instance, while semi-intensive shrimp farms require a minimal use of chemicals, mostly fertilizers and liming materials (Boyd and Massaut 1999; Choo 2001; Gräslund and Bengtsson 2001), as shrimp production is intensified, management becomes more problematic, and the number and diversity of chemical compounds largely increases (Gräslund and Bengtsson 2001). Intensive pond culture also requires a higher diversity of chemicals when compared to cage systems, which mostly use disinfectants, antifoulants and veterinary medicines (Kelly and Elberizon 2001; Read and Fernandes 2003).

The main environmental risks associated with the use of chemical compounds relate to: i) deterioration of water quality, ii) interference on biogeochemical processes, iii) direct toxicity to wild fauna and flora, iv) development of resistance by pathogenic organisms, and v) reduction of the prophylactic efficiency of therapeutants (Costello et al. 2001). The improper use of chemical compounds may also affect the safety of aquaculture products, constituting a threat to human health (Choo 2001, Islam et al. 2004).

Since many of the chemicals used in aquaculture were not originally developed for this industry, their effects on the aquatic environment are not fully known.

2.4 Spread of parasites and diseases

The dissemination of parasites and diseases from farmed species to wild stocks, principally through water, escapees or diseased seed (Nash 2005), constitutes an important constraint to the sustainability of the aquaculture industry, not only from the ecological point of view but also from the economical perspective because it affects the investors' confidence, the commercialization of aquatic products and profit margins (Choo 2001; Kaiser 2001; Pearson and Black 2001; Subasinghe and Phillips 2002). Even though this was usually considered a localized problem in the past, with the expansion and globalization of the aquaculture industry, pathogens and parasites restricted to one region are now rapidly spreading over the world. For instance, the introduction of post larvae and broodstock from areas affected by the White Spot Syndrome Virus and Taura Syndrome Virus caused mass mortalities in a wide range of shrimp species in Asia and Latin America countries (Choo 2001). Wild salmon and sea trout cultivated in marine cage farms are also thought to be at risk due to the spread infective larval sea lice from salmon farms (SECRU 2002). The level of risk for disease or parasites transfer is usually difficult to quantify not only because hosts may carry pathogenic organisms without showing any symptoms but also because a wide range of parasitic worms, pathogenic bacteria (*Salmonella*, *Escherichia*, *Vibrio*, and others) and viruses are already present in natural waters, being common to both wild and cultured species. Many of these pathogenic organisms may also be introduced by other human activities besides aquaculture, like livestock, human waste and aquatic products transportation (SECRU 2002). Besides the environmental risks, the propagation of parasites and diseases also constitutes a risk to human health although it can be minimized or even completely eliminated, through the implementation of strict sanitary and food safety regulations (e.g. HACCP) to commercial aquaculture (World Bank 2006).

2.5 Habitat destruction and modification

The loss or degradation of habitats, in particular of coastal habitats such as mangrove systems and other wetlands (seagrass meadows, saltmarshes, coastal lagoons, estuaries) is one of major adverse impacts of aquaculture (Wu 1995; Deb 1998; Naylor et al. 2000; Black 2001; Páez-Osuna 2001b; Ruiz et al. 2001; Pérez et al. 2008). Studies carried out in marine cage farms on the Mediterranean coastline reported the destruction/degradation of *Posidonia oceanica* meadows, as a consequence of the high organic and nutrient loading from fish farming activities. Conversion of mangrove forests into shrimp farms (Deb 1998; Choo 2001; Páez-Osuna 2001b) has mainly caused the loss of feeding, nursery, shelter and spawning grounds for a wide variety of marine and terrestrial animals (Ruiz et al. 2001; Pérez et al. 2008), and the loss of natural protection against floods, storms and hurricanes (Deb 1998; Choo 2001; Páez-Osuna 2001b). Coastal lowlands, such as mangroves and saltmarshes, play a significant role in shore protection by deflecting and reducing the energy of water masses, and by being important routes of water discharge (Deb 1998; Choo 2001; Páez-Osuna 2001b). The construction of channels and dikes for inland aquaculture has also irreversibly altered the hydrological conditions (e.g. water discharge rates and sediment loads) of many coastal systems and the shore geomorphology (Deb 1998; Primavera 2006). Habitat modification caused by bivalve farming during harvesting or the preparation of cultivation grounds (usually by addition of gravel, sand and protecting nets), may additionally change the sedimentary processes and the biogeochemistry of farming sites. This disruption of bottom communities (e.g. benthic fauna or seagrasses) may have negative consequences for the higher trophic levels, for example, by affecting the feeding behaviour of wadding birds and of marine mammals (Kaiser 2001; Watson-Capps and Mann 2005). Other potentially adverse impacts on marine mammals include for example, the death or injury through entanglement in gear, habitat displacement, and disruption of migration pathways, especially for large cetaceans (Watson-Capps and Mann 2005).

2.6 Introduction of new species and new genetic varieties

The deliberate or inadvertent introduction of new species or genetic varieties should be a key aspect when assessing the environmental impacts of aquaculture. The main impacts of introductions fall into two categories: i) ecological, including biological and genetic effects, and ii) socio-economic (cf. – Section 2.8), that can be interrelated. Despite providing significant social and economic benefits (e.g. supply of animal protein and disease control), the use of exotic species may also seriously affect ecosystem functioning. The main negative ecological impacts resulting from the introduction of new species and genetic varieties include: i) loss of biodiversity, due to direct biological interactions such as predation and competition; ii) loss of genetic diversity in wild populations, mainly due to breeding of alien organisms with local strains or species; iii) transmission or spread of diseases to which indigenous species are more vulnerable; iv) and habitat modification (Black 2001). A case reporting the hazards of species introductions is that of the Nile perch in Lake Victoria, which became the dominant species of the lake’s fauna. Even though the introduction of Nile perch generally provided relevant economic benefits for some entrepreneurs (may be not so for the population depending directly on lake biodiversity), the arrival of the invasive water hyacinth blocked waterways and the access to riparian villages and fishing grounds, causing major economic losses (World Bank

2006). Whirling disease, a virus infection that affects rainbow trout, was introduced in North America through the importation of European brown trout that was immune to the virus (World Bank 2006). Other vectors for species introduction include for example the ships ballast water or the feces and digestive tracks of commercialised bivalves, which may transport the resting cysts of toxic phytoplanktonic species and of seaweeds species (Kaiser 2001).

The release of cultivated organisms to the natural environment, either by accident or natural catastrophes, not only poses a risk for the structure of wild populations but also to the regional economies (Youngson et al. 2001; Read and Fernandes 2003). Most of the negative ecological impacts resulting from the interaction between cultivated and wild species result from the genetic interaction of wild organisms with their aquaculture conspecifics. The genetic impacts of escapes on wild populations are a complex subject, but the fundamental problem rests on the genetic differences between wild and farmed species (Kapuscinski and Brister 2001). As part of the evolutionary strategy, wild species possess higher genetic diversity both within and between populations (SECRU 2002). Escapees that survive and spread to spawning grounds can interbreed with wild organisms (Kapuscinski and Brister 2001; SECRU 2002; Naylor et al. 2005), posing two types of hazards: firstly, outbreeding depression (i.e. loss of fitness in the offspring) that mainly reduce the survival fitness and efficiency of wild organisms and secondly, the homogenization of genetic differences which increases the vulnerability of individuals to environmental changes, and compromise the sustainability of wild populations (Kapuscinski and Brister 2001). Even though domesticated species, such as the farmed Atlantic salmon, are generally less fit for survival and breeding (mainly due to a lower ability to participate in breeding and to a poorer quality and quantity of gametes), when a substantial proportion of escapees secure mating with wild fish, outbreeding depression may cause the decline of wild populations (Kapuscinski and Brister 2001; SECRU 2002; Naylor et al. 2005) due to the loss of environmental adaptive genotypes which determine the species success. These risks are greater for small populations that are already threatened and, whenever genetic modified organisms (GMOs) are used. The growing development of GMOs to increase the quantity and quality of aquatic products may seriously jeopardize the genetic integrity of wild stocks and ecosystems functioning (Spreij 2004).

2.7 Harvest of wild stocks as feed or seed/broodstock to aquaculture operations

The depletion of wild resources and biodiversity to produce animal feeds or to supply seed/broodstock to aquaculture can cause significant damages to aquatic ecosystems (Deb 1998; Choo 2001; Kaiser 2001; Páez-Osuna 2001b). Fish species of low commercial value (e.g. Japanese anchovy and chub mackerel) are mainly targeted to be processed into feeds for carnivorous fish, or as supplements for other species, like for example, shrimp, tilapia and milkfish (Black 2001). The use of this so-called “trash fish” puts even more pressure on the already overexploited wild fish stocks. The broad collection of wild seed (e.g. of eel, grouper, yellowtail, and tuna aquaculture) and broodstock for aquaculture purposes also contributes to the decline of natural populations. The collection of wild shrimp and shellfish seed is particularly environmentally-damaging because not only it threatens the wild stocks of target species (e.g. by affecting species recruitment) but also affects the stocks of other living resources (other shrimp species, macrozooplankton, finfish and shellfish juveniles and larvae) that are indiscriminately killed. This reduces the food availability for other organisms such as aquatic birds, reptiles and mammals linked through the trophic web, and may subsequently increase their mortality at the same time that it reduces their breeding success (Choo 2001). Harvest of wild species may also cause genetic degradation of native populations and the destruction and modification of natural habitats, causing further disturbances on the aquatic food web (Deb 1998, Primavera 1998, Islam et al. 2004, World Bank 2006). This activity is particularly dangerous for heavily fished species and for species with low reproductive capacities (World Bank 2006), but probably as long as the production of broodstock in captivity remains costly, the purchase of wild spawners will continue, causing environmental damages in ecosystems around the world (Nash 2005; World Bank 2006).

2.8 Socio-economic impacts

Despite the negative impacts that it might have on the environment, aquaculture may also provide important socio-economic benefits. For instance, aquaculture is foreseen to become the major source of animal protein (Naylor et al. 2000; Sugiura et al. 2006; World Bank 2006).

The commercialisation of aquaculture products is also an important source of incomes (Biao et al. 2004; Primavera 2006) and largely contributes to the country’s economic development (Table 2.2). For instance, since 1970, the aquaculture sector has increased at an average annual rate of 10.4% in developing countries (World Bank 2006) while in developed countries it grew on average 4% per year. The trade of aquaculture products is particularly important for developing countries and to low profit-food deficit countries (e.g. Bangladesh, Indonesia, Vietnam) because it considerably increases their revenues. Besides contributing to the development of national economies, aquaculture has also allowed the stabilization and strengthen of populations from remote regions or marginalised social groups (mainly in Asia and Africa), by increasing rural development and reducing poverty and hunger (Black 2001; World Bank 2006). Aquaculture production may also contribute to the reduction of fish prices, at the same time that it increases the access to fish products by poor households. An example of pro-poor aquaculture has been implemented in Asia, where it was developed under two models: one in which commercial opportunities have been opened for enterprises, and the other consisting in using public support to generate enough critical mass for smallholders. The enterprise model not only generated growth and employment in poor regions where alternative employment is scarce as also increased the stability of local communities (Black 2001). For example, this sector employs more than 12 million people in China, Indonesia, and Bangladesh alone (FAO 2007). Many of these people are rural dwellers and some, such as wild shrimp seed collectors, are among the poorest and most marginalized (Deb 1998). On the other hand, public support extended profit opportunities to smallholders in China, Vietnam, and Bangladesh mainly by combining a supportive policy (e.g. microcredit) with the dissemination of knowledge on proven technologies (e.g. polyculture). This strategy has also proven to be an effective mean of targeting the landless poor (e.g. rice farmers) mainly by improving their livelihoods (World Bank 2006; FAO 2007). A surplus in households may turn into a social benefit because it improves the nutritional state of poor populations and provides an opportunity to invest in education.

Table 2.2 – Top ten aquaculture producer countries in 2006 and its respective aquaculture revenues.

Country	million tons	%	US$ billions	%
China	34.4	67.7	38.4	48.8
India	3.12	6.05	3.43	4.36
Vietnam	1.66	3.20	3.32	4.21
Thailand	1.39	2.68	2.22	2.81
Indonesia	1.29	2.50	2.25	2.86
Bangladesh	0.892	1.73	1.36	1.73
Chile	0.802	1.55	4.43	5.62
Japan	0.734	1.42	3.10	3.93
Norway	0.708	1.37	2.72	3.45
Philippines	0.623	1.21	0.981	1.25

Source: FAO Fishstat, ftp://ftp.fao.org/fi/stat/summary/default.htm

Other social benefits provided by aquaculture include for example, women empowerment. In Bangladesh and Vietnam, more than 50 percent of workers in seed collection, fish markets and processing plants are women, and although salaries of these workers are still quite low ($1–$3 per day), they are significantly higher than salaries earned in agricultural activities (World Bank 2006). In the Mekong delta aquaculture has also contributed to a decrease in urban migration by young women and prevented women from being forced into prostitution, reducing the risks of spreading sexual diseases (FAO 2007).

Although responsible aquaculture can provide significant economic benefits, uncontrolled and irresponsible aquaculture operations can cause a wide range of negative socio-economical impacts, particularly when the ecosystem functioning is radically altered and the resources that support other human activities are affected. For instance, pandemics outbreaks have devastated shrimp farming in many producing countries (Deb 1998). Other adverse effects result from the introduction of new species. For example, the introduction of the golden apple snail into Asian countries, mainly with the purpose of developing an export industry, resulted in high damages to rice farmers, since this snail consumed large quantities of paddy-rice (World Bank 2006). The import of crayfish and oysters from North America also destroyed the European crayfish and oyster industries mainly due to the introduction of pathogens hosted on the imported organisms (World Bank 2006). Conversely, in Chile, the introduction of the Pacific and Atlantic salmon in the 1970s turned into an economic benefit, since the country is now the world’s leader in salmon production. Tilapia, a group of species originating in Africa, is also cultured worldwide and provides income and high-quality protein to many rural areas, especially in developing countries.

The inexistence of an ecosystemic approach for the management of the aquaculture industry, often lead to conflicts over common resources such as land and water. For instance, the conversion of mangrove forests into commercial shrimp farms led to the loss of forest products and fisheries (Primavera 2006), affecting principally the poor populations. The conversion of residential, agriculture (rice and pastures) and common lands in Asian countries (Thailand, Bangladesh and Philippines) has also raised serious conflicts between agriculture and shrimp farmers (Deb 1998; Choo 2001; Primavera 2006). Conflicts over water use are particularly frequent because aquaculture effluents may contaminate the water used by other aquaculture units downstream (Deb 1998; Gräslund and Bengtsson 2001). On the other hand, aquaculture itself may be subjected to water contamination due to urban waste and agricultural pollution. Saltwater intrusion caused by aquaculture activities, either from the percolation of water discharged from brackish/marine cultivation ponds or from active pumping of groundwater, has also several negative socio-economic repercussions, including, for example, the loss of agricultural crops, land subsidence, decrease in fish production or the occurrence of freshwater crisis that cause gastrointestinal diseases (Deb 1998; Choo 2001; Páez-Osuna 2001b). Other negative impacts resulting from the massive introduction of aquaculture structures (ponds, cages, or rafts) include the blocked access to coastal resources, navigational hazards, privatisation of public lands and waterways, and fisheries decline (Primavera 2006). Conflicts over common resources generally lead to serious social problems and even in some cases, to human rights abuse (World Bank 2006).

An ecosystem approach to the management of the aquaculture industry is therefore crucial for its sustainability. Letting aquaculture development proceed irresponsibly or taking only partial approaches to its management incurs a risk that the negative impacts may counteract any benefits from aquaculture or that it will not produce the expected benefits.

3 The concept of carrying capacity in aquaculture

In a broad sense, carrying capacity (CC) may be defined as the capacity of a natural or man-made system to hold a certain pressure without driving its structure and function above Limits of Acceptable Change (LAC) (Duarte 2003). Whilst this general and simple definition may be appropriate as a first approach, it is important to apply the concept to some specific areas and to develop more precise definitions. CC may be defined within the scope of any activity implying some sort of environmental, social or economical impact. The LAC concept has long been used in tourism management (e.g. Wearing and Neil 1999). The goal is to be able to use natural and man-made ecosystems without compromising their capacity to continue providing the goods and services that people need. The definition of LACs is not straightforward, because though some of these limits may be defined on a quantitative way, others are rather subjective and depend on people’s perception about the environment. For example, water quality parameters may be used to establish quantitative limits on aquaculture outflows to prevent ecosystem degradation. However, it may be more arguable to establish limits in relation to scenic or habitat quality (GESAMP 2001).

The concept of CC is a central theme in aquaculture and it may be related to the amount of natural resources available for aquaculture operations, such as food and space, the services provided by natural ecosystems, such as organic matter mineralization and nutrient cycling, or the economic yield of aquaculture and its economic and social effects. When CC is exceed, negative-feedbacks affect aquaculture operations and may result in yield losses.

Policy makers must take management decisions that may affect the sustainability of natural resources. Having at hand the relevant CC indicators, is the way to prevent them from taking decisions that will jeopardize options for future usages. Whenever possible, these indicators should be quantitative, such as the area that may be allocated for aquaculture, the standing stock of fish that may be kept in a fish culture area, etc.

Given the multiple exploitation possibilities of aquatic ecosystems and their synergic effects, it is clear that CC must be accessed for different activities taking into account their interactions. For example, if a coastal zone is used for sewage dispersal, it’s CC for aquaculture may be limited, because not all areas will have the necessary water quality for aquaculture and also because the impact of the sewage outfall may limit ecosystems resilience to assimilate organic loads from aquaculture leases. These complex set of interactions between different uses and the ambiguities of resource ownership leads to the idea of including aquaculture within the framework of Integrated Coastal Zone Management (GESAMP 2001). Concerning inland aquaculture, similar integrated approaches are needed integrating other activities such as agriculture, tourism, nature conservation, etc.

3.1 CC categories and definitions

The CC definition and classification defined by Inglis et al. (2000), adopted by McKindsey et al. (2006) and adapted by Gibbs (2009), regarding coastal aquaculture development, was followed in the present work for aquaculture in general:

(i) physical CC – the total area of farms that can be accommodated in the available physical space;

(ii) production CC – the stocking density of cultured organisms at which harvests are maximized;

(iii) ecological CC – the stocking or farm density which causes unacceptable ecological impacts;

(iv) economic CC – the biomass that investors are willing to establish and maintain;

(v) social CC – the level of farm development that causes unacceptable social impacts or that community is willing to allow.

Some of the above categories are defined differently by different authors. For example, according to Jiang and Gibbs (2005), production CC is the theoretical maximum culture that could be supported in an embayment. Alternatively, production CC was defined as the maximum sustainable yield of culture that can be produced within a region, whereas ecological CC was defined as the level of culture that can be supported without leading to significant changes to ecological processes, species, populations or communities in the growing environment (Gibbs 2007). Therefore, in defining production CC, most authors choose to express it as a stock measure (e.g. Carver and Mallet 1990; Bacher et al. 1998; Inglis et al. 2000; Jiang and Gibbs 2005), whereas others define it as a yield measure (e.g. Gibbs 2007). Therefore, it is important to agree on some common measurements for the sake of comparability within and across different aquaculture areas. Since stock and yield are related, although differently in different aquaculture areas, and since stock is easier to regulate, perhaps it is the most straightforward way to quantify CC.

3.2 CC and limiting factors

The CC categories above reflect some of the most common limiting factors for aquaculture development. However, it must be emphasised that, in most instances, these categories are interlinked. In the case of physical CC, space may be limiting due to the lack of sheltered areas and to other competing uses such as sewage dispersal, harbour activities, fisheries, tourism, nature conservation and water availability (in the case of inland aquaculture). For example, the Southeast Asia’s seas are under several threatens – 11% of coral reefs collapsed, whilst 80% face risks, mangroves – one of the most threatened tropical environments (Valiela et al. 2001) – have lost 70% of their cover, seagrass beds' loss ranged from 20 to 60%, urbanization is predicted to increase and there are tens of pollution hot spots (PEMSEA 2003). Whilst urbanization and resulting pollution may limit geographically aquaculture development, aquaculture itself has been one of the reasons for mangrove destruction in most tropical countries (Primavera 2006) – in relation to ecological CC. In the case of Thailand, a ban on mangrove destruction in the early 1990s was followed by a shift from salt water to low-salinity inland shrimp farming, leading to competition for soil resources between rice and shrimp farmers and to soil salinization (GESAMP 2008). Competition between farmers may be a result of overcoming social CC.

Regarding production CC, limiting factors depend mostly on the culture type. In the case of extensive and semi-intensive cultures, stocks may be limited by food availability and water quality. A typical example of extensive systems, fully dependent on natural food (phytoplankton and organic detritus) is the cultivation of bivalve suspension-feeders. Both the quantity and the quality of these food items are important for bivalve growth (Bayne 1993; Hawkins et al. 1998). Production CC for bivalve cultivation depends on the renewal rate of available food. Suspension feeders have a remarkable capacity to filter the water column such that they are food limited at higher culture density. Therefore, water residence times and phytoplankton doubling times may limit CC (Dame and Prins 1998).

The relationship between bivalve production and bivalve standing stock is parabolic (Figure 2.1), as demonstrated by the theoretical model described in Bacher et al. (1998) and the results of Ferreira et al. (1998) and Duarte et al. (2003). There is an initial increase in production, but as available space becomes filled up with stock, individual bivalve growth rate is depressed and mortality increases due to several factors associated with overcrowding. The overall result of these effects is a strong reduction of harvest yields above a certain stock threshold.

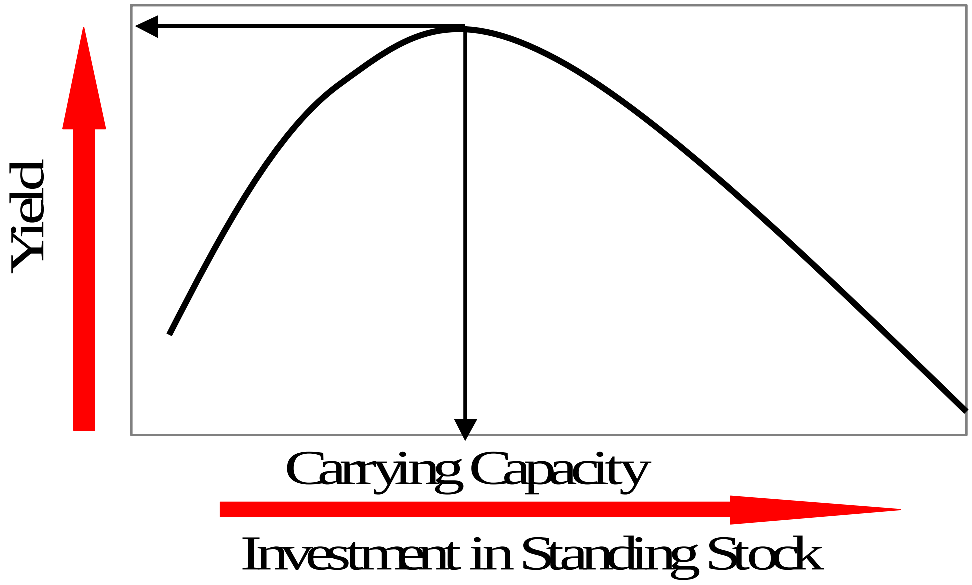


Figure 2.1 – The parabolic relationship between stock and yield in bivalve culture. Carrying capacity increases with stock up to a point above which individual growth is severely compromised due to food limitation (see text).

In semi-intensive and intensive systems, production CC may be limited by water quality, namely, by dissolved oxygen (DO) in some fish farms (Shin and Wu 2003). On the other hand, release of feces, uneaten food and excreta may increase biochemical oxygen demand (BOD) and nutrient concentrations that may overtake limits defined for ecological CC. According to Sarà (2007), available literature data on the effects of aquaculture leases on water quality present a convincing evidence for increases in ammonium, nitrite and nitrate and, to a lesser degree, dissolved phosphorus, in comparison to non-aquaculture sites. These “aquaculture effects” are most noticeable in sheltered water bodies with high residence times.

Another important limitation for bivalve production in coastal areas is Harmful Algal Blooms (HAB) that may cause bivalve contamination and mortality by harmful toxins (Hágaret et al.

2007) (cf. – Section 2.2). In most areas of the world bivalves are monitored for the occurrence of several toxins to prevent their commercialization.

In a recent paper, Gibbs (2009) discusses the role of social barriers to the establishment of aquaculture activities in suitable areas. According to this author, local stakeholders tend to be more environmentally conscious and demanding strong evidence about the environmental and economic sustainability of aquaculture development. This attitude is related to their perception that aquaculture benefits are diffused among the community and state, while costs are internalized locally, especially in coastal regions where recreational and amenity values are high.

3.3 Methods for determining CC

Physical CC may be analysed and estimated from physical, chemical and biological data, with the help of a Geographical Information System (GIS). These data may include geographic descriptors, sediment and vegetation types, depth, meteorology, hydrography, water quality, land use, etc. The interception of layers with this data types helps selecting areas that may potentially be used for different aquaculture types. For example, sensitive habitats may be excluded, as well as contaminated or other areas, where land use, management plans or political boundaries are not compatible with aquaculture development.

GIS may also be used to help assessing economic and social CC, if it contains information on relevant descriptors. For example, areas that are used for some other economic activities or where local stakeholders have a strong opposition to aquaculture developments may be excluded, reducing social conflicts.

Production and ecological CC may be approached at several spatial scales, such as the scale of the cultivation unit (farms, rafts, etc.) and the ecosystem scale. The former is directly relevant to farmers, whereas the latter is relevant for ecosystem management (Duarte 2003). In accordance to this, aquaculture leases produce “near-field” and “far-field” effects – the latter result from the cumulative effects of the former at the ecosystem scale. This scale may be easy to define in the case of estuaries, bays and fjords but more difficult for open coastal areas (Anderson et al. 2006).

Following the last authors, if the scale of the farm is large in comparison with the ecosystem scale, more important impacts are expected than in the opposite situation. Therefore, the definition of ecosystem boundaries is critical in evaluating aquaculture impacts. One possible approach is the analysis of impacts from the farmer scale to progressively larger scales, until they are no longer relevant. Such an approach is hardly achieved without a mathematical model.

One important point here is that whatever method is used to estimate aquaculture impacts or CC, it should allow resolving scales smaller than the ecosystem scale. The rationale beyond this statement is discussed in Duarte et al. (2005) in relation to bivalve culture, but concepts may be extended to other culture types. The general idea is that if CC is evaluated at a scale larger than the farm scale, “farm effects” are diluted over a relatively larger area. For example, in the case of bivalve suspension-feeders, food limitation may be underestimated, since local food depletion is ignored, with the result of overestimating production CC. Ecological CC may also be overestimated, since excreta from cultivated organisms are “diluted” over a larger area.

Ideally, the smaller scale resolved should be small enough for water residence time to be lower than the time needed for significant changes to occur in any chemical or biological factors related with CC. When this condition holds, water properties do not change much across the scale considered. Current speed measurements or a hydrodynamic model may be used to determine the mentioned smaller scale.

In Figure 2.2, a practical example of the above concepts is presented (for details see Duarte et al. 2003, 2005) regarding Sungo bay (People’s Republic of China). This bay is extensively used for kelp and bivalve culture. If a whole system bivalve production CC is estimated from water residence time, phytoplankton doubling time and bivalve clearance time (the time it takes for the bivalves to filter the water in the bay), as described by Dame and Prins (1998), the obtained result suggests that bivalve density may be doubled within the ecosystem (from ca. 44000 to ca. 88000 tonnes). In fact, Nunes et al. (2003), using a zero dimensional bay ecosystem model obtained even larger production CC estimates. On the other hand, Duarte et al. (2003), using a two dimensional hydrodynamic-biogeochemical model, with a finite-difference grid of 500 m resolution (Figure 2.2) – in line with considerations above on the need to resolve scales smaller than the ecosystem scale - obtained much lower CC estimates. Given average current velocities in Sungo Bay, water residence time within the 500 X 500 m grid cells depicted in Figure 2.2, is smaller than the time bivalves need to filtrate the water within the cells, considering their large densities within cultivated areas (Duarte et al. 2005).

Considering the complex feedbacks between cultivated species and environmental variables, the cumulative effects of many aquaculture activities and the various dimensions of CC, an ecosystem model is necessary for a description of the problem. However, any model is just a pale description of the real system with many limitations, as discussed by Gibbs (2009). Whenever there are no available data and models for a reliable estimate of CC, an adaptive approach should be used by being conservative, according to the precautionary principle, monitoring relevant variables and processes over time and being able to make any adjustments to avoid permanent damage to natural and man-made systems. In fact, this adaptive approach should be followed even in the presence of detailed data and validated models, due to the limitations mentioned above (Gibbs 2009). The confrontation of model results and observations allows model improvements over time, as more knowledge is accumulated about the ecosystem under study. Furthermore, uncertainties associated with model parameters and results may help defining sampling strategies and experiments to fill the gaps.

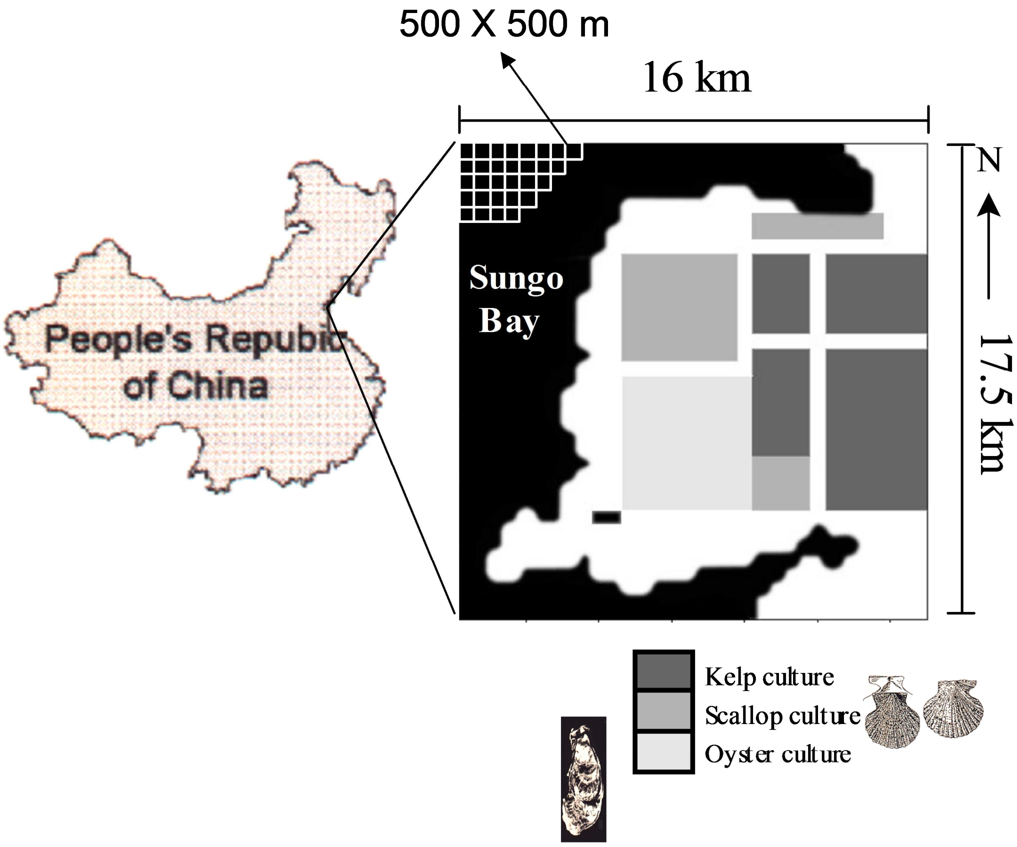


Figure 2.2 – Areas cultivated in Sungo Bay since 1999 with kelps (*Laminaria japonica*), oysters (*Crassostrea gigas*) and scallops (*Chlamys farreri*), including part of a two dimensional model grid (upper left corner), for which the spatial step is 500 m (refer text).

A model capable of predicting production and ecological CC should include a transport and a biogeochemical sub-model. Ideally, it should also include a thermodynamic sub-model, for water temperature calculations, and biological sub-models for relevant species or species groups. The transport sub-model should be able to predict current speeds and water mixing (or simply to read and return current speed time series measured or obtained with another model) and calculate the transport of dissolved substances and particles. It may be forced by wind, river flows, tidal height variability at sea boundaries, etc. The biogeochemical sub-model should reproduce biogeochemical cycles of elements that are most likely to become limiting, such as phosphorus and nitrogen, that may limit primary production of phytoplankton and cultivated plants, oxygen, that may limit fish survival, etc. This sub-model should include a pelagic and a benthic compartment, especially when water-sediment interactions are more important, as in shallow water ecosystems. The biological sub-models should simulate growth, production and biological interactions of most relevant species or species groups. It should also simulate nutrient production/consumption and link these with the biogeochemical sub-model. The spatial resolution of the model should follow considerations above. For some examples see Duarte et al. (2003, 2007), Ferreira et al. (2007), Grant et al. (2007), and Shin and Wu (2003). For a review of recent CC models see McKindsey et al. (2006).

The above sub-models should be forced with time series obtained at their boundaries (e.g. river or sea boundaries) for the simulated water column variables. It is also important to have time series of meteorological data on: solar radiation, air temperature, wind speed, and relative humidity. For very large areas, it may be necessary to nest more detailed models within the grid of larger scale models, with the latter providing boundary conditions for the former.

Having a model to estimate production and ecological CC it is then necessary to simulate several aquaculture scenarios regarding density of organisms, their geographical distribution and different rearing techniques, for example. The analysis of obtained results concerning predicted production and water quality variables may then be used to evaluate the different scenarios. Typically, an increase in production leads to changes in water quality variables and deciding whether these are acceptable or not, depends on the availability of some criteria. For example, Duarte et al. (2007) simulated water and sediment quality as a function of bivalve density in Ria Formosa (Portugal) and compared scenarios on the basis of bivalve production and water quality using the IFREMER water and sediment classification scheme (e.g. Austoni et al. 2004).

Ideally, a Decision Support System (DSS) should be used, integrating also economic and social descriptors (for an example see Pereira et al. 2007). It is important to involve local stakeholders in the decision process. At this point, economic and social CC may be revised by stakeholders, since obtained results may change their initial perspectives (Figure 2.3). It may also be necessary to try other scenarios and iteratively reach a good solution.

4 Aquaculture and environmental management towards sustainable development

In the next paragraphs, some possible mitigation measures and methodological approaches are suggested to reduce and anticipate, respectively, aquaculture impacts. Management aspects that may help reducing the direct ecological impacts of aquaculture leases are discussed in section 4.1. The advantages of Ecological Aquaculture are discussed in section 4.2. Remaining sections present methodological approaches do help stakeholders and decision makers defining potential problems of aquaculture developments and deciding on alternative scenarios.

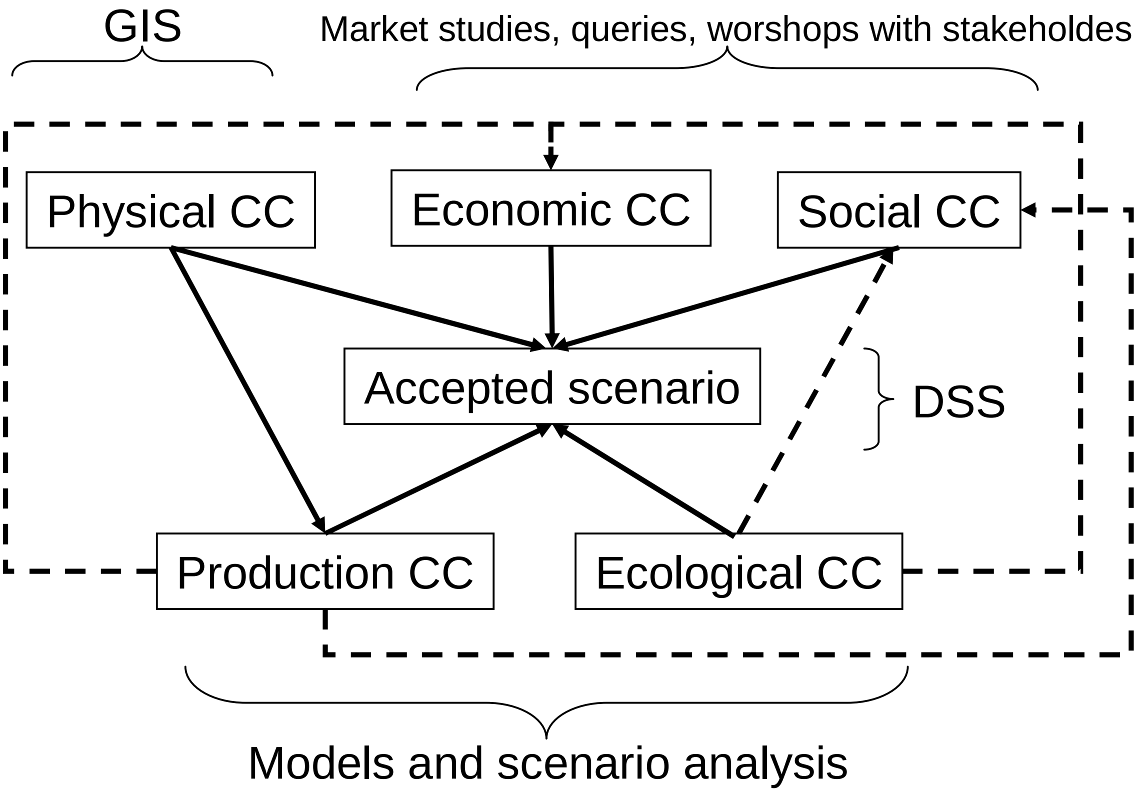


Figure 2.3 – Diagram showing the interactions and feedbacks among different carrying capacity categories towards an accepted aquaculture scenario, and the tools used for physical, production and ecologic CC and for the scenario selection. Continuous lines show direct influences of CC over the accepted scenario or other CC category. Physical CC limits production CC. Dashed lines showing feedbacks from production and ecological CC to economic and social CC (see text).

4.1 Mitigation measures

The sustainable development of the aquaculture industry depends largely on the preservation of natural resources and on ecosystem CC (Read et al. 2001). The adoption of an ecosystem approach to aquaculture (EEA) is probably the way to overcome the problems related to its increasing growth and intensification, in particular those associated with the use and allocation of common resources. The implementation of an EAA requires a partnership among aquaculture organizations (e.g. producers associations), governmental agencies (e.g. fisheries administration, rural, urban and industrial development organizations) and the public sector (e.g. NGO’s), for the development of appropriate regulatory frameworks and efficient enforcement mechanisms. As an alternative to legal frameworks, the aquaculture industry has developed self-regulation instruments, such as Codes of Conduct (e.g. the FAO Code of Conduct for Aquaculture Practices and the International Aquatic Animal Health Code) and Codes of Practice, to ensure the sustainable development of the activity. Compliance to the norms and principles defined in these codes may also contribute for the minimisation of the negative impacts of aquaculture. At the farm and at the ecosystem levels, an efficient use of Environmental Impact Assessment (EIA) or other decision-making tools (e.g. Decision Support Systems during the planning phase of aquaculture operations together with the implementation of mitigation measures (e.g. environmental monitoring) for activities that already exist, may also contribute to a more environmentally-friendly activity. Some of the important decisions that can be made are mainly related to site selection, species selection (exotic *versus* native), definition of stocking densities and proper farming systems or technologies and, on the socio-economic relevance of aquaculture projects (Read and Fernandes 2003). During the operational phase of aquaculture units, specific proactive measures may also be adopted to safeguard the ecosystems integrity. Some of these mitigation measures are presented in the following paragraphs.

4.1.1 Interference in biogeochemical processes

Given that the impacts on bottom sediments are the most obvious form of pollution resulting from aquaculture activities, the reduction of the amount of wastes and effluents released into the environment is crucial for avoiding that the ecological CC is exceeded (Giles et al. 2006). The effects of organic and inorganic waste discharges can be significantly reduced by careful site selection. The specific hydrographic conditions (hydrodynamics, water residence time, and tidal regime), topography, geography and the ecological CC of the receiving body (Buschmann et al. 1996; Pearson and Black 2001; Choo 2001; Gräslund and Bengtsson 2001; Primavera 2006), strongly influence the behaviour of all type of wastes released into the water column. For instance, the impacts of wastes discharges from marine cage farming may be minimized by avoiding regions of restricted water exchange, such as enclosed bays or fjords (Pearson and Black 2001). Site rotation allows the seabed to return to normal conditions. Site selection is also crucial for managing the environmental impacts of shrimp farming since aquaculture units are usually established in mangrove areas and tidal wetlands, which in addition to their high ecological value are also characterised by acidic soils and high organic loadings, that may contribute to the deterioration of water quality and to disease outbreaks (Kongkeo 1997; Boyd and Clay 1998).

Organic sediment enrichment can also cause severe environmental impacts if the scale of the farm operation is not suitable for the aquaculture site, i.e., if organic and nutrient loadings are above the ecological CC of the water body. Hence, the limitation of stocking densities may contribute to a significant reduction in the amount of wastes released into the environment, particularly in sensitive habitats, such as mangrove systems and salt marshes (Buschmann et al. 1996; Kautsky et al. 2000; Gräslund and Bengtsson 2001; Páez-Osuna 2001a; Primavera 2006).

Improving of feeding husbandry techniques (e.g. meal timing or methods for feed supply) and of feed formulation may also be an effective strategy for reducing organic loadings and to prevent the hypernutrification of aquatic systems (Buschmann et al. 1996; Páez-Osuna et al. 1998; Páez-Osuna 2001a; Pearson and Black 2001). In marine cage farms or pens, the installation of feeding devices with hydrosensors that detect the reduction of fish activity or the use of acoustic feed detectors to reduce the loss of feed pellets, may prevent overfeeding and excessive waste production (Pearson and Black 2001). Other mitigation measures for open systems include for example, the use of settling devices for collection of fecal pellets and food wastes under the cages and the use of pumps for the dispersion of solid elements (Gowen and Bradbury 1987; Buschmann et al. 1996). Improvement of feed pellet technology, either by increasing the stability of feeds or reducing its sinking rates may also be a way to maximise the amount of feed ingested, and thereby to minimize waste production (Choo 2001; World Bank 2006). The development of appropriate feeds (with optimal protein/energy ratio) for each species and respective developmental stages further reduces the organic and inorganic loadings to the environment. Since energy requirements can generally be satisfied by lipids and carbohydrates, diets with a higher content of these compounds, increase protein retention and improve feed conversion rates (World Bank 2006). Feeds with high FCRs, like the ones currently used by the Atlantic salmon industry (FCR = 1:1.1, i.e., 1 kg aquatic product per kg of feed), not only reduce the amount of nutrients (nitrogen and phosphorus) released into the environment as also minimize the costs with feeds, since protein is mainly used for body tissue construction (Black 2001; Choo 2001; World Bank 2006). The use of formulated artificial feed instead of “trash fish” (i.e. fish unfit to human consumption), in shrimp and carnivorous finfish culture, is also desirable not only in terms of its nutritional value and supply but also in terms of waste loadings (World Bank 2006). Furthermore, aquacultures activities depending on these resources are particularly vulnerable to collapse since a reduction in fisheries, will most likely increase feed prices and consequently cause a loss of profits (Black 2001; World Bank 2006).

4.1.2 Interference with the life cycles of wild species

Water-related best management practices (BMPs) may also minimize the risks associated with the introduction and dissemination of viruses and other pathogens (Kongkeo 1997).

Recirculation Aquaculture Systems (RAS) systems in particular, not only reduce the possibility of pathogen introduction in freshwater systems as may be an alternative method for the production of healthy seed for marine aquaculture systems (Gutierrez-Wing and Malone 2006). The compliance to other BMPs related to environmental control, as for example careful species selection, limitation of stocking densities and use of proper feeds to avoid deterioration of water quality, or to disease prevention and/or control BMPs like the use of effective vaccines or other prophylactic agents (e.g. probiotics), use of approved medicines and development of disease free strains by selective breeding (Dunham et al. 2001; Primavera 2006; World Bank 2006), may also mitigate the negative environmental impacts of aquaculture. Diseases spread through trade and transboundary movements can also be managed by veterinary control or strict regulations for the movement of living aquatic organisms (either eggs, seeds, juveniles or adults) and by the use of certified disease-free organisms (Argue et al. 2002; SEACASE 2009). Other measures such as the implementation of environmental programmes, e.g. the Hazard Analysis and Critical Control Point (HACCP) method may also minimize the deleterious effects of disease transmission, and ensure the safety of aquatic products. The reduction of disease incidence is a key aspect for the environmental sustainability of aquaculture because not only it reduces the use of chemicals (e.g. antibiotics) and the requirements for land and water, as also improves the efficiency and viability of the farming activity (Hulata 2001; Argue et al. 2002).

As intensification progresses and new species are cultured, seed-based aquaculture is likely to expand, and thereby every effort should be made to reduce the dependence on wild seed. Control/regulation of wild seed by-catch through the establishment of suitable sites, periods, catch efforts, and the production of commercial hatchery post-larvae (Páez-Osuna 2001b; World Bank 2006), may minimize the interference of seed/broodstock harvest in the life cycle of wild species and potential adverse effects on the ecosystems food-webs. These measures should be accompanied by alternatives to minimize the social-economic effects of the reduction of wild seed collection in traditional aquaculture systems and in particular to low livelihood farmers.

4.1.3 Impacts of introduction of new species or genetic varieties

Some of the negative environmental impacts associated with the introduction of new species and new genetic varieties, including the loss of ecosystem integrity and genetic diversity, may be avoided or substantially mitigated through the effective implementation of the existing Codes of Practice and guidelines on this issue. Risk assessment and the application of preventive measures to species introductions (World Bank 2006), namely quarantine systems and cooperation between neighbouring countries before introducing non-native species into transboundary aquatic ecosystems, may also contribute to a responsible use of these species for aquaculture purposes (World Bank 2006). These limitations may easily be overcome by the use of RAS because farmed species are physically contained in these systems, eliminating the risk of escapes (Black 2001; Gutierrez-Wing and Malone 2006).

4.1.4 Degradation of genetic diversity

Another option to minimize the potential loss of genetic diversity due to the interaction of farmed and wild species is to ensure that escapees cannot breed. This is done successfully with rainbow trout by sterilising the females through the induction of a chromosomal abnormality called triploidy (SECRU 2002). Additional preventive measures proposed for cage aquaculture include the improvement of cage design, anchoring, net management, regulation of near-farm operations, deployment of fish cages at a safe distance from wild populations and the development of contingency plans in case of escapes, including for example the capture of escapees identified by genetic markers or tags (Pearson and Black 2001; SECRU 2002). Current methods to reduce Atlantic salmon escapes from cage farms also include the reduction of net damage from predators by using acoustic deterrents (SECRU 2002), however these method may negatively affect and even exclude marine species with high sensitivity to underwater acoustic noise, such as whales and dolphins (SECRU 2002). In restocking programmes to rebuild endangered species or depleted stocks, the utilisation of juveniles with minimal genetic divergence from their wild counterparts may minimize the loss of the species genetic pool (World Bank 2006). This can be achieved for example by using of a large number of breeders and genetic markers (World Bank 2006).

4.1.5 Modification and/or destruction of habitats

The problem of the destruction and/or modification of ecosystem structure, function and services by aquaculture activities may be generally solved by effective EIA. In the case of existing aquacultures, specific mitigation measures including the creation of buffer zones may also prevent or minimize the impacts of aquaculture operations on natural habitats (Choo 2001; Páez-Osuna 2001b). For shrimp aquaculture it has been also suggested that the use of abandoned ponds to restore mangrove systems and halophyte crop, or the conversion of shrimp ponds into salt ponds or for cultivation of other species (e.g. shellfish and crabs) (Páez-Osuna 2001b; Primavera 2006), may not only turn into an ecological benefit but also into an economic benefit.

4.2 Ecoaquaculture

Integrated aquaculture systems, either polyculture (e.g. fish and mussels, fish and seaweeds) or integrated aquaculture-agriculture systems (e.g. rice – fish farming), has also been considered an efficient and environmentally sound strategy for recycling aquaculture wastes (Buschmann et al. 1996; Pearson and Black 2001; Choo 2001; Gräslund and Bengtsson 2001; Páez-Osuna 2001a; Primavera 2006). Examples of the efficiency of these systems can be found worldwide. For instance, filter-feeders (e.g. oysters, mussels) and economically important seaweeds (e.g.

Gracilaria, kelp) cultured in the immediacy of finfish cages were proven to remove a significant part of the suspended organic matter and dissolved nutrients generated by cage aquaculture, alleviating waste loadings at the same time that it increase the farm productivity (Pearson and Black 2001). Polyculture with shellfish is particularly viable in eutrophic systems because these organisms can significantly reduce algal densities and nutrients loadings (Pearson and Black 2001), in a way that minimize the risks of eutrophication (cf. – Section 2.2). Coupling shrimp culture with bivalve molluscs and fish has also been considered (Sandifer and Hopkins 1996) a promising methodology to reduce the negative environmental effects resulting from the intensification of shrimp farming (Gräslund and Bengtsson 2001; Páez-Osuna 2001a; Biao et al. 2004; Primavera 2006). Another example of polyculture is the combined culture of the Chinese and Indian major carps in China, which has the added value that aquaculture wastes can be converted into agricultural wastes (World Bank 2006). Integrated aquaculture-agriculture practices are considered as an ecotechnology, particularly for inland aquaculture. For example, in Vietnam, the use of effluents from hybrid catfish aquaculture on rice farming was able to reduce 32% of total nitrogen and 24% of total P loadings (Lin and Yi 2003). Low-salinity effluents from inland shrimp farming were also used to irrigate melon crops in Brazil, and proved to be an efficient method for minimising the impacts of effluent discharges (Miranda et al. 2008). Integrated aquaculture-agriculture may also be used to remove nutrients from pond sediments (Lin and Yi 2003). According to these authors the use of rooted aquatic plants, such as lotus (*Nelumbo mucifera*) in semi-intensive cultures of tilapia (*Nile tilapia*) may remove up to 300 kg N and 43 kg P/ha/year. Besides its widely proven efficiency in removing aquaculture wastes, integrated aquaculture systems, may also reduce the risks of chemical contamination (Gifford et al. 2004; Primavera 2006). For instance, as aquaculture effluents naturally improve the fertilization of agriculture fields they reduce the use of environmentally damaging agriculture chemicals (e.g. pesticides, fertilizers), helping farmers to improve protein production and to ensure the economic viability of the activity (Lin and Yi 2003). Polyculture done with bivalves, that filter large volumes of water, may significantly lower the quantity of toxic contaminants released into the environment, acting as bioremediators of stressed coastal environments (Gifford et al. 2004). However, if human-consumed bivalves are involved carefully should be taken to avoid chemical and bacterial contamination (Gifford et al. 2004).

Another alternative to limit the impacts of effluents from pond aquaculture is the improvement of pond design. For example, ponds that are too shallow might be invaded by macrophytes, whereas in deeper ponds, the water may stratify, causing severe water quality problems, such as oxygen depletion (Boyd 1995a). The creation of buffer ponds (e.g. constructed wetlands) has also been proposed as a remediation measure for shrimp farming since it promotes the sedimentation of organic matter and the removal of other pollutants associated with suspended solids before the water is released into the surrounding environment (Boyd and Clay 1998; Kautsky et al. 2000; Páez-Osuna 2001a; Primavera 2006). An example from the Red Sea, considered as the third-generation of shrimp farms, consists of circular ponds with central drainage, in which more than 50% of the water surface (including upstream buffer ponds and wastewater treatment ponds) is dedicated to water quality control (Páez-Osuna 2001b). Reduction or elimination of water exchange rates between shrimp ponds and the adjacent water bodies has also been proposed to minimize the adverse effects of effluents discharge (Kongkeo 1997; Boyd and Clay 1998; Páez-Osuna 2001a; Primavera 2006). Restricted water exchange rates will not only lower the risk for sudden changes in water quality parameters, as may minimize the risks of water contamination by saltwater intrusion because it reduces the needs for groundwater. Other measures to reduce or even avoid saltwater intrusion include the utilisation of pond liners and of pond effluents to grow terrestrial halophytes in conjunction with natural filters such as mangroves (Páez-Osuna 2001a; Primavera 2006).

Recirculating Aquaculture Systems (RAS) may also be considered as an ecotechnology. The use of these systems has proven to reduce the amount of effluents by a factor of 500-1000 (Chen et al. 1997; Timmons et al. 2001), mainly because more than 90% of the water is recycled within the system (Black 2001). Even though the use of RAS does not always result in the overall reduction of discharges but rather on a relocation of wastes (Piedrahita 2003), these systems may facilitate effluent treatment, and thereby minimize potential negative impacts on the environment. Besides requiring fewer water resources, RAS allow a better control over waste discharges and diseases and may prevent the loss of genetic biodiversity (Black 2001; Piedrahita 2003; Gutierrez-Wing and Malone 2006). Because there is no possibility of interactions with wild stocks, this technology also allows the diversification and domestication of farmed species (Black 2001; SECRU 2002; Gutierrez-Wing and Malone 2006) and the intensification of aquaculture operations without seriously damaging the environment, and may contribute to an increase in the productivity and profitability of the aquaculture industry (Black 2001). On the other hand, the use of this technology may have significant economic drawbacks mainly related to the high capital expenditure and running costs (e.g. energy and maintenance) that it involves and due to the increased risk of failure if the systems are not adapted (in terms of biological and engineering concepts) to the species requirements (Black 2001).

As aquaculture grows, it extends its demands on environmental resources, making it urgent to develop new regulations that ensure the transition of the sector to more responsible and environmentally friendly practices. Sound policies, regulatory frameworks, codes of practice and BMPs, including EIAs, physical planning, and economic instruments (World Bank 2006), are among the tools that can be used to reduce the ecological footprint of aquaculture operations and to ensure the sustainability of this activity. Since a substantial component in this footprint is related to wastes production and to the use of fish meal/oils for the production of pelleted diets, the improvement of diet formulations is fundamental for the minimisation of the aquaculture environmental impacts. The development of ecofeeds rely largely on a vast understanding of the nutritional physiology and biochemistry of the different cultivated species (World Bank 2006), from which results a selection of very digestible ingredients that facilitate nutrient assimilation and promote the increase of FCRs. High FCRs have been shown to maximise protein retention and minimize the amount of solid wastes and nutrient loadings resulting from undigested, un-utilized and uneaten feeds (Black 2001; World Bank 2006). One of the current lines of investigations on ecofeeds consists for example, in the substitution of fishmeal protein from “trash fish” by a vegetable protein source (e.g. soya), in order to reduce the pressure on natural fisheries resources (Kaushik et al. 2004). However, vegetable substitutes often lack essential amino acids and fatty acids, which may constitute an impediment for the economic viability of aquaculture systems. Another constraint is the increasing consumer pressure so that these vegetable ingredients are GMO-free, i.e., not produced from genetically modified organisms (SEACASE 2009).

Given the necessity to ensure the safety of aquaculture products and the increasing consumers demand on food safety and welfare, the adoption of the environmentally friendly practices mentioned above becomes fundamental. The development of certification and ecolabeling schemes, attesting the character of the production processes and the quality of the products, may be an easy and efficient way to achieve the consumer perception and a mean to fulfil the market requirements and of adding value to aquaculture products (WorldBank 2006; SEACASE 2009).

4.3 Drivers, pressures, states, impacts and responses (DPSIR)

DPSIR is a causal framework for integrated environmental assessment, describing the interactions between society and the environment (UNEP/RIVM 1994, RIVM 1995). According to this framework, there is a chain of causal links with the following components: Driving forces, Pressures, States, Impacts and Responses. A Driving force results from a need, leading to activities that cause Pressures, affecting the state of the environment, causing Impacts that demand Responses from the society.

Table 2.3 is a possible example of an application of DPSIR to aquaculture development. It is important to have indicators to quantify each of the five DPSIR components, whenever possible. These indicators may be spatially resolved and integrated in a GIS. Suggested indicators for the example given in Table 2.3 could be: Driver - area allocated for fish farms; Pressure – Fluxes of nutrients, organic matter and xenobiotics, and differences in drag related to the presence of aquaculture leases; State – Concentrations in the water and in the organisms (regarding xenobiotics); Impact – changes in described rates; Response – seaweed production, area of sediments where pumping takes place, proportion of leases reallocated and changes in fish density within the farms, respectively.

Implementing the DPSIR framework may be useful to synthesize those indicators that should be included in a GIS for physical CC assessment, as well as those aspects that should be accounted for in CC models, including scenarios to be analyzed (cf. – Section 3.3). This framework may be used in more complex situations, when there are more drivers besides aquaculture.

Table 2.3 – Drivers, Pressures, States, Impacts and Responses for a hypothetical aquaculture development.

Driver	Pressure	State	Impact	Response
Fish farming	Increased nutrient fluxes	Increased nutrient and organic matter concentrations	Increased phytoplankton biomass/ eutrophication	Seaweeds production to remove excess nutrients
	Increased organic matter fluxes and oxygen	Decreased oxygen levels	Higher mortality of benthic organisms/ decreased benthic diversity	Bottom aeration
		Accumulation of organic matter in the sediments		
	Increased drag forces	Reduced flow-through and increased residence time	Increased sediment deposition	Reallocation to areas of more intense hydrodynamics
	Release of xenobiotics	Bioconcentration	Increased mortality of non-target species	Less intensive farming to reduce disease propagation

4.4 Decision support systems

A DSS is an information system that may bring together databases, models and other information sources to help the decision-making process. Considering the multiple interactions between aquaculture systems and other uses of natural resources, an important point about any DSS is to define for whom it is intended. Different actors and stakeholders are important in the decision process. This is well in line with the Integrated Water Resource Management (IWRM) concept, where a balance is to be found between economic and environmental objectives, and where public participation is a key issue (Agnetis et al. 2006).

A DSS should allow stakeholders and decision makers to analyse different aquaculture scenarios using geographic and socio-economic data, and model results. These data should reflect best knowledge about several aspects of CC, discussed before (cf. – Section 3.1). The DSS should include a methodology to evaluate those scenarios on some quantitative way towards an informed final decision (Agnetis et al. 2006; Pereira et al. 2007).

For example, let's assume that several scenarios were purposed regarding increasing the number of fish cages in a particular ecosystem. After conducting a DPSIR analysis - Drivers, pressures, states, impacts and responses (DPSIR) with stakeholders, decision-makers and scientists, potential shortcomings could be identified and used to define the responses needed from scenario analysis. Afterwards, an ecological model of the system under study could predict that increasing fish cages would increase fish production by a certain amount and decrease water quality (for example, though increases in ammonia concentrations and decreases in oxygen levels). An economic assessment of yields could reveal that the aquaculture income was not linearly related to fish production if market prices were not elastic. Therefore, at the end of the simulation process, several results regarding water quality, fish production and economic gains would have to be somehow weighted and compared. This could be done using the Analytic Hierarchical Process (AHP) methodology (Saaty 1980) as suggested in Agnetis et al. (2006) and obtaining a score for each scenario. This methodology allows for some subjectivity to be incorporated in the decision process, as a result of different sensitivities of stakeholders to environmental, economic and social aspects. For some examples see Agnetis et al. (2006), where this general approach was applied to several management scenarios (including aquaculture) for five coastal lagoons across southern Europe.

5 Concluding Remarks

Considering all the aspects discussed in the previous sections, some conclusions may be synthesized as follows:

(i) Aquaculture management should be participated by relevant stakeholders and viewed within the context of management plans, including other activities with which it may have positive and negative synergies;

(ii) Ideally, an ecosystemic approach in line with Ecological Engineering should be developed towards an "ecological aquaculture" to prevent going through the same mistakes as industrial agriculture and husbandry;

(iii) Low trophic level species should be preferred for a higher energy efficiency and low ecological footprint;

(iv) The Carrying Capacity concept is central to aquaculture sustainability in all its environmental, economic and social dimensions;

(v) There are several tools that may and should be used in aquaculture management and that have already widely been tested, such as GIS, the DPSIR framework, mathematical models and DSS.

Chapter 3

Physical, chemical and biological processes in semi-intensive fishponds

Chapter 3.1

Geochemical changes in white seabream (*Diplodus sargus*) earth ponds during a production cycle

---

Serpa D., Falcão M., Pousão-Ferreira P., Vicente M., Carvalho S. (2007). Geochemical changes in white seabream (*Diplodus sargus*) earth ponds during a production cycle. Aquaculture Research 38 (15), 1619-1626.

Abstract

The knowledge of geochemical processes in fishponds is important in defining farming strategies and the carrying capacity of these systems, and therefore essential for the management and sustainability of semi-intensive aquaculture in earth ponds. The main purpose of the present work, developed in the Aquaculture Research Station located in Ria Formosa, was to study the geochemical changes in semi-intensive earth ponds of white seabream *Diplodus sargus* L. during a production cycle, and relate it to farming conditions (fish biomass and feeding rate). Settled material and sediment samples were collected in a fish production pond and in a non-fish production pond during two years. Results obtained showed that particle-settling rates (S, g m^{-2} d^{-1}) increased linearly with time (t, days): $S = 0.7\ t - 34$, in the fishpond. Increasing deposition of particulate material increased the organic matter content of bottom sediments, particularly during the second production year. Organic matter mineralization, during periods of high temperatures, led to high nutrient concentrations in porewater (NH_4^+ – 965 µM; NO_3^- - 40 µM; HPO_4^{2-} - 39 µM) and subsequently to an increase in benthic primary production in the fishpond. The geochemical similarities between fishpond sediments and shallow coastal system’s sediments, along with the high fish survival rate (94%), suggests that for the assayed farming conditions there were no environmental constraints within the pond. However, some impact on bottom sediments namely, increase of settled material, organic matter deposition, nutrients in porewater and microphytobenthos production, was evident above a fish biomass of 500 g m^{-3} and a feeding rate of 150 kg $month^{-1}$, indicating that pond environmental conditions should be carefully monitored from this point on.

1 Introduction

In Portugal and all around southern Europe, marine fish farming is traditionally semi-intensive and carried out in earth ponds. Due to the increasing demand of consumers on food safety and welfare of cultivated species, semi-intensive culture products are coming back in the front scene. Nevertheless, the production costs of this type of aquaculture are often too high to maintain a sustainable economic activity due to the low productivity of these systems, which rely largely on traditional practices and are often managed on an intuitive basis (Giovannini and Piedrahita 1994). A better understanding of the earth pond environment is necessary to develop management practices that optimise the use of aquatic resources (Piedrahita 1988; Culberson and Piedrahita 1996).

Aquaculture ponds are complex systems. Survival and growth of fishes in ponds are determined by the physical and chemical characteristics of water (Piedrahita 1988). Water quality, in turn, is affected by external inputs, by the organisms present and by biogeochemical processes occurring in ponds. Studies on ponds geochemistry are emerging as an important area for fishpond management as sediments play an important role both as a source of various dissolved compounds and a sink for particulate material (Lefebvre et al. 2001).

Organic sediment enrichment caused mainly by settlement of plankton, uneaten food and fish waste products, induces changes on the biological and chemical environment of fishponds (Krom et al. 1985a; Hargreaves 1998; Jamu and Piedrahita 2001; Boyd et al. 2002). From a management perspective, accumulation of organic matter is undesirable, as it may accumulate to levels that can affect water quality and consequently fish yields, due to the release of toxic substances such as hydrogen sulphides, free ammonia and nitrites, which are toxic to fish even at low concentrations (Meade 1985). High organic matter deposition may also cause a high oxygen demand and lead to oxygen depletion (Boyd 1995b), which in turn affects fish production.

Since geochemical processes in earth ponds, depend mostly on fish biomass, food and feeding strategies, water temperature variations, water circulation and water depth, studies on this subject should be planned with a view toward the development of practical management procedures that enhance fish growth and production (Hargreaves 1998). The purpose of the present work was to study the geochemical changes in semi-intensive earth ponds of white seabream (*Diplodus* sargus L.) during a production cycle, in order to relate the environmental changes in fishponds bottom with farming conditions (fish biomass and feeding rate). This information may be useful for defining farming strategies and protocols to improve management of semi-intensive white seabream earth ponds, which are often managed intuitively, and consequently to optimise the production of this species that has been considered a potential candidate for Mediterranean aquaculture in terms of market preferences, economic value and flesh quality (Ozorio et al. 2006).

2 Material and Methods

2.1 Experimental system

The present study was developed in the Aquaculture Research Station (ARS), located in the Ria Formosa Natural Park, Southeast of Portugal (Figure 3.1.1). The experiment was carried out in two rectangular earth ponds, with an average surface area of 475 m^2 and a volume of 700 m^3. Experimental ponds were supplied with seawater pumped from a reservoir that fills up according to the lagoon tidal cycle. In May 2003, one of the ponds (Pf) was stocked with 3000 juveniles of white seabream while the other was left without fish (Pnf). The water flow rate varied from 25 to 100 $m^3 h^{-1}$ in Pf; whereas in Pnf, water flow rates were lower (4 to 10 $m^3 h^{-1}$). The fishpond was equipped with aerators (FORCE-7; 1.5 hp) to control dissolved oxygen levels that varied between 6.3 and 9.6 mg L^{-1}. Fish were fed a commercial food pellet (DOURASOJA®, manufactured by Sorgal) containing 51% of total protein, 29% fat and 1.2% total P, by a combination of automatic and manual feeding. Monthly ration varied throughout the experiment (Figure 3.1.2), according to fish biomass and feeding response.

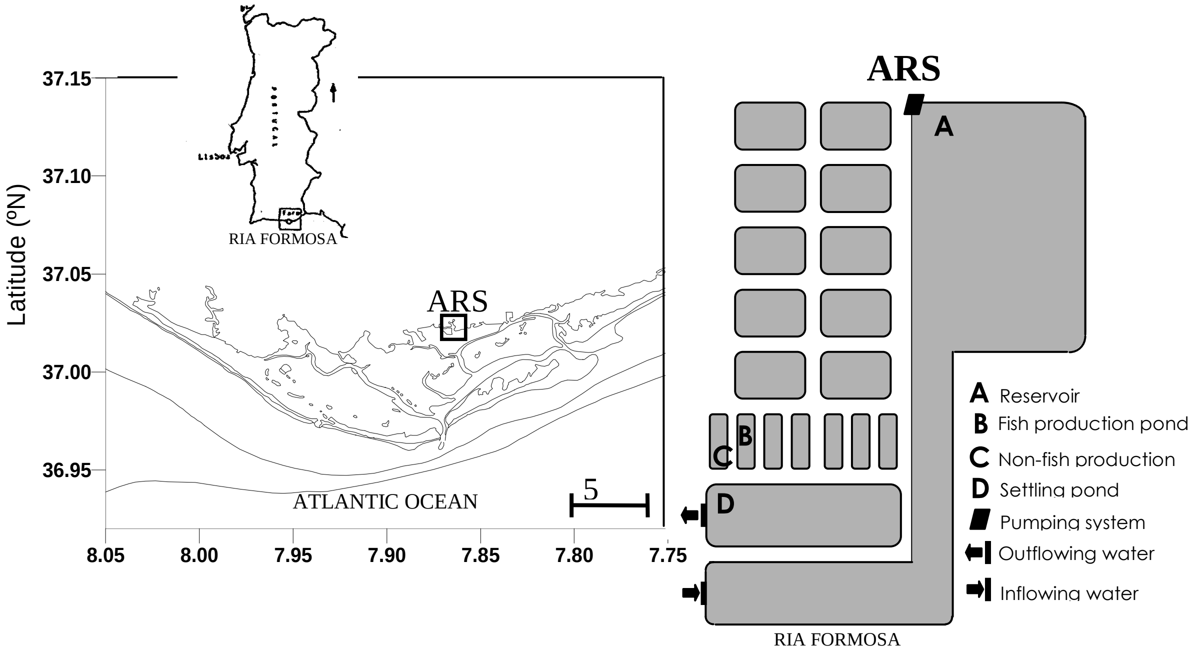


Figure 3.1.1 – Location of the Aquaculture Research Station (ARS).

2.2 Field sampling

Sampling was carried out in Pf and Pnf in June, July, August and November 2003; March, June and October 2004 and March 2005.

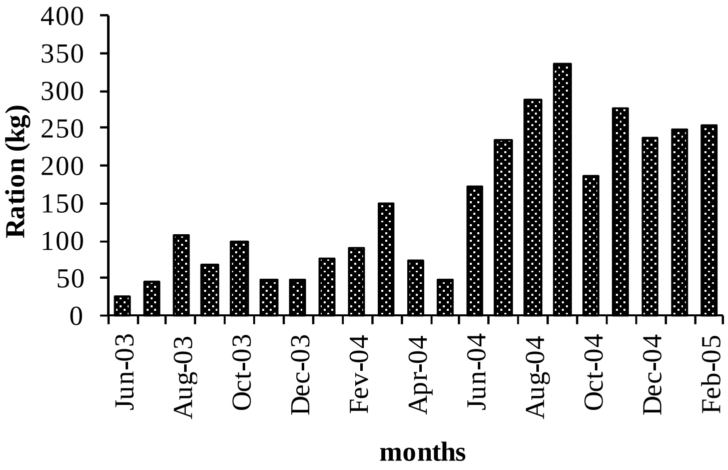


Figure 3.1.2 – Monthly ration (kg) supplied to white seabream during the production cycle.

2.2.1 Sediment-traps settled particles

A sediment-trap with 6 PVC tubes (5 cm diameter and 50 cm length) was placed in both ponds. The tubes containing particulate suspended material remained two weeks under water at each sampling period. The traps were allowed to settle and dripped out slowly, and then transported to the laboratory with minimal disturbance. Each tube was placed to dry in a stove at 70ºC.

2.2.2 Sediment cores

In each sampling, nine sediment cores were randomly collected by a diver in Pf and Pnf. Cores (15 cm length and 5 cm diameter) were collected by gently pushing the open-ended PVC tube into the sediment, to preserve sediment layers. Samples were transported to the laboratory under refrigerated conditions, to be sliced in 2 cm layers for porewater and solid fraction analysis.

2.3 Analytical methodology

2.3.1 Porewater samples

Sediment samples were centrifuged for 10 minutes at 3000 r.p.m. (1600 *g*) to separate porewater from the solid fraction and filtered with 0.45 µm Macherey-Nagel filters. Porewater samples were analysed for ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), silicon ($Si(OH)_4$) and phosphate (HPO_4^{2-}) using a "Skalar" autoanalyser according to the following methodology: NH_4^+ was determined by indophenol's blue colour at 630 nm (detection limit - 0.2 µM); NO_3^- was reduced to NO_2^-, by passing the sample through a cadmium column and afterwards by detection of the coloured complex formed with the ethylenediamine at 550 nm (detection limit - 0.1 µM). Silicon was determined using ascorbic acid as the reducer (detection limit - 0.5 µM), and HPO_4^{2-} with a blue antimony complex (detection limit - 0.08 µM). Dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) were oxidized by potassium peroxodisulphate ($K_2S_2O_8$) in autoclave (1.5 bar; 120º C) as described in Grasshoff (1983) and determined in the autoanalyser (detection limits - 0.1 µM and 0.08 µM, respectively). Chlorophyll *a* and phaeopigments (Phaeop) were extracted with acetone (90 %) and determined by fluorimetry according to Parsons et al. (1984).

2.3.2 Solid fraction

Particles collected in the sediment-trap and upper sediment layer (0-2 cm) were dried at 80ºC until a constant weight and ground to a fine powder for determining total organic carbon (TOC) and nitrogen (TON). Total and inorganic carbon and nitrogen were determined using a CNH analyser "NC 2500 CE instruments" with acetanilide as reference material (Byers et al. 1978), and organic carbon and nitrogen determined by difference between total and inorganic fractions. Total phosphorus was determined by digestion of dry sediment samples with HCl (1 N) during

20 minutes at 200ºC according to Andersen (1976). Phosphorus sorbed to litogenic particles was sequentially extracted: phosphorus weakly bound (wb-P), 1 hour stirring with NH_4Cl (1 N); phosphorus bound to calcium (Ca-bound P), 1 hour stirring with HCl (0.5 N) and phosphorus bound to iron (Fe-bound P), 17 hours stirring with NaOH (0.1 N) according to the methodologies of Chang and Jackson (1957) and, Hosomi and Sudo (1982). Organic phosphorus was determined by difference between the total and inorganic fraction. Chlorophyll *a* (Chl *a*) and phaeopigments (Phaeop) were extracted with acetone (90%) from the upper sediment layer and determined by fluorimetry according to Parsons et al. (1984).

2.4 Fish sampling

To evaluate white seabream condition, in each sampling, approximately 200 specimens were caught with a beach seine net to be measured and weighted under light anaesthesia (0.15 ml L^{-1} de 2-phenoxyethanol).

2.5 Data analysis

The variability of Pf and Pnf porewater samples was evaluated through a multivariate Correspondence Analysis (CA) using BRODGAR “Software for Univariate & Multivariate Analysis and Multivariate Time Series, Version 2.4.1”. Non-parametric Wilcoxon-Mann and Whitney tests were performed to the data in order to determine significant differences between Pf and Pnf (Zar 1999).

3 Results

3.1 Particulate material

Particles-settling rates were clearly higher in Pf than in Pnf, increasing from 26 ± 13 g m^{-2} d^{-1} in the first two weeks to 399 ± 14 g m^{-2} d^{-1} at the end of the production cycle (Figure 3.1.3). A significant linear relationship was found between particle-settling rates (S, g m^{-2} d^{-1}) and time (t, days): $S = 0.7t – 34$ ($R^2 = 0.88$; $n = 46$; $P<0.01$), whereas in Pnf, settling rates remained constant over time.

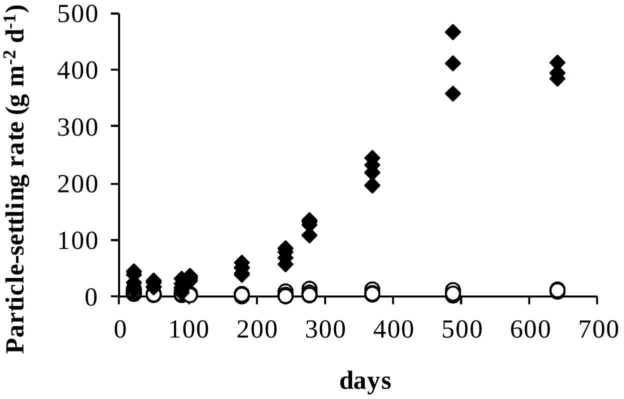


Figure 3.1.3 – Particle-settling rates (g m^{-2} d^{-1}) in Pf (♦) and Pnf (○), during the sampling period.

Based on the amount of particles deposited in sediment traps was estimated that about 90 kg d^{-1} of particulate matter is settled in the entire pond by the end of the first production year, doubling at the end of the production cycle, when the overall fish biomass reached 1 kg m^{-3} and feeding rate 275 kg $month^{-1}$.

Particle settling rates and organic content allowed the calculation of particulate organic carbon (POC), nitrogen (PON) and phosphorus (POP) deposition rates in Pf and Pnf (Table 3.1.1). Organic compounds settling rates were significantly higher in the fishpond (Wilcoxon Mann-Whitney test, *P*<0.01), increasing one order of magnitude after the first production year and two orders of magnitude at the end of the second year. In Pf, POC deposition rates ranged from 500 to 10500 mg m^{-2} d^{-1} during the production cycle, whereas PON and POP varied respectively between 40 to 2300 mg m^{-2} d^{-1} and 5 to 205 mg m^{-2} d^{-1}, corresponding to C: N and N: P ratios close to the Redfield ratio.

Table 3.1.1 - Deposition rates of particulate organic carbon (POC), particulate organic nitrogen (PON) and particulate organic phosphorus (POP) in the fish production pond (Pf) and in the non-fish production pond (Pnf), during the experiment.

Days	POC (mg m^{-2} d^{-1})		PON (mg m^{-2} d^{-1})		POP (mg m^{-2} d^{-1})	
	Pf	Pnf	Pf	Pnf	Pf	Pnf
19	608	350	39	17	6.2	0.19
48	601	232	56	29	5.2	0.12
89	525	226	56	6.0	0.06	0.62
101	729	295	62	9.6	0.11	0.37
177	961	84	74	7.3	1.9	0.80
241	1519	128	148	9.4	19	0.26
276	2413	192	189	16	39	1.3
368	8421	273	861	42	99	1.5
486	9368	230	1646	20	185	2.4
639	10500	256	2320	35	205	4.2

3.2 Sediment

Figure 3.1.4 illustrates the TOC, TOP, Fe-bound P and Chl *a* contents in superficial sediments of Pf and Pnf, during the experiment. Total organic carbon was clearly higher in the production pond throughout the experiment, whereas for TOP and Fe-bound P differences between ponds became more evident one year after the beginning of the experiment, reaching values 2 to 4 times higher than in Pnf. During the production cycle, sediments organic nitrogen content remained below the detection limit. Chlorophyll *a* concentrations were similar in both ponds during the first year. However after this period microphytobenthos production increased 2 to 9 times in Pf, being highly correlated to sediment organic carbon content ($R^2 = 0.62$; n = 27; $P<0.05$).

A Correspondence Analysis applied to physical (Temp), chemical (NH_4^+, NO_3^-, NO_2^-, HPO_4^{2-}, $Si(OH)_4$, DON, DOP) and biological (Chl *a* and Phaeop) data from porewater and superficial sediments of Pf and Pnf is represented in a two-dimensional space, defined by two ordination axis (Axis 1, Axis 2). Axis inertia was about 84% indicating that almost all data variability is explained by the analysis (Figure 3.1.5). Pf samples are well represented in the positive and negative Axis 1. The cluster close to the negative Axis 1 corresponds to parameters with strong affinity to higher temperatures (NH_4^+, NO_3^-, HPO_4^{2-}, Chl *a* and Phaeop) conversely, DON and DOP were well represented in the positive Axis 1 showing higher affinity to the months of lower temperatures. Pnf points (in positive Axis 2), showed high similarity and weak affinity for the analysed parameters whose concentrations were 1 to 2 orders of magnitude lower than in Pf. Besides the evident contrast between Pf and Pnf data points, the analysis also revealed a well-defined seasonal variability for Pf samples.

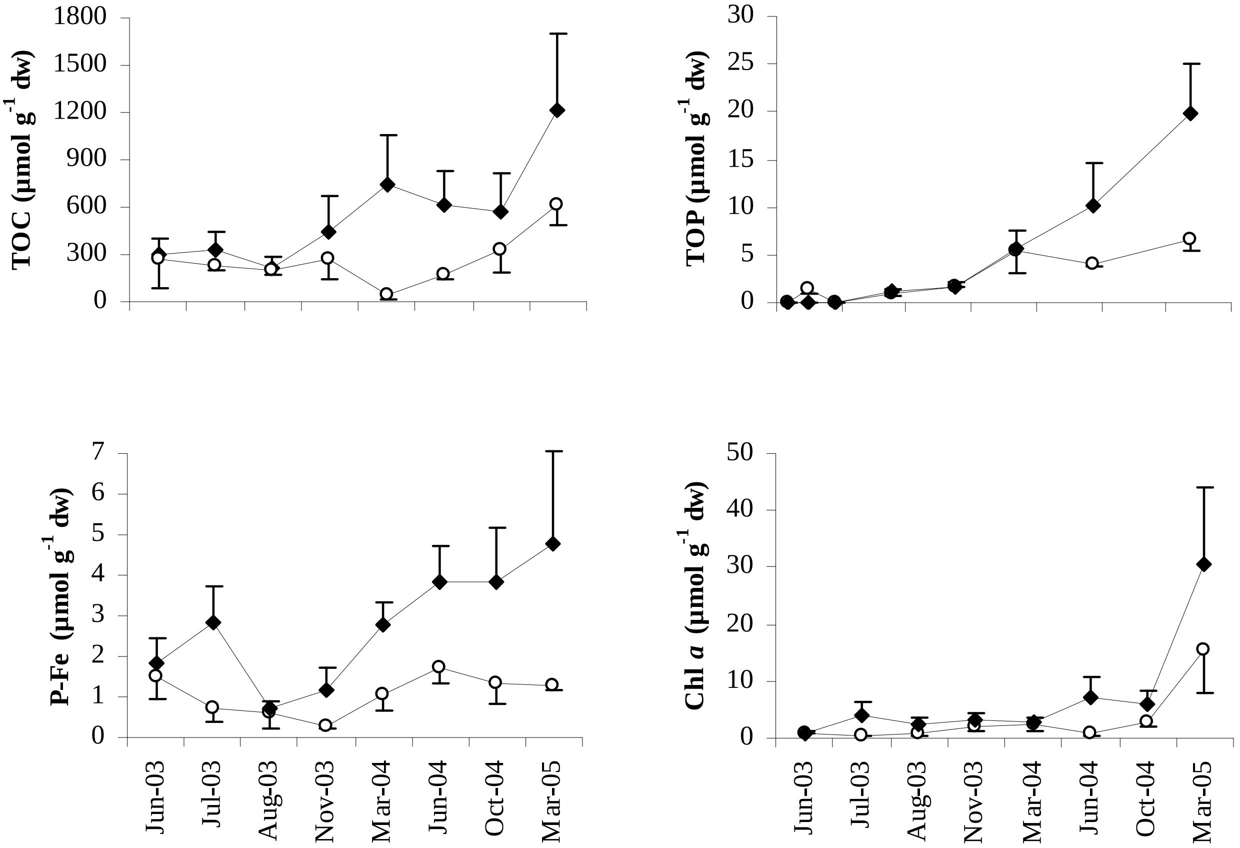


Figure 3.1.4 – Total organic carbon (TOC), total organic phosphorus (TOP), phosphorus bound-Fe (P-Fe) and chlorophyll *a* (Chl *a*) concentrations in superficial sediments (0-2 cm) of Pf (♦) and Pnf (○), during the sampling period.

3.3 Fish data

White seabream grew from 6.9 ± 1.2 to 24.9 ± 1.7 cm during the production cycle (Table 3.1.2). To this increase in length (L_t), corresponded an increase in body weight (W) according to the following equation: $W = 0.031*L_t^{2.86}$. The fish condition factor (k), determined according to the Ozorio et al. (2006) equation:

$$k = W*100/L_t^3$$

where W is the fish body weight (g) and L_t is the total fish length (cm), registered high values all over the production cycle (≥ 1.9) indicating the well being of cultivated fishes. During the experiment, white seabream mortality was less than 6%.

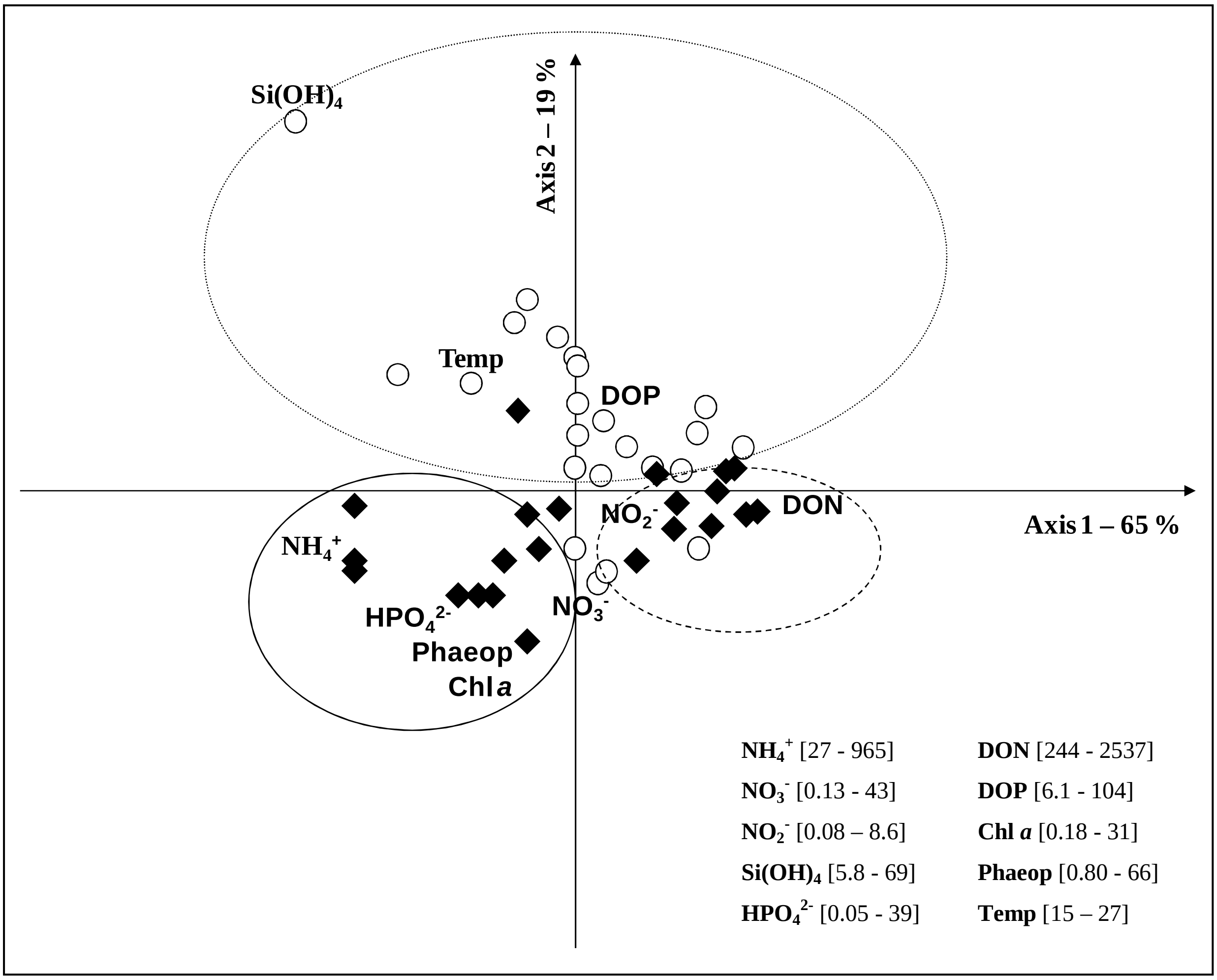


Figure 3.1.5 – Correspondence analysis ordination plot of physical (Temp – temperature, in ºC), chemical (NH_4^+ - ammonium; NO_3^- - nitrates; NO_2^- - nitrites; $Si(OH)_4$ – silicates; HPO_4^{2-} - phosphates; DON – dissolved organic nitrogen; DOP – dissolved organic phosphorus, in µM) and biological (Chl *a* – chlorophyll *a*; Phaeop – phaeopigments, in µg g^{-1}) parameters in porewater and superficial sediments of Pf (♦) and Pnf (O) during the sampling period. Parameters annual range in Pf.

Table 3.1.2 – Total length (cm), body weight (g) and condition factor (k) of white seabream throughout the production cycle.

Days	Total length (cm)	Body weight (g)	Condition factor (k)
1	6.9 ± 1.2	7.6 ± 1.3	2.3 ± 0.1
121	11.6 ± 2.1	60.0 ± 6.1	3.8 ± 0.2
211	16.5 ± 1.0	112.3 ± 16.2	2.4 ± 0.2
451	21.5 ± 1.5	179.6 ± 26.0	1.9 ± 0.1
651	24.9 ± 1.7	263.4 ± 51.0	1.7 ± 0.2

4 Discussion

In aquaculture systems where temperature favours fish growth throughout the year, organic matter loading is continuous (Steeby et al. 2004). Increasing fish size and feed input might explain increasing particle settling rates in fishponds (Hargreaves 1998; Avimelech 1999), as observed by the exponential relation of particle settling rates with fish biomass ($S = 18 * \exp^{0.003 * \text{biomass}}$; $R^2 = 0.94$; $P<0.01$), and feeding rate ($S = 17 * \exp^{0.01 * \text{feeding}}$; $R^2 = 0.74$; $P<0.05$), particularly above 500 g m^{-3} of fish biomass and 150 kg $month^{-1}$ of food. The settled material contributed largely to the organic enrichment of Pf sediments during the experiment, especially in biogenic material, since the C:N and N:P ratios of settled particles were close to the Redfield ratio.

In aquatic systems, organic matter in sediments is the difference between the rate of *in situ* production plus allochtonous material that reaches the bottom and the rate of organic matter mineralization in the sediment (Haas et al. 2002, Jiménez-Montealegre et al. 2002b). The increasing organic loading mainly in the form of fish wastes, uneaten feed and senescent phytoplankton (Hargreaves 1998) determined changes in organic carbon, nitrogen and phosphorus content of bottom sediments. While fine particles were incorporated in the upper sediment layers during the production cycle, organic carbon increased exponentially (TOC = 0.23 exp $^{0.38* \text{ day}}$; $R^2 = 0.85$; n = 27; $P < 0.05$) whereas organic nitrogen bound in form of proteins and other compounds in plants and animal bodies was practically irrelevant in Pf, probably because these materials are degraded faster than carbon and phosphorus (Ryther and Dunstan 1971; Nixon and Pilson 1983; Dale and Prego 2002). The extremely low values of total organic phosphorus observed during the first experimental year may be explained by the release of P from organic material and its retention in sediment due to an early diagenetic “sink-switching” to lithogenic forms (Slomp 1997), mainly Fe-bound P. In the second year, the exponential increase of organic phosphorus in the white seabream pond was likely related to deposition rates greater than losses from decomposition (Steeby et al. 2004). Organically richer sediment may favour benthic remineralization and consequently enhance microphytobenthos production in top-layer sediment (Brotas 1990; Gutiérrez et al. 2000), which explains the significant relationship between organic carbon and chlorophyll *a* in Pf (TOC = -0.98 Chl a^2 + 60 Chl a + 284; R^2 = 0.83; n = 27; $P < 0.05$). The disparity between high concentrations of NH_4^+, NO_3^-, NO_2^-, HPO_4^{2-} and $Si(OH)_4$ and low concentrations of organic compounds in Pf porewater during periods of high temperature, suggests seasonality in mineralization processes probably due to an increase in microbial metabolism which contributes for intense nutrient production to porewater (Nowicki and Nixon 1985; van Raaphorst et al. 1992; Kristensen 1993; Asmus et al. 2000).

The geochemical similarities between fishpond sediments and Ria Formosa intertidal sediments (Falcão and Vale 1998; Falcão et al. 2006), associated with high fish survival and high fish condition factor values, suggests that for these experimental farming conditions there were no environmental constraints within the pond. However when fish biomass and feeding rate exceeded 500 g m^{-3} and 150 kg $month^{-1}$ respectively, impacts on bottom sediments became evident, as settled material, organic matter deposition, nutrients in porewater and microphytobenthos production increased substantially, hence bottom sediments and fish quality should be watchfully monitored from this point on. This study, which quantifies geochemical changes in white seabream ponds and relates it to farming conditions, may be a starting point to define acceptable ranges for bottom sediment parameters and, to optimise feeding rates and pond carrying capacity for a species whose production in aquaculture may be important due to its high commercial value (Ozorio et al. 2006).

Acknowledgments

This work has been supported by Project PROMAR (INTERREG III A) “Cooperação Algarve – Andaluzia para a Promoção de Recursos Aquícolas Marinhos no Litoral Atlântico Sul and Project “Tecnologias de Produção Aquícola” MARE (22-05-01-FBR-00014-QCAIII). The authors would also like to thank M. L. Inácio for her help in field sampling and laboratory analysis, and to Isabel Ferreira for helping in the fish sampling.

Chapter 3.2

Contribution of benthic nutrient mineralization and inflowing water to the nutrient availability in semi-intensive fish earth ponds: consequences for pond water quality

Serpa D., Falcão M., Caetano M., Pousão-Ferreira P., Carvalho S., Duarte P., Cancela da Fonseca L., Dinis M. T. Contribution of benthic nutrient mineralization and inflowing water to the nutrient availability in semi-intensive fish earth ponds: consequences for pond water quality. *Submitted to Aquaculture International*.

Abstract

Water quality is critical for pond management not only because it strongly influences fish welfare and growth, but also because it defines the quality of effluent waters discharged back into the environment. In semi-intensive production systems, water quality is often dependent on the state of the sediments as well on the quality of source water. The main objective of the present study was to assess the contribution of benthic mineralization and inflowing water to the nutrient availability in a white seabream (*Diplodus sargus*) production pond. Experiments were carried out in a fishpond and in a control pond (without fish), from June 2003 to March 2005. Benthic fluxes (diffusive and biologically-mediated fluxes) of inorganic and organic nitrogen and phosphorus compounds were estimated. Diffusive fluxes were calculated according to Fick's 1st Law, using concentration gradients between porewater and overlying water measured in the field, whereas biologically-mediated fluxes were determined in laboratory incubation experiments. Estimates of inflowing water's nitrogen and phosphorus inputs to the system were based on the concentration of nutrients in the source water and on water exchange rates. Results showed that the input of labile organic matter (e.g. fish feces and feed wastes) over the 2-year production cycle led to maximum ammonium and phosphate diffusive fluxes (respectively, 150 ± 60 and 1.4 ± 0.5 nmol cm^{-2} d^{-1}) at the end of the trial. Benthic fauna considerably enhanced solute transport, since biologically-mediated fluxes were one order of magnitude higher than diffusive fluxes. Using data on benthic fluxes, inflowing water composition and other nutrient sources and sinks, a nutrient budget was constructed for the fishpond. The budget revealed that inflowing water was the major source of nutrients to the pond, accounting for 52% of the nitrogen supplied and 70% of the available phosphorus, while sediments accounted for respectively, 46% and 29% of the daily nitrogen and phosphorus inputs. This study provided

clear evidence that an efficient pond management strongly rely on sediment treatment between production cycles and on the optimization of water exchange rates.

1 Introduction

The rain of organic particles, resulting from uneaten feed, plankton sedimentation and fish metabolic wastes influence early diagenesis in earth ponds (Krom et al. 1995; Hargreaves 1998; Holmer et al. 2002). Settled organic matter is mineralised, generating nutrients (Hall et al. 1992; Mesnage et al. 2007) that are mobilized to the water column or downward to the deeper sediment layers (Lerat et al. 1990; Falcão and Vale 1998; Rao and Jahnke 2004; Serpa et al. 2007a).

The dominating transport mechanisms contributing to solute exchange between sediment and water column are diffusion and bioturbation (Berner 1980; Helder and Andersen 1987; Rao and Jahnke 2004; Nizzoli et al. 2007; Holmer and Heilskov 2008). Diffusive fluxes depend mainly on concentration gradients between porewater and overlying water, sediment characteristics and microbial activity (Berner 1980; Anschutz et al. 2000; Graca et al. 2006; Valdemarsen et al. 2009), whereas biologically-mediated fluxes are related to distinct and specific activity patterns of benthic organisms (Sandnes et al. 2000; Holmer and Heilskov 2008). As a consequence of sediment reworking by benthic fauna an intense mineralization of dissolved and particulate compounds is usually found (Aller and Aller 1992; Nizzoli et al. 2007) with enhanced benthic nutrient fluxes (Aller and Aller 1992; Sandnes et al. 2000; Falcão and Vale 2003; Holmer and Heilskov 2008).

Aside from sediment quality, water column nutrient concentrations in land-based aquaculture systems are substantially influenced by the characteristics of source water (Krom et al. 1985a;

Neori et al. 1989; Krom et al. 1995; Brambilla et al. 2007). In semi-intensive earth ponds, the water quality status is maintained by frequent water exchange, which varies as a function of stocking densities and water temperature. Thus, inflowing water plays a major role on the quality of fishpond water (Hopkins et al. 1993). Moreover, pond water is often discharged back into the source environment with little or no treatment, having negative consequences for the adjacent aquatic systems (Hopkins et al. 1993; Brambilla et al. 2007).

Quantifying the contribution of the major sources and sinks of nutrients to the water composition of semi-intensive systems is extremely important for pond management, because it helps to define strategies that ensure optimal water quality within the production ponds and reduce environmental impacts (Alongi et al. 2000; Papathyphon et al. 2005; Casillas-Hernández et al. 2006). The purpose of the present work was to estimate the contribution of benthic nutrient mineralization and inflowing water to dissolved nitrogen and phosphorus availability in a white seabream (*Diplodus sargus*) production pond. Being a new species in Mediterranean aquaculture (Golomazou et al. 2006; Pérez et al. 2007; Sá et al. 2007, 2008; Serpa et al. 2007b), additional research is needed for the optimization of white seabream production in semi-intensive ponds, namely on nutrient management. In order to achieve our goal, field and laboratory experiments were performed to estimate benthic fluxes (diffusive and biologically-mediated fluxes) and source water nutrient inputs. Diffusive fluxes were estimated according to the Fick's 1st Law (Anschutz 2000; Graca et al. 2006), from porewater and overlying water nutrient data collected during the 2-year production cycle of white seabream; biologically-mediated fluxes were determined in laboratory incubation experiments; and source water inputs were calculated based on the chemical composition of inflowing water and on water exchange rates. Additionally, a nutrient budget was built up for assessing the relative contribution of benthic mineralization, source water and other nutrient sources and sinks, for the N and P availability in fishpond water.

2 Material and Methods

2.1 Description of the system

The present study was developed at the IPIMAR's Aquaculture Research Center, located in a coastal lagoon (Ria Formosa), at Southeast Portugal (Figure 3.2.1). Experiments were carried out in two rectangular earth ponds, with an average surface area of 475 m^2 and a volume of 700 m^3. In May 2003, one of the ponds (Pf) was stocked with 8 g juveniles of white seabream at 6.7 fish per m^2, while the other (Pnf) was left without fish. Seawater was supplied to the fishpond (Pf) at rates varying from 25 to 100 $m^3 h^{-1}$, which corresponds to a 90 to 300% daily water exchange, whereas in the control pond (Pnf) water exchange rates were lower, varying from 4 to 10 $m^3 h^{-1}$. The fishpond was equipped with aerators (FORCE-7; 1.5 hp) in order to maintain dissolved oxygen above critical levels for fish survival (range: 6.3 to 9.6 mg L^{-1}). Fish were fed daily with a commercial feed pellet containing 51% of total protein, 29% fat and 1.2% total P, at 1.2% body wet weight per day in the first production year, and 0.8% in the second year. Monthly ration varied throughout the experiment, between 25 and 350 kg $month^{-1}$, according to fish biomass and feeding response (Serpa et al. 2007b). The input of allochthonous organic matter promoted the establishment of significantly different macrobenthic assemblages in the two ponds (Carvalho et al. 2007; Carvalho et al. 2009). Polychaetes, and in particular the species *Capitella spp.* and *Pseudopolydora paucibranchiata*, accounted for more than 90% of benthic organisms abundance in Pf, whereas insects (Chironomidae) and bivalves (*Cerastoderma spp.*) were visibly dominant in Pnf (Carvalho et al. 2009).

2.2 Field sampling

Sampling was performed in Pf and Pnf during the 2-year period of white seabream production. In the first trimester of the experiment, sampling was intensified (June, July and August 2003) due to the recent pond construction (May 2003), but from then on, was carried out seasonally, in November 2003; March, June, October 2004 and March 2005. In each sampling period, 9 sediment cores (PVC tubes of 15 cm height and 5 cm diameter) and overlying water samples were collected in each pond to measure the concentration gradients between porewater and overlying water, and further determine diffusive fluxes under field conditions. Sediment sampling was carried out by carefully pushing the open-ended PVC tube into the sediment to minimize disturbance of sediment layers, and overlying water was collected 2 cm above the sediment surface with pre-cleaned syringes. In addition to earth pond sampling, inflowing water samples were collected to determine nutrient concentrations. Immediately after collection, samples were refrigerated and transported to the laboratory.

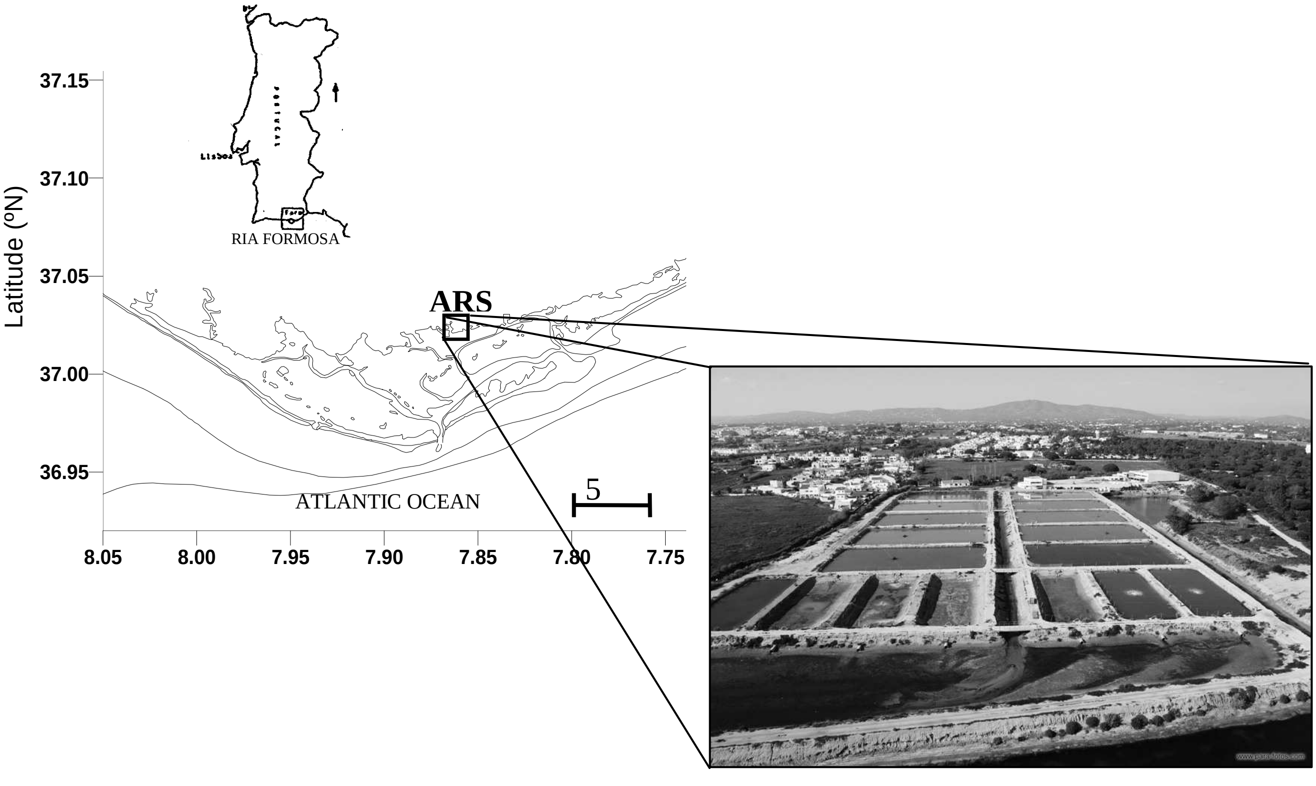


Figure 3.2.1 – Location of the Aquaculture Research Center.

2.3 Laboratory experiments

In June and November 2004, additional sediment samples were collected in Pf and Pnf, and immediately brought to the laboratory to perform incubation experiments, for evaluating biologically-mediated fluxes. The 3 cm upper sediment layer was sectioned and carefully transferred to 6 glass cubic (15 cm × 15 cm × 15 cm) incubation chambers (3 replicates per pond), to minimize disturbance of physical and chemical gradients as well as biological communities. Experiments were carried out at *in situ* temperatures (June-04: 21° C and November-04: 15°C) to test the influence of biological activity on solute transport. In each experiment, sediments were submerged in pond water up to ~10 cm height. All chambers were kept at constant temperature and aerated continuously to maintain *in situ* oxygen levels (> 6 mg L^{-1}), because these variables strongly influence sediment-water fluxes (Berner 1980; van der Loeff et al. 1984; Helder and Andersen 1987). After a stabilization period (4 h), overlying water samples were collected with a pre-cleaned syringe at regular time intervals (30 minutes during day time and 1 hour at night), for a 24 hour-period. Concomitantly, porewater samples were collected in the upper 2 cm sediment layer by a system of taps on the chambers.

2.4 Analytical procedures

Sediment samples collected during the production cycle were sliced into 2-cm layers and centrifuged for 10 minutes at 3000 r.p.m. (1600 *g*), to separate porewater. Prior to analysis, porewater and overlying water samples from both field and laboratory experiments were filtered with 0.45 µm polycarbonate filters. Samples were analysed for ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-) and phosphate (HPO_4^{2-}) using a “Skalar” autoanalyser according to the following methodology: NH_4^+ was determined by indophenol’s blue colour at 630 nm (detection limit - 0.2

μM); NO_3^- was reduced to NO_2^-, by passing the sample through a cadmium column and detection of the coloured complex formed with ethylenediamine at 550 nm (detection limit - 0.1 μM); and phosphate was determined with a blue antimony complex (detection limit - 0.08 μM). Dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) were oxidized by potassium peroxodisulphate ($K_2S_2O_8$) in autoclave (1.5 bar; 120º C) as described in Grasshoff (1983) and determined in the autoanalyser. Porosity was calculated from sediment weight loss after drying it at 105º C (Holmer et al. 2002).

2.5 Calculations

2.5.1 Diffusive fluxes

Earth pond sediments were mainly characterized by highly permeable sands (94%). As a result of easy water percolation, oxygen penetrated deeper into the sediments, allowing the use of a thicker depth resolution (2 cm) for the calculation of diffusive fluxes, in opposition to the narrower resolutions (mm) commonly used in fine grained-muddy sediments (Aller and Aller 1992; Falcão and Vale 1998; Serpa et al. 2007a). The diffusive fluxes (J_s) of each solute were calculated by applying the Fick's 1st Law of diffusion adapted to sediment conditions (Berner 1980; Burdige et al. 1992):

$$J_s = -\phi \times D_s \times (C_o - C_p) / \Delta x \tag{1}$$

where ϕ is sediment porosity (dimensionless), (C_o-C_p) is the concentration gradient (μM) between overlying water (C_o) and porewater (C_p), Δx is the distance across the interface where concentrations were measured (2 cm), and D_s is the whole sediment diffusion coefficient for each solute (cm^{-2} s^{-1}) corrected for tortuosity (eq. 2):

$$D_s = D_o / \left(1 - \ln \phi^2\right) \tag{2}$$

in which D_o is the solute diffusion coefficient in seawater (Boudreau 1997). D_o values of inorganic compounds at different temperatures were extracted from Schulz (2000), whereas those of organic compounds (DON and DOP) were calculated from an empirical relation between molecular weight and D_o given by Burdige et al. (1992) and adjusted to in situ temperatures using the Stokes-Einstein equation (Li and Gregory 1974). According to eq. 1, positive diffusive fluxes indicate that nutrients are transferred from sediments to the overlying water while negative fluxes mean that nutrients diffuse into the sediments.

2.5.2 Biologically-mediated fluxes

Total nutrient fluxes to overlying water (F_{total}) were determined according to the following equation:

$$F_{total} = \Delta C_{ow} / \Delta t \times h \tag{3}$$

where ΔC_{ow} is the variation of overlying water nutrient concentrations (µM) over the time of the experiment (t, in minutes), and h is the height of the water column in the incubation chambers (10 cm). This flux reflects both the contribution of diffusion and bioturbation processes. Biologically-mediated fluxes (F_{bio}) were calculated from the difference between F_{total} and the diffusive fluxes (Js) determined in the same chamber, using the methodology described above:

$$F_{bio} = F_{total} - J_s \tag{4}$$

2.5.3 Nutrient inflow rates

Daily inputs of dissolved nitrogen (NH_4^+, NO_2^-, NO_3^- and DON) and phosphorus (HPO_4^{2-} and DOP) compounds via inflowing water (*Nut_inflow*; µM d^{-1}) were calculated as follows:

$$Nut\_inflow = \frac{[Nut]in}{V} \times Water\_in \tag{5}$$

where *Nut* corresponds to a specific dissolved compound (NH_4^+, NO_2^-, NO_3^-, DON, HPO_4^{2-} and DOP), *[Nut]in* to the concentrations of dissolved compounds in inflowing water (µM), *V* is the pond volume (m^3) and *Water_in is* the water inflow rate (m^3 d^{-1}).

2.6 Data analysis

Correlation analyses were performed to evaluate relationships between porewater nutrient concentrations and temperature (Zar 1999). Similarities between experimental ponds were evaluated by non-parametric Wilcoxon tests (Zar 1999).

3 Results

3.1 Field experiments

3.1.1 Overlying and inflowing water

Overlying water NH_4^+ (1.2 – 13 µM) and NO_3^- (0.05 – 3.6 µM) concentrations were significantly higher in the fishpond (Wilcoxon test, n=8, $P<0.05$), unlike NO_2^- (0.05 – 0.7 µM) and DON concentrations (0.6 – 16 µM) that showed no significant differences between ponds (Wilcoxon test, n=8, $P>0.4$). Similar HPO_4^{2-} (<0.5 µM) and DOP (<1.6 µM) levels were found in the overlying waters of Pf and Pnf (Figure 3.2.2). For both overlying and inflowing water, no seasonal variation was observed. Nevertheless, inorganic nutrient concentrations in inflowing water followed a similar variation pattern than in Pf and Pnf (Figure 3.2.2).

3.1.2 Porewater

The low nutrient levels found in the water column of Pf and Pnf, contrasted with concentrations measured in porewater, which were up to three orders of magnitude higher (Table 3.2.1).

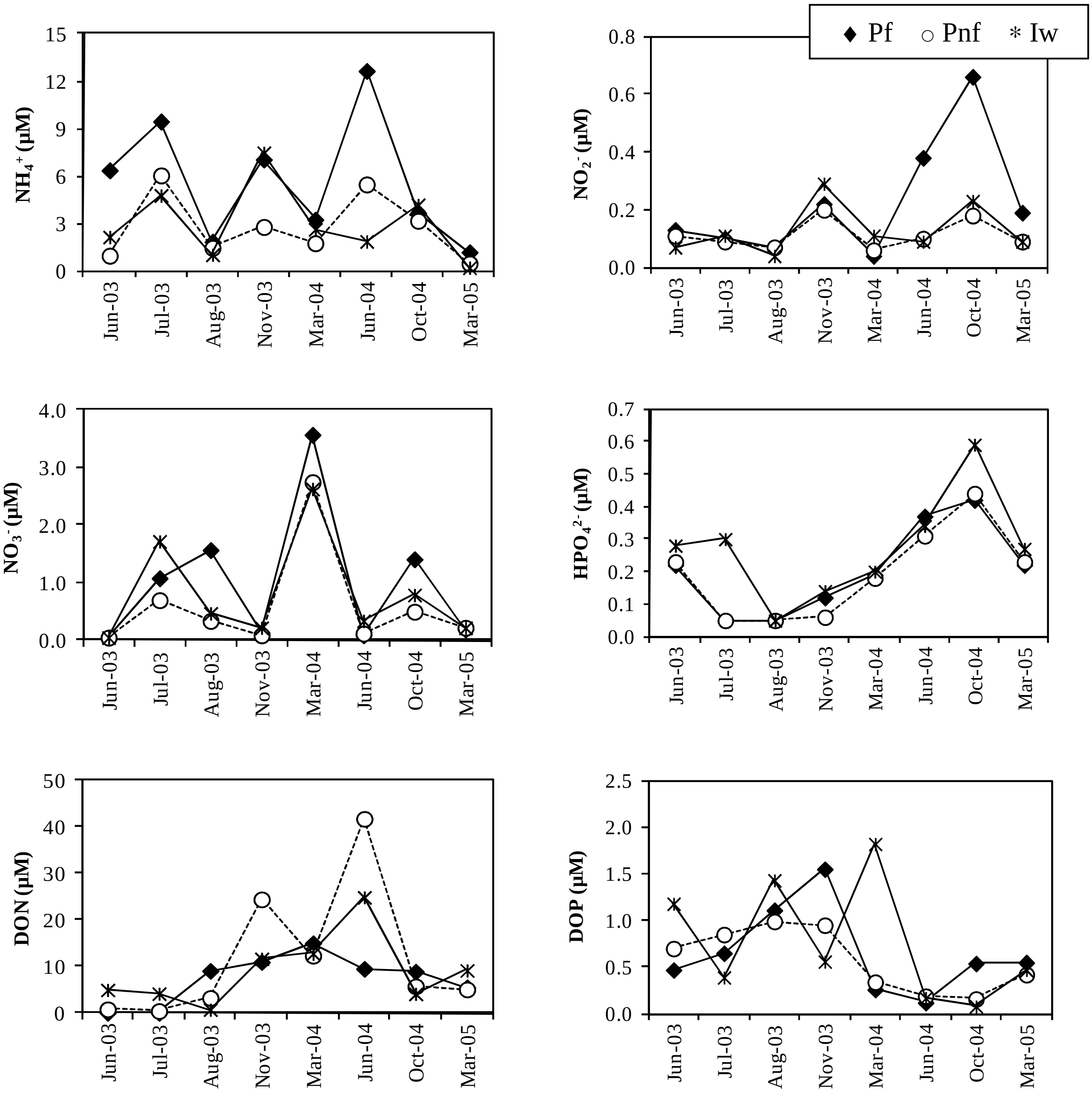


Figure 3.2.2 – Overlying water ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), phosphate (HPO_4^{2-}), dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) concentrations (µM) in the fishpond (Pf, ♦), control pond (Pnf, ○) and inflowing water (Iw, ∗), throughout the trial.

Differences between ponds were evident for NH_4^+ and DON (Wilcoxon test, n=8, $P<0.05$) but less significant for the other compounds (Wilcoxon test, n=8, $0.07<P<0.5$). During the production cycle, no clear seasonal variation pattern was found for most porewater compounds in Pf, with the exception of NO_2^- and NO_3^- that presented a significantly negative relationship with temperature (r = -0.57; n=8, $P<0.05$). Porewater NH_4^+ and HPO_4^{2-} concentrations increased sharply (≈ 60% to 80%) in Pf during the second year of the experiment, while DON and DOP decreased, 12% and 23%, respectively.

Table 3.2.1 – Porewater ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), phosphate (HPO_4^{2-}), dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) concentrations (mean ± *standard deviation*) in the upper sediment layers (2 cm) of the fishpond (Pf) and control pond (Pnf), during the experiment.

Year		Temp	Porosity	NH_4^+	NO_3^-	NO_2^-	HPO_4^{2-}	DON	DOP
		(ºC)		µM					
Pf									
03	Jun	21	0.35	128 ± *52*	0.05 ± *0.01*	0.63 ± *0.19*	5.2 ± *2.3*	398± *1*	10 ± *5*
	Jul	26	0.50	161 ± *61*	3.2 ± *0.1*	0.10 ± *0.02*	0.05 ± *0.01*	420 ± *153*	32 ± *10*
	Aug	27	0.40	88 ± *27*	7.7 ± *2.7*	1.6 ± *0.1*	0.08 ± *0.01*	1767 ± *753*	63 ± *31*
	Nov	16	0.42	124 ± *24*	37 ± *5*	4.9 ± *1.5*	1.2 ± *0.4*	1528 ± *440*	89 ± *17*
04	Mar	19	0.42	48 ± *22*	0.05 ± *0.01*	2.7 ± *0.6*	0.19 ± *0.01*	1189 ± *576*	46 ± *18*
	Jun	27	0.54	775 ± *93*	1.7 ± *0.3*	2.0 ± *0.2*	17 ± *4*	1311 ± *330*	23 ± *7*
	Oct	20	0.49	318 ± *28*	24 ± *1*	8.1 ± *1.3*	26 ± *11*	480 ± *52*	61 ± *24*
05	Mar	15	0.57	186 ± *59*	15 ± *4*	4.7 ± *1.4*	19 ± *4*	624 ± *275*	20 ± *8*
Pnf									
03	Jun	20	0.37	152 ± *22*	0.05 ± *0.01*	0.53 ± *0.24*	2.7 ± *0.6*	150 ± *1*	4.6 ± *1.6*
	Jul	26	0.40	80 ± *30*	3.7 ± *1.7*	0.23 ± *0.05*	0.05 ± *0.01*	163 ± *71*	38 ± *6*
	Aug	26	0.36	61 ± *12*	7.3 ± *3.5*	2.0 ± *0.3*	0.08 ± *0.02*	355 ± 147	38 ± *4*
	Nov	16	0.39	73 ± *31*	4.1 ± *0.8*	4.4 ± *0.8*	4.2 ± *2.1*	786 ± *258*	88 ± *28*
04	Mar	19	0.40	7.5 ± *0.1*	0.05 ± *0.01*	4.0 ± *0.8*	0.17 ± *0.02*	752 ± *208*	43 ± *4*
	Jun	26	0.40	115 ± *7*	3.9 ± *0.9*	3.3 ± *0.7*	7.1 ± *1.0*	750 ± *220*	32 ± *14*
	Oct	20	0.44	76 ± *34*	20 ± *5*	12 ± *2*	13 ± *3*	964 ± *271*	39 ± *14*
05	Mar	15	0.48	67 ± *24*	6.1 ± *1.2*	5.3 ± *1.6*	13 ± *2*	365 ± *138*	8.5 ± *2.1*

3.1.3 Diffusive fluxes

Throughout the production cycle, NH_4^+ fluxes were particularly higher in the fishpond (Wilcoxon test, n=8, $P<0.05$) during the second year of the experiment (Figure 3.2.3), reaching a maximum in June-04 (150 nmol cm^{-2} d^{-1}). For oxidized nitrogen forms, diffusion was directly related to temperature, as would be expected from porewater concentrations (Table 3.2.1). Diffusive fluxes of NO_3^- were lower and even negative (-33 nmol cm^{-2} d^{-1}) during spring/summer, increasing to maximum values during autumn/winter periods (5.1 nmol cm^{-2} d^{-1}). On the other hand, DON fluxes were highest in warmer months (29 nmol cm^{-2} d^{-1}) and always directed out of the sediments (Figure 3.2.4). The transport of HPO_4^{2-} to the water column was almost negligible in the first year of the experiment, but then increased sharply, up to 2 orders of magnitude, after June 2004 (Figures 3.2.3 and 3.2.4). Minor variations of DOP fluxes were found during the 2 years of the experiment ($0.54<DOP<1.7$ nmol cm^{-2} d^{-1}).

3.2 Laboratory experiments

3.2.1 Biologically-mediated fluxes

Biologically-mediated fluxes seemed closely coupled with temperature, since the sediment-water transport of most compounds was enhanced, up to 3 fold, at higher temperatures (Table 3.2.2). Nevertheless, the effect of bioturbation was more evident in the fishpond. Ammonium (4908 nmol cm^{-2} d^{-1}) and DON fluxes (1570 nmol cm^{-2} d^{-1}) were one order of magnitude higher in Pf than in the control pond (Wilcoxon test, n = 12, $P<0.05$). For HPO_4^{2-}, differences between ponds were only found at higher temperatures. In both ponds, biologically-mediated transport was one to two orders of magnitude higher than the diffusive fluxes evaluated in the incubation chambers (Table 3.2.2).

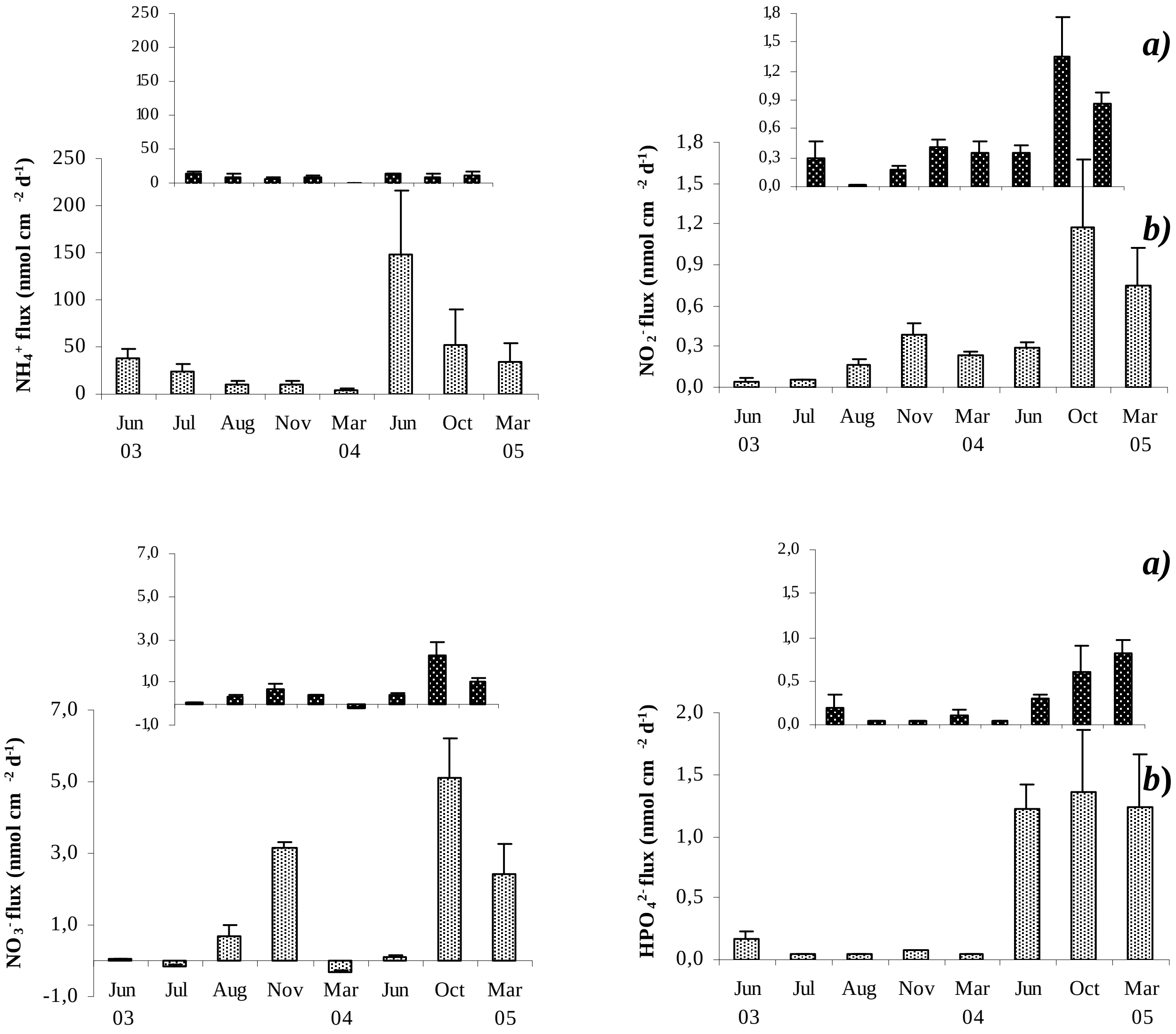


Figure 3.2.3 – Ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-) and phosphate (HPO_4^{2-}) molecular diffusive fluxes (mean ± *standard deviation*, nmol cm^{-2} d^{-1}) in the control pond (Pnf, ***a***) and in the fishpond (Pf, ***b***), throughout the trial.

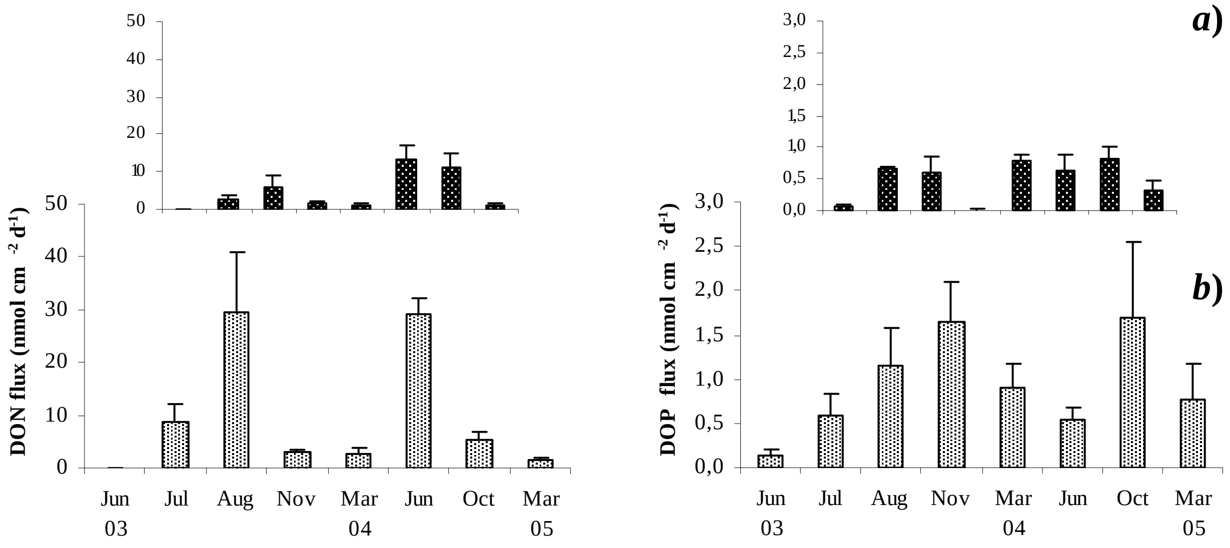


Figure 3.2.4 – Dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) molecular diffusive fluxes (mean ± *standard deviation*, nmol cm^{-2} d^{-1}) in the control pond (Pnf, ***a***) and fishpond (Pf, ***b***), throughout the trial.

Table 3.2.2 – Ammonium (NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), phosphate (HPO_4^{2-}), dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) biologically-mediated fluxes (F_{bio}) and diffusive fluxes (Js) for the fishpond (Pf) and control pond (Pnf) sediments (mean ± *standard deviation*) in incubation chambers at two different temperatures (T).

		F_{bio}		J_s		Porosity
		nmol cm^{-2} d^{-1}		nmol cm^{-2} d^{-1}		ϕ
		T = 15 ºC	T = 21º C	T = 15 ºC	T = 21º C	
Pf	NH_4^+	1993 ± *855*	4908 ± *908*	145 ± *45*	233 ± *17*	0.59 ± *0.03*
	NO_3^-	-68 ± *29*	23 ± *7*	-0.11 ± *0.01*	0.25 ± *0.09*	
	NO_2^-	15 ± *5*	31 ± *13*	-0.15 ± *0.04*	0.28 ± *0.09*	
	HPO_4^{2-}	11 ± *4*	64 ± *22*	5.2 ± *2.5*	1.0 ± *0.5*	
	DON	600 ± *255*	1570 ± *623*	8.6 ± *3.9*	13 ± *2*	
	DOP	35 ± *7*	102 ± *23*	1.7 ± *0.1*	3.5 ± *0.1*	
Pnf	NH_4^+	495 ± *187*	478 ± *198*	14 ± *4*	44 ± *20*	0.45 ± *0.04*
	NO_3^-	-31 ± *4*	14 ± *5*	-0.07 ± *0.03*	0.29 ± *0.04*	
	NO_2^-	5.0 ± *1.3*	15 ± *7*	-0.11 ± *0.01*	0.24 ± *0.07*	
	HPO_4^{2-}	11 ± *2*	6.6 ± *2.0*	2.6 ± *0.9*	0.56 ± *0.24*	
	DON	245 ± *66*	308 ± *96*	1.9 ± *0.8*	7.4 ± *1.3*	
	DOP	6.2 ± *2.3*	70 ± *20*	1.0 ± *0.1*	2.3 ± *0.1*	

4 Discussion

The increasing organic loading throughout the production cycle due to increasing fish biomass and feed inputs, contributed largely to bottom sediment enrichment, particularly in the second year of the experiment when fish biomass reached 500 g m^{-3} and the feeding rate was about 150 kg per month (Serpa et al. 2007b). The addition of labile organic material (e.g. fish feces and feed wastes), intensified organic matter breakdown (Lerat et al. 1990; Holmer and Kristensen 1996; Mesnage et al. 2007; Valdemarsen et al. 2009), leading to higher porewater nutrient concentrations in Pf. The decomposition of nitrogen rich labile substrates in fishpond sediments (Holmer and Kristensen 1996; Holmer et al. 2003) coupled with the low nitrification rates resulting from lower oxygen availability in organically richer sediments (Holmer et al. 2003) were most likely the causes for higher porewater NH_4^+ concentrations in the second year of the experiment. Reducing sediment conditions resulting from organic matter accumulation (Serpa et al. 2007b) promoted phosphorus desorption, leading to a sharp increase (80%) in porewater concentrations over the second production year (van Raaphorst and Kloosterhuis 1994; Slomp et al. 1998). The reverse process, phosphorus retention, occurred during the first year presumably due to the generation of fresh iron oxides under oxidized sediment conditions (Slomp 1997; Falcão et al. 2006; Serpa et al. 2007a). Following the increase of NH_4^+ and HPO_4^{2-} concentrations in porewater, the decrease of DON and DOP during the second year of the experiment suggests the decomposition of more refractory organic matter (Belias et al. 2007).

Concentration gradients between sediments and overlying water drive to a great extent the exchanges across the interface (Hall et al. 1992; Holmer et al. 2002; Wilson and Brennan 2004; Belias et al. 2007; Mesnage et al. 2007). In the present study, the increase in NH_4^+ and HPO_4^{2-}

porewater concentrations observed in the second production year (Table 3.2.1) was coupled to an increase in the diffusive fluxes of these compounds (Figure 3.2.3). Likewise, NO_2^- and NO_3^- fluxes (Figure 3.2.3) were inversely related with temperature, as would be expected from porewater concentrations (Table 3.2.1). The highest NO_2^- and NO_3^- fluxes were observed during colder periods (NO_2^-: 1.2 nmol cm^{-2} d^{-1}; NO_3^-: 5.1 nmol cm^{-2} d^{-1}) probably because nitrification processes are driven by high oxygen availability (Hall et al. 1996; Hargreaves 1998; Asmus et al. 2000). Conversely, higher temperatures promoted organic matter mineralization, leading to higher DON fluxes (≈30 nmol cm^{-2} d^{-1}) in warmer periods (Burdige and Zheng 1998; Wilson and Brennan 2004). For both inorganic and organic compounds, the estimated diffusive fluxes in the white seabream pond were lower than the ones measured in intensive fishponds and marine cage farms but similar to those reported for coastal ecosystems (Table 3.2.3). Lower stocking densities (≈1.5 kg m^{-3} at the end of the production cycle) and feed inputs (25 to 350 kg $month^{-1}$) in this semi-intensive fishpond system probably accounted for these results, confirming the environmental sustainability of farming conditions (Serpa et al. 2007b).

The presence of benthic fauna in fishpond sediments strongly induces the transport of solutes from porewater to overlying water (van der Loeff et al. 1984; Aller and Aller 1992; Nizzoli et al. 2007; Holmer and Heilskov 2008). As a result, biologically-mediated fluxes were up to one order of magnitude higher than diffusive fluxes (Table 3.2.2). Biological activities, such as burrowing, particle-reworking, ventilation and irrigation, promote solute transfer across the sediment-water interface (Nizzoli et al. 2007; Holmer and Heilskov 2008), either by increasing sediment diffusion coefficients (Berner 1980) and changing sediment porosity or, by porewater flushing through animal burrows (Nizzoli et al. 2007). The magnitude of these effects depends mostly on faunal abundance and functional traits, i.e. life habits, mobility, feeding type (Aller and Aller 1992), which is consistent with the results presented herein since biogeochemical dissimilarities (higher porewater concentrations and benthic fluxes) were found among experimental ponds with significantly different biological assemblages (Carvalho et al. 2007; Carvalho et al. 2009). In fact, the highly abundant polychaetes in fishpond sediments have been reported to enhance upward NH_4^+ and DON fluxes (Burdige and Zheng 1998; Nizzoli et al. 2007), by stimulating aerobic reactions such as organic matter oxidation and other early diagenetic reactions like nitrification/denitrification (Aller and Aller 1992; Nizzoli et al. 2007; Holmer and Heilskov 2008). On the other hand, the higher HPO_4^{2-} fluxes in Pf during summer months were most likely related to adsorption/desorption processes rather than to differences in macrofauna assemblages. Low oxygen availability during periods of higher temperatures has been referred to promote P release from organically-richer sediments in the Ria Formosa lagoon (Falcão et al. 2006; Serpa et al. 2007a). In both ponds, the effect of bioturbation was magnified at higher temperatures as a result of increased animal activity (van der Loeff et al. 1984; Aller and Aller 1992), which promoted nutrient fluxes between sediments and overlying water.

Table 3.2.3 – Brief review of diffusive fluxes (nmol cm^{-2} d^{-1}) in different types of ecosystems.

Nutrient	Ecosystem	Value	Reference
NH_4^+	Marine fishpond	63 – 84	Blackburn et al. (1988)
	Bay of Cadiz (Spain)	3660	Forja et al. (1994)
	Skagerrak Sea (N-E North Sea)	-6.1 – 4.5	Hall et al. (1996)
	Intensive fishpond (France)	523 – 7650	Lefebvre et al. (2001)
	Gazi Bay (Kenya)	-648 – 355	Mwashote and Jumba (2002)
	Mussel farm (New Zealand)	192 – 765	Giles et al. (2006)
	Baltic Sea	60 – 120	Graca et al. (2006)
	Thau lagoon (France)	20 – 1000	Mesnage et al. (2007)
	Ria Formosa lagoon (Portugal)	10 – 104	Serpa et al. (2007a)
NO_2^-	Skagerrak Sea (N-E North Sea)	-2.1 – 0.47	Hall et al. (1996)
	Gazi Bay (Kenya)	-140 – 144	Mwashote and Jumba (2002)
NO_3^-	Fourleague Bay (USA)	1680 (mean)	Teague et al. (1988)
	Skagerrak Sea (N-E North Sea)	-14 – 30	Hall et al. (1996)
	Gazi Bay (Kenya)	-190 – 100	Mwashote and Jumba (2002)
	Mussel farm (New Zealand)	7.4 – 52	Giles et al. (2006)
HPO_4^{2-}	Fourleague Bay (USA)	708 (mean)	Teague et al. (1988)
	Marine cage farm(Sweden)	16 – 675	Holby and Hall (1991)
	Skagerrak Sea (N-E North Sea)	-1.5 – 10	Hall et al. (1996)
	Intensive fishpond (France)	24 – 1392	Lefebvre et al. (2001)
	Gazi Bay (Kenya)	-190 – 180	Mwashote and Jumba (2002)
	Baltic Sea	7 – 14	Graca et al. (2006)
	Thau lagoon (France)	2 – 96	Mesnage et al. (2007)
	Ria Formosa lagoon (Portugal)	0.2 – 8	Serpa et al. (2007a)
DON	Chesapeake Bay (USA)	4 – 42	Burdige and Zheng (1998)

4.1 Nutrient budgets

Besides benthic fluxes, the other main sources of dissolved N and P to semi-intensive fishpond systems include, source water, fish excretion and dissolution of uneaten feed (Krom et al. 1985b;

Neori et al. 1989; Krom 1995; Hargreaves 1998; Brambilla et al. 2007). Conversely, nutrients are rapidly consumed by phytoplankton and microphytobenthos (Jordan et al. 1991; Hargreaves 1998) and a substantial fraction is lost from the system by outflowing water, especially in ponds with continuous water exchange (Lefebvre et al. 2001).

Nutrient budgets are helpful tools for understanding the relative contribution of the former nutrient sources and sinks to the water composition of fish production ponds (Krom et al. 1985b; Holby and Hall 1991; Hall et al. 1992). Since the impact of fish biomass on the pond environment, particularly on bottom sediments, was more relevant in the second production year, the daily contributions of the different nutrient sources and sinks were calculated on a yearly basis, in order to understand how sources and sinks interact with each other (Table 3.2.4 and Figure 3.2.5 - *top*). Benthic fluxes represented, on average, 46% of the daily N inputs in the first and second production years (i.e. 16.14 and 24.49 µM d^{-1}, respectively), which is indicative that sediments play a significant role in pond nutrient dynamics, and are thereby crucial for the quality of pond water (Helder and Andersen 1987; Blackburn et al. 1988; Sandnes et al. 2000; Nizzoli et al. 2007). The contribution of sediments to P availability was very similar between the two production years (respectively, 0.56 and 0.60 µM d^{-1}), most likely due to the high retention of this element in pond sediments (Slomp 1997; Slomp et al. 1998). The major source of dissolved nutrients to the system was inflowing water, which supplied on average 52% (19 to 27 µM d^{-1}) of dissolved N in pond water and most (65% in first year and 76% in the second production year) of the available P. The higher contribution of inflowing water in the second production year was probably related to higher water exchange rates as a consequence of higher fish biomass and feed inputs in this period (Serpa et al. 2007b). Fish excretion and dissolution of uneaten feed represented a small fraction (respectively 1.6% and 1.1%) of total N and P inputs to the system most likely as a result of the low stocking densities in the fishpond (ca. 1.5 kg m^{-3} at the end of the production cycle). Nitrogen (75-89%) and phosphorus (82-93%) losses occurred principally via outflowing water, however 2.74 to 8.7 µM d^{-1} (7-18%) of dissolved N and 0.11 to 0.27 µM d^{-1} (11-22%) of dissolved P, was consumed by primary producers. When comparing the results of the fishpond budget with the one constructed for the pond without fish (Table 3.2.4 and Figure 3.2.5-*bottom*), it becomes evident that the contribution of the different nutrient sources and sinks was higher in the fishpond, as would be expected from the higher water exchange rates and higher amount of organic matter resulting from fish activity (Serpa et al. 2007b).

As the water quality of semi-intensive fishpond systems seems dependent on the sediment redox status and on water exchange rates, pond management should consider these aspects to avoid poor water quality that may compromise commercial fish production. The treatment of bottom sediments (e.g. drying) between production cycles is particularly important to promote the decomposition of organic matter accumulated as a result of fish activity, helping to prevent water quality problems in the next cycle. The fact that the water composition profile within the production pond was similar to that of the control pond and of inflowing water, confirms the inexistence of limiting fish growth conditions as referred by Serpa et al. (2007b), suggesting that the present farming conditions (i.e. water exchange rates, feeding rates, stocking density) may be used as guidelines for the optimization of white seabream production in semi-intensive systems. As these systems are often managed intuitively, this information may be important to the aquaculture sector in order to ensure the sustainability of semi-intensive fish farms.

Table 3.2.4 – Description of the main sources and sinks of dissolved N and P in the fishpond: source data (annual averages) and brief explanation of the calculations. Values for the control pond are given between brackets.

Sources	Description	N		P		Unit	Reference data
		Year 1	Year 2	Year 1	Year 2		
Diffusive fluxes	Estimated from daily diffusive fluxes calculated in the field experiment, and extrapolated for the entire pond by assuming constant fluxes in space.	27 (9.9)	93 (21)	1.0 (0.5)	2.3 (1.2)	nmol cm^{-2} d^{-1}	Present study
Biologically-mediated fluxes	Estimated from nutrient fluxes measured in the laboratory experiments and extrapolated for the entire pond area.	3093 (356)	4536 (1202)	117 (9)	113 (20)	nmol cm^{-2} d^{-1}	Present study
Inflowing water	Estimated by multiplying the daily average water inflow rates (dm^3 d^{-1}) by the nutrients concentrations (µM) in source water .	19 (1.2)	27 (1.6)	1.0 (0.08)	2.1 (0.13)	µM d^{-1}	Present study
Fish excretion (urinary + gill) and dissolution of uneaten feed	Microcosm experiments were carried out in order to estimate white sea bream excretion and feed dissolution rates on an hourly basis. The results of these experiments were then extrapolated for the entire system by taking into account the number of daily meals (3) and stocking density in the production pond.	0.4	0.8	0.01	0.02	µM d^{-1} $fish^{-1}$	PROMAR (2006) – EU Project
Sinks							
Phytoplankton uptake	Data obtained from *in situ* productivity incubation experiments using the oxygen method. Hourly oxygen production rates were converted to N and P uptake rates by the Redfield ratio, and extrapolated for one day by assuming a photoperiod of 12 hours.	1.8 (0.57)	7.0 (1.4)	0.05 (0.02)	0.17 (0.06)	µM d^{-1}	PROMAR (2006) – EU Project
Microphytobenthos uptake	Data obtained from *in situ* productivity incubation experiments (oxygen method) in the Ria Formosa lagoon. Since chlorophyll *a* concentrations were similar to those found in Pf and Pnf sediments (6-15 µg Chl *a* g^{-1}), similar uptake rates were assumed and extrapolated for one day (by assuming an average photoperiod of 8 hours d^{-1}) and for the entire pond surface.	3.0 (1.4)	5.4 (2.9)	0.19 (0.09)	0.32 (0.16)	µM m^{-2} h^{-1}	OARRE (2001) – EU Project
Outflowing water	Estimated by multiplying the daily average water outflow rates (m^3 d^{-1}) by the nutrients concentrations (µM) in Pf and Pnf. Water outflow rates were assumed equal to inflow rates because pond water level changes slightly.	23 (1.4)	26 (1.5)	1.3 (0.09)	1.5 (0.08)	µM d^{-1}	Present study

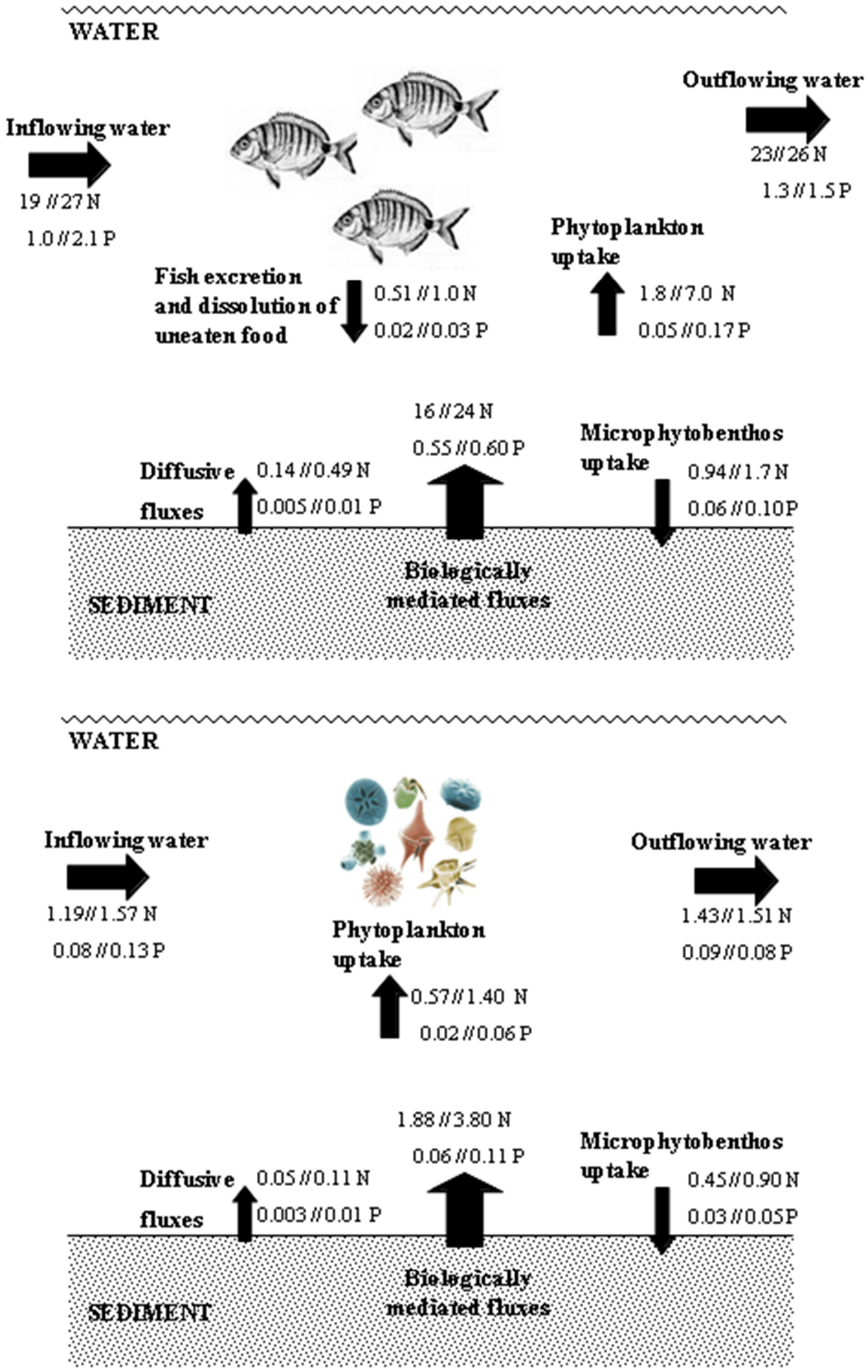


Figure 3.2.5 – Average daily contributions (µM d^{-1}) of different nutrient sources (benthic fluxes, inflowing water, fish excretion and dissolution of uneaten food) and sinks (phytoplankton uptake, microphytobenthos uptake and outflowing water), for dissolved nitrogen (N) and phosphorus (P) availability in pond water, Pf (*top*) and Pnf (*bottom*), during the first//second year of the trial.

5 Conclusions

This study clearly provided evidence of the importance of bottom sediments and water exchange rates to the water quality of semi-intensive fish production systems. Therefore, monitoring sediment quality and optimizing water exchange rates are possible approaches toward efficient pond management. The budget is a starting point for the development of a mathematical model to predict water and sediment quality in fishponds under different farming conditions. Such an ecological model may be a useful tool for defining practical management strategies to maintain a healthy pond environment and minimize the environmental impacts of inshore aquaculture.

Acknowledgments

The present work was supported by Project PROMAR (INTERREG III-A) “Cooperação Algarve – Andaluzia para a Promoção de Recursos Aquícolas Marinhos no Litoral Atlântico Sul” and by a PhD grant from Fundação para a Ciência e Tecnologia (SFRH/BD/27643/2006). The authors would specially like to thank Maria de Lurdes Santos for her help in field sampling and laboratory work, and also to Christian Campeau for his assistance with the English language.

Chapter 4

Modelling biogeochemical processes in semi-intensive fish earth ponds: model development and calibration

---

Serpa D., Pousão Ferreira P., Caetano M.., Cancela da Fonseca L., Dinis M. T., Duarte P. Modelling biogeochemical processes in semi-intensive fish earth ponds: model development and calibration. *In preparation*.

Abstract

The biogeochemistry of fish earth ponds is a complex subject due to the interactions between several water column and sediment compounds, particularly nutrient species. Models can improve our ability to understand such complexity. This paper combines existing knowledge on biogeochemical processes in earth ponds into a model that calculates the concentrations of the compounds that are more likely to negatively affect fish production and cause undesirable environmental impacts, such as nitrogen, phosphorus, and oxygen. Aside from inorganic nutrient forms, organic compounds were included in the model due to their relevance for the nutrient cycles in aquatic systems. The model couples the pelagic and benthic compartments, due to the importance of sediment-water interactions in shallow earth ponds. In this first approach in modelling the fishpond environment, the feedbacks between cultivated species and the environment were not accounted for in the model, to reduce its complexity and easily identify the interactions between water column and sediment variables and processes. The model was calibrated for an earth pond without fish, using data sets collected during a 2-year trial. The variability of water column compounds was generally well predicted ($p<0.01$), however the model could not fully reproduce ammonium and dissolved organic phosphorus concentrations. In sediments, organic phosphorus was accurately simulated ($p<0.05$) while nitrogen and carbon pools were occasionally over or under-estimated. Model limitations regarding sediment variables are most likely related to the effects of benthic primary producers and macrofauna activity in earth ponds biogeochemistry. Future applications of the model developed herein include its coupling to a fish Dynamic Energy Budget (DEB) model to be used as a predictive tool for fishpond management.

1 Introduction

The biogeochemical processes occurring in earth ponds are essentially the same as in other aquatic systems (Chapelle 1995; Wang et al. 2003; Burford and Lorenzen 2004; Kittiwanich et al. 2007; Serpa et al. 2007a, b). However, in shallow earth ponds, the interactions between pelagic and benthic systems are more intense because most autochthonous particulate organic matter is rapidly settled, being mineralized in the top sediment layer (Hargreaves 1998; Serpa et al. 2007b). Organic matter decomposition generates a pool of organic and inorganic nutrients (Kittiwanich et al. 2007; Worsfold et al. 2008), which are intensely transported to the water column, becoming available for the biota (Kittiwanich et al. 2007; Worsfold et al. 2008).

Although several studies on earth pond biogeochemistry have been produced (Hargreaves 1998; Alongi et al. 1999; Montoya et al. 2000; Lefebvre et al. 2001; Burford et al. 2003; Burford and Lorenzen 2004; Boyd et al. 2006; Muendo 2006; Xinglong and Boyd 2006; Mukherjee et al. 2008), linkage between early diagenetic processes and the interactions between compounds are complex and poorly understood. Furthermore, biogeochemical processes are affected by abiotic (e.g. dissolved oxygen, temperature, pH and light intensity) and biotic parameters (e.g. structure of microbial and benthic macrofauna communities) that interact in a complex way (Moriarty 1997; Hargreaves 1998; Peng et al. 2007), making it difficult to predict the variability of the different compounds.

Mathematical models can improve our ability to understand the complexity of such systems by integrating physical, chemical and biological processes occurring in earth ponds. Models are also powerful tools to predict the effects of management strategies on pond biogeochemistry (Montoya et al. 2000; Li and Yakupitiyage 2003; Burford and Lorenzen 2004; Piedcausa et al.

2010), providing useful information on how to improve water quality and to reduce the environmental impacts of fish farms. Several mathematical models have been developed for aquaculture ponds (Piedrahita et al., 1984; Kochba et al. 1994; Culberson and Piedrahita 1996; Montoya et al. 2000; Lefebvre et al. 2001; Li and Yakupitiyage 2003; Jiménez-Montealegre et al. 2002a; Burford and Lorenzen 2004; Mukherjee et al 2008). Some of these models were specifically used for analysing nitrogen (Kochba et al. 1994; Hargreaves 1997; Jiménez-Montealegre et al. 2002a; Burford and Lorenzen 2004) and phosphorus dynamics (Montoya et al. 2000), while less effort has been made to develop more comprehensive predictive models (Piedrahita et al., 1984; Lefebvre et al. 2001; Li and Yakupitiyage 2003; Mukherjee et al. 2008).

The general objective of this work was to develop a mathematical model for the main biogeochemical processes in fish earth ponds, namely for the elements that are more likely to negatively affect fish production and cause undesirable environmental impacts due to their excess, such as nitrogen (N) and phosphorus (P), or deficit, such as oxygen (DO) (Serpa and Duarte 2008). Given the importance of sedimentation and diffusion processes in shallow aquatic systems, the model developed herein couples the pelagic and benthic compartments to simulate the interactions between them. However, feedbacks between fish and the environment were not considered in this work because this would substantially increase model complexity, making it difficult to calibrate the model and evaluate its performance regarding the simulation of other biogeochemical processes. The specific objectives of this study were to:

(1) evaluate model sensitivity to changes in individual processes;

(2) identify the main sources and sinks of nutrients in the system;

(3) identify those processes needing further study,

The model described herein is the first step towards a complete fish pond model after its coupling with a Dynamic Energy Budget (DEB) model.

2 Methodology

2.1 Description of the system

Data for model calibration was collected during a 2-year white seabream growth trial (Chapter 3), carried out in the earth ponds of the IPIMAR Aquaculture Research Center (ARC), located in the Ria Formosa lagoon (Southeast Portugal). In this trial, a rectangular earth pond with a surface area of 495 m^2 (33 m × 15 m) and 1.5 m depth (height of the water column) was used as a control pond (without fish). The model developed herein was calibrated against water column and sediment data from this pond. The water flow rate to the pond varied from 1 to 3 L s^{-1}, depending on the lagoon tidal cycles and water temperature.

2.2 Model description

Given the small dimensions and the absence of stratification in the pond, a zero dimensional (0D) model was developed, assuming water column and sediments as two homogeneous boxes. The biogeochemical model consists of a pelagic and a benthic module describing the main processes occurring in the water column and sediments of earth ponds.

2.2.1 Pelagic module

In the pelagic system there were five state variables for particulate matter: Total Particulate Matter (TPM), Particulate Organic Matter (POM) and its carbon, nitrogen and phosphorus components (POC, PON and POP, respectively); three variables for dissolved organic forms: Dissolved Organic Carbon, Nitrogen and Phosphorus (DOC, DON and DOP, respectively); and four variables for inorganic nutrients: Ammonium (NH_4^+), Oxidized Nitrogen forms (NO_x), Phosphate (HPO_4^{2-}) and Dissolved Oxygen (DO).

The main source of particulate matter for the system was inflowing water. A fraction of the total particulate matter (TPM) entering the system, is organic. Although in shallow systems, most part of particulate organic matter (POM) is settled to the bottom (Hargreaves 1998), the model considers that a fraction of POC, PON and POP may be either mineralized or dissolved in the water column.

A conceptual model for the main N and P forms and processes in the water column is shown in Figure 4.1. Unlike in other studies, DON was included in the present model because it represents a large fraction (60–69%) of total dissolved N in aquatic systems (Kittiwanich et al. 2007; Worsfold et al. 2008), particularly on fishponds, due to the addition of formulated feeds (Burford and Lorenzen 2004). Part of DON is mineralized by proteolitic, heterotrophic bacteria to NH_4^+ (Hargreaves 1998), which undergoes a series of transformations. In well oxygenated pond waters, nitrification is dominant over denitrification, whereas the latter process becomes more significant under anoxic conditions (Chapelle 1995). Because only 40% of oxidized nitrogen forms (NO_x) has been reported to be reduced to NH_4^+ (Chapelle 1995), the model also considers denitrification into gaseous forms such as, dinitrogen (N_2) and nitrous oxide gas (N_2O).

In what concerns P compounds, besides POP and phosphate (HPO_4^{2-}), which is usually the most abundant form of dissolved P in aquatic systems (Worsfold et al. 2008), DOP was also included in the pelagic module not only because this P-form can be at least as abundant as inorganic P (Worsfold et al. 2008), but also because it can be further mineralized into HPO_4^{2-} (Kittiwanich et al. 2007).

Since most of the biogeochemical processes occurring in the water column (e.g. organic matter mineralization, nitrification and denitrification) are not only temperature-dependent but also oxygen-dependent (Chapelle et al. 1995), the model also simulates the concentrations of dissolved oxygen (DO) in the water column.

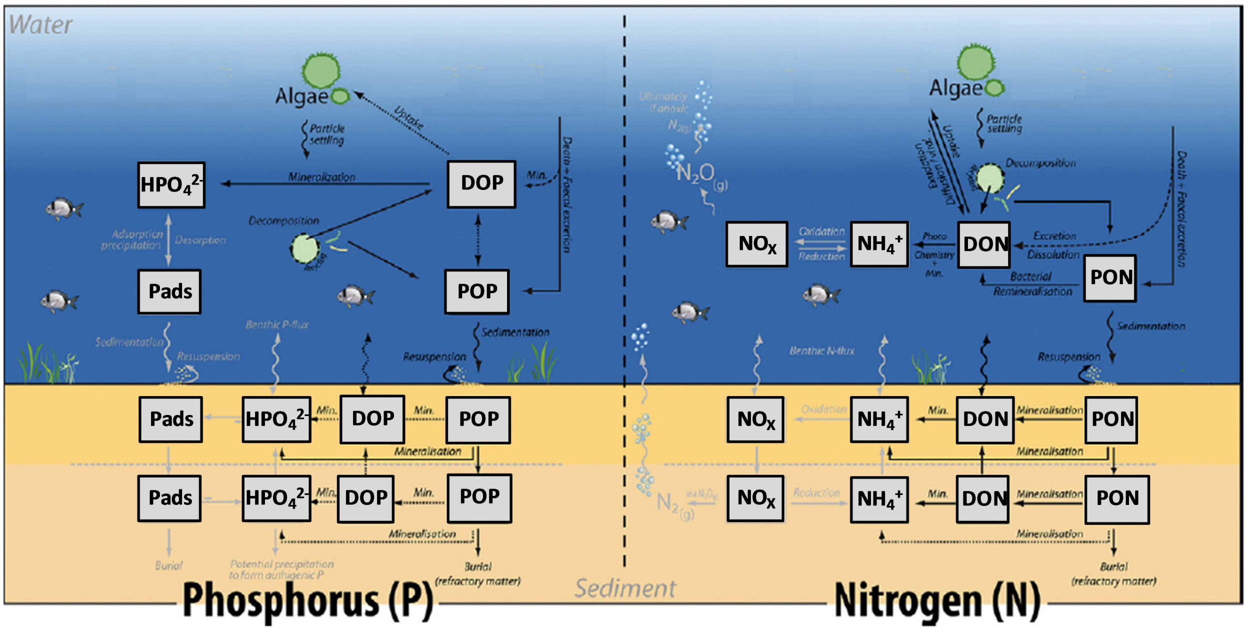


Figure 4.1 – Conceptual model for nitrogen (N) and phosphorus (P) transformations in the water column and sediments of fish earth ponds. POP: particulate organic phosphorus; DOP: dissolved organic phosphorus; HPO_4^{2-}: phosphate; Pads: inorganic phosphorus adsorbed to sediments; PON: particulate organic nitrogen; DON: dissolved organic nitrogen; NH_4^+: ammonium; NO_X: oxidized nitrogen forms. Adapted from Worsfold et al. (2008).

2.2.2 Benthic module

Most of the variables and processes described for the pelagic module were common to the benthic system (Figure 4.1). However, there are some specificities in this module. For instance, the fraction of POC, PON and POP that is not mineralized or dissolved is now buried into the sediments. Moreover, as P reacts with a wide variety of compounds, being taken up and released from biogenic and abiogenic particles (van Raaphorst and Kloosterhuis 1994), there is a new state variable for P in this module to account for the fraction of inorganic phosphorus that is sorbed onto sediment particles (Pads) (Chapelle 1995, Serpa et al. 2007a). Phosphorus sorption/desorption processes are not only controlled by temperature and oxygen availability (Chapelle 1995, Slomp et al. 1998; Falcão et al. 2006; Serpa et al. 2007a), but also by sediment composition, particularly in what concerns to iron contents (Slomp et al. 1998; Falcão et al. 2006; Serpa et al. 2007a, b).

Depending on the concentration gradients between pond sediments and overlying water, dissolved compounds are subjected to diffusion processes according to the Fick's First Law, being transferred to the water column or into the sediments (Lefebvre et al. 2001).

Hereafter, chemical species include the subscript “w” or “s” with respect to water column and sediments (including porewater), respectively.

2.3 Model equations

Appendixes A and B show the symbols, units and differential equations for every state variable in the pelagic and benthic modules, respectively. Rate equations and parameters used to describe biogeochemical processes in the model are presented in Appendixes C and D, respectively.

2.4 Model implementation

The 0D model developed in this work was implemented with EcoDynamo (Pereira et al. 2006), a software that uses object-oriented programming (OOP). Each object simulates several state variables (Table 4.1) and processes, and interacts with the other objects by means of a shell or server (Pereira et al. 2006). The shell interface allows the user to define the model setups – time steps, output formats (file, graphic and tables), objects to be used and variables to be visualized (Pereira et al. 2006). As input data, the model required complete data sets on average daily water temperature and wind speed (Figure 4.2), water inflow and outflow rates (assumed constant over time = 0.001 $m^3 s^{-1}$), and particulate matter and nutrient concentrations in inflowing water (cf. Chapters 3.1 and 3.2). Whenever there were incomplete data sets for forcing functions, interpolations were carried out using the available information. After definition of initial and boundary conditions and input of model parameters, state variables were simulated over time (time step = 6 minutes) using the Euler integration method. Simulations were run for a period of 651 days, which corresponds to the trial's duration (cf. Section 2.1).

2.5 Model calibration

The present model was calibrated with water column and sediment data collected in an earth pond without fish (cf. – Section 2.1). Environmental parameters measured during the former trial were used as forcing functions for the present model (cf. – Section 2.4) and state variables values at the beginning of the experiment were used as initial values for the simulations. Calibration was based on parameter values determined experimentally or taken from the literature (Appendix D), adjusted until achieving the best fit between simulated and observed data.

2.6 Sensitivity analysis

In order to evaluate which model parameters and initial conditions are more likely to affect each variable, a sensitivity analysis was performed to the model. This analysis consisted in changing one parameter at a time by ± 20%, comparing the results with a standard simulation using the calibrated parameters set (Appendix D) and expressing the difference in percentage variation with respect to the standard simulation.

Table 4.1 – EcoDynamo objects implemented for earth ponds and respective state variables.

Object type	**Object name**	**Object outputs**
Forcing functions	Wind object	Wind speed
	Water temperature object	Water temperature
State variables	Suspended matter object	Total particulate matter (TPM_w), particulate organic matter (POM_w), particulate organic carbon (POC_w), particulate organic nitrogen (PON_w) and particulate organic phosphorus (POP_w)
	Dissolved substances object	Dissolved inorganic ($NH_4^+{}_w$, $NO_{x\,w}$) and organic nitrogen (DON_w), phosphate ($HPO_4^{2-}{}_w$), organic phosphorus (DOP_w) and oxygen (DO_w)
	Sediment biogeochemistry object	Porewater dissolved inorganic ($NH_4^+{}_s$, NO_{xs}) and organic nitrogen (DON_s), phosphate ($HPO_4^{2-}{}_s$), organic phosphorus (DOP_s) and oxygen (DO_s) Sediment organic carbon (POC_s), nitrogen (PON_s) and phosphorus (POP_s), and inorganic phosphorus adsorbed to sediments (Pads)

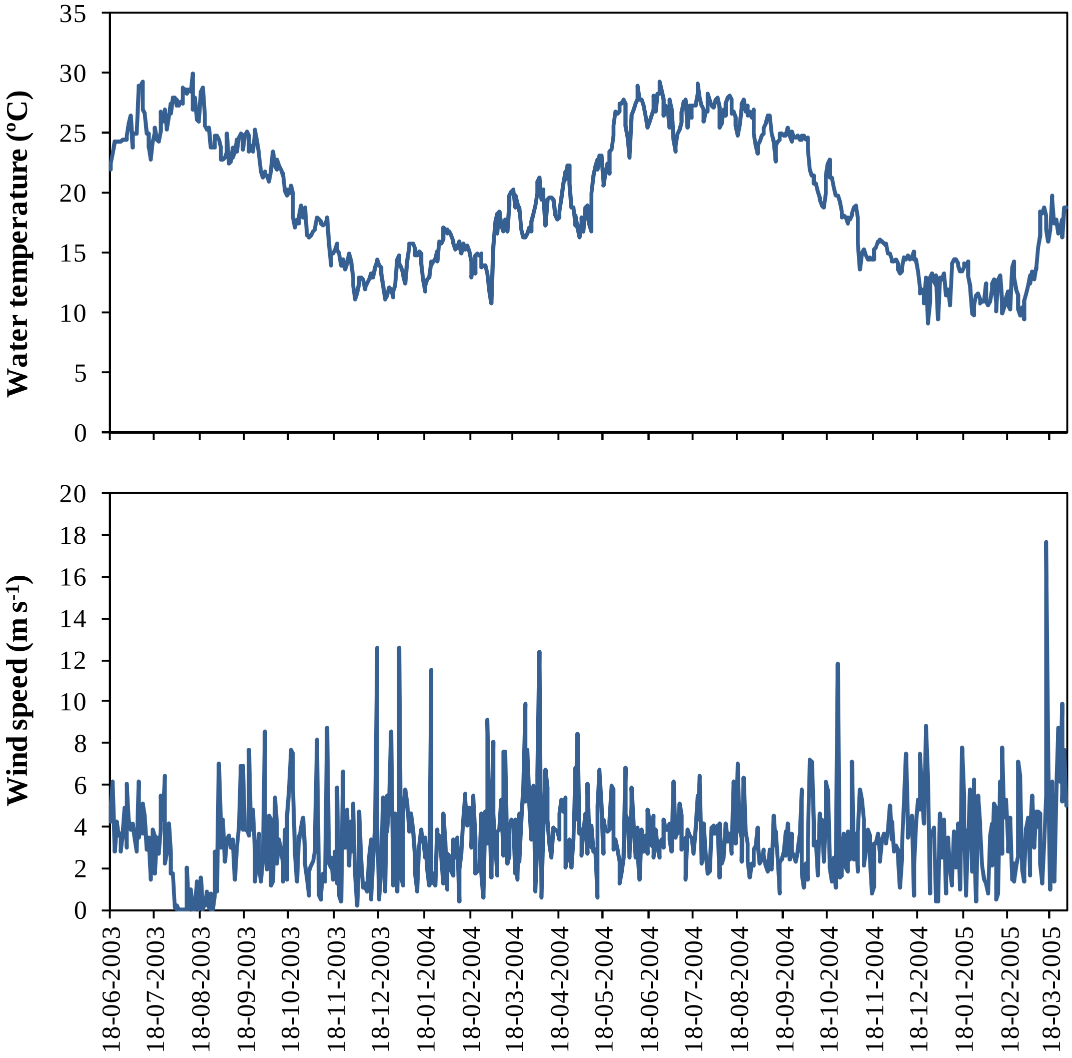


Figure 4.2 – Average daily water temperature (ºC) and wind speed (m s^{-1}) in an earth pond without fish, from June 2003 to March 2005.

2.7 Statistical analysis

Model performance was evaluated by model II regressions between predicted and observed values of state variables, as described by Laws and Archie (1981). According to these authors, a good model fit (p<0.05), implies that the slope of the regressions is not different from one and the y-intercept is not different from zero. A slope that significantly differs from one indicates a difference between observed and simulated values which is proportional to the observed values. If the slope is not significantly different from one but the y-intercept significantly differs from zero there is a systematic difference between observations and simulations.

3 Results

3.1 Model calibration

3.1.1 Water column variables

The comparison between model simulations and observations for water column variables is presented in Figures 4.3 and 4.4. Model II regressions between predicted and measured values (Table 4.2) suggest that the model was able to accurately predict (*p*<0.01) the variability of POM_w and $HPO_4^{2-}{}_w$ in pond water. Nevertheless, a systematic overestimation was found for these variables (Figures 4.3 and 4.4), since the slope of the regressions was not significantly different from one but the y-intercept significantly differs from zero. A significant (*p*<0.01) part of TPM_w, NO_{xw}, DON_w and DO_w variability was also explained by the model (Table 4.2), even though it could under or over estimate the concentrations of these compounds (Figure 4.3 and 4.4). Water column NH_4^+ and DOP were poorly simulated (*p*>0.05), despite the model could reproduce the majority (5 out of 8) of data points (Figure 4.4).

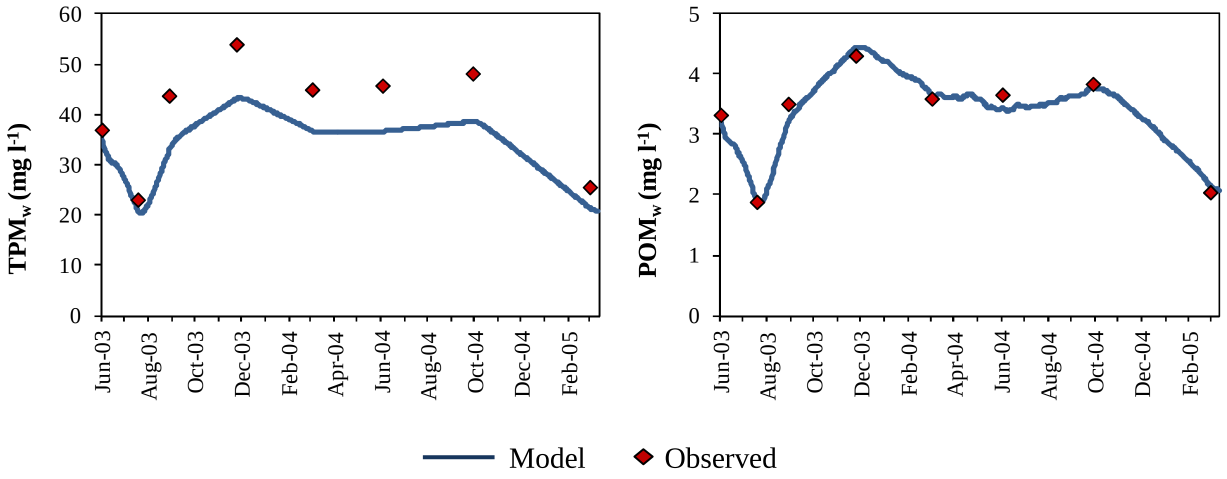


Figure 4.3 – Predicted (line) and observed (diamonds) values of total particulate matter (TPM, mg L^{-1}) and particulate organic matter (POM, mg L^{-1}) in the water column of an earth pond without fish.

3.1.2 Porewater variables

Figures 4.5 and 4.6 compare predicted and measured values of porewater N and P compounds. According to the regression analysis, the model presented reduced accuracy (p>0.05) for most porewater variables (Table 4.3). Nevertheless, it was able to explain most of NH_4^+ and HPO_4^{2-} variability (p<0.05). This exercise was not carried out for porewater oxygen because this variable was not measured in field experiments.

3.1.3 Sediment variables

The results of the regression analysis for sediment variables are presented in Table 4.4. The model explained a significant part of sediment organic phosphorus (POP$_s$) variability (p<0.05), but could not fully predict (p>0.05) sediment organic nitrogen (PON$_s$) and carbon (POC$_s$) concentrations (Figure 4.7).

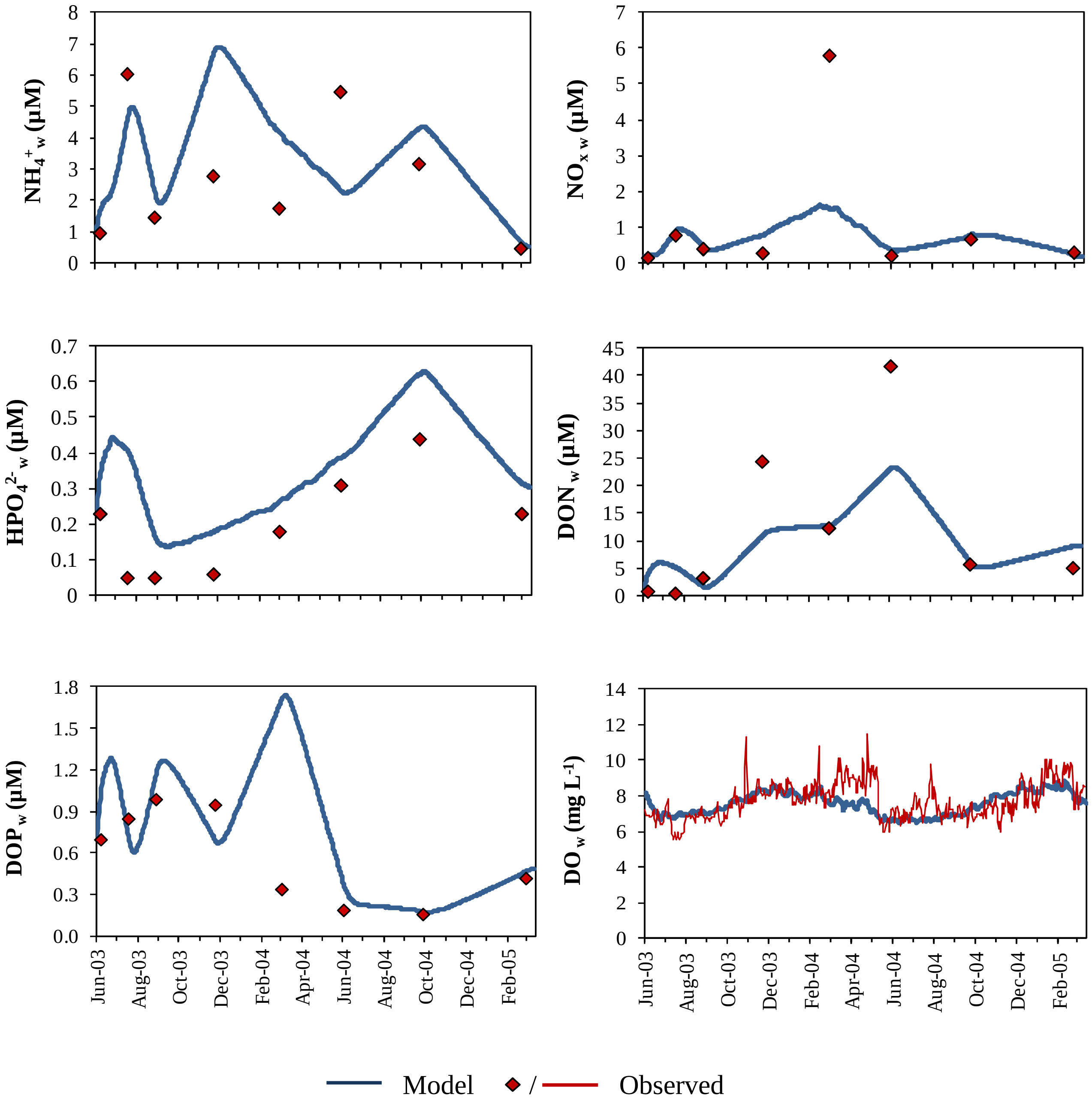


Figure 4.4 – Predicted (blue line) and observed (red diamonds and line) ammonium ($NH_4^+{}_w$), oxidized nitrogen forms (NO_{xw}), dissolved organic nitrogen (DON_w), phosphate ($HPO_4^{2-}{}_w$), dissolved organic phosphorus (DOP_w) and dissolved oxygen (DO_w) concentrations in the water column of an earth pond without fish.

Table 4.2 – Results of model II regressions for water column variables.

		TPM_w mg L^{-1}	POM_w mg L^{-1}	$NH_4^+{}_w$ µM	NO_{Xw} µM	$HPO_4^{2-}{}_w$ µM	DON_w µM	DOP_w µM	DO_w mg L^{-1}
slope	Value	0.76	1.00	0.97	0.20	1.10	0.47	2.86	0.44
	Upper 95% limit	0.98	1.22	-0.56	0.33	3.79	0.70	-0.20	0.49
	Lower 95% limit	0.59	0.83	-1.92	0.07	0.37	0.27	-0.51	0.39
y-intercept	Value	5.19	0.22	0.56	0.43	0.11	3.19	-0.90	4.17
	Upper 95% limit	12.28	0.81	8.56	0.57	0.25	5.49	1.04	4.54
	Lower 95% limit	-3.35	-0.48	4.81	0.29	-0.41	0.49	0.86	3.79
p	Value	<0.01	<0.01	0.29	0.01	0.04	<0.01	0.48	<0.01

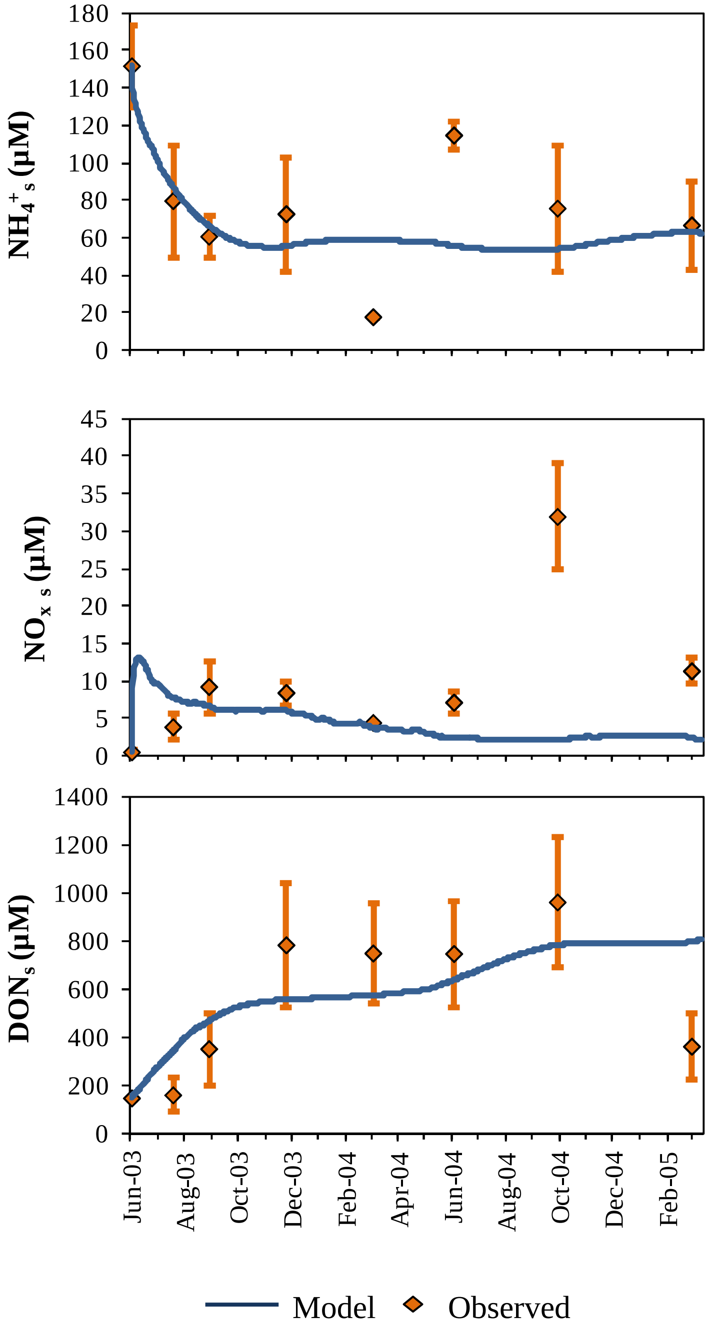


Figure 4.5 – Predicted (line) and observed ± standard deviation (diamonds) ammonium ($NH_4^+{}_s$), oxidized nitrogen forms (NO_{xs}) and dissolved organic nitrogen (DON_s) porewater concentrations in an earth pond without fish.

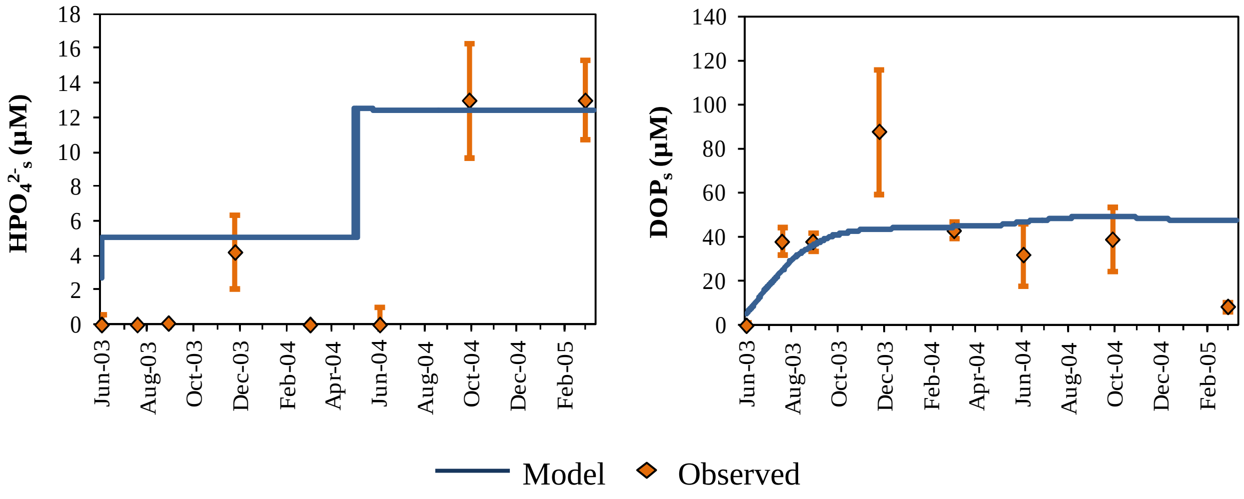


Figure 4.6 – Predicted (line) and observed ± standard deviation (diamonds) phosphate ($HPO_4^{2-}{}_s$) and dissolved organic phosphorus (DOP_s) porewater concentrations in an earth pond without fish.

Table 4.3 – Results of model II regressions for porewater variables.

		$NH_4^+{}_s$ µM	NO_{Xs} µM	$HPO_4^{2-}{}_s$ µM	DON_s µM	DOP_s µM
slope	Value	0.64	-0.05	0.72	0.60	0.34
	Upper 95% limit	1.26	0.24	1.26	1.62	1.93
	Lower 95% limit	0.02	-0.35	0.37	0.06	-0.47
y-intercept	Value	21.76	4.47	3.86	281.04	24.62
	Upper 95% limit	76.74	7.33	5.56	588.43	54.08
	Lower 95% limit	-33.22	1.69	1.16	-26.35	-33.27
p	Value	0.04	0.67	<0.01	0.06	0.33

Table 4.4 – Results of model II regressions for sediment variables.

		POC_s $\mu g\ g^{-1}$ dw	PON_s $\mu g\ g^{-1}$ dw	POP_s $\mu g\ g^{-1}$ dw
slope	Value	0.09	0.05	0.75
	Upper 95% limit	0.22	0.12	1.10
	Lower 95% limit	-0.04	-0.03	0.39
y-intercept	Value	3320.27	31.25	0.13
	Upper 95% limit	3727.93	36.29	38.0
	Lower 95% limit	2903.14	26.18	-37.8
p	Value	0.15	0.20	<0.01

3.2 Sensitivity analysis

A summary of the sensitivity analysis performed to the model for water column, porewater and sediment variables is presented in Tables 4.5, 4.6 and 4.7, respectively. Model parameters, initial conditions and forcing functions producing a variation lower than 4% were omitted. This analysis has one particularity in what concerns water inflow (*Water_in*) and outflow (*Water_out*) rates. Instead of individually testing the sensitivity of model variables to these rates, their combined effect was evaluated (i.e. *Water exchange*) because as the water level in the earth ponds is constant over time, if the water inflow rate increases so does the outflow rate.

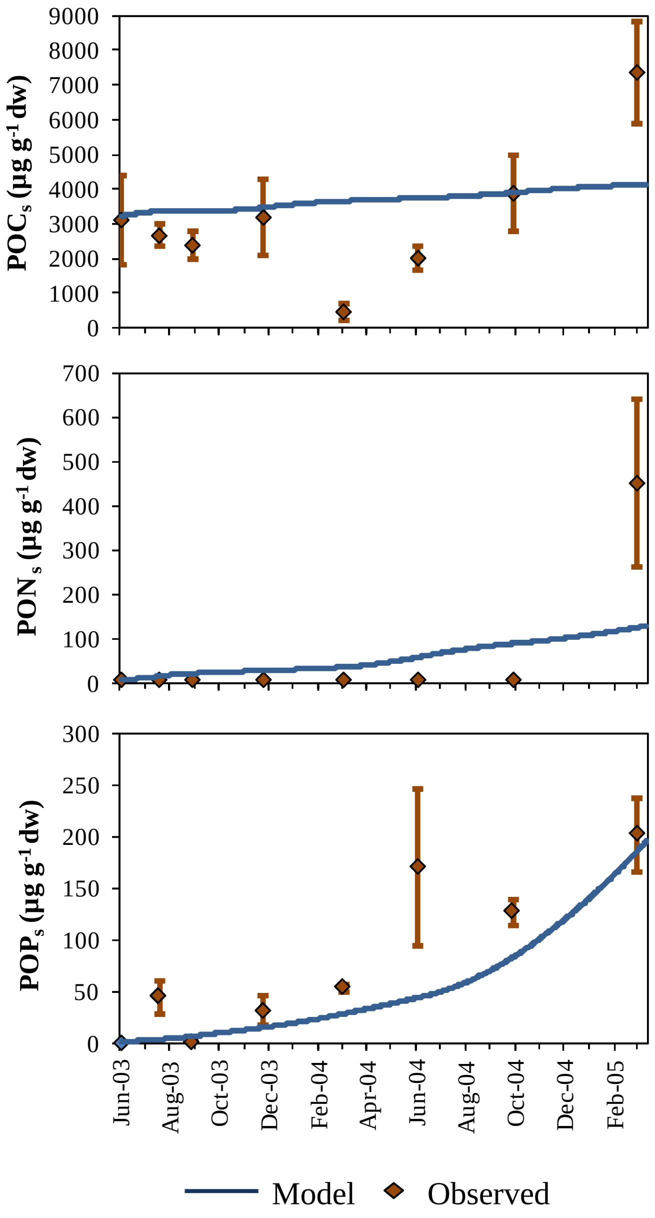


Figure 4.7 – Predicted (line) and observed ± standard deviation (diamonds) organic carbon (POC$_s$), nitrogen (PON$_s$) and phosphorus (POP$_s$) content in the sediments of an earth pond without fish.

3.2.1 Water column variables

A 20% variation (increment and decrement) in model parameters had practically no effect on TPM$_w$ concentrations, but POM$_w$ was positively affected by water exchange rate (*Water exchange*) and negatively affected by its sinking velocity (ν'), which produced on average a 5%

variation on this variable. Water column NH_4^+ concentrations were negatively influenced by pond volume (*V*) and water exchange rate. A 20% increase in pond volume resulted in a 10% decrease in $NH_4^+{}_w$ levels (Table 4.5), while a similar variation in the water exchange rate resulted in a 9% variation in the concentrations of this solute. Oxidized nitrogen forms, on the other hand, were mainly affected by parameters related to nitrification-denitrification processes, such as $knit_w$, $kdenit_w$ and $kdenitO_{2w}$ (Appendix D). A 20% increase in denitrification-related parameters ($kdenit_w$ and $kdenitO_{2w}$) led to a 5 to 6.4% decrease in NO_x levels, whereas $knit_w$ produced the inverse effect (Table 4.5). Other water column variables were only slightly affected by model parameters.

Table 4.5 – Sensitivity analysis for water column state variables. Results are expressed as the % of variation relative to the average value in the standard simulation.

Parameter		**$NH_4^+{}_w$**	**NO_{xw}**
V	+20%	-9.7	-3.0
	-20%	11	4.8
Water exchange	+20%	-9.2	2.8
	-20%	14	-1.9
$knit_w$	+20%	-1.7	4.6
	-20%	1.8	-4.8
$kdenit_w$	+20%	1.9	-6.4
	-20%	-2.2	7.4
$kdenitO_{2w}$	+20%	1.5	-5.0
	-20%	-1.8	6.0

3.2.2 Porewater variables

Table 4.6 summarizes the sensitivity of each porewater variable to selected model parameters. Porewater organic N and P compounds were substantially affected by POM sinking velocity (ν’) and dissolution (αdis_s). A 20% increase in the latter parameter, led to a 14 to 19% increase on porewater DON and DOP, while inorganic N and P forms were practically insensitive to this parameter (< 1% variation). POM sinking velocity (ν’), on the other hand, affected almost all porewater compounds, with the exception of $NH_4^+{}_s$ and $HPO_4^{2-}{}_s$. Sediment characteristics, in particular sediment density (*Sed_density*) and sediment-water ratio (*SedWaterRatio*) had a strong effect on porewater variables, except for $NH_4^+{}_s$ that was mainly affected by porosity (ϕ). Porewater NH_4^+ was also sensitive to diffusion processes, since a 20% increase on the oxygen diffusion coefficient (*DsDO*) increased $NH_4^+{}_s$ concentrations by 16%, while a 20% decrease in the ammonium diffusion coefficient ($DsNH_4^+$) increased its concentrations by 5%. Porewater oxidised nitrogen forms (NO_{xs}) were strongly affected by nitrification-denitrification related-parameters ($knit_s$, $kdenit_s$ and $kdenitO_{2s}$), but temperature can be equally determinant for these compounds since a 20% variation on water temperature produced a 23 to 33% change in NO_x porewater concentrations. Temperature also had a significant impact on dissolved oxygen availability (DO_s) in porewater (maximum variation of 36%). In fact, these two variables seemed closely related since they were generally affected by the same model parameters. For instance, the sediment oxygen diffusion coefficient (*DsDO*) affected both NO_x and DO porewater concentrations; however, the impact of this parameter was higher on NO_{xs} levels (average 34% variation) than on DO_s (20% variation). Both variables were also substantially influenced by DON mineralization-related parameters, such as *MinNd* and $kminO_2d$. A 20% increase in *MinNd* led to a 15% decrease on average NO_{xs} and DO_s levels, whereas $kminO_2d$ produced the reverse effect. With regards to $HPO_4^{2-}{}_s$, concentrations were mostly affected by adsorption/desorption processes, since a 20% variation on k'_a (adsorption rate in anoxic conditions), k_d (desorption rate) and *Pmax* (maximum P adsorption capacity) and on the initial concentrations of Pads produced, on average, a 20% variation on this variable.

Table 4.6 – Sensitivity analysis for porewater state variables. Results are expressed as the % of variation relative to the average value in the standard simulation.

Parameter		**$NH_4^+{}_s$**	**NO_{xs}**	**DON_s**	**$HPO_4^{2-}{}_s$**	**DOP_s**	**DO_s**
Pond_depth	+20%	-1.0	-17	7.4	-2.6	2.4	-16
	-20%	1.0	26	-9.9	3.8	-3.6	25
V	+20%	-0.4	7.0	-4.4	-0.1	-5.9	7.3
	-20%	0.4	-6.4	4.4	0.1	5.9	-6.6
Sed_density	+20%	-1.0	15	-15	-1.0	-12	13
	-20%	1.0	-16	23	1.0	18	-14
Water exchange	+20%	1.0	-5.4	5.0	0	5.1	-5.5
	-20%	-1.0	7.9	-6.6	0	-6.9	8.2
ϕ	+20%	10	18	12	0.4	8.9	9.6
	-20%	-11	-21	-9.0	-0.4	-6.3	-13
SedWaterRatio	+20%	-1.0	-16	21	20	22	-16
	-20%	1.1	25	-21	-20	-22	24
k_T	+20%	0.1	-4.0	2.7	0	3.4	-4.0
	-20%	-0.1	4.1	-2.5	0	-3.3	4.2
k'_a	+20%	0	0	0	-17	0	0
	-20%	0	0	0	25	0	0
Pmax	+20%	0	0	0	-17	0	0
	-20%	0	0	0	26	0	0
k_d	+20%	0	0	0	20	0	0
	-20%	0	0	0	-20	0	0
$DsNH_4^+$	+20%	-4.5	-3.3	0	0	0	0
	-20%	4.8	3.6	0	0	0	0
DsDO	+20%	16	37	-1.7	0.3	0	21
	-20%	-16	-30	1.7	-0.3	0	-20

Table 4.6 – (continued)

Parameter		**$NH_4^+{}_s$**	**NO_{xs}**	**DON_s**	**$HPO_4^{2-}{}_s$**	**DOP_s**	**DO_s**
$knit_s$	+20%	-0.1	19	0	0	0	-0.3
	-20%	0.1	-19	0	0	0	0.3
$kdenit_s$	+20%	0	-16	0	0	0	0
	-20%	0	25	0	0	0	0
$knitO_{2s}$	+20%	0.1	23	0	0	0	0.2
	-20%	-0.2	-16	0	0	0	-0.3
minNd	+20%	0.1	-15	0	0	0	-15
	-20%	-0.2	23	0	0	0	22
$kminO_2d$	+20%	-0.1	18	0	0.3	0	17
	-20%	0.2	-18	0	-0.3	0	-17
v'	+20%	0.1	-11	15	0	12	-9.5
	-20%	-0.2	16	-16	0	-13	13
αdis_s	+20%	0	-1	14	0	19	-1
	-20%	0	1	-14	0	-19	1
Initial conditions							
$NH_4^+{}_s$	+20%	4.1	5.7	0	0	0	-0.2
	-20%	-4.1	-5.7	0	0	0	0.2
Pads	+20%	0	0	0	21	0	0
	-20%	0	0	0	-20	0	0
PON_s	+20%	0	-4.0	0	0	0	-5.4
	-20%	0	4.5	0	0	0	6.5
Forcing functions							
Water temperature	+20%	-1.1	-23	-1.2	0	-1.2	-24
	-20%	1.0	33	1.0	0	1.0	36

3.2.3 Sediment variables

The organic C, N and P contents in earth pond sediments were mostly sensitive to POM sinking velocity and water temperature (Table 4.7). A 20% increase in *v*' increased POC_s, PON_s and POP_s in 5 to 16%, whereas a similar variation on water temperature negatively affected sediment variables, on average by 24%. Sediment characteristics influenced organic C, N and P concentrations in different ways. For instance, a 20% increase in sediment density tends to decrease the organic C, N and P content of pond sediments, while porosity (ϕ) positively affected these variables. Other model parameters also influenced PON_s, POP_s and POC_s but at a lower extent (< 8%) than the parameters referred above.

Table 4.7 – Sensitivity analysis for sediment state variables. Results are expressed as the % of variation relative to the average value in the standard simulation.

Parameter		**POC_s**	**PON_s**	**POP_s**
V	+20%	-2.1	-6.7	-5.7
	-20%	2.1	7.1	5.8
Sed_density	+20%	-5.1	-16	-14
	-20%	7.6	23	21
Water exchange	+20%	1.8	5.9	4.8
	-20%	-2.4	-7.7	-6.5
ϕ	+20%	4.6	14	13
	-20%	-3.5	-11	-9.8
v'	+20%	5.1	16	14
	-20%	-5.4	-17	-15
Forcing functions				
Water temperature	+20%	-25	-23	-23
	-20%	36	30	31

4 Discussion

Like other mathematical models that have successfully predicted water quality in earth ponds (Piedrahita et al. 1984; Hargreaves 1997; Jiménez-Montealegre et al. 2002a; Burford and Lorenzen 2004), the model developed herein was able to reproduce the variability of most water column variables, with the exception of $NH_4^+{}_w$ and DOP_w (Figure 4.4 and Table 4.2). As the variation pattern predicted by the model was principally determined by the concentrations of these compounds in inflowing water (cf. Chapter 3.2), the results suggest that the influence of biogeochemical processes on $NH_4^+{}_w$ and DOP_w concentrations, may superimpose the contribution of inflowing water, to water quality.

The intensification of upward NH_4^+ fluxes (up to 50%) as a result of increased biological activity at the pond bottom during warmer periods (Riise and Roos 1997; Hargreaves 1998), might explain the model inability to predict the $NH_4^+{}_w$ summer peaks since diffusion-related parameters (ϕ and $DsNH_4^+$) were constant over time. To test this hypothesis the previous parameters were increased to values reported in the literature (Aller 1992; Chapelle 1995), but no substantial improvement (< 3% increase) was observed on model results. Besides increasing solute exchange, sediment reworking by benthic fauna also promotes organic matter mineralization at the sediment-water interface, which often induce an increase of water column NH_4^+ concentrations, during summer months (Hargreaves 1998). However, this process does not seem to explain the model inability to reproduce $NH_4^+{}_w$ at warmer periods because after increasing benthic N mineralization rates ($MinN_s$ and $MinNd_s$), no substantial increase (< 2%) was observed in $NH_4^+{}_w$. Likewise, neither water column mineralization or any other biogeochemical process considered in the pelagic module (i.e. nitrification and denitrification), could have been responsible for the lower $NH_4^+{}_w$ predictions in summer months, because when increasing *MinRate* and *kdenit*$_w$ or decreasing *knit*$_w$, the model remained unable to predict $NH_4^+{}_w$ concentrations. Thus, the alternative explanation for these findings is sediment resuspension. Wind-driven water turbulence in shallow earth ponds or benthic fauna activity may increase NH_4^+ concentrations in the water column by promoting its desorption from sediment particles (Riise and Roos 1997; Hargreaves 1998). Although in the present study, sediment resuspension was not calculated due to the lack of data on bottom currents, this process can be easily included in future studies because it is already implemented in the EcoDynamo software (Duarte et al. 2007). One hypothesis for the model overestimation of $NH_4^+{}_w$ concentrations in November 2003 and March 2004 (Figure 4.4) would be primary producers consumption, however the low chlorophyll *a* concentrations in pond water (<3.3 µg L^{-1}) (Serpa, *unpublished results*), exclude phytoplankton uptake. Green macroalgae (*Enteromorpha* spp.), on the other hand, could have been responsible for $NH_4^+{}_w$ uptake because their biomass increases in the Ria Formosa lagoon during the autumn and spring seasons (Aníbal 1998). Moreover, macroalgal photosynthetic activity might explain the discrepancy between measured and predicted DO_w concentrations in March 2004. The higher oxygen availability in field conditions presumably promoted nitrification processes (Hargreaves 1997, 1998; Burford and Lorezen 2004), leading to a NO_{xw} peak that was underestimated by the model. In March 2004, the model also inadequately predicted a maximum in DOP_w concentrations, reducing the accuracy of the simulations. This high value traduces a peak in inflowing water DOP concentrations (cf. Chapter 3.2), which was probably consumed by the biota since DOP can be used as a source of P via enzymatic hydrolysis and/or bacterial decomposition (Sañudo-Whilhemy 2006).

Particulate organic matter (POM_W) was simulated with high accuracy ($p<0.01$; Table 4.2), which is extremely pertinent because settled organic material is the most important source of porewater nutrients in earth ponds (Hargreaves 1998). This idea was also supported by the results of the sensitivity analysis since most porewater variables were strongly affected by POM sinking velocity (ν'). Dissolved organic compounds (DON_s and DOP_s) were particularly sensitive to POM dissolution (Table 4.6), in agreement with what has been described by Hargreaves (1998) and Worsfold et al. (2008). According to these authors, the hydrolysis of POM, including settled phytoplankton cells, is the main source of porewater dissolved organic N and P compounds in aquatic systems. Although this process was accounted for the model, the variability of DON_s and DOP_s concentrations was not well reproduced (Figures 4.5 and 4.6), increasing as organic N and P accumulated in sediments (Figure 4.7). A possible explanation for these results is the absence of bioturbation effects in the model, because the activity of benthic organisms may promote the decrease in DON_s and DOP_s concentrations through the intensification of mineralization and diffusion processes (Burdige and Zheng 1998), or act as source of these compounds as a result of animal excretion (Burdige and Zheng 1998).

In opposition, the variability of $NH_4^+{}_s$ concentrations was predicted with reasonable accuracy, except in March and June 2004 (Figure 4.5) when these values were, respectively, over and underestimated by the model. Ammonium adsorption to negatively charged sediment colloids (Berner 1980; Hargreaves 1998) and bottom seepage (Boyd 1990) have been referred as sinks for porewater NH_4^+, and might explain the extreme low value measured in March 2004. However, as seepage is reduced over time (Boyd 1990), the last hypothesis might be set aside because as ponds had been recently constructed (May 2003), its effects should have been particularly noticeable at the beginning of the experiment. On the other hand, the higher $NH_4^+{}_s$ concentrations measured in June 2004 could have been related to increased benthic fauna activity in warmer periods (cf. Chapter 3.2), since bioturbation is known to stimulate organic matter mineralization (Aller and Aller 1992; Nizzoli et al. 2007; Holmer and Heilskov 2008). However, as porewater NH_4^+ was weakly affected by parameters related to mineralization (Table 4.6), the model could not reproduce the effects of bioturbation on this process. What seems to dictate $NH_4^+{}_s$ concentrations in the model is sediment-water oxygen diffusion (Table 4.6). An increase in the sediment oxygen diffusion coefficient (*DsDO*), increases DO_s concentrations (Table 4.6), which in turn promotes mineralization processes (Appendix A), leading to an increase in $NH_4^+{}_s$. As DO diffusion across the sediment-water interface strongly depends on benthic fauna activities such as ventilation and irrigation (Aller and Aller 1992; Nizzoli et al. 2007), the constant value assumed for *DsDO* presumably prevented the model to simulate the $NH_4^+{}_s$ peak in June 2004. In earth ponds, maximum $NH_4^+{}_s$ concentrations during periods of higher temperatures are usually related to low porewater oxygen solubility since anaerobic conditions promote denitrification processes (Hargreaves 1997, 1998). From the denitrification equation used in the model (Appendix C, eq. 43-44) it would be expected that low DO_s concentrations (Figure 4.8) would lead to increased $NH_4^+{}_s$ concentrations at higher temperatures. However, the low porewater NO_x concentrations predicted by the model must have limited denitrification. Low oxygen predictions (<0.1 mg L^{-1}, Figure 4.8) after June 2004 prevented the model to accurately reproduce NO_{xs} concentrations, since oxygen availability also restricts nitrification rates (Appendix C, eq. 45-46). The inexistence of an extra source of oxygen in the model, either from benthic fauna (cf. Chapter 3.2) or microphytobenthos photosynthetic activity (Serpa et al. 2007b), is an hypothesis for low oxygen predictions (Figure 4.8). The relationship between porewater NO_x and DO concentrations is supported by the results of the sensitivity analysis, since several parameters related to oxygen-dependent processes (e.g. nitrification, DO diffusion and DON mineralization) substantially affected both variables (Table 4.6).

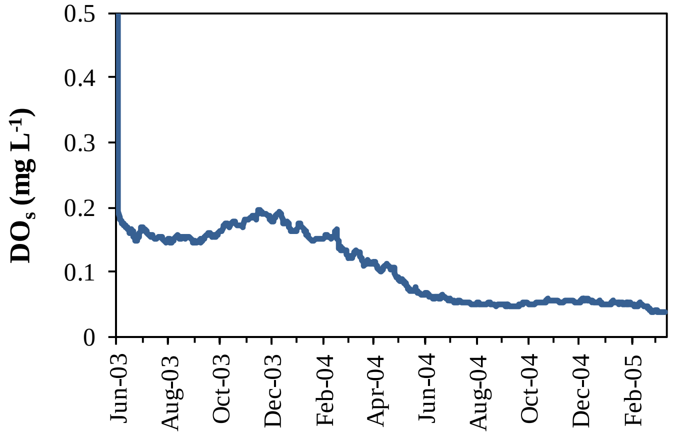


Figure 4.8 – Predicted dissolved oxygen porewater (DO$_s$) concentrations in an earth pond without fish.

According to the model results, oxygen availability also influenced porewater HPO_4^{2-} concentrations. In the sensitivity analysis (Table 4.6), $HPO_4^{2-}{}_s$ was deeply affected by parameters related to adsorption/desorption processes (e.g. k'_a, *Pmax* and k_d). The fact that the adsorption coefficient for anoxic conditions (k'_a) produced a more pronounced effect on $HPO_4^{2-}{}_s$ concentrations than the one for oxic conditions (k_a), suggests that DO$_s$ concentrations (Figure 4.8) were below the oxygen threshold value for P adsorption under anoxic conditions (< 0.1 mg L^{-1}), after June 2004. The overestimation of $HPO_4^{2-}{}_s$ concentrations during the first year of the experiment, suggests that the adsorption rates in field conditions were higher than those calculated by the model, probably because higher oxygen availability in bottom sediments promotes the formation of iron oxides, trapping P into sediments (van Raaphorst and Kloosterhuis 1994; Slomp et al. 1998; Serpa et al. 2007b).

A better model fit was found for organic P in bottom sediments than for carbon and nitrogen (Figure 4.7; Table 4.4). The continuous accumulation of POC$_s$ and PON$_s$ (Figure 4.7) suggests a positive balance between their sources (organic matter deposition) and sinks (mineralization and dissolution). Conversely, in field conditions, PON_s was extremely low during most of the experiment, pointing to a faster degradation kinetics than in the model (Dale and Prego 2002). Another possible explanation is that sediment organic nitrogen might have incorporated in the biota (Jiménez-Montealegre et al. 2002a). In March 2005, the sedimentation of N compounds exceeded decomposition processes, presumably as a result of lower microbial efficiency in poorly oxygenated organically richer sediments (Moriarty 1977; Avnimelech et al. 1992, Avnimelech 1999), which led to a PON_s peak that was not predicted by the model. The discrepancy between predicted and measured POC_s may be associated to microphytobenthos activity. This hypothesis is in line with the findings of other authors that found a close link between the organic C content of aquatic sediments and benthic primary production (Gutiérrez et al. 2000; Serpa et al. 2007b). Bioturbation may also introduce POC_s variations since benthic fauna play an important role in the supply as well as in the mineralization of organic matter in bottom sediments (Heilskov and Holmer 2001; Holmer and Heilskov 2008). Considering the potential effects of benthic primary producers and benthic fauna activity on pond biogeochemistry further combined experimental and modelling studies on these processes are required to improve model performance.

Using the average daily predicted N and P fluxes for the 2-year simulation period, a model-based nutrient-budget approach was followed to identify the main sources and sinks of nutrients in the system (Figure 4.9 and 4.10). Most (>50%) dissolved N compounds available in pond water were supplied by inflowing water, unlike what was previously estimated in a nutrient budget constructed for the same pond (cf. Chapter 3.2), in which sediment-water transport was the main source of dissolved N to the system. The differences between the results of the two approaches may be related to an overestimation of biologically-mediated fluxes in the previous study since these fluxes were extrapolated from a laboratory experiment carried out at two different temperatures, or to the fact that no chemical transformations were considered in the former budget. For dissolved P compounds (Figure 4.10), the model predicted that the major source of P to the system was inflowing water, similarly to what was estimated in the budget presented in Chapter 3.2, but the relative contributions differed between studies. In both approaches, the major pathway of N and P removal from the system was effluent water, even though other sinks, namely phytoplankton and microphytobenthos nutrient uptake, were considered in the budget presented in Chapter 3.2.

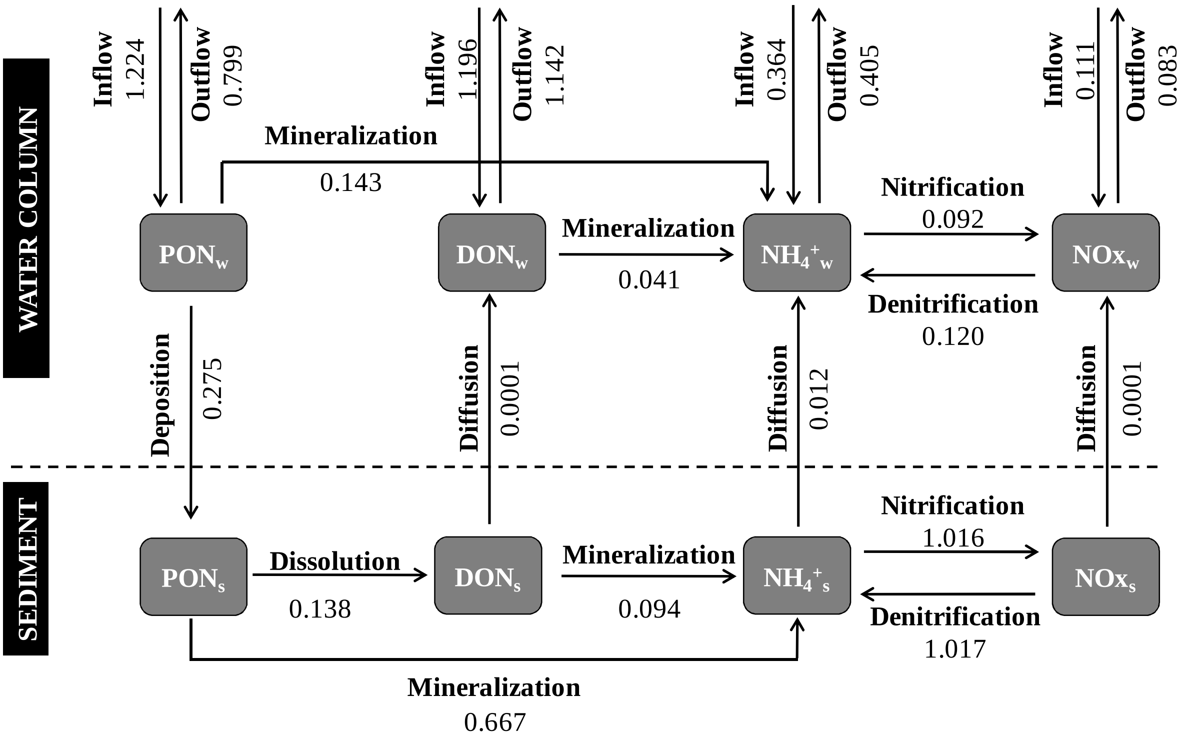


Figure 4.9 – Average daily nitrogen fluxes (µM N d^{-1}), for the 2-year simulation period, in an earth pond without fish.

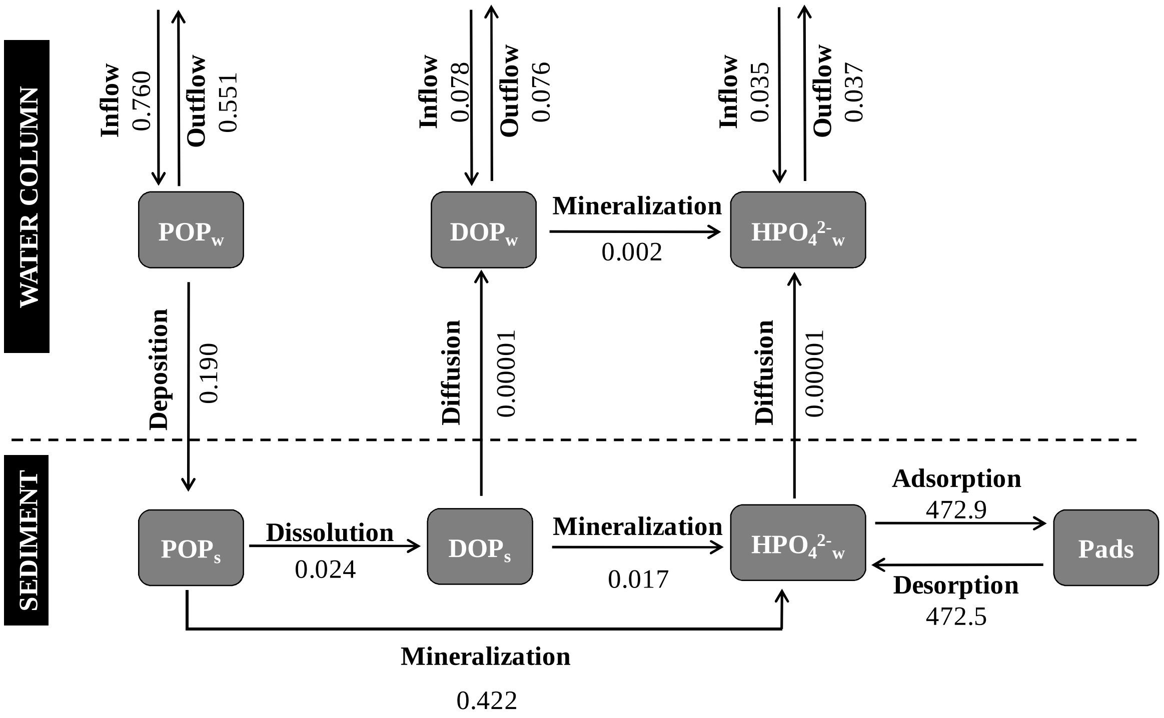


Figure 4.10 – Average daily phosphorus fluxes (µM P d^{-1}), for the 2-year simulation period, in an earth pond without fish.

One of the advantages of using model-based nutrient budgets is that it is possible to investigate the dynamics of the different N and P compounds because chemical transformations are taking into account. For instance, water column NH_4^+ is produced within the system mostly through particulate organic matter mineralization, unlike the other N forms that seemed to be consumed since inputs from inflowing water exceeded the outputs via outflowing water (Figure 4.9). In the benthic system, inorganic nutrients were mostly regenerated by mineralization of particulate organic matter (PON_S and POP_S) settled to the pond bottom (Hargreaves 1998; Alongi et al. 1999; Burford and Lorenzen 2004), whereas the mineralization of dissolved organic N and P forms (DON and DOP), had a minor contribution to porewater $NH_4^+{}_s$ and $HPO_4^{2-}{}_s$ concentrations (respectively, 12 and 3%). Although nitrification and denitrification fluxes (respectively, 1.016 and 1.017 µM N d^{-1}) were relatively tighter, the balance between the two processes favours

$NH_4^+{}_s$ production, unlike what was reported for other aquatic systems (Chapelle 1995; Kittiwanich et al. 2007). The low oxygen availability predicted by the model might account for these differences (Figure 4.8), since it promotes denitrification over nitrification (Appendix C, eq. 43-46). Adsorption/desorption processes was a pathway of inorganic P removal, since average daily adsorption fluxes (472.2 µM P d^{-1}) were higher than desorption fluxes (471.8 µM P d^{-1}), as described by Chapelle (1995) in a Mediterranean coastal lagoon.

Besides biogeochemical processes, the water and sediment quality in earth ponds, might also be affected by the pond structural features (e.g. height of the water column and volume) and by operational parameters such as, water exchange rate (Hargreaves 1998; Burford and Lorenzen 2004). The effects of these parameters on water and sediment variables were tested with the sensitivity analysis (Tables 4.5, 4.6 and 4.7). Increasing pond volume and water exchange rates by 20% promoted the reduction of $NH_4^+{}_w$, on average by 10%, mainly due to a dilution effect, whereas pond depth had little influence on water column variables. In what concerns the sediments, a 20% increase in pond depth was found to negatively affect NO_{xs} and DO_s concentrations by 25%, whereas pond volume and water exchange rates affected mainly the organic C, N and P contents of bottom sediments, although in a divergent way. While, a 20% increase in pond volume reduced the organic C, N and P contents of bottom sediments by 2%, 7% and 6%, respectively, an identical variation in water exchange rate led to an increase in sedimentary compounds because increasing the water flow promotes the increase of organic suspended matter that is “trapped” by deposition within the pond. Information on pond design and management is particularly relevant for Environmental Impact Assessment (EIA) studies for the implementation of new aquaculture units, to ensure the maximization of fish production and minimization of environmental impacts.

In summary, the approach followed in this work resulted in a better understanding of earth pond dynamics, and was crucial for identifying some of the processes (e.g. primary productivity, sediment resuspension and bioturbation) that should be further investigated to overcome model limitations and to ensure its future application to other systems.

5 Conclusions

The model developed herein simulated fairly well the water and sediment quality in an earth pond without fish, constituting a basis for understanding the biogeochemistry of fish earth ponds. During calibration it became clear that, in general, changes in model parameters would not substantially improve model performance, which suggests that further studies are needed on the effects of unaccounted processes such as sediment resuspension as well as primary producers and benthic fauna activity, on nutrient dynamics. Future applications of the present model include its linkage to a fish Dynamic Energy Budget (DEB) model, for which this work will be particularly helpful, since the results of the sensitivity analysis will facilitate model calibration. The similarities between an earth pond without fish and a wastewater treatment pond (Peng 2007) also make this model eligible to be used in this type of systems, to help increasing the removal efficiencies of particulate matter and dissolved N and P compounds.

Acknowledgments

This research was funded by a PhD fellowship from Fundação para a Ciência e Tecnologia (SFRH/BD/27643/2006). Special thanks to Maria de Lourdes Santos for her help in the laboratory work.

Appendix A – General differential equations for the state variables in the pelagic module. The subscript *w* stands for water column variables and processes.

Total particulate matter – TPM$_w$ (mg L^{-1})

$$\frac{dTPM_w}{dt} = TPMin - TPMdep - POM_w \min - TPMout \qquad (1)$$

TPMin	Total particulate matter inflow rate	mg L^{-1} time^{-1}
TPMdep	Total particulate matter deposition rate	
POM$_w$ min	Water column particulate organic matter mineralization	
TPMout	Total particulate matter outflow rate	

Particulate organic matter – POM$_w$ (mg L^{-1})

$$\frac{dPOM_w}{dt} = POMin - POMdep - POMdis_w - POM_w \min - POMout \qquad (2)$$

POMin	Particulate organic matter inflow rate	
POMdep	Particulate organic matter deposition rate	
POM$_w$dis	Water column particulate organic matter dissolution	mg L^{-1} time^{-1}
*POM$_w$*min	Water column particulate organic matter mineralization	
POMout	Particulate organic matter outflow rate	

Water column particulate organic carbon – POC$_w$ (mg L^{-1})

$$\frac{dPOC_w}{dt} = POCin - POCdep - POC_w dis - POC_w \min - POCout \qquad (3)$$

POCin	Particulate organic carbon inflow rate	
POCdep	Particulate organic carbon deposition rate	
POC$_w$dis	Water column particulate organic carbon dissolution	mg L^{-1} time^{-1}
*POC$_w$*min	Water column particulate organic carbon mineralization	
POCout	Particulate organic carbon outflow rate	

Appendix A – (continued)

Water column dissolved organic carbon – DOC$_w$ (µM)

$$\frac{dDOC_w}{dt} = DOCin + \frac{POC_w dis}{Cmass} \pm DOCSedWaterDif - DOC_w \min - DOCout \tag{4}$$

DOCin	Dissolved organic carbon inflow rate	µM time^{-1}
POC$_w$dis	Water column particulate organic carbon dissolution	µM time^{-1}
Cmass	Carbon molar mass	g mol^{-1}
DOCSedWaterDif	Dissolved organic carbon sediment-water diffusion	µM time^{-1}
*DOC$_w$*min	Water column dissolved organic carbon mineralization	µM time^{-1}
DOCout	Dissolved organic carbon outflow rate	µM time^{-1}

Water column particulate organic nitrogen – PON$_w$ (mg L^{-1})

$$\frac{dPON_w}{dt} = PONin - PONdep - PON_w dis - PON_w \min - PONout \tag{5}$$

PONin	Particulate organic nitrogen inflow rate	
PONdep	Particulate organic nitrogen deposition rate	
PON$_w$dis	Water column particulate organic nitrogen dissolution	mg L^{-1} time^{-1}
*PON$_w$*min	Water column particulate organic nitrogen mineralization	
PONout	Particulate organic nitrogen outflow rate	

Water column dissolved organic nitrogen – DON$_w$ (µM)

$$\frac{dDON_w}{dt} = DONin + \frac{PON_w dis}{Nmass} \pm DONSedWaterDif - DON_w \min - DONout \tag{6}$$

DONin	Dissolved organic nitrogen inflow rate	µM time^{-1}
PON$_w$dis	Water column particulate organic nitrogen dissolution	µM time^{-1}
Nmass	Nitrogen molar mass	g mol^{-1}
DONSedWaterDif	Dissolved organic nitrogen sediment-water diffusion	µM time^{-1}
*DON$_w$*min	Water column dissolved organic nitrogen mineralization	µM time^{-1}
DONout	Dissolved organic nitrogen outflow rate	µM time^{-1}

Appendix A – (continued)

Water column ammonium – $NH_4^+{}_w$ (µM)

$$\frac{dNH_4{}^+{}_w}{dt} = NH_4{}^+in + \frac{PON_w \min}{Nmass} + DON_w \min + Denit_w \pm NH_4{}^+SedWaterDif - Nitrif_w - NH_4{}^+out \quad (7)$$

NH_4^+in	Ammonium inflow rate	µM time^{-1}
PON_wmin	Water column particulate organic nitrogen mineralization	µM time^{-1}
Nmass	Nitrogen molar mass	g mol^{-1}
DON_wmin	Water column dissolved organic nitrogen mineralization	µM time^{-1}
$Denit_w$	Water column denitrification	µM time^{-1}
$NH_4^+SedWaterDif$	Ammonium sediment-water diffusion	µM time^{-1}
$Nitrif_w$	Water column nitrification	µM time^{-1}
NH_4^+out	Ammonium outflow rate	µM time^{-1}

Water column nitrogen oxidized forms – NO_{xw} (µM)

$$\frac{dNO_{xw}}{dt} = NO_x in + Nitrif_w \pm NO_x SedWaterDif - Denit_w - Nred_w - NO_x out$$

(8)

NO_xin	Oxidized nitrogen forms inflow rate	
$Nitrif_w$	Water column nitrification	
$NO_xSedWaterDif$	Oxidized nitrogen forms sediment-water diffusion	µM time^{-1}
$Denit_w$	Water column nitrate reduction to ammonium	
$Nred_w$	Water column nitrate reduction to gaseous forms	
$NO_x out$	Oxidized nitrogen forms outflow rate	

Water column particulate organic phosphorus – POP_w (mg L^{-1})

$$\frac{dPOP_w}{dt} = POPin - POPdep - POP_w dis - POP_w \min - POPout \quad (9)$$

POPin	Particulate organic phosphorus inflow rate	
POPdep	Particulate organic phosphorus deposition rate	
POP_wdis	Water column particulate organic phosphorus dissolution	mg L^{-1} time^{-1}
POP_wmin	Water column particulate organic phosphorus mineralization	
POPout	Particulate organic phosphorus outflow rate	

Appendix A – (continued)

Water column dissolved organic phosphorus – DOP_w (µM)

$$\frac{dDOP_w}{dt} = DOPin + \frac{POP_w dis}{Pmass} \pm DOPSedWaterdif - DOP_w \min - DOPout \qquad (10)$$

DOPin	Dissolved organic phosphorus inflow rate	µM $time^{-1}$
$POP_w dis$	Water column particulate organic phosphorus dissolution	µM $time^{-1}$
Pmass	Phosphorus molar mass	g mol^{-1}
DOPSedWaterDif	Dissolved organic phosphorus sediment-water diffusion	µM $time^{-1}$
DOP_wmin	Water column dissolved organic phosphorus mineralization	µM $time^{-1}$
DOPout	Dissolved organic phosphorus outflow rate	µM $time^{-1}$

Water column phosphate – $HPO_4^{2-}{}_w$ (µM)

$$\frac{dHPO_4{}^{2-}{}_w}{dt} = HPO_4{}^{2-}in + \frac{POP_w \min}{Pmass} + DOP_w \min \pm HPO_4{}^{2-}SedWaterDif - HPO_4{}^{2-}out \qquad (11)$$

$HPO_4^{2-}in$	Phosphate inflow rate	µM $time^{-1}$
POP_wmin	Water column particulate organic phosphorus mineralization	
DOP_wmin	Water column dissolved organic phosphorus mineralization	
$HPO_4^{2-}SedWaterDif$	Phosphate sediment-water diffusion	
$HPO_4^{2-}out$	Phosphate outflow rate	

Water column dissolved oxygen – DO_w (mg L^{-1})

$$\frac{dDO_w}{dt} = DOin \pm DOAirWaterDif \pm DOSedWaterDif - Min_w Cons - Nitrif_w Cons - DOout \qquad (12)$$

DOin	Dissolved oxygen inflow rate	mg L^{-1} $time^{-1}$
DOAirWaterDif	Dissolved oxygen air-water diffusion	
DOSedWaterDif	Dissolved oxygen sediment-water diffusion	
$Min_w Cons$	Water column oxygen consumption in mineralization processes	
$Nitrif_w Cons$	Water column oxygen consumption in nitrification processes	
DOout	Dissolved oxygen outflow rate	

Appendix B – General differential equations for the state variables in the benthic module. The subscript *s* stands for sediment variables and processes.

Organic carbon in sediments – POC$_s$ (µg g^{-1} dw)

$$\frac{dPOC_s}{dt} = POCdep \times \frac{Pond\_depth}{Sed\_density \times Sed\_depth} - POC_s\min - POC_s dis \tag{13}$$

POCdep	Particulate organic carbon deposition rate	mg L^{-1} time^{-1}
Pond_depth	Height of the water column	dm
Sed_density	Sediment density	g dm^{-3}
Sed_depth	Sediment layer depth	dm
*POC$_s$*min	Sediment organic carbon mineralization	µg g^{-1} time^{-1}
POC$_s$dis	Sediment organic carbon dissolution	µg g^{-1} time^{-1}

Porewater dissolved organic carbon – DOC$_s$ (µM)

$$\frac{dDOC_s}{dt} = POC_s dis \times \frac{SedWaterRatio}{Cmass} \pm DOCSedWaterDif - DOC_s\min \tag{14}$$

POC$_s$dis	Sediment organic carbon dissolution	µg g^{-1} time^{-1}
SedWaterRatio	Ratio between sediment mass and porewater volume	g L^{-1}
Cmass	Carbon molar mass	g mol^{-1}
DOCSedWaterDif	Dissolved organic carbon sediment-water diffusion	µM time^{-1}
*DOC$_s$*min	Sediment dissolved organic carbon mineralization	µM time^{-1}

Organic nitrogen in sediments – PON$_s$ (µg g^{-1} dw)

$$\frac{dPON_s}{dt} = PONdep \times \frac{Pond\_depth}{Sed\_density \times Sed\_depth} - PON_s\min - PON_s dis \tag{15}$$

PONdep	Particulate organic nitrogen deposition rate	mg L^{-1} time^{-1}
Pond_depth	Height of the water column	dm
Sed_density	Sediment density	g dm^{-3}
Sed_depth	Sediment layer depth	dm
*PON$_s$*min	Sediment organic nitrogen mineralization	µg g^{-1} time^{-1}
PON$_s$dis	Sediment organic nitrogen dissolution	µg g^{-1} time^{-1}

Appendix B – (continued)

Porewater dissolved organic nitrogen – $\mathbf{DON_s}$ (µM)

$$\frac{dDON_s}{dt} = PON_s dis \times \frac{SedWaterRatio}{Nmass} \pm DONSedWaterDif - DON_s \min \tag{16}$$

$PON_s dis$	Sediment organic nitrogen dissolution	µg g^{-1} $time^{-1}$
$SedWaterRatio$	Ratio between sediment mass and porewater volume	g L^{-1}
$Nmass$	Nitrogen molar mass	g mol^{-1}
$DONSedWaterDif$	Dissolved organic nitrogen sediment-water diffusion	µM $time^{-1}$
$DON_s\min$	Sediment dissolved organic nitrogen mineralization	µM $time^{-1}$

Porewater ammonium – $\mathbf{NH_4^+{}_s}$ (µM)

$$\frac{dNH_4^+{}_s}{dt} = PON_s \min \times \frac{SedWaterRatio}{Nmass} + DON_s \min + Denit_s \pm NH_4^+ SedWaterDif - Nitrif_s \tag{17}$$

$PON_s\min$	Sediment organic nitrogen mineralization	µg g^{-1} $time^{-1}$
$SedWaterRatio$	Ratio between sediment mass and porewater volume	g L^{-1}
$Nmass$	Nitrogen molar mass	g mol^{-1}
$DON_s\min$	Sediment dissolved organic nitrogen mineralization	µM $time^{-1}$
$Denit_s$	Sediment denitrification	µM $time^{-1}$
$NH_4^+SedWaterDif$	Ammonium sediment-water diffusion	µM $time^{-1}$
$Nitrif_s$	Sediment nitrification	µM $time^{-1}$

Porewater oxidized nitrogen forms – $\mathbf{NO_{x\,s}}$ (µM)

$$\frac{dNO_{xs}}{dt} = Nitrif_s \pm NO_x SedWaterDif - Denit_s - Nred_s \tag{18}$$

$Nitrif_s$	Sediment nitrification	
$NO_xSedWaterDif$	Oxidized nitrogen forms sediment-water diffusion	µM $time^{-1}$
$Denit_s$	Sediment nitrate reduction to ammonium	
$Nred_s$	Sediment nitrate reduction to gaseous forms	

Appendix B – (continued)

Organic phosphorus in sediments – POP$_s$ (µg g^{-1} dw)

$$\frac{dPOP_s}{dt} = POPdep \times \frac{Pond\_depth}{Sed\_density \times Sed\_depth} - POP_s\min - POP_s dis \tag{19}$$

POPdep	Particulate organic phosphorus deposition rate	mg L^{-1} time^{-1}
Pond_depth	Height of the water column	dm
Sed_density	Sediment density	g dm^{-3}
Sed_depth	Sediment layer depth	dm
POP_smin	Sediment organic phosphorus mineralization	µg g^{-1} time^{-1}
POP_sdis	Sediment organic phosphorus dissolution	µg g^{-1} time^{-1}

Porewater dissolved organic phosphorus – DOP$_s$ (µM)

$$\frac{dDOP_s}{dt} = POP_s dis \times \frac{SedWaterRatio}{Pmass} \pm DOPSedWaterDif - DOP_s\min \tag{20}$$

POP_sdis	Sediment organic phosphorus dissolution	µg g^{-1} time^{-1}
SedWaterRatio	Ratio between sediment mass and porewater volume	g L^{-1}
Pmass	Phosphorus molar mass	g mol^{-1}
DOPSedWaterDif	Dissolved organic phosphorus sediment-water diffusion	µM time^{-1}
DOP_smin	Sediment dissolved organic phosphorus mineralization	µM time^{-1}

Porewater phosphate – HPO$_4^{2-}$$_s$ (µM)

$$\frac{dHPO_4^{2-}{}_s}{dt} = POP_s\min \times \frac{SedWaterRatio}{Pmass} + DOP_s\min + Pdesorp \pm HPO_4^{2-}SedWaterDif - Padsorp \tag{21}$$

POP_smin	Sediment organic phosphorus mineralization	µg g^{-1} time^{-1}
SedWaterRatio	Ratio between sediment mass and porewater volume	g L^{-1}
Pmass	Phosphorus molar mass	g mol^{-1}
DOP_smin	Sediment dissolved organic phosphorus mineralization	µM time^{-1}
Pdesorp	Phosphate desorption	µM time^{-1}
$HPO_4^{2-}SedWaterDif$	Phosphate sediment-water diffusion	µM time^{-1}
Padsorp	Phosphate adsorption	µM time^{-1}

Appendix B – (continued)

Phosphate adsorbed to sediment particles – Pads (µg g^{-1} dw)

$$\frac{dPads}{dt} = Padsorp \times \frac{Pmass}{SedWaterRatio} - Pdesorp \tag{22}$$

Padsorp	Phosphate adsorption	µM time^{-1}
Pdesorp	Phosphate desorption	µg g^{-1} time^{-1}
Pmass	Phosphorus molar mass	g mol^{-1}
SedWaterRatio	Ratio between sediment mass and porewater volume	g L^{-1}

Porewater dissolved oxygen – DO$_s$ (mg L^{-1})

$$\frac{dDO_s}{dt} = \pm DOSedWaterDif - Min_sCons - Nitrif_sCons \tag{23}$$

DOSedWaterDif	Dissolved oxygen sediment-water diffusion	
Min_sCons	Sediment oxygen consumption in mineralization processes	mg L^{-1} time^{-1}
$Nitrif_sCons$	Sediment oxygen consumption in nitrification processes	

Appendix C – Equations for rate processes in the model.

Particulate matter and dissolved compounds inflow rates

$$(Solute)in = \frac{(Solute)\_in}{V} \times Water\_in \qquad (24)$$

(Solute)	*TPM, POM, POC, DOC, PON, DON,* NH_4^+, NO_X, *POP, DOP,* HPO_4^{2-} and *DO*
(Solute)_in	Particulate matter (mg L^{-1}) and dissolved compounds concentrations (µM) in inflowing water
V	Pond volume (m^3)
Water_in	Water inflow rate (m^3 s^{-1})

TPM deposition rate (*TPMdep*)

$$TPMdep = \frac{v}{Pond\_depth} \times TPM_w \qquad (25)$$

v	Particulate matter sinking velocity (m d^{-1})
Pond_depth	Height of the water column (m)

POM mineralization (*POM*$_w$min)

$$POM_w \min = MinRate \times e^{k_{T_w} \times T} \times POM_w \times f(O_2)_w \qquad (26)$$

MinRate	Particulate organic matter mineralization rate (d^{-1})
k_{Tw}	Temperature increasing rate in the water column (ºC^{-1})
T	Water temperature (ºC)

$$f(O_2)_w = \frac{DO_w}{DO_w + k\min O_{2w}} \qquad (27)$$

$kminO_{2w}$	Half-saturation coefficient for O_2 limitation of mineralization in the water column (mg L^{-1})

Adapted from Kittiwanich et al. (2007)

Appendix C – (continued)

Particulate matter and dissolved compounds outflow rates

$$(Solute)out = \frac{(Solute)\_out}{V} \times Water\_out \tag{28}$$

(Solute)	*TPM, POM, POC, DOC, PON, DON,* NH_4^+, NO_X, *POP, DOP,* HPO_4^{2-} and *DO*
(Solute)_out	Particulate matter (mg L^{-1}) and dissolved compounds concentrations (µM) in outflowing water

POM deposition rate (*POMdep*)

$$POMdep = \frac{v'}{Pond\_depth} \times POM_w \tag{29}$$

v'	Particulate organic matter sinking velocity (m d^{-1})

POM dissolution ($POM_w dis$)

$$POMdis = \alpha dis_w \times e^{k_{T_w} \times T} \times POM_w \tag{30}$$

αdis_w	Fraction of particulate organic matter that dissolves in the water column (d^{-1})

From Kittiwanich et al. (2007)

POC deposition rate (*POCdep*)

$$POCdep = \frac{v'}{Pond\_depth} \times POC_w \tag{31}$$

From Chapelle (1995)

Appendix C – (continued)

POC dissolution (*POCdis*)

$$POC_{w,s}dis = \alpha dis_{w,s} \times e^{k_{Tw,s} \times T} \times POC_{w,s} \tag{32}$$

$\alpha dis_{w,s}$	Fraction of particulate organic matter that is dissolved in the water column ($_w$) and in sediments ($_s$)
$k_{Tw,s}$	Temperature increasing rate in the water column ($_w$) and in sediments ($_s$) (ºC^{-1})

POC mineralization (*POC*min)

$$POC_{w,s}\min = MinC_{w,s} \times e^{k_{T\,w,s} \times T} \times POC_{w,s} \times f(O_2)_{w,s} \tag{33}$$

$MinC_{w,s}$	Water column ($_w$) and benthic ($_s$) mineralization rate of organic C at 0ºC (d^{-1})

$$f(O_2)_{w,s} = \frac{DO_w}{DO_w + k\min O_{2w,s}} \tag{34}$$

$kminO_{2w,s}$	Half-saturation coefficient for O_2 limitation of mineralization in the water column ($_w$) and in sediments ($_s$) (mg L^{-1})

Adapted from Chapelle (1995)

Dissolved compounds sediment-water diffusion

$$(Solute)SedWaterDif = \frac{Ds(Solute) \times \left[\left((Solute)_s - (Solute)_w\right)\right] \times \phi}{Sed\_depth \times \Delta x} \tag{35}$$

(Solute)	DOC, DON, NH_4^+, NO_x, DOP, HPO_4^{2-}, DO
Ds(Solute)	Dissolved compounds diffusion coefficients in sediments (cm^2 d^{-1})
$(Solute)_s$	Dissolved compounds concentration in porewater (µM)
$(Solute)_w$	Dissolved compounds concentration in the water column (µM)
ϕ	Porosity
Sed_depth	Sediment layer depth (cm)

$$\Delta x = \frac{Pond\_depth}{2} + \frac{Sed\_depth}{2} \text{ (cm)} \tag{36}$$

Adapted from Berner (1980)

Appendix C – (continued)

DOC mineralization (*DOC*min)

$$DOC_{w,s}\min = MinCd_{w,s}\times e^{k_{T\,w,s}\times T}\times DOC_{w,s}\times g(O_2)_{w,s} \tag{37}$$

$MinCd_{w,s}$	Water column ($_w$) and benthic ($_s$) mineralization rate of DOC (d^{-1})

$$g(O_2)_{w,s} = \frac{DO_{w,s}}{DO_{w,s} + k\min O_2 d_{w,s}} \tag{38}$$

$kminO_2d_{w,s}$	Half-saturation coefficient for O_2 limitation of dissolved compounds mineralization in the water column ($_w$) and in sediments ($_s$) (mg L^{-1})

From Kittiwanich et al. (2007)

PON deposition rate (*PONdep*)

$$PONdep = \frac{v'}{Pond\_depth}\times PON_w \tag{39}$$

From Chapelle (1995)

PON dissolution (*PONdis*)

$$PON_{w,s}dis = \alpha dis_{w,s}\times e^{k_{Tw,s}\times T}\times PON_{w,s} \tag{40}$$

PON mineralization (*PON*min)

$$PON_{w,s}\min = MinN_{w,s}\times e^{k_{T\,w,s}\times T}\times PON_{w,s}\times f(O_2)_{w,s}\times FractionMineralizedtoNH_4{}^{+}{}_{w,s} \tag{41}$$

$MinN_{w,s}$	Water column ($_w$) and benthic ($_s$) mineralization rate of organic N at 0ºC (d^{-1})
$FractionMineralizedtoNH_4{}^{+}{}_{w,s}$	Fraction of PON mineralized to NH_4^+ in the water ($_w$) and in sediments ($_s$)

Adapted from Chapelle (1995)

Appendix C – (continued)

DON mineralization (*DON*min)

$$DON_{w,s}\min = MinNd_{w,s} \times e^{k_{T\,w,s} \times T} \times DON_{w,s} \times g(O_2)_{w,s} \tag{42}$$

$MinNd_{w,s}$	Water column ($_w$) and benthic ($_s$) mineralization rate of DON (d^{-1})

From Kittiwanich et al. (2007)

Nitrate reduction to ammonium (*Denit*)

$$Denit_{w,s} = kdenit_{w,s} \times NO_{x\,w,s} \times e^{k_{T\,w,s} \times T} \times h(O_2)_{w,s} \tag{43}$$

$kdenit_{w,s}$	Water column ($_w$) and benthic ($_s$) denitrification rate at 0ºC (d^{-1})

$$h(O_2)_{w,s} = 1 - \frac{DO_{w,s}}{DO_{w,s} + kdenitO_{2\,w,s}} \tag{44}$$

$kdenitO_{2w,s}$	Half-saturation coefficient for O_2 limitation of denitrification in the water column ($_w$) and in sediments ($_s$) (mg L^{-1})

From Chapelle (1995)

Nitrification (*Nitrif*)

$$Nitrif_{w,s} = knit_{w,s} \times NH_4{}^{+}{}_{w,s} \times e^{k_{T_{w,s}} \times T} \times j(O_2)_{w,s} \tag{45}$$

$knit_{w,s}$	Water column ($_w$) and benthic ($_s$) nitrification rate at 0ºC (d^{-1})

$$j(O_2)_{w,s} = \frac{DO_{w,s}}{DO_{w,s} + knitO_{2w,s}} \tag{46}$$

$knitO_{2w,s}$	Half-saturation coefficient for O_2 limitation of nitrification in the water column ($_w$) and in sediments ($_s$) (mg L^{-1})

From Chapelle (1995)

Appendix C – (continued)

Nitrate reduction to gaseous forms (*Nred*)

$$Nred_{w,s} = Denit_{w,s} \times \alpha denit_{w,s} \tag{47}$$

$\alpha denit_{w,s}$	% of N denitrified to N_2 in the water column ($_w$) and in sediments ($_s$)

From Chapelle (1995)

POP deposition rate (*POPdep*)

$$POPdep = \frac{v'}{Pond\_depth} \times POP_w \tag{48}$$

POP dissolution (*POPdis*)

$$POP_{w,s}dis = \alpha dis_{w,s} \times e^{k_{Tw,s} \times T} \times POP_{w,s} \tag{49}$$

Adapted from Kittiwanich et al. (2007)

POP mineralization (*POP*min)

$$POP_{w,s}\,\mathrm{min} = MinP_{w,s} \times e^{k_{T_{w,s}} \times T} \times POP_{w,s} \times f(O_2)_{w,s} \times FractionMineralizedtoHPO_4^{\;2-}{}_{w,s} \tag{50}$$

$MinP_{w,s}$	Water column ($_w$) and benthic ($_s$) mineralization rate of organic P at 0ºC (d^{-1})
$FractionMineralizedtoHPO_4^{2-}{}_{w,s}$	Fraction of POP mineralized to HPO_4^{2-} in the water column ($_w$) and in sediments ($_s$)

Adapted from Chapelle (1995)

Appendix C – (continued)

DOP mineralization (*DOPmin*)

$$DOP_{w,s}\min = MinPd_{w,s} \times e^{k_{Tw,s}\times T} \times DOP_{w,s} \times g(O_2)_{w,s} \qquad (51)$$

$MinPd_{w,s}$	Water column ($_w$) and benthic ($_s$) mineralization rate of DOP (d^{-1})

From Kittiwanich et al. (2007)

DO air-water diffusion (*DOAirWaterDif*)

$$DOAirWaterDif = k_{ar} \times (SatDO - DO_w) \qquad (52)$$

$SatDO$	Saturation deficit (mg L^{-1})

From Burns (2000)

Oxygen consumption in mineralization processes (*MinCons*)

$$Min_wCons = \left(\frac{PON_w\min}{Nmass} + DON_w\min\right) \times O_2toN\min \qquad (53)$$

$$Min_sCons = \left(PON_s\min \times \frac{SedWaterRatio}{Nmass} + DON_s\min\right) \times O_2toN\min \qquad (54)$$

$O_2toN\min$	Ratio of O_2 consumed per N mineralised (mg O_2 per µmol N)

Adapted from Chapelle (1995)

Oxygen consumption in nitrification processes (*NitrifCons*)

$$Nitrif_{w,s}Cons = Nitrif_{w,s} \times O_2toNnitrif \qquad (55)$$

$O_2toNnitrif$	Ratio of O_2 consumed per N nitrified (mg O_2 per µmol N)

From Chapelle (1995)

Appendix C – (continued)

Phosphate desorption (*Pdesorp*)

$$Pdesorp = k_d \times \frac{Pads}{P\max} \tag{56}$$

k_d	Desorption rate (µg g^{-1} d^{-1})

From Chapelle (1995)

Phosphate adsorption (*Padsorp*)

If $DO_s > O_2thr$ then

$$Padsorp = k_a \times \left(1 - \frac{Pads}{P\max}\right) \times HPO_4{}^{2-}{}_s \tag{57}$$

If $DO_s < O_2thr$ then

$$Padsorp = k_a' \times \left(1 - \frac{Pads}{P\max}\right) \times HPO_4{}^{2-}{}_s \tag{58}$$

O_2thr	Oxygen threshold value for anoxic conditions (mg L^{-1})
k_a	Adsorption rate in oxic conditions (d^{-1})
$Pmax$	Maximum P adsorption capacity for sediments (µg g^{-1} dw)
k_a’	Adsorption rate in anoxic conditions (d^{-1})

From Chapelle (1995)

Appendix D – Model parameters and conversion factors.

Parameter	**Definition**	**Units**	**Calibrated Value**	**Literature Value**	**Reference**
Pond_depth	Height of the water column	m	1.5		
Sed_density	Sediment density	g cm^{-3}	2.3		Field measurements
Sed_depth	Sediment layer depth	cm	2		
SedWaterRatio	Ratio between sediment mass and porewater volume	g L^{-1}	1700		Field measurements
V	Pond volume	m^{3}	742		
Water_in	Water inflow rate	m^{3} s^{-1}	0.001		Field measurements
v	Particulate matter sinking velocity	m d^{-1}	0.02		Field measurements
MinRate	POM mineralization rate	d^{-1}	0.001	0.05	Jørgensen et al. (1991)
k_{Tw}	Temperature increasing rate in the water column	ºC^{-1}	0.07	0.07	Ruardji and van Raaphorst (1995)
$kminO_{2w}$	Half-saturation coefficient for O_2 limitation of mineralization in the water column	mg L^{-1}	0.5	0.5	Henriksen and Kemp (1988)
Water_out	Water outflow rate	m^{3} s^{-1}	0.001		Field measurements
v'	Particulate organic matter sinking velocity	m d^{-1}	0.06		Field measurements
αdis_w	Fraction of POM that is dissolved in the water column	d^{-1}	0	0.02	Kawamiya et al. (1995)
k_{Ts}	Temperature increasing rate in sediments	ºC^{-1}	0.01	0.07	Ruardji and van Raaphorst (1995)
$MinC_w$	Water column mineralization of organic C at 0ºC	d^{-1}	0.001		
$MinC_s$	Benthic mineralization of organic C at 0ºC	d^{-1}	0.001		
$kminO_{2s}$	Half-saturation coefficient for O_2 limitation of mineralization in sediments	mg L^{-1}	0.5	0.5	Henriksen and Kemp (1988)

Appendix D – (continued)

Parameter	Definition	Units	Calibrated Value	Literature Value	Reference
$DsDON$	Diffusion coefficient for dissolved organic nitrogen in sediments	$cm^{-2}\ d^{-1}$	0.00864	0.00864	Burdige et al. (1992)
				0.8	Kittiwanich et al. (2007)
$DsNH_4^+$	Diffusion coefficient for ammonium in sediments	$cm^{-2}\ d^{-1}$	5.788	5.788	Laboratory experiments
				0.847	Kittiwanich et al. (2007)
$DsNO_x$	Diffusion coefficient for nitrate in sediments	$cm^{-2}\ d^{-1}$	0.432	4.32	Chapelle (1995)
				1.642	Jiménez-Montealegre et al. (2002a)
				0.845	Kittiwanich et al. (2007)
$DsDOP$	Diffusion coefficient for dissolved organic phosphorus in sediments	$cm^{-2}\ d^{-1}$	0.00864	0.00864	Burdige et al. (1992)
				0.7	Kittiwanich et al. (2007)
$DsHPO_4^{2-}$	Diffusion coefficient for phosphate in sediments	$cm^{-2}\ d^{-1}$	0.1728	4.32	Chapelle (1995)
				0.5	Kittiwanich et al. (2007)
$DsDO$	Diffusion coefficient for oxygen in sediments	$cm^{-2}\ d^{-1}$	17.28	21.6	Chapelle (1995)
				0.2	Kittiwanich et al. (2007)
ϕ	Porosity		0.4		Field measurements
$MinCd_w$	Water column mineralization rate of DOC	d^{-1}	0		Calibrated
$MinCd_s$	Benthic ($_s$) mineralization rate of DOC	d^{-1}	0.005		Calibrated
$kminO_2d_w$	Half-saturation coefficient for O_2 limitation of dissolved compounds mineralization in the water column	$mg\ L^{-1}$	0.5	0.5	Chapelle et al. (2000)

Appendix D – (continued)

Parameter	Definition	Units	Calibrated Value	Literature Value	Reference
$kminO_2d_s$	Half-saturation coefficient for O_2 limitation of dissolved compounds mineralization in sediments	mg L^{-1}	0.5	0.5	Chapelle et al. (2000)
$MinN_w$	Water column mineralization of organic N at 0ºC	d^{-1}	0.005	0.01	Kawamiya et al. (1995)
$MinN_s$	Benthic mineralization of organic N at 0ºC	d^{-1}	0.001	0.005	Billen and Lancelot (1988)
				0.003 - 0.05	Jamu and Piedrahita (2002)
$FractionMineralizedtoNH_4{}^+{}_w$	Fraction of PON mineralized to NH_4^+ in the water column		1		
$FractionMineralizedtoNH_4{}^+{}_s$	Fraction of PON mineralized to NH_4^+ in sediments		0.6		
αdis_s	Fraction of POM that is dissolved in sediments	d^{-1}	0.0001	0.07	Kittiwanich et al. (2007)
$MinNd_w$	Water column mineralization rate of DON	d^{-1}	0	0.01	Kawamiya et al. (1995)
$MinNd_s$	Benthic mineralization rate of DON	d^{-1}	0.005	0.9	Kittiwanich et al. (2007)
$kdenit_w$	Water column denitrification rate at 0ºC	d^{-1}	0.2	0.08	Kittiwanich et al. (2007)
$kdenit_s$	Benthic denitrification rate at 0ºC	d^{-1}	0.2	0.3	Billen and Lancelot (1988)
				0.05-10	Jamu and Piedrahita (2002)
				9.0	Kittiwanich et al. (2007)
$kdenitO_{2w}$	Water column half-saturation coefficient for O_2 limitation of denitrification	mg L^{-1}	2	2	Kittiwanich et al. (2007)

Appendix D – (continued)

Parameter	Definition	Units	Calibrated Value	Literature Value	Reference
$kdenitO_{2s}$	Sediment half-saturation coefficient for O_2 limitation of denitrification	mg L^{-1}	2	2	Chapelle (1995)
$knit_w$	Water column nitrification rate at 0ºC	d^{-1}	0.01	0.01	Jiménez-Montealegre et al. (2002a)
				0.08	Kittiwanich et al. (2007)
$knit_s$	Benthic nitrification rate at 0ºC	d^{-1}	0.5	0.2	Henriksen and Kemp (1988)
				0.05-0.1	Jamu and Piedrahita (2002)
				0.24	Jiménez-Montealegre et al. (2002a)
				7	Kittiwanich et al. (2007)
$knitO_{2w}$	Half-saturation coefficient for O_2 limitation of nitrification in the water column	mg L^{-1}	4	2	Kittiwanich et al. (2007)
$knitO_{2s}$	Half-saturation coefficient for O_2 limitation of nitrification in sediments	mg L^{-1}	4	4	Chapelle (1995)
$\alpha denit_w$	% of N denitrified into N_2 in the water column		0	0.01	Kittiwanich et al. (2007)
$\alpha denit_s$	% of N denitrified into N_2 in sediments		0.4	0.6	Billen and Lancelot (1988)
$MinP_w$	Water column mineralization of organic P at 0ºC	d^{-1}	0	0.02	Kawamiya et al. (1995)
$MinP_s$	Benthic mineralization of organic P at 0ºC	d^{-1}	0.001	0.005	van der Molen (1991)
$FractionMineralizedtoHPO_4{}^{2-}{}_w$	Fraction of PON mineralized to HPO_4^{2-} in the water column		1		

Appendix D – (continued)

Parameter	**Definition**	**Units**	**Calibrated Value**	**Literature Value**	**Reference**
$FractionMineralizedtoHPO_4^{2-}{}_s$	Fraction of PON mineralized to HPO_4^{2-} in sediments		1		
$MinPd_w$	Water column mineralization rate of DOP	d^{-1}	0	0.02	Kawamiya et al. (1995)
$MinPd_s$	Benthic mineralization rate of DOP	d^{-1}	0.005	0.7	Kawamiya et al. (1995)
k_{ar}	Raeration coefficient	d^{-1}	0		
$SatDO$	Saturation deficit	$mg\ L^{-1}$			
$O_2toNmin$	Ratio of O_2 consumed per N mineralised	mg O_2 per µmol N	0.212	0.212	Chapelle (1995)
$O_2toNnitrif$	Ratio of O_2 consumed per N nitrified	mg O_2 per µmol N	0.064	0.064	Chapelle (1995)
k_d	Desorption rate	$\mu g\ g^{-1}$ dw d^{-1}	80	80	Furumai et al. (1989)
O_2thr	Oxygen threshold value for anoxic conditions	$mg\ L^{-1}$	0.08	0.2	Chapelle (1995)
k_a	Adsorption rate in oxic conditions	d^{-1}	100	200	Furumai et al. (1989)
$Pmax$	Maximum P adsorption capacity for sediments	$\mu g\ g^{-1}$ dw	300	685	Furumai et al. (1989)
				500	Serpa et al. (2007a)
k_a’	Adsorption rate in anoxic conditions	d^{-1}	40	40	van Raaphorst et al. (1992)

Chapter 5

Modelling the growth of white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) in semi-intensive earth ponds using the Dynamic Energy Budget approach

Serpa D., Pousão Ferreira P., Ferreira H., Cancela da Fonseca L., Dinis M. T., Duarte P. Modelling the growth of white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) in semi-intensive fishponds using the Dynamic Energy Budget approach. *Submitted to Journal of Sea Research*.

Abstract

Fish growth models may help understanding the influence of environmental, physiological and husbandry factors on fish production, providing crucial information to maximize the growth rates of cultivated species. The main objectives of this work were to: i) develop and implement an Individual Based Model using a Dynamic Energy Budget (IBM-DEB) approach to simulate the growth of two commercially important Sparidae species in semi-intensive earth ponds, the white seabream (*Diplodus sargus*) which is considered as a potential candidate for Mediterranean aquaculture and the gilthead seabream (*Sparus aurata*) that has been cultivated since the early 80's; ii) evaluate which model parameters are more likely to affect fish performance, and iii) investigate which parameters might account for the growth differences between the two Sparidae. The model may be run in two modes: the "state variable" mode, in which an average fish is simulated with a particular parameter set and the "Individual Based Model" (IBM) mode that simulates a population of *n* fishes, each with its specific parameter set assigned randomly. The IBM mode has the advantage of allowing a quick model calibration and an evaluation of the parameter sets that produce the best fit between predicted and observed fish growth. Observations on the variation of white seabream and gilthead seabream length and weight during a production cycle were obtained from growth trials carried out in earth ponds. Results revealed that the model reproduces reasonably well the growth of the two species. Fish performance was mainly affected by parameters related to reserves utilization and feed ingestion/absorption, suggesting that special attention should be taken in the estimation of these parameters, particularly when applying this model to other species. Comparing the DEB parameters for the two seabreams, it seems that the white seabream low growth rates are a reflection of higher maintenance costs and a lower feed absorption efficiency. Hence, the development of new feed formulations may determine the success of white seabream production in semi-intensive earth ponds.

1 Introduction

During the last decade, Mediterranean semi-intensive fish farming has been struggling with economic difficulties as a consequence of the overproduction and saturation of the market with traditionally cultivated fish species, like the gilthead seabream (*Sparus aurata*) and the seabass (*Dicentrarchus labrax*) (SEACASE 2009; Barazi-Yeroulanos 2010). This situation has led to a sharp decrease in market prices for these species, to a point that revenues hardly compensate for the production costs (Barazi-Yeroulanos 2010). Some of the solutions that have been proposed by researchers and commercial groups to overcome these problems include the optimisation of production for traditionally cultivated species (e.g. by investing in higher market sizes) and product diversification (SEACASE 2009; Barazi-Yeroulanos 2010).

By predicting the effects of environmental (e.g. temperature), physiological (e.g. assimilation and excretion rates) and husbandry factors (e.g. feeding rates) on fish performance, growth models may be of considerable help for the aquaculture industry, to maximize the growth rates and efficiencies of cultivated fish (Alunno-Bruscia et al. 2009; van der Veer et al. 2009). Several bioenergetic models have been developed to simulate fish growth. Most of these models (Yi 1998; Hernández et al. 2003; Zhou et al. 2005; Libralato and Solidoro 2008; Moss et al. 2009) are Static Energy Budgets (SEB) consisting of a set of allometric functions that describe the relationships between the physiological rates (e.g. food consumption, growth) and size of a particular species as a function of abiotic factors (e.g. temperature, salinity) (van der Veer et al. 2009). Despite being widely applied in the aquaculture context, SEB models lack generality as they are not based on physiological principles (van der Veer et al., 2009).

Dynamic Energy Budget (DEB) models, on the other hand, are mechanistic models that rely on simple physiological principles common to all species (Alunno-Bruscia et al. 2009), so that they are able to predict the development, growth and reproduction of an organism in a dynamic environment (Pecquerie et al. 2011). Another advantage of DEB models is that the diversity between species can be captured in differences in a small number of parameters (Kooijman 2000). Given its non-species specificity and mechanistic rules, the DEB theory has been widely applied and successfully tested for a large number of fish species and for various research purposes. Some of the applications of this approach include the simulation of growth and reproduction for flatfish (van der Veer et al. 2003, 2009) and anchovy (Pecquerie et al. 2009) within variable environmental conditions, the forecasting of the metabolic effects of radionuclids (uranium) on individual zebrafish and the consequences at the population level (Augustine et al. 2011), the description of biological factors influencing persistent organic pollutants (PCBs) bioaccumulation in European hake (Bodiguel et al. 2009) and common sole (Eichinger et al. 2010) through fish ontogeny and, the description of the full lifecycle of ecologically and economically important migratory fish species such as, the Pacific salmon (Pecquerie et al. 2011) and the Pacific bluefin tuna (Jusup et al. 2011).

To build a DEB model it is necessary to define parameters related to physiological processes such as feeding, assimilation, storage, maintenance and growth (Kooijman 2000, 2010), which can be inferred from experimental as well as literature data (van der Veer et al. 2001; Bodiguel et al. 2009; Eichinger et al. 2010). However, for some species there is a lack of information to estimate all parameters. To surpass this problem, Duarte et al. (2010) proposed the use of an Individual Based Model (IBM) as a practical calibration procedure, mostly because when an IBM is run for a large number of individuals, each with its own parameter set, it is possible to select the parameter sets that produce the best fit between predicted and observed fish data.

Besides facilitating model calibration, when the proper parameter set is selected, the IBM may be run for a single individual that will represent the average fish in a population, making them suitable to be used in a more complex ecosystem model (Duarte et al. 2010).

In the present work, an IBM using the DEB framework was implemented to simulate the growth of two commercially important Sparidae: the white seabream (*Diplodus sargus*) and the gilthead seabream. While the latter species is the most important fish in Mediterranean aquaculture (Barazi-Yeroulanos 2010; FAO 2011b), with a well developed production technology and maximized growth rates, the white seabream is still a candidate species since only small scale production has been achieved so far (Sá et al. 2006; Golomazou et al. 2006). When cultivating a new species, the lack of knowledge on its optimal physical conditions and nutritional requirements often compromises its growth in captivity (Cejas et al. 2004; Golomazou et al. 2006). Nevertheless, the white seabream is still able to captivate fish farmer’s interest due to its high market value and highly appreciated flesh (Cejas et al. 2004; Sá et al. 2006). The main purpose in building the present model was to provide information on how to optimize fish production in semi-intensive systems, by using the fully validated model to: i) evaluate which of the DEB parameters are more likely to affect fish performance and to ii) investigate which parameters might account for the growth differences between the two Sparidae species.

2 Species information

The white seabream and the gilthead seabream are two members of the Sparidae family that may be found along the Mediterranean Sea and the eastern Atlantic Ocean (FishBase 2010). These sparids are demersal fishes living in a variety of costal habitats, from rocky shores to sandy bottoms (Bargelloni et al. 2005). Feeding strategies differ between the two species; while the white seabream is an omnivorous species, feeding on benthic invertebrates (mainly small crustaceans, molluscs and sea urchins) and on algae (Figueiredo et al. 2005; Leitão et al. 2007), the gilthead seabream is mainly carnivorous feeding on molluscs, crustaceans and fish (Aksnes et al. 1997). In its natural environment, the gilthead seabream may reach a maximum total length and weight of respectively, 70 cm and 17.2 kg (FishBase 2010), whereas the white seabream is more of a slow grower since the maximum total length and weight ever recorded for this species in the wild was 49 cm and 1.87 kg (FishBase 2010).

3 Methodology

The fish growth model consists of an Individual Based Model (IBM) based on the standard DEB theory (Kooijman 2000; 2010). The model may be run in two modes: i) the IBM mode that simulates a population of *n* fishes, each with its specific parameter set assigned randomly and ii) the “state variable” mode, in which an average fish is simulated with a particular parameter set (Duarte et al. 2010). The strategy in this work was to use the IBM mode for model calibration, and then to select the parameter sets that produced the best fit between predicted and observed fish growth to calibrate the “state variable” model.

3.1 The DEB theory

The DEB theory (Kooijman 2000, 2010) describes the rates at which organisms utilize and assimilate energy from food for maintenance, growth, development and reproduction (Figure 5.1). These rates depend on the state of the organism (size, sex and maturity status) and vary as a function of food availability and temperature (Kooijman 2000). In a standard DEB model, food ingestion depends on food availability and is proportional to the organisms’ structural area. Part of the energy in the ingested food is lost through feces while the remainder is assimilated and stored in the reserves compartment. A fixed fraction κ of mobilised reserves is allocated for growth and somatic maintenance, with a priority for maintenance. The remaining energy fraction

(1−κ) is spent on maturity maintenance plus maturation in embryos and juveniles or, reproduction (gamete production and spawning), in adults. The flux of reserves that is allocated to maturation and reproduction is temporally stored in a buffer (reproductive tissue), which is emptied during spawning. In the DEB theory, the embryos correspond to the life stage in which individuals neither feed nor reproduce; the juveniles feed but do not reproduce and adults, both feed and reproduce. At each life stage, it is assumed that an individual can be characterized by two main body fractions: structural biovolume (somatic tissue) and reserves. The chemical composition of each fraction is assumed to remain constant according to the "strong homeostasis" concept (Kooijman 2010). In the DEB theory, the chemical composition of the individual, as a whole, is also maintained constant as long as substrate availability in the environment remains constant, a concept that is known as "weak homeostasis" (Kooijman 2010).

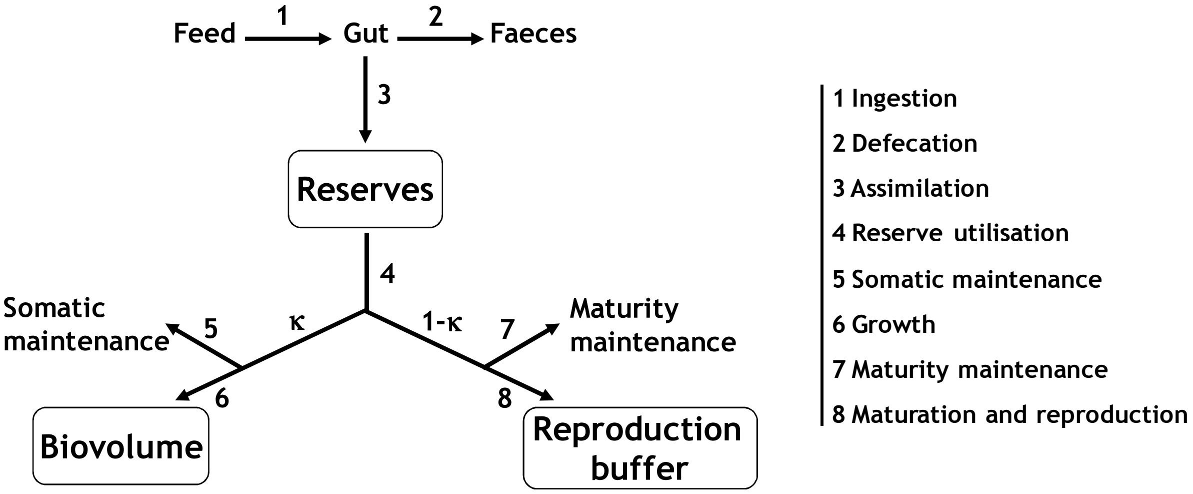


Figure 5.1 – Energy flow through an organism in a standard DEB model. Arrows indicate rates and squares are state variables.

3.2 Model development

The model developed herein simulates fish growth during its grow-out phase in semi-intensive earth ponds. In this phase, fishponds are stocked with juveniles (≈7 g) of white seabream and gilthead seabream, which are fattened until they reach the commercial size (350 - 400g). As sexual maturity, for both species, is attained at the end of the production cycle (FishBase 2010), reproduction was not included in the model.

3.2.1 Model equations

The notation and symbols presented in the next sections follow that of Kooijman (2000). Variables are expressed per unit of structural volume with square brackets [ ], or per unit of surface area with braces { }. All rates have dots, indicating the dimension per time.

3.2.1.1 Ingestion

The ingestion rate, $\dot{P}_X$ (J d^{-1}), is proportional to the surface area of the structural body volume ($V^{2/3}$, in cm^2) and depends upon food density in the environment and on temperature Kooijman 2000):

$$\dot{P}_X = \{\dot{P}_{Xm}\} \times f \times V^{2/3} \times k(T) \qquad (1)$$

where $\{\dot{P}_{Xm}\}$ is the maximum surface area-specific ingestion rate (J cm^{-2} d^{-1}); *f* is the scaled functional response (dimensionless) that varies between 0 (starvation) and 1 (optimal feeding conditions) and *k(T)* is the Arrhenius temperature limitation as described below (cf. - Section 3.2.1.5).

In the standard DEB theory, the scaled functional response (*f*) is defined by a Holling-type II equation:

$$f_1 = \frac{X}{X + X_K} \tag{2}$$

where X represents food density and X_K is the half-saturation constant, i.e. the food density at which ingestion rate is half its maximum value. However, in fishponds, the amount of food supplied does not necessarily represent the one that is available for fish because of pellets sedimentation and decay. Unless cultivated species can feed at the bottom, pellets become unavailable for fish as they reach the sediments. This is particularly relevant for rations with high sinking rates (0.035 ± 0.030 m s^{-1}) and for low-depth ponds (1.5 m), as is the case of the present study, because feeding is restricted to short time periods. Due to the difficulty to determine food density, the scaled functional response (*f*) was estimated using two different approaches. One of these approaches consisted in assuming a constant food density and then changing X_K until the average scaled functional response (f_1) that best described the growth of the two species was found (Cardoso et al. 2006; Freitas et al. 2009; Zimmer et al. 2011). In the second approach, it was assumed that pellets concentration decrease according to a first order process as a function of food sedimentation rate. Therefore, f_2 was calculated as follows:

$$f_2 = \frac{X_0 \exp^{(-FoodSedRate \times t)}}{X_0 \exp^{(-FoodSedRate \times t)} + X_K} \tag{3}$$

The time integrated f_2 is:

$$\bar{f}_2 = \frac{1}{\Delta t} \int_{t=t_0}^{t=t_1} \frac{X_0 \exp^{(-FoodSedRate \times t)}}{X_0 \exp^{(-FoodSedRate \times t)} + X_K} dt \tag{4}$$

With the following solution:

$$\bar{f}_2 = \frac{\ln\left[X_0 \exp^{(-FoodSedRate \times t_0)} + X_K\right] - \ln\left[X_0 \exp^{(-FoodSedRate \times t_1)} + X_K\right]}{FoodSedRate \times \Delta t} \tag{5}$$

where X_0 is the food density (mg L^{-1}) at the beginning of each time step (t_0, seconds), *FoodSedRate* is the pellets sedimentation rate (0.035 m s^{-1}), X_K is the food density at which ingestion rate is half its maximum (Tables 5.1 and 5.2), t_1 is the time corresponding to the end of each time step (seconds), and Δt is the time step in the simulations (1 hour). For this approach, the model requires hourly time series of food concentrations, in which a value exists for the specific hours that food is supplied.

Model simulations using f_1 and f_2 will be referred ahead as Simulations1 and 2, respectively.

3.2.1.2 Assimilation

The assimilation rate, $\dot{p}_A$ (J d^{-1}), i.e. the rate at which food is converted into reserves, depends on food availability and on the organisms' structural surface area, being calculated as follows:

$$\dot{p}_A = \{\dot{p}_{Am}\} \times f \times V^{2/3} \times k(T) \tag{6}$$

where $\{\dot{p}_{Am}\}$ is the maximum surface area-specific assimilation rate (J cm^{-2} d^{-1}). Since the ratio $\{\dot{p}_{Am}\}/\{\dot{P}_{Xm}\}$ gives the absorption efficiency (κ_X), the assimilation rate may be expressed as:

$$\dot{p}_A = \kappa_X \times \{\dot{P}_{Xm}\} \times f \times V^{2/3} \times k(T) \tag{7}$$

3.2.1.3 Reserves utilization

Assimilated energy is integrated in a reserve pool, E, whose dynamics is given by the equation:

$$\frac{dE}{dt} = \dot{p}_A - \dot{p}_C \tag{8}$$

where $\dot{p}_C$ (J d^{-1}) denotes the energy mobilisation rate. The rate, at which energy is mobilized from the reserves and is allocated to somatic maintenance and growth (κ fraction) and to development or reproduction ($1-\kappa$), may be calculated according to the following equation (Kooijman 2000):

$$\dot{P}_C = \frac{[E]}{[E_G] + \kappa[E]} \left(\frac{\{\dot{p}_{Am}\}[E_G]V^{2/3}}{[E_m]} + [\dot{p}_M]V \right) k(T) \tag{9}$$

where $[E]$ stands for reserve density ($[E]$= E/V, J cm^{-3}), $[E_G]$ is the volume-specific costs for structural growth (J cm^{-3}), κ is the fraction of energy spent on soma (i.e. somatic maintenance plus growth), $[E_m]$ is the maximum energy density in the reserve compartment (J cm^{-3}), [$\dot{p}_M$] is the volume-specific costs for maintenance (J cm^{-3} d^{-1}) and V is the structural body volume (cm^3).

3.2.1.4 Growth

Growth is the conversion of reserves into structure (Kooijman 2000, 2010). This process ceases whenever maintenance costs cannot be covered by reserves since maintenance has priority over growth. From the DEB energy allocation rule (κ-rule), the growth of structural biovolume is given by:

$$\frac{dV}{dt}=\frac{\kappa\{\dot{p}_{Am}\}[E]V^{2/3}}{[E_m]([E_G]+\kappa[E])}-\frac{[\dot{p}_M]V}{[E_G]+\kappa[E]} \tag{10}$$

For an organism that does not change in shape during growth, i.e. an isomorphic organism (Kooijman 2010), total length (L, cm) may be related to structural volume using the shape coefficient (δ_m, dimensionless):

$$L=\frac{V^{1/3}}{\delta_m} \tag{11}$$

In a standard DEB model, it is assumed that δ_m is species-specific and is constant for isomorphs, however the DEB theory also allows for changes in shape (Kooijman 2010).

Fish wet weight, W (g), was obtained by summing the 2 compartments, the reserves, E (J), and structural volume, V (cm^3), after conversion into mass:

$$W=V\rho_1+\left(\frac{E}{\alpha_E}\rho_2\right) \tag{12}$$

where ρ_1 is the specific density (1 g cm^{-3}) of the somatic tissue (van der Veer et al. 2001), ρ_2 is the density assumed for the reserves tissue (1 g cm^{-3}) and α_E is the reserves energetic value per unit of volume, which was assumed to be equal to the $[E_G]$ value.

3.2.1.5 Temperature limitation

Since temperature influences all physiological rates, the Arrhenius limitation was used to simulate the temperature effect (van der Veer et al. 2001):

$$k(T) = \dot{k}_1(T)\exp\left\{\frac{T_A}{T_1} - \frac{T_A}{T}\right\} \tag{13}$$

where $\dot{k}_1$ is a physiological rate (i.e ingestion, assimilation and maintenance rates), T is ambient temperature (K), T_1 is a chosen reference temperature (K) and T_A is a species-specific coefficient, the so-called Arrhenius temperature (K).

3.2.2 Parameter estimation

In order to run the model on the IBM mode, it was necessary to define ranges for each of the DEB parameters (Tables 5.1 and 5.2). In the present work, these ranges were defined primarily from experimental data as described in the following sections, but literature data was also used.

3.2.2.1 Maximum surface area-specific ingestion rate ($\{\dot{P}_{Xm}\}$)

The maximum surface area-specific ingestion rate of white seabream was estimated from microcosm experiments, using daily food intake data as a function of the volumetric length ($V^{2/3}$) of white seabreams fed *ad libitum* at constant temperatures, from 17ºC to 25ºC. A similar approach was used for gilthead seabream, using the data of Lupatsch et al. (2003).

Table 5.1 – DEB parameters for white seabream.

Parameter	Description	Value or Range	Units	Source
$\{\dot{P}_{Xm}\}$	Maximum surface-area specific ingestion rate	930 – 1504$^{(a)}$ 1200 – 1600$^{(b)}$	J cm^{-2} d^{-1}	Experimental data Calibrated
X_K	Half-saturation coefficient	0.5– 7.5$^{(a)}$ 0.1 – 0.6$^{(b)}$	mg L^{-1}	Calibrated
κ_X	Absorption efficiency	0.70 – 0.80		Experimental data
$\{\dot{p}_{Am}\}$	Maximum surface-area specific assimilation rate	651 – 1203	J cm^{-2} d^{-1}	Experimental data
κ	Fraction of reserves allocated to soma	0.58 – 0.85		Bodiguel et al. (2009) van der Veer et al. (2001, 2009)
$[E_G]$	Volume-specific costs for growth	5600 – 7563	J cm^{-3}	Bodiguel et al. (2009) van der Veer et al. (2009)
$[E_m]$	Maximum energy storage density	1505 – 2903	J cm^{-3}	Bodiguel et al. (2009) Eichinger et al. (2010)
$[\dot{p}_M]$	Volume-specific costs for maintenance	52 – 60	J cm^{-3} d^{-1}	Experimental data
δ_m	Shape coefficient	0.255 – 0.264	Dimensionless	Field data
T_A	Arrhenius temperature	7674 – 10914	K	Experimental data
T_1	Reference temperature	293.15	K	Fixed

^{a} Range used in Simulations1; ^{b} Range used in Simulations2

Table 5.2 – DEB parameters for gilthead seabream.

Parameter	Description	Value or Range	Units	Source
$\{\dot{P}_{Xm}\}$	Maximum surface-area specific ingestion rate	892 – 1066$^{(a)}$ 1000 – 1500$^{(b)}$	J cm^{-2} d^{-1}	Lupatsch (2003) Calibrated
X_K	Half-saturation coefficient	0.5 – 10$^{(a)}$ 0.6 – 0.9$^{(b)}$	mg L^{-1}	Calibrated
κ_X	Absorption efficiency	0.80 - 0.84		Experimental data
$\{\dot{p}_{Am}\}$	Maximum surface-area specific assimilation rate	714 - 895	J cm^{-2} d^{-1}	Experimental data
κ	Fraction of reserve allocated for soma	0.58 – 0.85		Bodiguel et al. (2009) van der Veer et al. (2001, 2009)
$[E_G]$	Volume-specific costs for growth	5600 - 7563	J cm^{-3}	Bodiguel et al. (2009) van der Veer et al. (2009)
$[E_m]$	Maximum energy storage density	1505 - 2903	J cm^{-3}	Bodiguel et al. (2009) Eichinger et al. (2010)
$[\dot{p}_M]$	Volume-specific costs for maintenance	33 - 37	J cm^{-3} d^{-1}	Guinea and Fernandez (1997)
δ_m	Shape coefficient	0.242 – 0.255	Dimensionless	Field data
T_A	Arrhenius temperature	7032 - 8206	K	Experimental data
T_1	Reference temperature	293.15	K	Fixed

[a] Range used in Simulations1; [b] Range used in Simulations2

Conversion from feed weight (g) into energy (J) was made according to the manufacturer´s information on the ration energy content, using a value of 23800 J g^{-1} dry weight of feed (Sorgal 2011). The $\{\dot{P}_{Xm}\}$ values of both species at the reference temperature, were obtained by a linear regression between the ingestion rate (J d^{-1}) and volumetric length ($V^{\frac{2}{3}}$), calculated from body volume data (Figure 5.2). The estimated $\{\dot{P}_{Xm}\}$ value for white seabream, 1450 J cm^{-2} d^{-1} ($n = 6$; $R^2 = 0.99$) was higher than the value found for gilthead seabream, 965 J cm^{-2} d^{-1} ($n = 6$; $R^2 = 1.00$). For both species, maximum and minimum $\{\dot{P}_{Xm}\}$ values in Figure 5.2, were used to define the ranges used in the IBM (Tables 5.1 and 5.2).

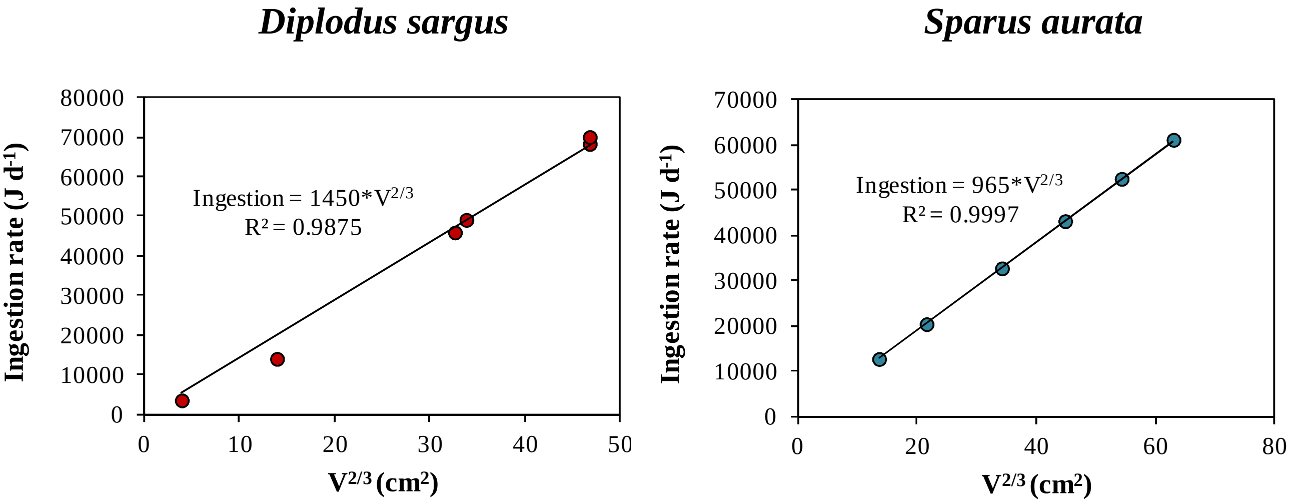


Figure 5.2 – Ingestion rate (J d^{-1}) - volumetric length (cm^2) relationships for white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) at the reference temperature (20ºC = 293.15 K). The slope of the linear regression between these two variables corresponds to maximum surface area-specific ingestion rate $\{\dot{P}_{Xm}\}$ for these species.

3.2.2.2 Absorption efficiency (κ_X)

In the present study, κ_X was calculated as: $1-\frac{FishFaecesMass}{MassOfFeedIngested}$, however this is a rough estimate because such an approach only provides an upper limit for this parameter (Kooijman 2010). The average κ_X value (± standard deviation) for white seabream, estimated in the same laboratory experiments used to determine $\{\dot{P}_{Xm}\}$ (cf. Section 3.2.2.1), was 0.75 ± 0.05 whereas for gilthead seabream, the absorption efficiency was slightly higher, 0.82 ± 0.02 (Dias et al. 2009). For both species, the $\{\dot{P}_{Am}\}$ values range used in the IBM (Tables 5.1 and 5.2) was calculated from κ_X and $\{\dot{P}_{Xm}\}$ ranges, according to eq. 7.

3.2.2.3 Fraction of reserves allocated for soma (κ), volume-specific costs for growth ($[E_G]$) and maximum energy storage density ($[E_m]$)

The ranges for some DEB parameters that are not directly measurable such as, κ, $[E_m]$ and $[E_G]$ were defined according to literature values for other fish species (Tables 5.1 and 5.2).

3.2.2.4 Volume-specific costs for maintenance ($[\dot{p}_M]$)

In the present study, $[\dot{p}_M]$ values were estimated based on the oxygen consumption rates of non feeding, immature white seabream individuals at constant temperatures in sealed aquariums (Table 5.3), as described by Eichinger et al. (2010). For gilthead seabream, the resting oxygen consumption rates measured by Guinea and Fernandez (1997) were used (Table 5.4), since it is commonly agreed that it reflects the energy spent on maintenance activities (Guinea and Fernandez 1997; Eichinger et al. 2010). For both species, oxygen consumption rates (mg O_2 g^{-1} d^{-1}) were converted into energy values (J cm^{-3} d^{-1}) using a conversion factor of 13.84 mg O_2 J^{-1} (Brafield and Llewellyn 1982) and a fish density of 1 g cm^3. At the reference temperature (20 ºC

= 293.15 K), the average ± standard deviation [$\dot{p}_M$] value for white seabream was higher (56 ± 4 J cm^{-3} d^{-1}) than the value found for gilthead seabream (35 ± 2 J cm^{-3} d^{-1}).

Table 5.3 – White seabream oxygen consumption rates (mg O_2 g^{-1} d-1, mean ± sd) at different temperatures (17 to 25 ºC) and corresponding volume-specific costs for maintenance ([$\dot{p}_M$], J cm^{-3} d^{-1}).

Temperature (ºC)	Oxygen consumption rate (mg O_2 g^{-1} d^{-1})	[$\dot{p}_M$] (J cm^{-3} d^{-1})
17	2.39 ± 0.48	33.0 ± 6.8
18	3.77 ± 0.60	52.0 ± 8.1
19	4.44 ± 0.37	61.4 ± 5.1
20	4.14 ± 0.37	55.9 ± 3.5
21	7.27 ± 0.15	100 ± 2
22	8.97 ± 0.63	124 ± 9
25	5.93 ± 0.63	82.1 ± 8.9

Table 5.4 – Resting oxygen consumption rates (mg O_2 g^{-1} d^{-1}, mean ± sd) for gilthead seabream (Guinea and Fernandéz 1997) and corresponding volume-specific costs for maintenance ([$\dot{p}_M$], J cm^{-3} d^{-1}).

Temperature (ºC)	Oxygen consumption rate (mg O_2 g^{-1} d^{-1})	[$\dot{P}_M$] (J cm^{-3} d^{-1})
16	0.070 ± 0.002	24.0 ± 1.2
21	0.110 ± 0.005	34.9 ± 2.3
23.5	0.115 ± 0.002	37.7 ± 0.7
26	0.125 ± 0.030	34.1 ± 10.6

3.2.2.5 Shape coefficient (δ_m)

The shape of an individual determines how a specific length measure relates to structural body mass. The shape coefficient of juvenile and adult white seabream as well as gilthead seabream were obtained by fitting a volume-length relationship to the available biometric data for these species (Figure 5.3). The slope of the linear regression between these two variables corresponds to the shape coefficient (δ_m) at the different life stages. The estimated δ_m value for juvenile white seabream (0.255) was lower than for adults (0.264). Likewise, slight differences were found between juvenile (δ_m = 0.242) and adult gilthead seabream (δ_m = 0.255). The δ_m values at the different life stages define the range of values for each species (Tables 5.1 and 5.2).

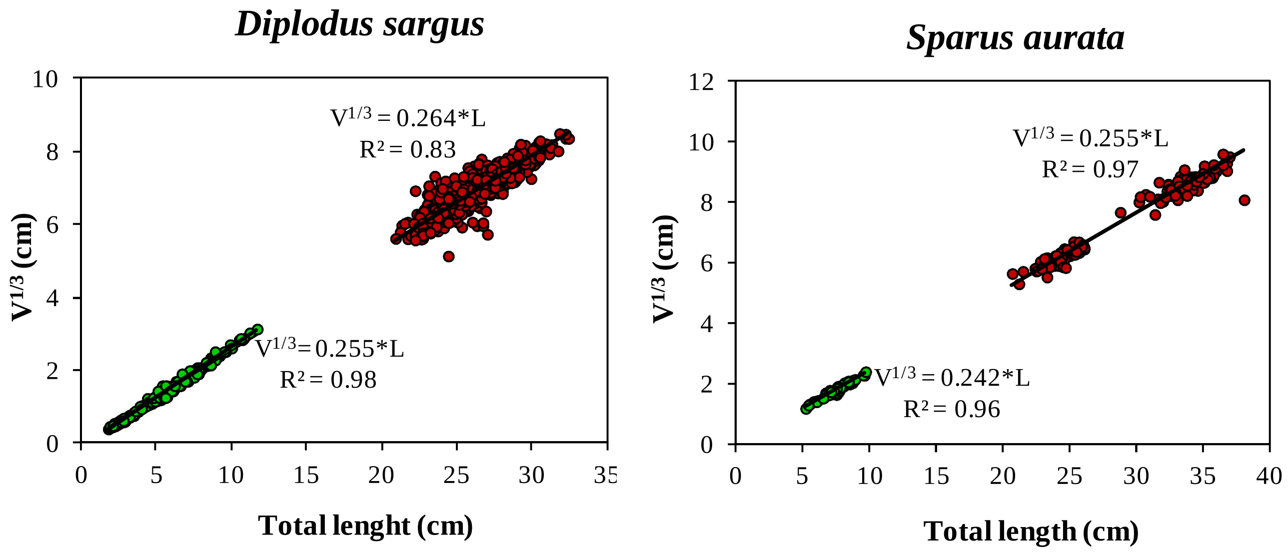


Figure 5.3 – Structural volume ($V^{1/3}$, cm) – total length (L, cm) relationships for juvenile (green dots) and adult (red dots) white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) individuals. The slope of the linear regression between these two variables corresponds to the shape coefficient (δ_m) at the different life stages.

3.2.2.6 Arrhenius temperature (T_A)

The slope of the linear regression between the logarithm (*ln*) of fish oxygen consumption rate ($\dot{k}_1$) and T^{-1} (T = absolute temperature) (Figure 5.4) gives the Arrhenius temperature - T_A (van der Veer et al. 2006). For the white seabream, this parameter was estimated from the data presented in Table 5.3, whereas for gilthead seabream, the T_A range was estimated from the oxygen consumption data in Requena et al. (1997). The estimated T_A ± standard error values for white seabream, 9294 ± 1620 ºK (n = 30; R^2 = 0.54) and gilthead seabream, 7619 ± 587 ºK (n = 10; R^2 = 0.95) defined the ranges used in the IBM.

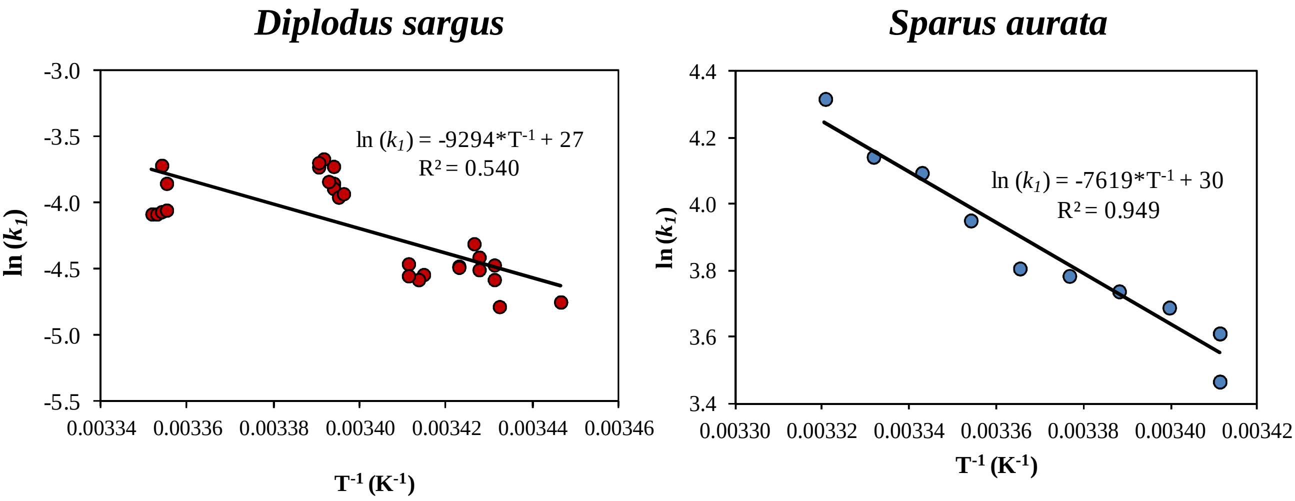


Figure 5.4 – Arrhenius plot (ln rate *versus* T^{-1}) for the oxygen consumption data of white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*). The slope of the linear regression corresponds to the Arrhenius temperature (T_A).

3.2.3 Model forcing and calibration

The main environmental factors taken into account by the DEB theory are, temperature and food availability, which act as forcing variables for the model (van der Veer et al. 2006). The IBM-DEB model was run using daily data sets of water temperature and food inputs (Figures 5.5) registered over the production cycles of white seabream (18th June 2003 to 31th March 2005) and gilthead seabream (2nd May 2005 to 22nd March 2007) at the IPIMAR's Aquaculture Research Center (Figure 5.6), in the Southeast of Portugal. Daily food input varied as a function of water temperature and fish biomass, being supplied 3 to 5 times a day, from 7 a.m. to 8 p.m. During the trials, seabream growth was also monitored by individual biometric measurements of total length (L, cm) and wet weight (W, g) at regular intervals. Fish data was later used for model calibration.

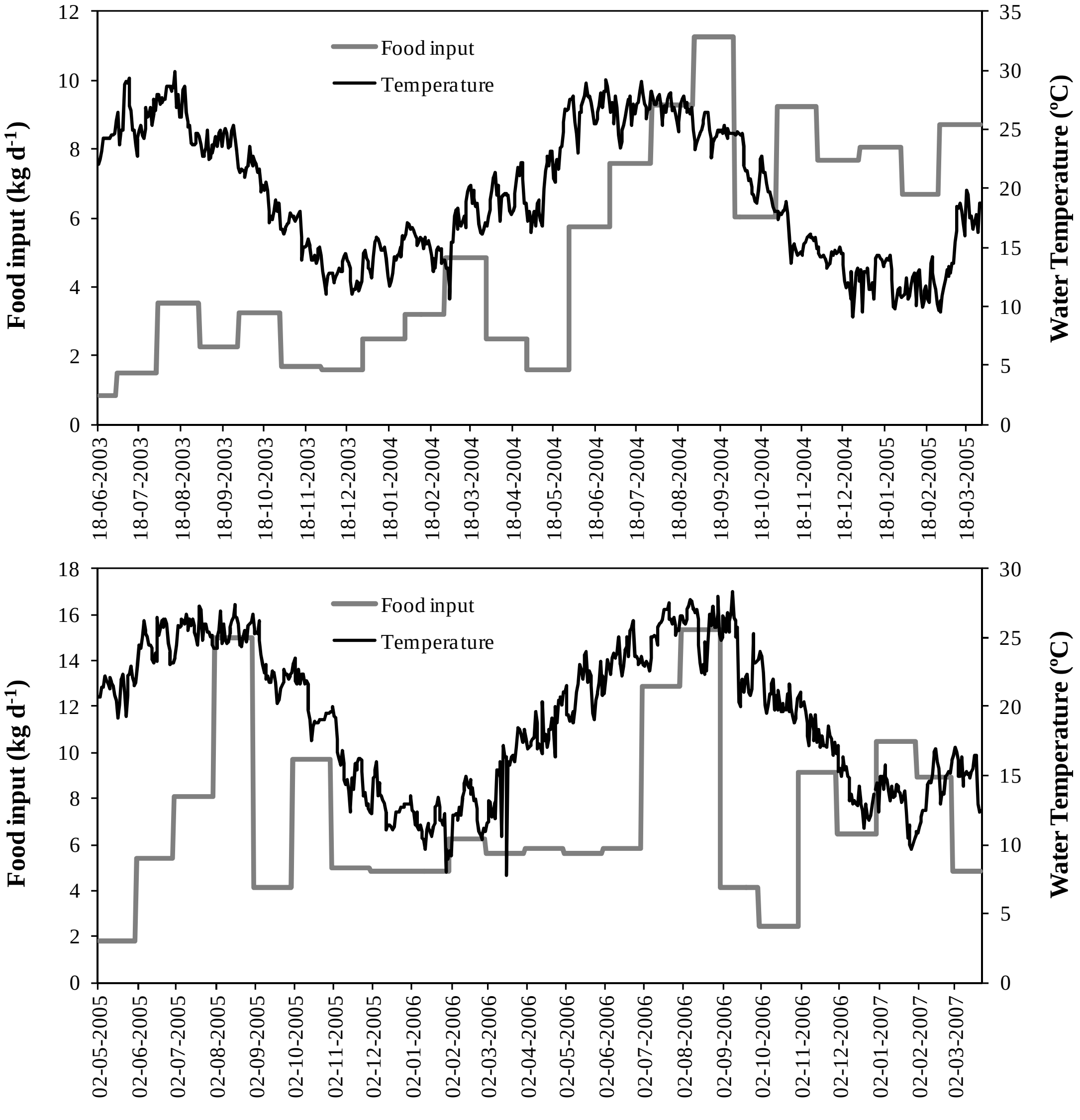


Figure 5.5 – Daily time series of food input (kg d^{-1}) and water temperature (ºC) registered during the white seabream (top graph) production cycle (from 18th June 2006 to 31th March 2005) and the gilthead seabream (bottom graph) production cycle (2nd May 2005 to 22nd March 2007).

Figure 5.6 – Aquaculture Research Center.

3.2.4 Model implementation

The IBM-DEB model developed in this work was implemented with EcoDynamo (Pereira et al. 2006), which uses object-oriented programming (OOP) written in C^{++}. In the EcoDynamo, each object simulates several state variables and processes, and interacts with the other objects by means of a shell or server (Pereira et al. 2006). The shell interface allows the user to define the model setups – time steps, output formats (file, graphic and tables), objects to be used and variables to be visualized (Pereira et al. 2006). After definition of initial and boundary conditions and input of model parameters, variables were computed over time (time step = 1 hour) using the Euler integration method.

3.2.5 Sensitivity Analysis

3.2.5.1 IBM model

Based on the outputs of the IBM model, which include a file with fish parameter sets and growth variables, a Principal Component Analysis (PCA) was performed using the final total length and wet weight of the 10000 fish as variables, and individuals as observations. Since to each individual corresponds a specific parameter set, this analysis may be used to assess the relative contribution of each parameter to growth differences among the fish population and to evaluate the relative contribution of each parameter to fish growth performance. By including the mean square deviation (MSD) between observed and predicted growth data in the analysis, it was also possible to evaluate which data sets produced the best and worst fits.

3.2.5.2 “State variable” model

A simple sensitivity analysis was carried out with the “state variable” model in order to test which of the DEB parameters were more likely to affect fish performance (total length and wet weight). The analysis consisted in changing one parameter at a time by ± 25% and comparing the results with a standard simulation using the parameter sets calibrated with the IBM model.

4 Results

Figures 5.7 and 5.8 show the results of Simulations1 for a virtual population of 10000 white seabream and gilthead seabream, each with its specific parameter set. Besides the predicted values (blue diamonds) of total length and wet weight for the 10000 individuals, these figures present the measured values (red dots) and the average ±1 standard deviations of predicted values (green line and whiskers). The fish growth variability predicted by the IBM model was clearly higher than the observed variability, suggesting that some parameter combinations were biologically unrealistic. Despite this high growth variability, the average predicted values of total length and wet weight for white seabream were in general very close to measured values (Figure 5.7), suggesting that the DEB parameter ranges defined in Table 5.1 lead to a good estimation of white seabream growth. On the other hand, for gilthead seabream there was a slight deviation between the average predict values and observed data (Figure 5.8), indicating that parameter ranges depicted in Table 5.2 lead to some underestimation of its growth.

Results from Simulations2 (Figures 5.9 and 5.10) were similar to those from Simulations1 in the sense that fish growth ranges were much wider than that expected from observed data. However, in these simulations, the predicted ranges for fish length and wet weight for both species were much narrower than those obtained with Simulations1. Even though measured values were between the average and standard deviation of predicted values for both species, in this approach, the parameter ranges depicted in Tables 5.1 and 5.2 seem to underestimate fish growth in certain stages of the production cycle and to overestimate it in other stages.

In order to understand which DEB parameters might explain the growth differences amongst the virtual fish population, a Principal Component Analysis (PCA) was performed with the IBM results of Simulations1 (Figure 5.11) and Simulations2 (Figure 5.12). The white seabream's PCA from Simulations1 (Figure 5.11 – top graph), showed that fish length and weight along with the MSD between predicted and observed fish growth, gave the largest contributions for the negative part of factor 1, the factor explaining most part (23 to 29%) of the analysis. The DEB parameters contributing mostly to the negative part of this factor were κ; $\{\dot{p}_{Am}\}$ and κ_X, which indicates that fish reaching larger sizes were the ones allocating a higher fraction of reserves to soma, assimilating higher amounts of food per unit of surface area and that are able to absorb food more efficiently. The gilthead seabream's PCA from Simulations1 (Figure 5.11 – bottom graph) was very similar to that of white seabream except for the fact that the MSD between predicted and observed fish growth appeared on the positive semi-axis of factor 1.

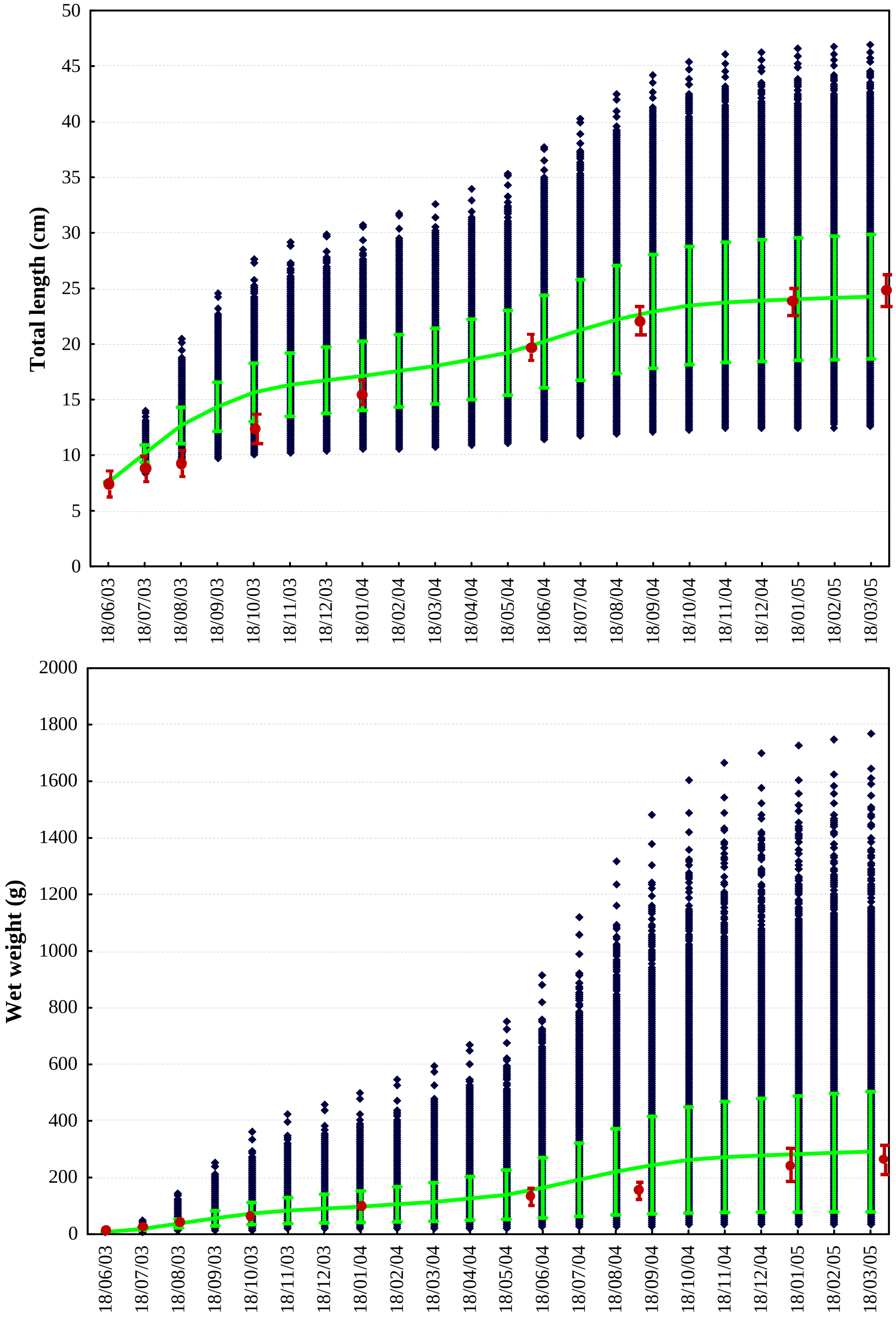


Figure 5.7 – Predicted (blue diamonds) total length and wet weight values for 10000 white seabreams (*Diplodus sargus*) from Simulations1. The green line and whiskers represent the average predicted values ± 1 sd while red dots and whiskers correspond to average measured values ± sd.

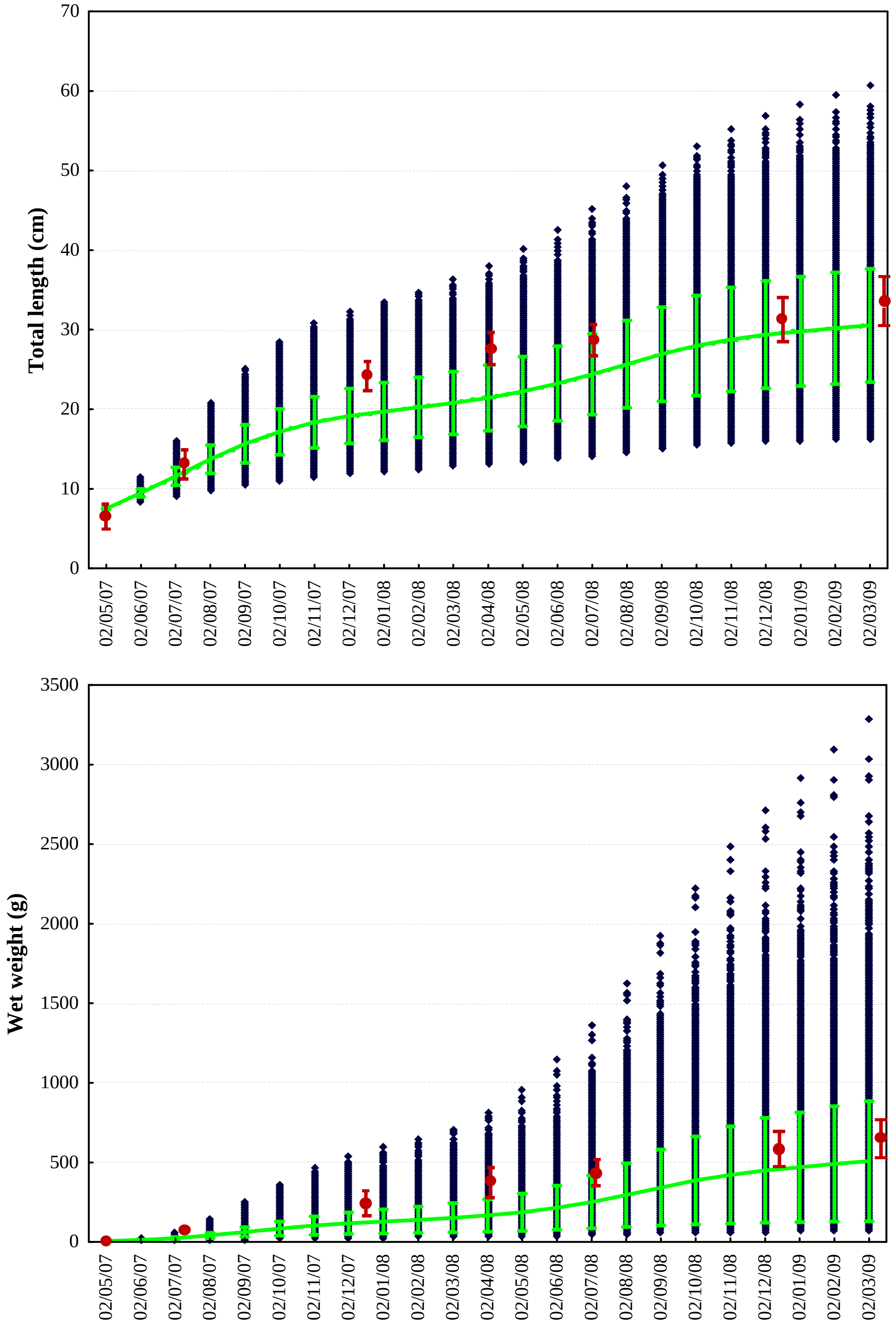


Figure 5.8 – Predicted (blue diamonds) total length and wet weight values for 10000 gilthead seabreams (*Sparus aurata*) from Simulations1. The green line and whiskers represent the average predicted values ± 1 sd while red dots and whiskers correspond to average measured values ± sd.

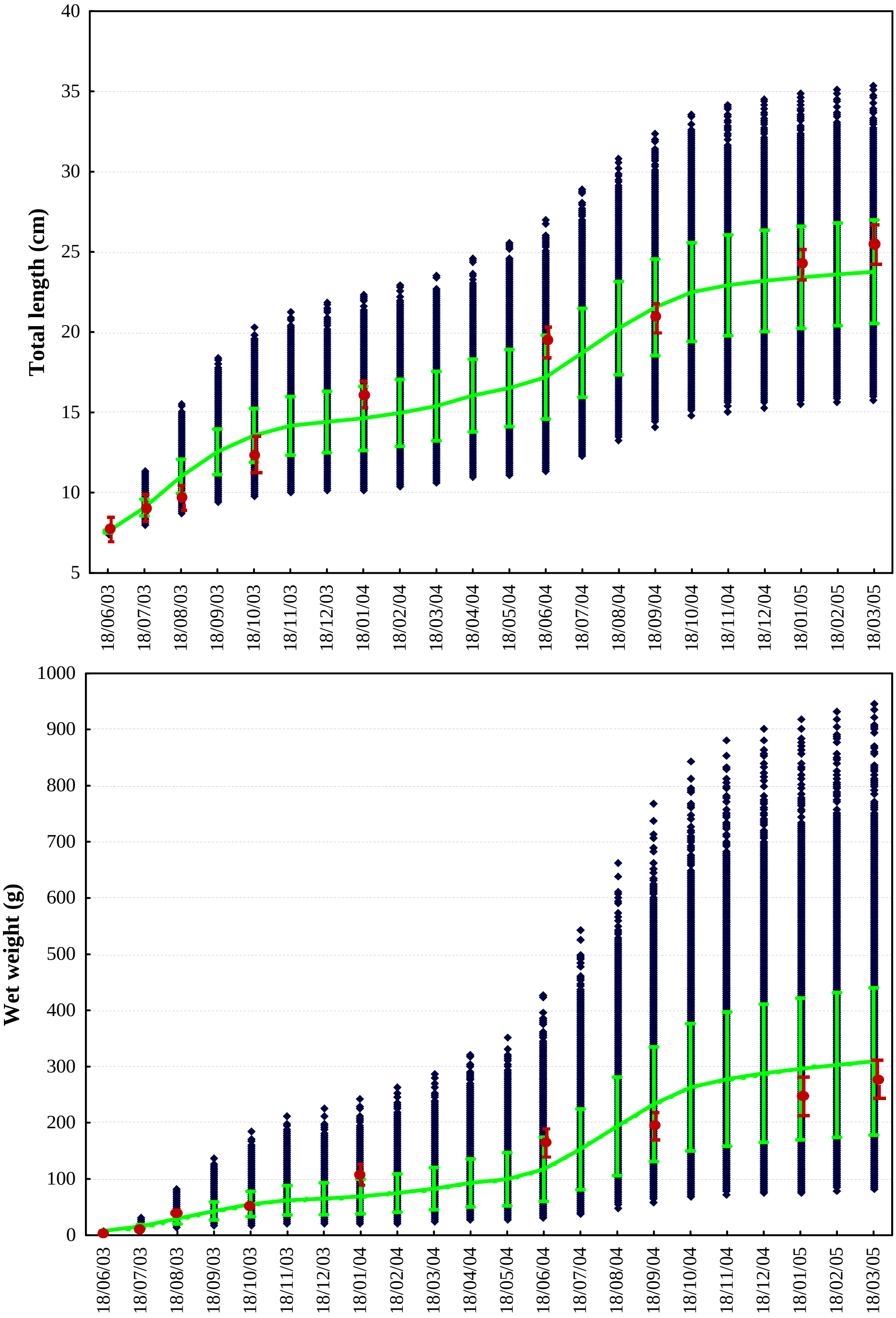


Figure 5.9 – Predicted (blue diamonds) total length and wet weight values for 10000 white seabreams (*Diplodus sargus*) from Simulations2. The green line and whiskers represent the average predicted values ± 1 sd while red dots and whiskers correspond to average measured values ± sd.

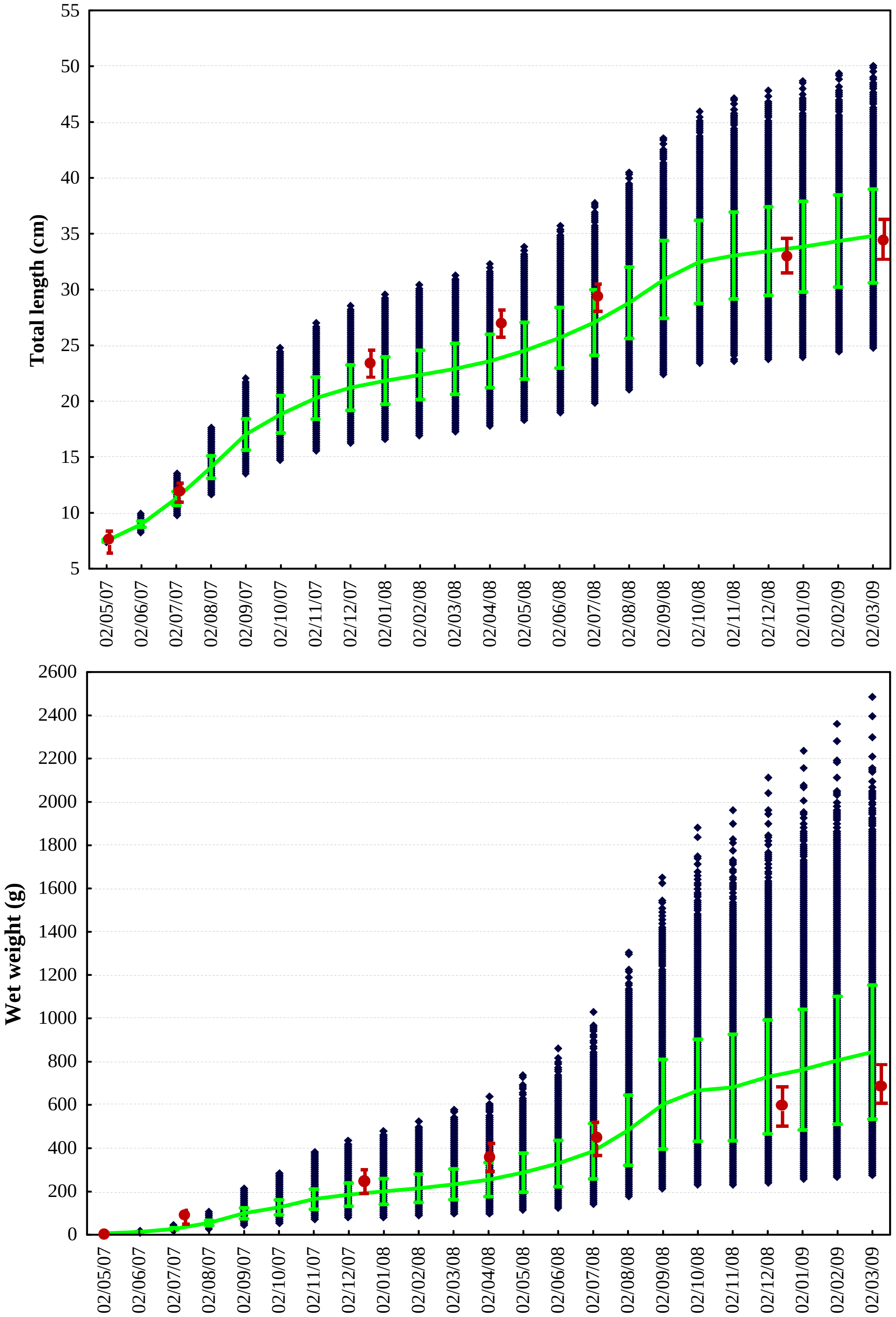


Figure 5.10 – Predicted (blue diamonds) total length and wet weight values for 10000 gilthead seabreams (*Sparus aurata*) from Simulations2. The green line and whiskers represent the average predicted values ± 1 sd while red dots and whiskers correspond to average measured values ± sd.

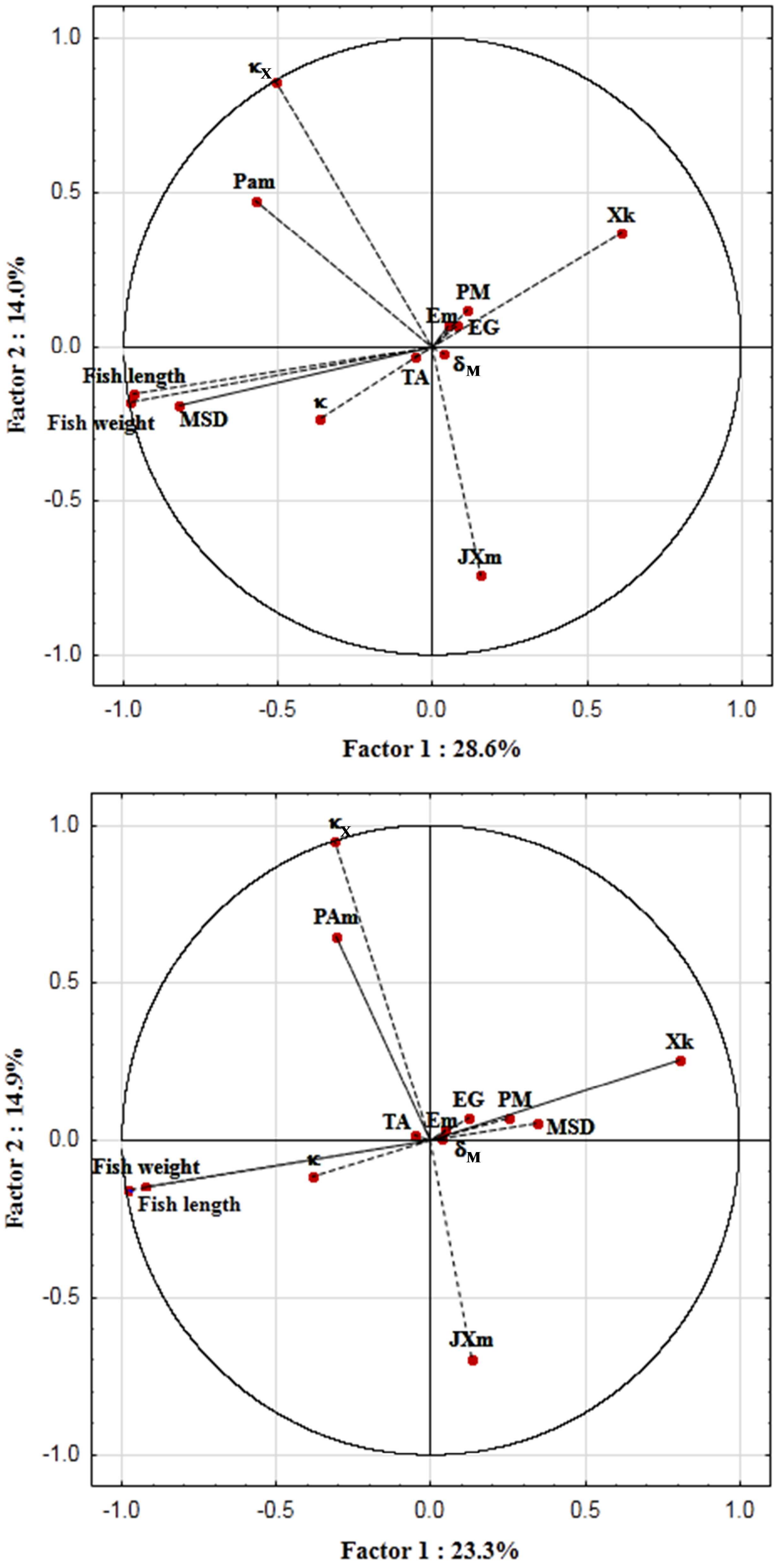
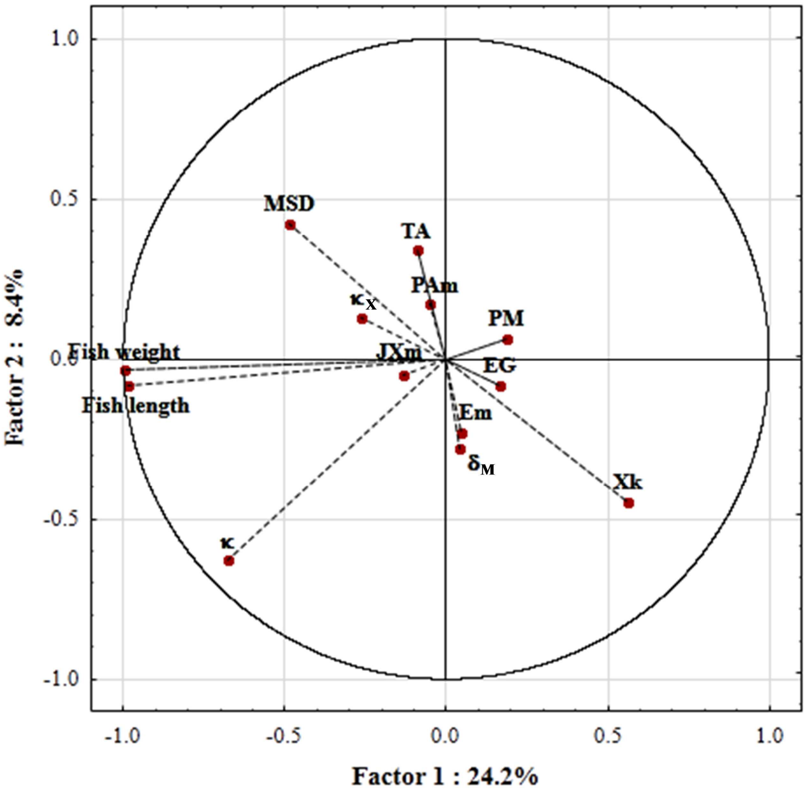


Figure 5.11 – Principal Component Analysis (PCA) showing the contribution of DEB parameters to white seabream (top graph) and gilthead seabream (bottom graph) growth in Simulations1.



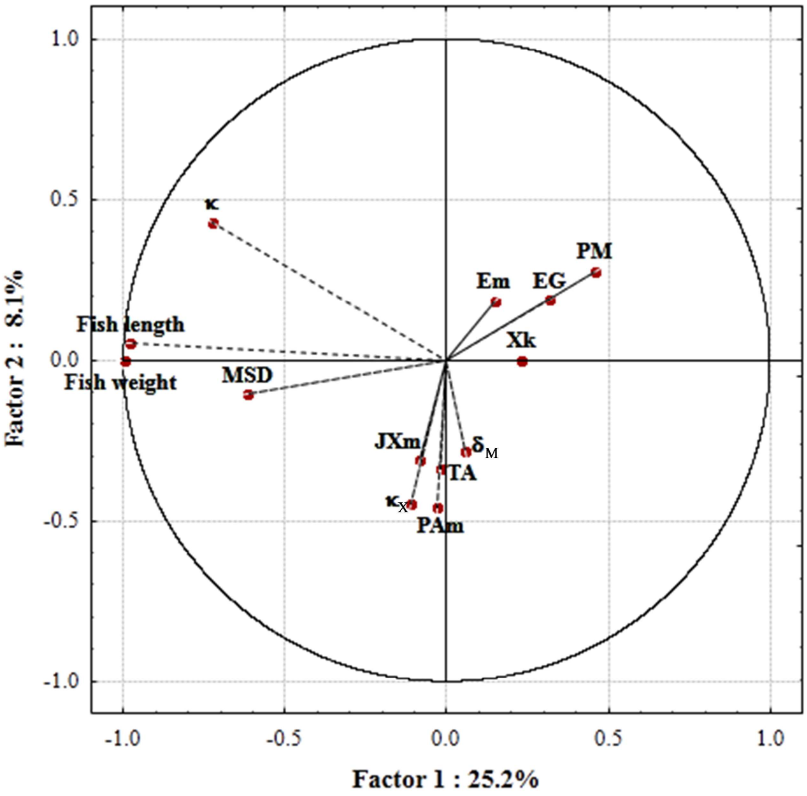


Figure 5.12 – Principal Component Analysis (PCA) showing the contribution of DEB parameters to white seabream (top graph) and gilthead seabream (bottom graph) growth in Simulations2.

Similarly to the white seabream's PCA for Simulations1, the PCA results for Simulations2 (Figure 5.12) showed that growth variables (total length and wet weight) and the MSD between predicted and observed fish growth contributed mostly to the negative part of factor 1. Once again, the parameter giving the largest contribution to the negative part of this factor was κ; indicating that bigger fishes had higher κ values. Some other parameters such as $\{\dot{P}_{Xm}\}$ and κ_X might have also contributed to extreme fish sizes due to their contributions to the same semi-axis. Conversely, natural occurring fish sizes seem to be explained by higher values of $[\dot{p}_M]$, $[E_G]$ and $[E_m]$ since these parameters were on the opposite semi-axis of growth variables and MSD.

Using the parameter sets that produced the best fits to observed data (Tables 5.5 and 5.6), the growth of white seabream and gilthead seabream was simulated by running the model on the "state variable" mode. Figure 5.13 represents the observed and predicted total length and wet weight values for respectively, a white seabream and a gilthead seabream from Simulations1. Both observed and predicted data show that the gilthead seabream gained twice as much weight (Δ = 620g, in 689 days) as the white seabream in almost the same period of time (Δ = 260g, in 651 days). A linear regression between predicted and observed data revealed a good model fit for both species. Nevertheless, white seabream weight ($R^2 = 0.96$, n=8) was simulated more accurately than length ($R^2 = 0.92$; n=8), whereas for gilthead seabream, length ($R^2 = 0.98$; n=7) was more accurately predicted than weight ($R^2 = 0.94$; n=7). Results from Simulations2 (Figure 5.14) were very similar to those from Simulations1 in terms of predicted growth ranges. However, model fits appeared to be slightly better with the second approach, for both white seabream ($0.94<R^2<0.96$; n=7) and gilthead seabream ($0.95<R^2<0.98$; n=7).

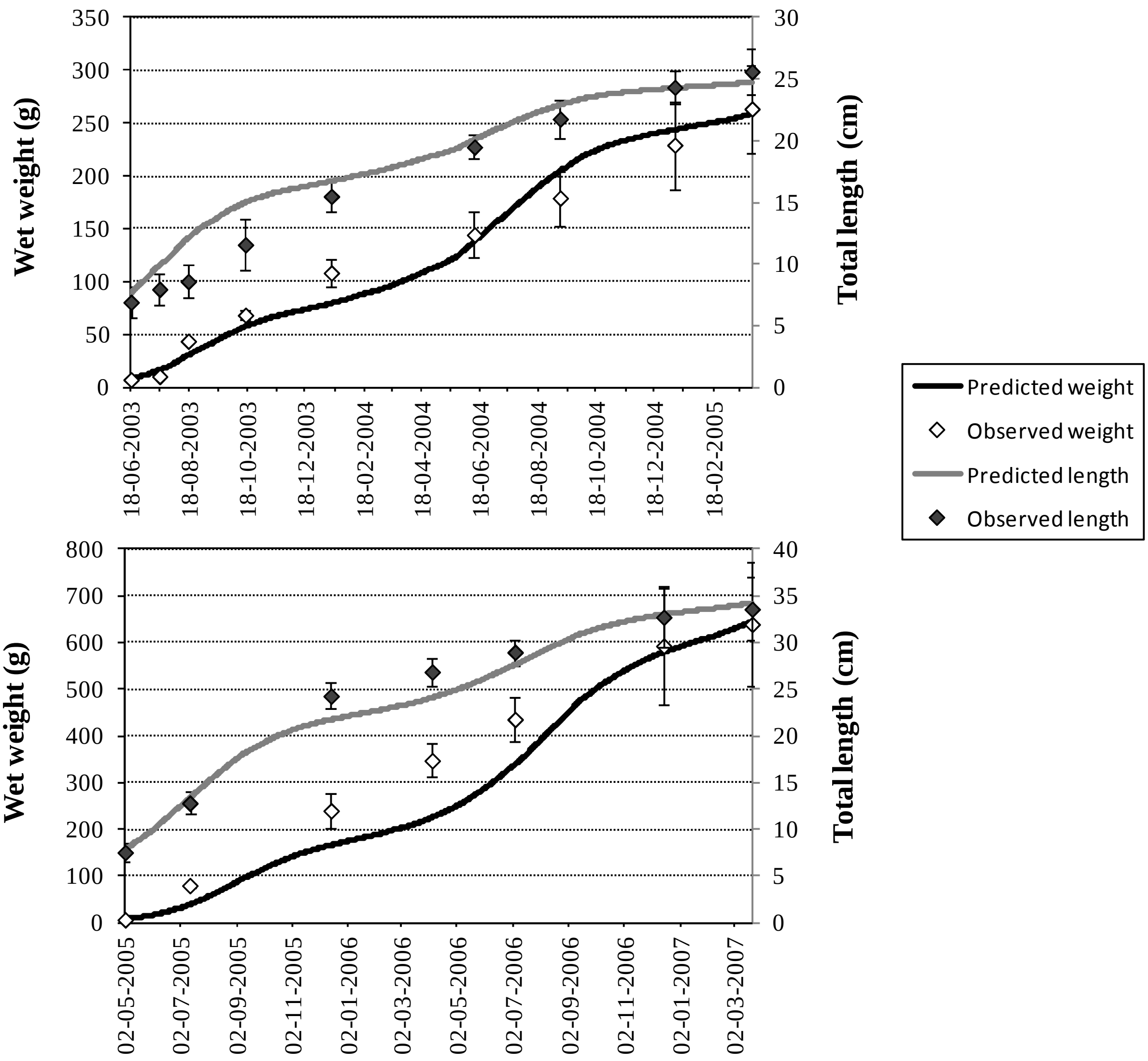


Figure 5.13 – Average ± sd measured (diamonds) and predicted values (lines) of wet weight (g) and total length (cm) for white seabream (top graph) and gilthead seabream (bottom graph) in Simulations1.

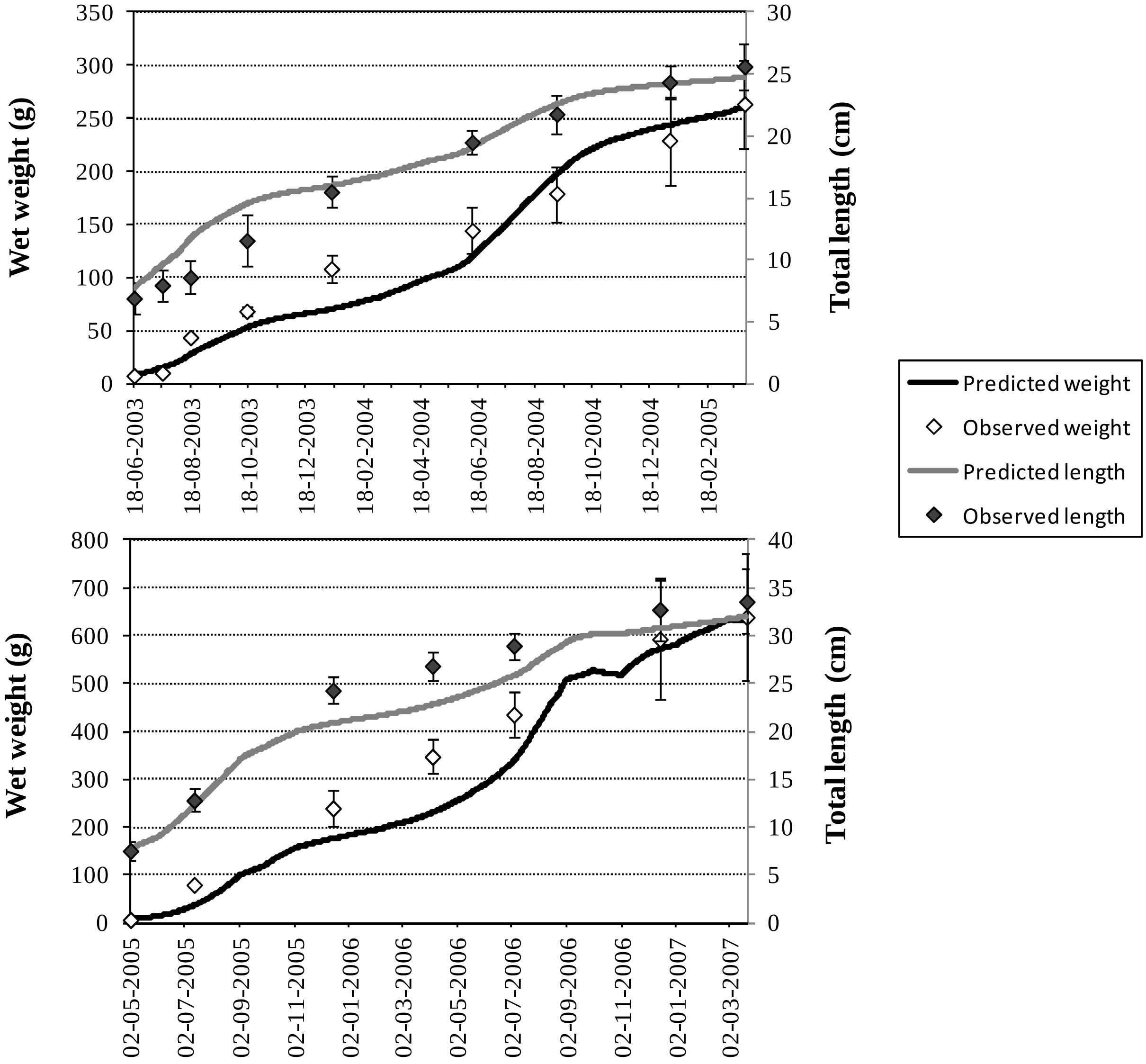


Figure 5.14 – Average ± sd measured (diamonds) and predicted values (lines) of wet weight (g) and total length (cm) for white seabream (top graph) and gilthead seabream (bottom graph) in Simulations2.

A sensitivity analysis carried out with the calibrated model parameters from Simulations1 (Table 5.5) revealed that fish wet weight is more likely to be affected by parameters such as κ; $\{ \dot{p}_{Am} \}$ and κ_X, in agreement to the results of the PCA analysis (Figure 5.11). A +25% change in the former parameters resulted in an increase of ca. 62% in mean wet weight. To a lesser extent the model also seemed to be sensitive to $[E_G]$ and $[\dot{p}_M]$ since a 25% variation in these parameters caused a variation of respectively, 24 % and 43% in mean wet weight. In what concerns fish length it seems that this variable was mostly affected by the shape coefficient (δ_m) as a 25% decrease in this parameter resulted in a 33% increase in mean total length. Nevertheless, other parameters like, κ; $\{\dot{p}_{Am}\}$ and κ_X may also strongly influence fish length, as observed in the PCA analysis (Figure 5.11), since a 25% increase in these parameters yielded a ca. 17% increase in this variable. Regarding Simulations 2, the results of the sensitivity analysis performed with the calibrated model parameters (Table 5.6) were very similar to those from Simulations1, except for the fact that the maximum-specific ingestion rate ($\{\dot{P}_{Xm}\}$) was much more relevant to fish growth than the assimilation rate ($\{\dot{p}_{Am}\}$), in agreement to the PCA results (Figure 5.12). Apart from that the relative contribution of the other DEB parameters was very similar in the two approaches.

Table 5.5 – Calibrated DEB parameters for white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) in Simulations1. Sensitivity analysis results as % of variation in white seabream mean total length and wet weight after changing each parameter by ±25%.

Parameter	*Diplodus sargus*	*Sparus aurata*	Total length		Wet weight	
			-25%	+25%	-25%	+25%
$\{\dot{P}_{Xm}\}$	1073	998	0	0	0	0
X_K	0.0002	0.0004	+1	-2	+3	-6
κ_X	0.75	0.83	-18	+17	-48	+62
$\{\dot{p}_{Am}\}$	808	870	-18	+17	-48	+62
κ	0.60	0.68	-18	+17	-48	+62
$[E_G]$	7203	6098	+8	-7	+24	-18
$[E_m]$	2251	1547	+2	-1	+4	-3
$[\dot{p}_M]$	54	33	+11	-9	+43	-28
δ_m	0.257	0.252	+33	-20	0	0
T_A	7807	7425	-2	+3	-6	+7

Table 5.6 – Calibrated DEB parameters for white seabream (*Diplodus sargus*) and gilthead seabream (*Sparus aurata*) in Simulations2. Sensitivity analysis results as % of variation in white seabream mean total length and wet weight after changing each parameter by ±25%.

Parameter	*Diplodus sargus*	*Sparus aurata*	Total length		Wet weight	
			-25%	+25%	-25%	+25%
$\{\dot{P}_{Xm}\}$	1300	1336	-19	+19	-50	+75
X_K	0.13	0.66	+3	-2	+8	-7
κ_X	0.71	0.83	-19	+19	-50	+75
$\{\dot{p}_{Am}\}$	922	809	-1	+1	-4	+2
κ	0.72	0.61	-20	+21	-50	+75
$[E_G]$	7296	5613	+9	-7	+27	-19
$[E_m]$	1815	1506	+1	-1	+3	-3
$[\dot{p}_M]$	59	34	+13	-11	+53	-32
δ_m	0.258	0.254	+33	-20	0	0
T_A	7925	7637	-3	+3	-6	+8

5 Discussion

The simple mechanistic principles and generality of DEB models (Kooijman 2010) make them ideal to forecast fish growth in aquaculture systems because the same model can be applied to different species and different production cycles. To implement these models a suitable parameter set must be selected for the cultivated species. In the present study, despite a lot of effort was invested in parameter estimation, particularly for white seabream which is a poorly studied species from the physiological point of view, there were still uncertainties regarding some of the DEB parameters. One of the main concerns was related with food density (X) because even though there were registers of the amount of food supplied to the system there was no easy way to convert this into the amount of food available for fish, due to pellets sedimentation. Immeasurable DEB parameters such as κ, $[E_m]$ and $[E_G]$, that were taken from the literature, were also a matter of concern. Therefore, instead of developing a simple DEB model, an IBM based approach was used to allow a fast assessment of the parameter sets producing the best fit between predicted and observed fish growth (Duarte et al. 2010). As a result of the stochastically assignments of DEB parameters to each individual in the IBM, not every parameter combination gave completely reliable predictions and, as a result, fish growth fell outside the ranges observed in nature for both Simulations (Figures 5.7, 5.8, 5.9 and 5.10). Nevertheless, this simple methodology has proven to be effective, as in Duarte et al. (2010), since it was possible to select several parameter sets for each approach that predicted white seabream and gilthead seabream growth with reasonable accuracy (Figures 5.13 and 5.14).

The similar results obtained with the two *f* formulations (cf. Section 3.2.1.1) suggest that both may be used to simulate seabream's growth. One of the advantages of the approach used in Simulations1 is that it enables the evaluation of food conditions over a fish production cycle, which is a crucial information for semi-intensive systems since they are often managed intuitively (Giovannini and Piedrahita 1994; SEACASE 2009). The average *f* describing the growth of white seabream was 0.85, which is indicative that food conditions were not maximal (Kooijman 2000). Nevertheless, lower *f* values were reported for hake (0.72) in its natural environment (Bodiguel et al. 2009) and for juvenile sole (0.68) reared in tanks (Eichinger et al. 2010). Since at some stages of the production cycle, weight predictions were below (November 2004 to April 2004) and above (September 2004 to December 2004) observations (Figure 5.13), this indicates that *f* values might be respectively, below and above the average value in these periods. Moreover, the lack of agreement in the discrepancies between predicted and observed weight and total length values for white seabream, suggests that food conditions were not responsible for the length mismatches observed at the beginning of the production cycle. Instead, these results may be explained by the higher shape coefficient value (0.260) used in the model (Table 5.5) comparatively to the estimated δ_m value (0.255) for juvenile white seabream (Figure

5.3). In fact, Loy et al. (2001) have referred that during their juvenile life, specimens of white seabream undergo important shape changes, which are consistent with habitat transition. For the gilthead seabream, the average f estimated by the model, 0.70, was similar to the values reported for other fish species (Bodiguel et al. 2009; Eichinger et al. 2010), but lower than the value estimated for white seabream. As described for the latter species, at a certain point (December 2005 to July 2006), the gilthead seabream's weight and length predictions were also below the observed growth data (Figure 5.14), suggesting that f was below the estimated average value. The fact that f values were on average lower than 1 for both species, may be indicative of inefficient feeding strategies (e.g. meal timing and frequency or feed supply methods) (Black 2001) or/and of substantial amounts of wasted feed as a result of pellets decay (Choo 2001; World Bank 2006). This last hypothesis is partially corroborated by more accurate growth predictions in Simulations 2 (cf. – Section 3.2.1.1) than in Simulations 1. The other advantage of using f_2 is that, if the model would be coupled to a biogeochemical model it would be possible to directly estimate the contribution of uneaten feed to the organic matter enrichment of earth pond sediments (Serpa et al. 2007b).

Regardless the approach, the IBM-DEB model may be used to investigate the reasons behind growth variability among fish. The PCAs from each simulation revealed that a relatively small number of parameters explained a large fraction of this variability. These were basically the parameters governing reserves allocation (κ), food ingestion ($\{\dot{P}_{Xm}\}$) and assimilation ($\{\dot{p}_{Am}\}$ and κ_X). An interesting result was that the contribution of these parameters depended on the approach used to estimate the scaled functional response. In the PCAs from Simulations 1, the main parameters explaining growth variability were, by order of importance, κ, $\{\dot{p}_{Am}\}$ and κ_X whereas in Simulations2 the $\{\dot{p}_{Am}\}$ was replaced by $\{\dot{P}_{Xm}\}$, probably because as food pellets settle very rapidly, the amount of food that an individual is able to ingest in a small amount of time becomes crucial for its success. In fact, according to Kooijman (2009), the mechanism behind size differences among fish in supply systems, is the way of feeding since the whole brood is usually fed few times a day. As a consequence of restricted food availability, the social interactions during feeding are intensified and size differences which initially are very small are amplified over time, as was the case for the present study (Figures 5.13 and 5.14), because largest animals take priority over smaller ones (Kooijman 2009). The consistency between the PCA (Figures 5.11 and 5.12) and the sensitivity analysis results (Tables 5.5 and 5.6), suggests that as potentially important parameters for fish growth, there should be an investment in the estimation of κ, $\{\dot{P}_{Xm}\}$,$\{\dot{p}_{Am}\}$and κ_X when applying the present model to other fish species.

Even though fish performance is more likely to be affected by DEB parameters governing food consumption and energy partioning, as found by other authors (van der Veer et al. 2003; Bodiguel et al. 2009), fish weight was also sensitive to parameters like $[\dot{p}_M]$ and $[E_G]$, and fish length was particularly affected by δ_m (Tables 5.5 and 5.6). According to Bodiguel et al. (2009), the reason for having volume-depending parameters $[\dot{p}_M]$ and $[E_G]$ affecting fish weight in a negative way is related to the fact that higher energetic costs for maintenance and growth imply a slower increase of body volume. Fish length, on the other hand, was mostly affected by the shape coefficient, as expected from eq. 11.

Taking into account the results of Simulations2 (higher accuracy in growth simulations), the values of DEB parameters substantially affecting white seabream and gilthead seabream growth were compared (Table 5.6), to assess which of these parameters would explain more of the interspecies growth variability. From the differences between the DEB parameter values of these two Sparidae and the results of the sensitivity analysis, it appears that the parameter explaining the white seabream low growth rates is $[\dot{p}_M]$. Since the $[\dot{p}_M]$ value for white seabream was 42%

higher than that of the gilthead seabream, if the value of the latter species was used in the white seabream model, its final weight would more than duplicate, as shown in the sensitivity analysis (Table 5.6). Unlike in other studies in which interspecies variability was reflected in energy partioning (van der Veer et al. 2001), in the present work the fraction of reserves allocated for growth (κ) could not account for the growth differences between the two species because this value was higher for the white seabream. A possible explanation for these results is that the impact of $[\dot{p}_M]$ in white seabream growth was much larger than the variation in κ, so that the extra energy invested in soma could not compensate for the higher maintenance costs. Regarding the ingestion rates ($\{\dot{P}_{Xm}\}$), as the difference between the values of the two species was of only 3%, this parameter could not be have been responsible for interspecific variability, as found for flatfish species (van der Veer et al. 2001). Conversely, the food absorption efficiency (κ_X) was able to explain part (≈33%) of the growth differences between these two seabreams, since a 12% variation in this parameter yielded a 117g increase in final fish weight.

This last finding might be important for the aquaculture sector because it suggests that if feed absorption efficiency is increased, it would be possible for white seabream to attain its commercial size (350-400g) in less than 2 years. Being a newly cultivated species, the white seabream has been fed with rations that were optimised for gilthead seabream (Cejas et al. 2004; Sá et al. 2006), so the development of new feed formulations is crucial for white seabream production. Even though the physiological characteristics of the former species seem to prevent it from reaching the growth rates of gilthead seabream, if the white seabream would attain the market size in a shorter time period this might be sufficient to ensure the economic viability of its production in semi-intensive systems because this species reaches market prices substantially higher (5-6 €/kg, for 350 g size fish) (FAO 2011c) than the fast growing gilthead seabream (3-4 €/kg, for 350 g size fish) (Barazi-Yeroulanos 2010).

6 Conclusions

In the present work, an IBM-DEB model was developed and implemented to simulate the growth of two seabream species, the white seabream and the gilthead seabream, during a production cycle in semi-intensive fishponds. The model uses two different approaches to estimate the scaled functional response (*f*), in trying to account for uncertainties in food availability. Running the IBM model with a large number of individuals, each with its own parameter set was found to be a practical and easy way to calibrate the model, and to find the parameter sets that best described the growth of the two species. Model simulations globally matched the total length and wet weight observations of white seabream and gilthead seabream over a production cycle. However, more data sets are needed to test its applicability in different farming conditions. Comparing the DEB parameter values for these two sparids, it seems that the most likely reason for white seabream's low growth is a higher energy demand for maintenance. However, a lower feed absorption efficiency might also explain part of the growth differences between these species. As the white seabream is currently being fed with rations optimised for gilthead seabream, an improvement in white seabream feeds might not only increase its production but also minimize the aquaculture environmental impacts by reducing the amount of waste. Future applications of the IBM-DEB model include its coupling to a biogeochemical model to reproduce the pond environment, and thereby create an efficient tool for pond management.

Acknowledgments

The present work was supported by the EU Project – SEACASE "Sustainable extensive and semi-intensive coastal aquaculture in southern Europe" (FP6-2005-SSP5A) and by a PhD grant from Fundação para a Ciência e Tecnologia (SFRH/BD/27643/2006).

Chapter 6

Mathematical modelling as a tool for managing semi-intensive production systems

---

Serpa D., Pousão Ferreira P., Caetano M.., Cancela da Fonseca L., Dinis M. T., Duarte P. Mathematical modelling as a tool for managing semi-intensive production systems. *In preparation*.

Abstract

The sustainability of semi-intensive aquaculture relies on management practices that simultaneously improve production efficiency and minimize the environmental impacts of this activity. The purpose of the present work was to develop a mathematical model that reproduced the dynamics of a fish earth pond, to simulate different management scenarios for optimizing fish production in semi-intensive systems. The modelling approach consisted of coupling a biogeochemical model that simulated the dynamics of the elements that are more likely to affect fish production and cause undesirable environmental impacts (namely nitrogen, phosphorus and oxygen), to a fish growth model based on the Dynamic Energy Budget approach. The biogeochemical sub-model successfully simulated most water column and sediment variables. A good model fit was also found between predicted and observed white seabream (*Diplodus sargus*) growth data over a production cycle. Model outputs were used to construct nutrient budgets for evaluating the efficiency of food utilization and quantifying daily nutrient discharges for standard farming conditions. Budgets revealed that almost 50% of food supplied is not eaten by fish. The fraction of nutrients that is effectively retained by fish represented only 24.5% and 27.3% of ingested N and P, which suggests low feed conversion efficiency for the cultivated species. In terms of effluent discharges, a 43.1 $\mu M\ d^{-1}$ of N and 2.9 $\mu M\ d^{-1}$ of P were estimated to be discharged into the Ria Formosa lagoon. In order to optimize fish production, different management scenarios were analysed with the model (e.g. increase stocking densities, decrease/increase water exchange rates, decrease/increase feeding rates, decrease phosphorus content in fish feeds, increase feed absorption efficiency and decrease pellets sinking velocity) to test their effects on the pond environment as well as on fish yields and effluent nutrient discharges. Scenarios were quantitatively evaluated and compared using the Analytical Hierarchical Process (AHP) methodology, which returns a score for each scenario. The best

management options for maximizing production while maintaining a good pond environment and minimizing the impacts on the adjacent coastal system were to double standard stocking densities and improve feed absorption efficiency.

1 Introduction

To turn aquaculture into a more productive activity with improved profit margins, fish farmers worldwide have been intensifying production (World Bank 2006). Intensification implies that fish are cultivated at high densities and using formulated feeds, which increases the ecological footprint of this activity (Folke et al. 1998). The lower environmental risks of semi-intensive aquaculture (Kautsky et al. 2000; Banas et al. 2008), together with the increasing demand of consumers on food safety and on cultivated species welfare, have brought semi-intensive aquaculture products back into the front scene (SEACASE 2009). Nevertheless, this industry has been struggling with economic difficulties as a result of high production costs (e.g. labour, energy and land costs) and low productivity. A low profitability together with the increasing market competition with low-price intensive aquaculture products (SEACASE 2009), may seriously compromise the economic viability of this activity.

Among the solutions that have been proposed to increase the competitiveness of semi-intensive aquaculture is the optimisation of fish production (SEACASE 2009). The latter proposal implies that not only production is maximized but also that sound environmental conditions are maintained within fishponds and in the receiving coastal waters. But how can this be achieved?

One of the possible approaches would be to optimize fish stocking densities. For instance, Ferreira et al. (2010) reported that doubling the standard stocking density of semi-intensive gilthead seabream (*Sparus aurata*) farms, from 1.5 kg m^{-3} to 3 kg m^{-3}, had no major environmental impacts in the adjacent coastal waters, which suggests the feasibility for productivity enhancement under sound environmental conditions. Nevertheless, optimum stocking densities strongly depend on several factors, such as the spatial and behavioural needs of fish species (Ashley 2007), pond water quality (Shin and Wu 2003) and the assimilative capacity of the surrounding aquatic environment (Sarà 2007), which are known for limiting the carrying capacity of production systems.

Water management is also an important issue for the sustainability of semi-intensive aquaculture because water quality in these systems, which is crucial for fish welfare, is often determined by water exchange rates (Hopkins et al. 1993; Brambilla et al. 2007). While some authors have suggested the reduction of water exchange rates as a way to lower the risk for sudden changes in water quality parameters and to minimize the adverse effects of effluents discharge (Páez-Osuna 2001a, b; Primavera 2006), others recommended the increase of water exchange as an effective strategy for reducing the concentrations of potentially toxic compounds, suggesting that such action may also allow substantially higher stocking densities (Burford and Lorenzen 2004).

Aquaculture feeds are determinant for the success of fish culture since they play a major role in fish growth and in the environmental impacts of farm effluents (Tacon and Forster 2003; Nwanna 2003; Bascinar et al. 2007; Booth et al. 2008). As feeding requirements by fish depend on a number of factors such as size, health status and the quality of the holding environment, the definition of optimal feeding rates (i.e. the amount of daily feed supplied as a function of fish biomass) is essential to avoid situations of over- or underfeeding (Wing-Keong et al. 2000; Nwanna 2003), which might not only compromise fish production, but in the case of overfeeding affect the environmental sustainability of this activity. The development of food formulations based on the nutritional requirements and bioenergetics of cultivated species, may also be an effective strategy for improving food assimilation efficiency and consequently food conversion rates (FCRs) as well as to minimize the amount of solid wastes and nutrient loadings resulting from undigested, un-utilized and uneaten feeds (Black 2001; Islam 2005; World Bank 2006; SEACASE 2009). Other authors (Kaushik et al. 2004; SEACASE 2009; Ferreira et al. 2010) have suggested the replacement of fishmeal protein and fat sources of commercial fish feeds by others of vegetable origin (e.g. soya), as a way to produce more “environmentally friendly” feeds (i.e. ecofeeds). Such diet formulations will not only reduce the dependence of aquaculture on fisheries products (Kaushik et al. 2004) but also minimize nutrients concentration in effluent waters due to the elimination of additional protein provided by fish oils (Kaushik et al. 2004) and lower content of soluble nutrients, like phosphorus (SEACASE 2009; Ferreira et al. 2010), thereby contributing for the reduction of semi-intensive aquaculture’s ecological footprint. Other key issue in food management is the improvement of food pellet technology, either by increasing the stability of pellets or reducing its sinking rates (Vassalo et al. 2006; Piedecausa et al. 2009), since it may simultaneously maximise the amount of food ingested and minimize waste production (Choo 2001; Nwanna 2003; World Bank 2006).

Testing the effects of these management options under field conditions is extremely time consuming and often unfeasible from the logistic point of view. Ecological models are a powerful tool to assist in this task because they are able to reproduce fishpond dynamics. Furthermore, models may be used to accurately determine organic matter and nutrient wastes resulting from fish activity, an information that is crucial for dimensioning Integrated Multi-Trophic Aquaculture (IMTA) systems, which have been widely referred as a sustainable option for reducing the environmental impacts of fish farming while increasing its economic efficiency

(Whitmarsh et al. 2006; Bunting and Shpigel 2009; Gazi et al. 2009; Troell et al. 2009; Nobre et al. 2010).

The purpose of this work was to develop a mathematical model that was capable of reproducing fishpond dynamics in order to simulate different management scenarios: i) increase of stocking densities; ii) decrease/increase of water exchange rates; iii) decrease/increase of fish feeding rates, iv) decrease of phosphorus content in fish rations; v) increase of food absorption efficiency and vi) decrease of food pellets sinking velocity. The final goal of this study was to evaluate which scenarios would lead to maximum fish production with minimum impacts for the environment.

2 Methodology

2.1 Description of the system

Data for model calibration was collected during a 2-year white seabream growth trial, carried out in the earth ponds of the IPIMAR Aquaculture Research Center (ARC), located in the Ria Formosa lagoon (Southeast Portugal). A rectangular earth pond with a surface area of 450 m^2 and an approximate volume of 650 m^3 was stocked with 3000 juveniles of white seabream (*Diplodus sargus*). Seawater was supplied to the fishpond at rates varying from 25 to 100 m^3 h^{-1}. The pond was equipped with aerators (FORCE-7; 1.5 hp) in order to maintain dissolved oxygen above critical levels for fish survival. Fish were fed daily with a commercial food pellet containing 51% of total protein, 29% fat and 1.2% total P at 1.2% body wet weight per day in the first production year, and 0.8% in the second year. Monthly ration varied throughout the experiment, between 0.83 and 11.7 kg d^{-1}, according to fish biomass and feeding response (Serpa et al. 2007b).

2.2 Model development

The first step for the development of the model presented herein was to build a biogeochemical model (cf. Chapter 4) that reproduced the cycles of the elements that are more likely to negatively affect fish production and cause undesirable environmental impacts due to their excess (e.g. nitrogen and phosphorus), or deficit (e.g. oxygen). The second step was to build a Dynamic Energy Budget (DEB model) to simulate white seabream growth as a function of the amount of food supplied and water temperature (cf. Chapter 5). The final step for creating a suitable tool to simulate the fishpond system was to couple the two models to account for the interactions between fish and the environment. As the biogeochemical and DEB models were extensively described in the previous chapters (cf. Chapters 4 and 5), the next paragraphs will be devoted to clarify models coupling.

Coupling consists in using the outputs of the DEB model as inputs for the biogeochemical model, and vice-versa (Figure 6.1). In the biological model, not all the food supplied was available to fish due to pellets sedimentation and decay (cf. Chapter 5). Uneaten food is assumed to be an extra source of particulate organic matter (POM) to pond sediments, settling at velocities of 0.035 ± 0.030 m s^{-1}. When reaching the bottom, uneaten food is directly integrated in the benthic organic carbon (POC_s), nitrogen (PON_s) and phosphorus (POP_s) pools according to the C:N:P ratio in fish feeds (Table 6.1), which provides the substrate for the diagenetic processes simulated by the benthic module of the biogeochemical model (cf. Chapter 4).

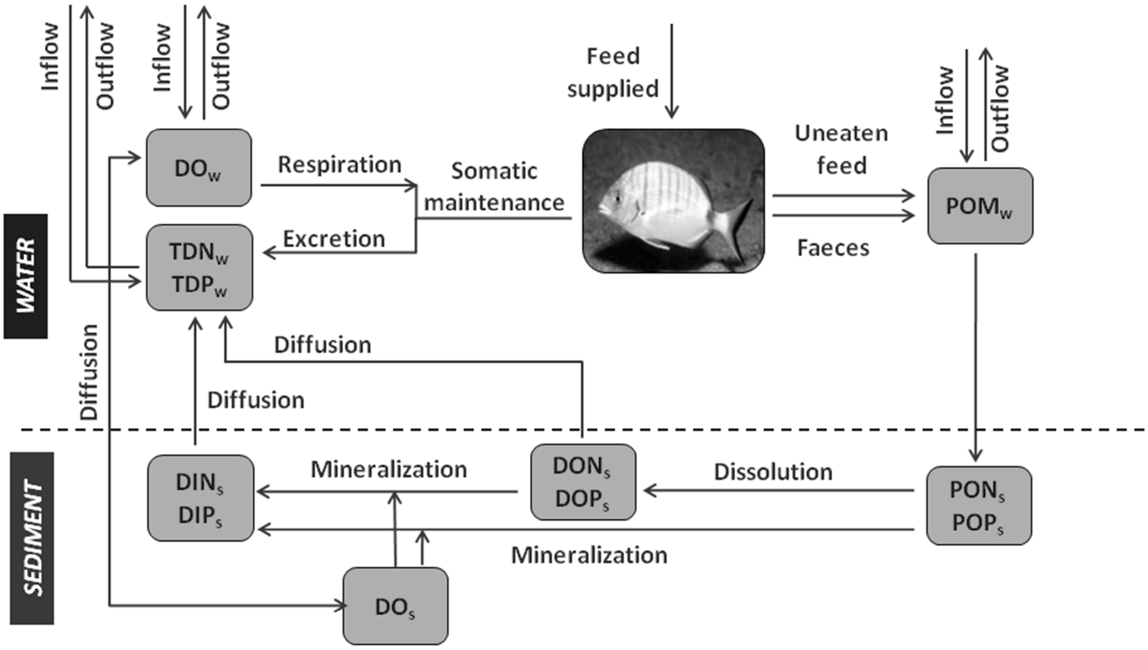


Figure 6.1 – Coupling of the biogeochemical and DEB models. POM_w – Particulate Organic Matter in the water column; PON_s and POP_s – Particulate Organic Nitrogen and Phosphorus in sediments; DON_s and DOP_s – Dissolved Organic Nitrogen and Phosphorus in porewater; DIN_s and DIP_s – Dissolved Inorganic Nitrogen and Phosphorus in porewater; DO_w and DO_s – Dissolved Oxygen in the water column and in porewater; TDN_w and TDP_w – Total Dissolved Nitrogen and Phosphorus in the water column.

Fish activity also influences the environment through respiration and excretion (urinary and fecal) processes. According to the DEB theory (Kooijman 2000, 2010), oxygen consumption is proportional to the energy mobilization rate ($[\dot{p}_C]$). In the model, $[\dot{p}_C]$ values (J cm^{-3} d^{-1}) are converted to DO values (mg) using a conversion factor of 13.84 J mg^{-1} O_2 (Brafield and Llewellyn 1982), so that the oxygen consumption for a single individual (*Resp*; mg O_2 d^{-1} per fish) could be estimated as:

$$Resp = \frac{[\dot{p}_C]}{ConversionJ\_mgO_2} \times V \qquad (1)$$

where $ConversionJ\_mgO_2$ is the conversion factor from Joules to oxygen units and V corresponds to the fish structural body volume (cm^3). Individual results are then extrapolated for the whole pond in order to determine the feedbacks of fish respiration on water column dissolved oxygen (DO$_w$) concentrations. Global fish respiration (*TotalResp*, in mg DO L^{-1} d^{-1}) is calculated as follows:

$$TotalResp = \frac{Resp \times Fish\_nr}{PondVolume \times 1000} \qquad (2)$$

where *Fish_nr* is the number of fish in the pond (3000) and *PondVolume*, the volume of the fishpond (650 m^3). During the respiration process, carbon dioxide (CO_2) is released to the water column. The amount of carbon released during respiration can be estimated using the respiration quotient (RQ), i.e. the ratio between carbon dioxide production and oxygen consumption. In the DEB theory, the RQ depends on the organism's composition in terms of proteins, lipids and polysaccharides (Kooijman 2000). Due to the lack of this information, in the present study it was assumed that the composition of organic matter metabolized by an organism is similar to fish food (Table 6.1), to guarantee that the organism's stoichiometry is not changed by food absorption. Therefore, the RQ value used in the model (0.82) is a weighted average of the RQs of food proteins, lipids and polysaccharides (Kooijman 2000):

$$RQ = \frac{ProtRQ * ProtContent + LipidRQ * LipidContent + PolysaccharRQ * PolysaccharContent}{ProtContent + LipidContent + PolysaccharContent}$$

(3)

where *ProteinsRQ* is the respiration quotient for proteins (0.84), *ProteinsContent* is the protein content in feeds (51%), *LipidsRQ* is the respiration quotient for lipids (0.67), *LipidsContent* is the lipids content in feeds (29%), *PolysaccharidesRQ* is the respiration quotient for polysaccharides (1) and *PolysaccharidesContent* is the polysaccharides content in feeds (20%). The carbon loss by respiration for a single individual (C_{resp}, in mgC d^{-1} per fish) is calculated according to the following equation:

$$C_{resp} = \frac{Resp \times RQ \times CarbonAtomicWeight}{OxygenMolecularWeight} \quad (4)$$

where *Resp* is the oxygen consumption for a single individual and *RQ*, the respiration quotient (0.82). Extrapolation for the whole pond was done by multiplying C_{resp} by the number of fish in the pond (3000).

Table 6.1 – Diet composition, relatively to food dry weight.

	Food (Sorgal® Balance5)
Organic composition (%)	
Proteins	51
Lipids	29
Polysaccharides	20
Elemental composition (%)	
Carbon	41.3 ± 1.2
Nitrogen	6.6 ± 0.6
Phosphorus	1.2 ± 0.3

Source: www.sorgal.pt

The contribution of urinary excretion to the nitrogen (N) and phosphorus (P) pools in pond water is calculated according to the weak homeostasis concept defined in the DEB theory (Kooijman 2000, 2010). In agreement with this concept, carbon and nutrients loss must be directly proportional to feed C, N and P inputs to ensure that the organism's C:N:P ratio is maintained

(Table 6.2). Therefore, nitrogen (N_{excr}, in mg N d^{-1} per fish) and phosphorus excretion (P_{excr}, in mg P d^{-1} per fish) is calculated as follows:

$$N_{excr} = \frac{C_{resp}}{\left(\frac{C}{N}\right)_{food}} \tag{5}$$

$$P_{excr} = \frac{C_{resp}}{\left(\frac{C}{P}\right)_{food}} \tag{6}$$

where $\left(\frac{C}{N}\right)_{food}$ and $\left(\frac{C}{P}\right)_{food}$ correspond to the C:N and C:P ratios in fish food (Table 6.1). The calculated N_{excr} and P_{excr} values for a single individual were multiplied by the number of fish in the pond (3000), for extrapolations to the whole pond.

Table 6.2 – Juvenile and adult white seabream (*Diplodus sargus*) elemental composition, relatively to fish dry weight.

Elemental composition (%)	Juvenile	Adult
Carbon	41.1 ± 2.3	43.9 ± 3.0
Nitrogen	7.11 ± 0.64	7.03 ± 0.56
Phosphorus	1.13 ± 0.05	1.20 ± 0.08

Fecal excretion (or egestion) is assumed to be a source of particulate organic matter to the water, being integrated in POC_w, PON_w and POP_w pools. Since in the DEB model, the egestion rate (*Egestion_rate*) is calculated by the difference between ingestion and assimilation rates (Kooijman 2000, 2010), the *Egestion_rate* had to be converted into mass units to be an input for the biogeochemical model. In the coupled model, egestion (*Egestion*; mg L^{-1} d^{-1}) is calculated according to eq. 7:

$$Egestion = \frac{Egestion\_rate}{FoodEnergyContent \times PondVolume \times 1000} \tag{7}$$

where *FoodEnergyContent* is the energy content of fish feeds (23.8 J mg^{-1} of dry food) and *PondVolume*, the volume of the fishpond (650 m^3). The C, N and P inputs (in mg L^{-1} d^{-1}) from fish feces (respectively, C_{egest}, N_{egest}, P_{egest}) to the suspended particulate organic C, N and P pools, were calculated as follows:

$$C_{Egest} = Egestion \times C_{food} \tag{8}$$

$$N_{Egest} = Egestion \times N_{food} \tag{9}$$

$$P_{Egest} = Egestion \times P_{food} \tag{10}$$

where C_{food}, N_{food}, P_{food}, are respectively the C, N and P contents in fish feeds (Table 6.1).

2.3 Model forcing and calibration

As input data the model requires complete data sets on average daily water temperature (cf. Chapter 4), wind speed (cf. Chapter 4), water inflow and outflow rates (varying from 0.007 to 0.027 m^3 s^{-1}), particulate matter and nutrient concentrations in inflowing water (cf. Chapter 3.2) and food input (cf. Chapter 5). Since both the biogeochemical and DEB models have been previously calibrated (cf. Chapters 4 and 5), only a few parameters were adjusted after models coupling to achieve the best fit between simulated and measured water and sediment quality data. Model parameters needing adjustment were related to water column processes such as, denitrification ($kdenit_w$ = 0.1 d^{-1}) and particulate matter sedimentation (ν = 0.1 m d^{-1}; ν' = 0.24 m d^{-1}), and to benthic processes like nitrification ($knit_s$ = 0.7 d^{-1}) and diffusion ($DsDO$ = 250 cm^2 d^{-1}; $DsNH_4^+$ = 15 cm^2 d^{-1}). For the pelagic compartment, the need for parameters adjustment is explained by fish activity, since the biogeochemical model had been previously calibrated for a pond without fish (cf. Chapter 4). Fish respiration affects dissolved oxygen availability in the water column, which in turn influences denitrification processes (Hargreaves 1998; Burford and Lorenzen 2004), thereby justifying the need for an adjustment in $kdenit_w$. On the other hand, the higher amount of particulate matter in the fishpond, as a result of uneaten food and fish feces as well as higher water exchange rates, justifies the increase in total and particulate matter settling velocities (cf. Chapter 4). Bottom sediment enrichment implies a decrease in oxygen availability, which restricted nitrification processes (Serpa et al. 2007b). Therefore, to ensure a good model fit between predicted and observed porewater NO_x concentrations a change in $knit_s$ was required. The increase in DO and NH_4^+ diffusion coefficients was mostly necessary to account for the effects of bioturbation, either from benthic fauna activity (cf. Chapter 3.2) or the foraging activity of fish (Riise and Roos 1997; Hargreaves 1998).

2.4 Model implementation

The coupled biogeochemical – fish DEB model was implemented with EcoDynamo (Pereira et al. 2006), and state variables were computed over time using the Euler integration method with a time step of 10 minutes.

2.5 Model performance

Model performance was evaluated by model II regressions between predicted and observed values of state variables, as described by Laws and Archie (1981). According to these authors, a good model fit ($p<0.05$) implies that the slope of the regressions is not different from one and the y-intercept is not different from zero. A slope that significantly differs from one indicates a difference between observed and simulated values which is proportional to the observed values. If the slope is not significantly different from one but the y-intercept significantly differs from zero there is a systematic difference between observations and simulations.

2.6 Scenario analysis

After calibration of the fishpond model, several management scenarios (Table 6.3) were simulated with the purpose of optimizing white seabream production in semi-intensive earth ponds. These scenarios were defined based on the management options proposed by several authors (cf. Section 1) for maximizing fish production while minimizing aquaculture environmental impacts. The effects of each scenario on pond water and sediment quality were evaluated by comparing scenario simulations with the standard simulation (after model calibration). These comparisons were made on the basis of average values for several model variables, integrated over the second year of the production cycle. The evaluation of pond water and sediment quality for the different scenarios was based on the IFREMER classification schemes (Austoni et al. 2004). However, for water quality, this scheme had to be modified because model outputs did not include turbidity. Instead, total particulate matter (TPM) were used as a water quality parameter, using as reference values for this variable, the Ria Formosa lagoon values measured in areas under different levels of anthropogenic pressure (Falcão 1997). Besides the quality of pond environment, each scenario was evaluated for its effects on final fish biomass and for impacts on the adjacent water body, to comply with the objectives of the present study. Environmental impacts were evaluated by total N and P discharges during a white seabream production cycle, which were obtained by multiplying total dissolved nitrogen (TN) and phosphorus (TP) concentrations in pond water by daily water outflow rates.

Table 6.3 – Scenarios and management options for semi-intensive pond aquaculture. Current rearing conditions (standard simulation): Stocking density = 1.2 kg m^{-3}; Water exchange rate = 25 to 100 m^3 h^{-1}; Feeding rate = 0.8 to 1.2% of fish body weight; Phosphorus content in feeds = 1.2%; Food absorption efficiency = 72%; Pellets sinking velocity = 0.035 m s^{-1}.

Scenario	**Management options**
Change stocking density	Increase stocking density to 3 kg m^{-3} (**Scenario1**)
Change water exchange rate	Decrease water exchange rate by 25% (**Scenario2**) Increase water exchange rate by 25% (**Scenario3**)
Change feeding rate	Decrease feeding rate to 0.4 to 0.6% of fish body weight per day (**Scenario4**) Increase feeding rate to 1.6 to 2.4% of fish body weight per day (**Scenario5**)
Change diet formulations	Decrease phosphorus content in feeds to 1.0% (**Scenario6**)
Change food absorption efficiency	Increase food absorption efficiency to 83% (**Scenario7**)
Change pellets sinking velocity	Decrease pellets sinking velocity to 0.018 m s^{-1}(**Scenario8**)

2.6.1 Scenario classification

The selection of best and worst scenarios was based on the Analytical Hierarchical Process (Saaty 1980), a mathematical technique for multicriteria decision making. In the present work, the decision elements or objectives, were: i) quality of the pond environment, ii) environmental impacts and iii) fish production. The quality of the pond environment was evaluated based on water column and sediment variables (cf. Section 2.6), combined into a single indicator by applying an AHP, in which each variable is a decision element with equal relevance for the pond environment. A similar procedure was followed for the second decision element, but in this case, total N and P discharges during a white seabream production cycle were the only variables defining the indicator. Fish production was exclusively evaluated based on the final fish biomass predicted by the model.

The Analytical Hierarchical Process (AHP) was conducted in several steps:

1. *Performance of pairwise comparisons* – comparison of the relative importance (i.e. weight) of each decision element according to the following scale: 1 – equally important; 2 – moderately more important and 3 – strongly more important. In one situation it was assumed that the 3 decision elements were equally important (A_1); whereas in the other, the most important decision element was considered to be fish production (A_2) for economic reasons. The different pairwised combinations for each situation formed two different reciprocal matrices (3 × 3), respectively A_1 and A_2 (see below), to ensure the consistency of the results.

$$A_1 = \begin{array}{cccl} \text{Pond qual.} & \text{Environ. impact} & \text{Fish prod.} & \\ 1 & 1 & 1 & \text{Pond qual.} \\ 1 & 1 & 1 & \text{Environ. impact} \\ 1 & 1 & 1 & \text{Fish prod.} \end{array} \qquad A_2 = \begin{array}{cccl} \text{Pond qual.} & \text{Environ. impact} & \text{Fish prod.} & \\ 1 & {}^{1}\!/_{2} & {}^{1}\!/_{3} & \text{Pond qual.} \\ 2 & 1 & {}^{1}\!/_{2} & \text{Environ. impact} \\ 3 & 2 & 1 & \text{Fish prod.} \end{array}$$

2. *Matrix normalization* – matrices A_1 and A_2 were normalized with eq. 13:

$$\overline{ajk} = \frac{ajk}{\sum_{j=1}^{m} ajk} \tag{13}$$

where *m* is the number of lines in the matrix.

3. *Computation of the relative weights* – the “average” weight for each decision element/indicator was calculated by eq. 14:

$$W_j = \frac{\sum_{l=1}^{n} ajl}{n} \tag{14}$$

4. Standardization – the matrix containing the values of the different indicators for each scenario (*I)* was standardized to allow the comparison between indicators. Standardization was done according to the equations of Agnetis et al. (2006), as follows:

- When the indicator value was directly proportional to quality and $I_{i,j} > I_{h,j}$, eq. 15 was used:

  $$b_{ih}^{j} = 8\frac{I_{ij-}I_{hj}}{I_{j,max}-I_{j,min}} + 1 \quad (15)$$

  If $I_{i,j} <= I_{h,j}$, then $b_{ih} = 1/b_{hi}$

- When the indicator was inversely proportional to quality and $I_{i,j} < I_{h,j}$, eq. 16 was applied:

  $$b_{ih}^{j} = 8\frac{I_{hi-}I_{ij}}{I_{j,max}-I_{j,min}} + 1 \quad (16)$$

  If $I_{i,j} >= I_{h,j}$, then $b_{ih} = 1/b_{hi}$

The outcome of the standardization exercise was a matrix (9 × 9) for each indicator. Each of these matrices was further normalized using eq. 13, generating 3 vectors that constituted an *S* matrix (9 × 3), in which columns correspond to indicators and lines to scenarios.

5. Calculation of global scores - the score (v) of each scenario was obtained by multiplying the *S* matrix by the *W* matrix (3 × 1), as follows:

$$v = S.W \quad (17)$$

3 Results

The first part of this section (Model performance) synthesizes some comparisons between observed and simulated data (from the standard simulation). The second part deals with scenario analysis, where results obtained with the different scenarios (Table 6.3) are compared to those of the standard one.

3.1 Model performance

Comparisons between predicted and observed values for the water column, porewater, sediment and fish variables are shown in Figures 6.2 – 6.8.

3.1.1 Water column variables

The results of model II regressions between predicted and measured values of water column variables (Table 6.4), show that the model could not fully predict (p>0.05) the variability of particulate matter (TPM_w and POM_w) concentrations over the white seabream production cycle (Figure 6.2). A poor model fit (p>0.05) was found for ammonium ($NH_4^+{}_w$) in pond water (Table 6.4), even though the model was able to predict the majority (5 out of 8) of data points (Figure 6.3). In opposition, a significant part (p<0.04) of oxidized nitrogen forms (NO_{xw}) and phosphate ($HPO_4^{2-}{}_w$) variability was explained by the model (Table 6.4), despite some discrepancies between model predictions and observations (Figure 6.3). Differences between dissolved organic compounds (DON_w and DOP_w) predicted and measured values (Figure 6.3), prevented the model from accurately simulating these variables (p>0.05). A significant part of DO_w variability was explained by the model (p<0.01 - Table 6.4), despite some underestimation of DO values during spring and summer months (Figure 6.3).

Table 6.4 – Results of model II regressions for water column variables.

		TPM$_w$	POM$_w$	NH$_4^+$$_w$	NO$_{xw}$	HPO$_4^{2-}$$_w$	DON$_w$	DOP$_w$	DO$_w$
		mg L^{-1}	mg L^{-1}	µM	µM	µM	µM	µM	mg L^{-1}
slope	Value	0.53	0.72	0.02	0.56	1.38	1.71	0.30	0.38
	Upper 95% limit	3.04	-8.95	0.64	0.79	4.94	0.30	1.35	0.43
	Lower 95% limit	-0.29	-0.47	-0.60	0.37	0.57	-4.42	-0.54	0.34
y-intercept	Value	19.49	2.07	4.20	0.18	0.04	-4.36	0.51	4.30
	Upper 95% limit	50.62	39.70	7.70	0.42	0.21	41.30.1	1.06	4.64
	Lower 95% limit	-76.28	6.69	0.63	-0.10	-0.69	6.15	-0.51	3.95
p	Value	0.20	0.23	0.94	<0.01	0.03	0.17	0.98	<0.01

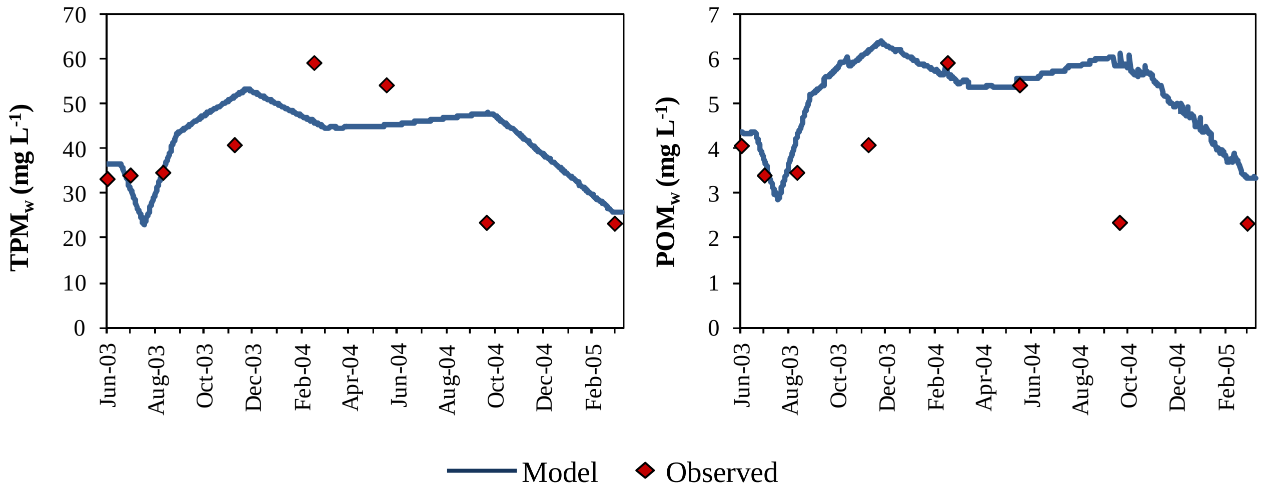


Figure 6.2 – Predicted (line) and observed (diamonds) values of total particulate matter (TPM$_w$, mg L^{-1}) and particulate organic matter (POM$_w$, mg L^{-1}) in the water column of a white seabream production pond.

3.1.2 Porewater variables

In general, the model could not fully predict (p>0.05) the variation pattern of porewater variables (Figures 6.4 – 6.5; Table 6.5), however for some variables like DON$_s$ and NO$_{xs}$, predicted values were very close to measured data (Figure 6.4). In fact, for the latter variable, small discrepancies were found between model predictions and observations, except in October 2004, when the model was unable to simulate a peak in NO$_{xs}$ concentrations. On the other hand, the model was able to explain a significant part (p<0.01) of porewater HPO$_4^{2-}$$_s$ variability (Table 6.5), despite some over- or underestimation of its concentrations (Figure 6.5).

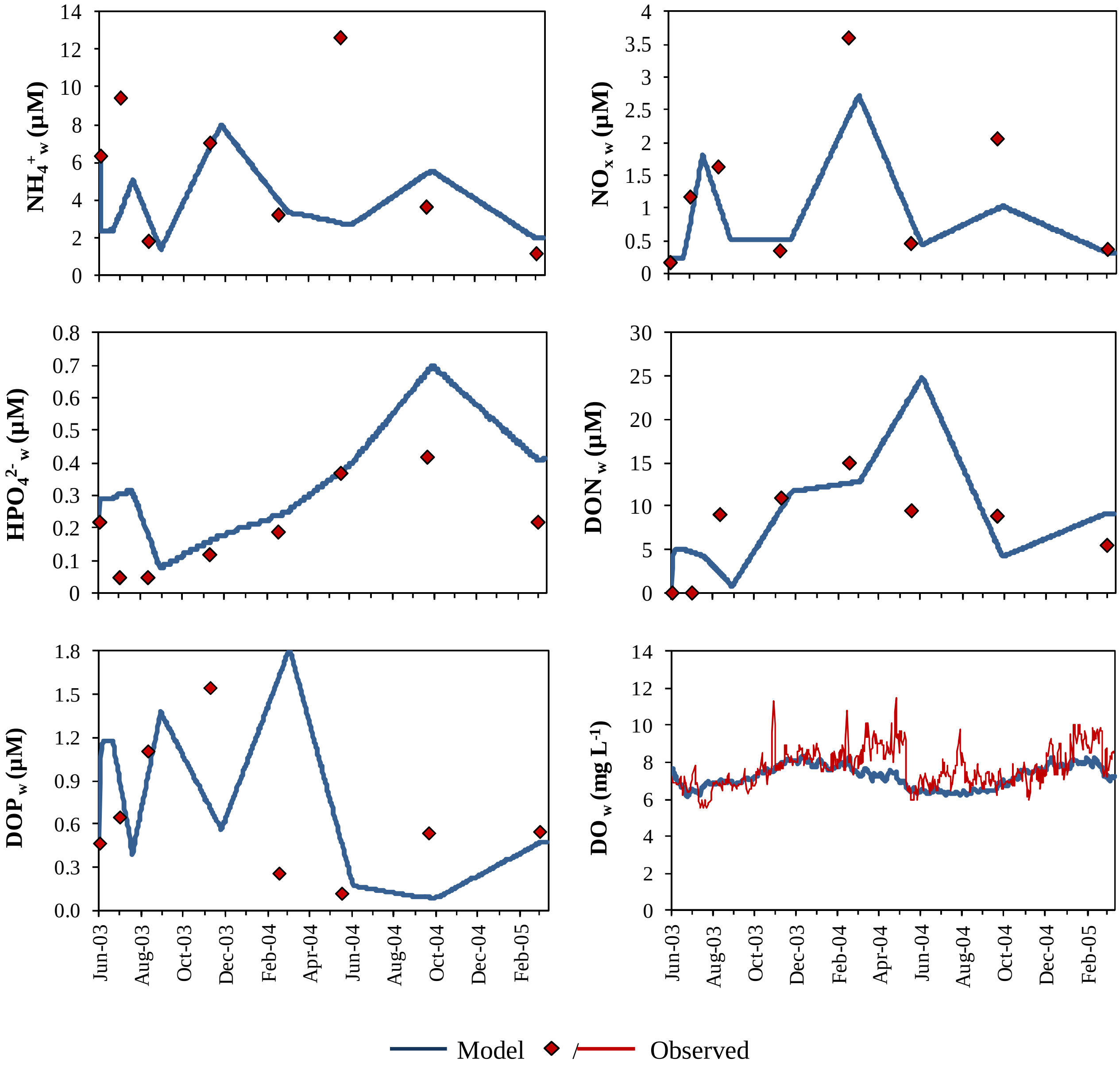


Figure 6.3 – Predicted (blue line) and observed (red diamonds and line) ammonium ($NH_4^+{}_w$), oxidized nitrogen forms (NO_{xw}), dissolved organic nitrogen (DON_w), phosphate ($HPO_4^{2-}{}_w$), dissolved organic phosphorus (DOP_w) and dissolved oxygen (DO_w) concentrations in the water column of a white seabream production pond.

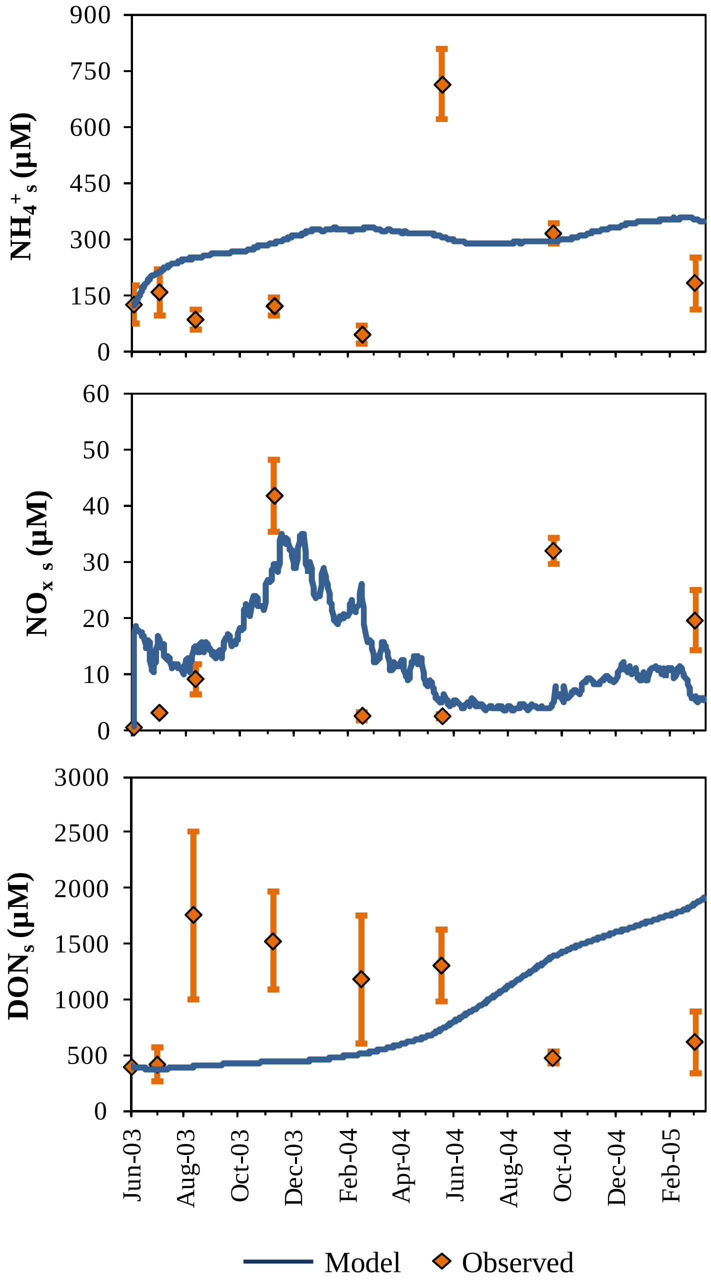


Figure 6.4 – Predicted (line) and observed ± standard deviation (diamonds) ammonium ($NH_4^+{}_s$), oxidized nitrogen forms (NO_{xs}) and dissolved organic nitrogen (DON_s) porewater concentrations in a white seabream production pond.

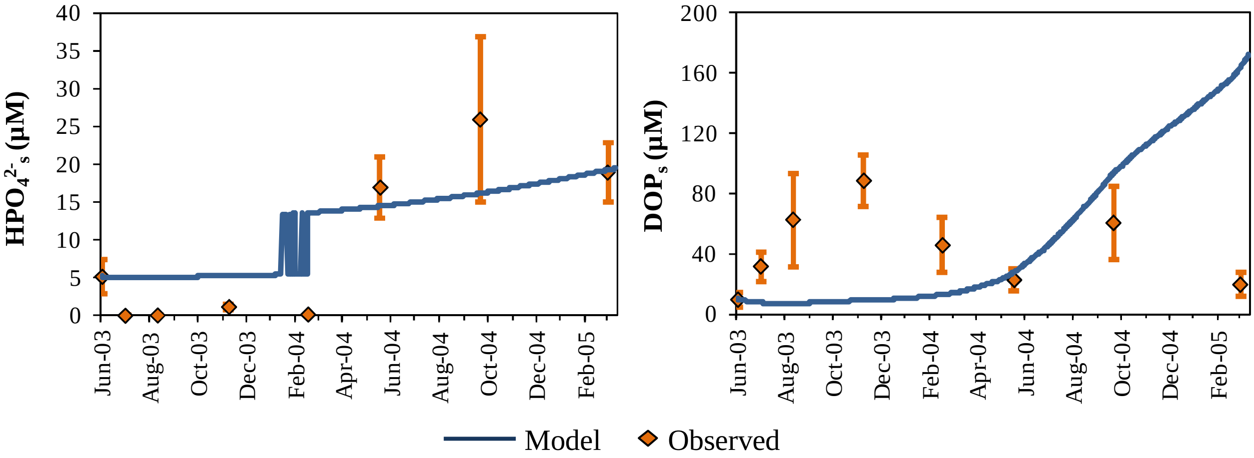


Figure 6.5 – Predicted (line) and observed ± standard deviation (diamonds) phosphate ($HPO_4^{2-}{}_s$) and dissolved organic phosphorus (DOP_s) porewater concentrations in a white seabream production pond.

Table 6.5 – Results of model II regressions for porewater variables.

		$NH_4^+{}_s$ µM	NO_{xs} µM	$HPO_4^{2-}{}_s$ µM	DON_s µM	DOP_s µM
slope	Value	0.09	0.30	0.56	-1.03	-9.33
	Upper 95% limit	0.49	-2.28	0.79	3.54	2.46
	Lower 95% limit	-0.29	-5.97	0.37	0.25	-1.46
y-intercept	Value	249.54	8.59	4.62	1753.53	448.04
	Upper 95% limit	333.07	96.72	6.29	518.21	109.91
	Lower 95% limit	159.98	44.80	2.64	-2658.80	-57.80
p	Value	0.57	0.52	<0.01	0.36	0.60

3.1.3 Sediment variables

Figure 6.6 presents the comparison between predicted and observed values of sediment variables, POC_s, PON_s and POP_s. From this figure it becomes evident that there was little discrepancy between model simulations and measured data, which is indicative of a good model fit (p<0.01) as confirmed by the results of the regression analysis (Table 6.6).

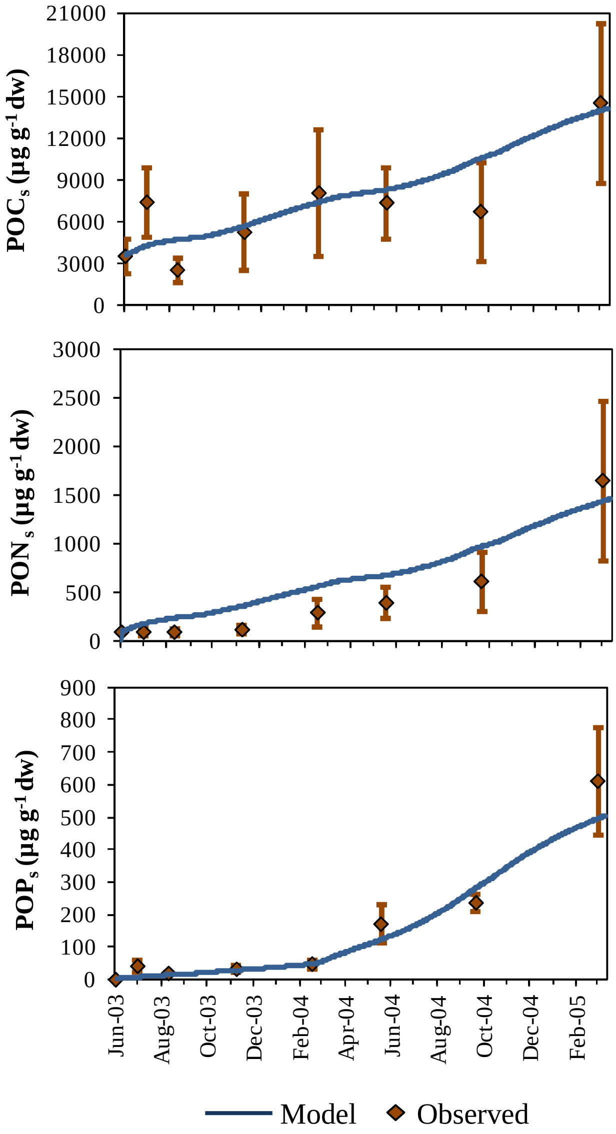


Figure 6.6 – Predicted (line) and observed ± standard deviation (diamonds) organic carbon (POC$_s$), nitrogen (PON$_s$) and phosphorus (POP$_s$) content in the sediments of a white seabream production pond.

Table 6.6 – Results of model II regressions for sediment variables.

		POC_s µg g^{-1} dw	PON_s µg g^{-1} dw	POP_s µg g^{-1} dw
slope	Value	0.97	0.85	0.85
	Upper 95% limit	1.97	1.21	1.04
	Lower 95% limit	0.47	0.58	0.70
y-intercept	Value	549.91	203.02	-0.94
	Upper 95% limit	4038.73	316.42	21.98
	Lower 95% limit	-6426.57	49.13	-28.04
p	Value	<0.01	<0.01	<0.01

3.1.4 Biological variables

Model II regressions between predicted and measured values of fish wet weight and total length (Table 6.7), revealed that the model was able to explain a significant part ($p<0.01$) of growth data variability. Nevertheless, a systematic overestimation of fish weight was found (Table 6.7), whereas for total length the model could over-or underestimate (Figure 6.7) this variable depending on its value.

Table 6.7 – Results of model II regressions for fish growth variables.

		Fish weight	Total length
		g	cm
slope	Value	1.02	0.77
	Upper 95% limit	1.20	0.87
	Lower 95% limit	0.86	0.69
y-intercept	Value	-20.99	3.14
	Upper 95% limit	-1.25	4.53
	Lower 95% limit	-44.41	1.61
p	Value	<0.01	<0.01

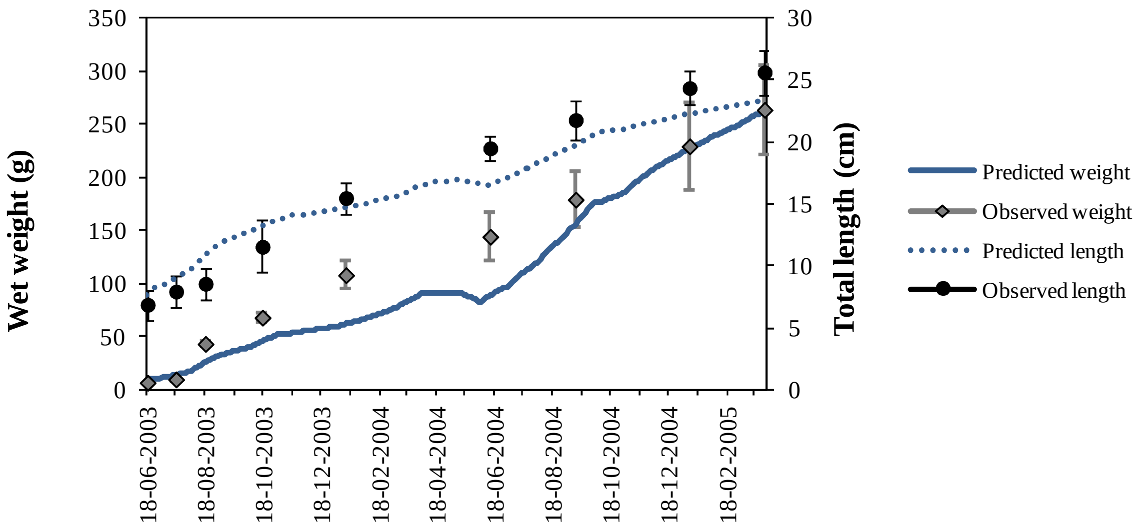


Figure 6.7 – Average ± standard deviation measured and predicted values of white seabream wet weight (respectively, diamonds and solid line) and total length (respectively, circles and dashed line) over a production cycle.

3.2 Scenario analysis

3.2.1 Water quality

In the standard simulation and in each of the scenarios analysed in the present work, most water quality variables fell into the category “High” and “Good”, except for TPM_w that was in the “Moderate” category in all situations (Table 6.8). Despite the fact that there were no water quality constraints regarding the scenarios simulated with the present model, the results suggest that some management options such as, increase of fish stocking densities (Scenario1) and decrease of water exchange rates (Scenario2), negatively affect pond water quality. In Scenario1, $NH_4^+{}_w$ and $HPO_4^{2-}{}_w$ concentrations increased 32 and 20%, respectively, compared to the standard simulation, whereas in Scenario2 these compounds increased 18 and 25%, respectively. The management option that is more likely to improve water quality is the decrease of feeding rates (Scenario4), since $NH_4^+{}_w$ and $HPO_4^{2-}{}_w$ concentrations decreased 16% and 9%; respectively, whereas the opposite scenario, i.e. the increase of fish feeding rates (Scenario5), not only increased the availability of $NH_4^+{}_w$ and $HPO_4^{2-}{}_w$ by 11 and 14%, respectively, as also produced a 4% decrease in DO_w. For the remaining scenarios, water column variables were very similar to the standard simulation.

Table 6.8 – Water quality assessment for the standard simulation and for each of the management scenarios considered in the present work. Quality status: BLUE – High; GREEN – Good; YELLOW – Moderate; ORANGE – Poor; and RED – Bad. Adapted from: Austoni et al. (2004).

	Variables	Units		🟦		🟩		🟨		🟧		🟥
Standard Simulation	TPM	mg L^{-1}	0		20		40	**40.61**	60		80	
	DO	%	100	**93.46**	80		70		60		50	
	NH_4^+	µM	0	**4.01**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.68**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.56**	1		1.5		4	
	TN	µM	0	**14.84**	50		75		100		120	
	TP	µM	0	**0.78**	1		2		5		8	
Scenario1	TPM	mg L^{-1}	0		20		40	**40.97**	60		80	
	DO	%	100	**87.37**	80		70		60		50	
	NH_4^+	µM	0	**5.31**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.69**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.67**	1		1.5		4	
	TN	µM	0	**16.15**	50		75		100		120	
	TP	µM	0	**0.89**	1		2		5		8	
Scenario2	TPM	mg L^{-1}	0		20		40	**40.73**	60		80	
	DO	%	100	**91.09**	80		70		60		50	
	NH_4^+	µM	0	**4.71**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.69**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.62**	1		1.5		4	
	TN	µM	0	**15.55**	50		75		100		120	
	TP	µM	0	**0.84**	1		2		5		8	

Table 6.8 – (continued)

	Variables	Units		🟦		🟩		🟨		🟧		🟥
Scenario3	TPM	mg L^{-1}	0		20		40	**40.56**	60		80	
	DO	%	100	**94.86**	80		70		60		50	
	NH_4^+	µM	0	**3.60**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.68**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.53**	1		1.5		4	
	TN	µM	0	**14.43**	50		75		100		120	
	TP	µM	0	**0.75**	1		2		5		8	
Scenario4	TPM	mg L^{-1}	0		20		40	**40.40**	60		80	
	DO	%	100	**95.90**	80		70		60		50	
	NH_4^+	µM	0	**3.66**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.68**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.51**	1		1.5		4	
	TN	µM	0	**14.49**	50		75		100		120	
	TP	µM	0	**0.73**	1		2		5		8	
Scenario5	TPM	mg L^{-1}	0		20		40	**41.15**	60		80	
	DO	%	100	**89.68**	80		70		60		50	
	NH_4^+	µM	0	**5.02**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.68**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.64**	1		1.5		4	
	TN	µM	0	**15.85**	50		75		100		120	
	TP	µM	0	**0.86**	1		2		5		8	

Table 6.8 – (continued)

	Variables	Units		🟦		🟩		🟨		🟧		🟥
Scenario6	TPM	mg L^{-1}	0		20		40	**40.62**	60		80	
	DO	%	100	**83.45**	80		70		60		50	
	NH_4^+	µM	0	**4.01**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.68**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.51**	1		1.5		4	
	TN	µM	0	**14.84**	50		75		100		120	
	TP	µM	0	**0.73**	1		2		5		8	
Scenario7	TPM	mg L^{-1}	0		20		40	**40.46**	60		80	
	DO	%	100	**92.12**	80		70		60		50	
	NH_4^+	µM	0	**4.27**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.68**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.58**	1		1.5		4	
	TN	µM	0	**15.10**	50		75		100		120	
	TP	µM	0	**0.80**	1		2		5		8	
Scenario8	TPM	mg L^{-1}	0		20		40	**40.64**	60		80	
	DO	%	100	**93.36**	80		70		60		50	
	NH_4^+	µM	0	**4.03**	7		10		20		30	
	$NO_2^- + NO_3^-$	µM	0	**0.68**	7.5		11		25		40	
	HPO_4^{2-}	µM	0		0.3	**0.56**	1		1.5		4	
	TN	µM	0	**14.86**	50		75		100		120	
	TP	µM	0	**0.78**	1		2		5		8	

3.2.2 Sediment quality

Regarding sediment quality, model outputs suggest an upgrade in the case of Scenario4 (i.e. decrease of feeding rates), with all variables falling into the categories “High” (Table 6.9). Quantitatively, this improvement was the result of a decrease, of respectively 15, 23 and 13%, in the organic matter (OM), total nitrogen (TN) and total phosphorus (TP) sediment contents relatively to the standard simulation. Conversely, sediment quality appears to worsen under Scenario5 (i.e. increase of feeding rates) since TP fell into the category “Poor” instead of “High” as in the standard simulation (Table 6.9). Besides leading to a 30% increase in TP, doubling feeding rates produced a substantial increase in the organic matter and total nitrogen contents of pond sediments, which increased 34 and 55%, respectively, compared to the standard simulation. Similarly to Scenario4, increasing the floatability of food pellets (Scenario8) upgraded TN to the category “High”. These results reflect a substantial decrease, respectively of 12, 20 and 10% in organic matter, total nitrogen and total phosphorus contents in bottom sediments. In all the other scenarios, OM, TN and TP varied slightly compared to the standard simulation (Table 6.9).

Table 6.9 – Sediment quality assessment (Austoni et al. 2004) for the standard simulation and for each of the management scenarios considered in the present work. Quality status: BLUE – High; GREEN – Good; YELLOW – Moderate; ORANGE – Poor; and RED – Bad.

	Variables	Units		🟦		🟩		🟨		🟧		🟥
Standard Simulation	OM	%	0	**3.15**	3.5		5.0		7.5		10	
	TN	g kg^{-1} dw	0		1.0	**1.07**	2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0	**362**	400		500		600		700	
Scenario1	OM	%	0		3.5	**3.81**	5.0		7.5		10	
	TN	g kg^{-1} dw	0		1.0	**1.43**	2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0		400	**428**	500		600		700	
Scenario2	OM	%	0	**3.26**	3.5		5.0		7.5		10	
	TN	g kg^{-1} dw	0		1.0	**1.13**	2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0	**372**	400		500		600		700	
Scenario3	OM	%	0	**3.07**	3.5		5.0		7.5		10	
	TN	g kg^{-1} dw	0		1.0	**1.02**	2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0	**354**	400		500		600		700	

Table 6.9 – (continued)

	Variables	Units		🟦		🟩		🟨		🟧		🟥
Scenario4	OM	%	0	**2.69**	3.5		5.0		7.5		10	
	TN	g kg^{-1} dw	0	**0.82**	1.0		2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0	**316**	400		500		600		700	
Scenario5	OM	%	0		3.5	**4.22**	5.0		7.5		10	
	TN	g kg^{-1} dw	0		1.0	**1.66**	2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0		400	**470**	500		600		700	
Scenario6	OM	%	0	**3.15**	3.5		5.0		7.5		10	
	TN	g kg^{-1} dw	0		1.0	**1.07**	2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0	**340**	400		500		600		700	
Scenario7	OM	%	0	**3.05**	3.5		5.0		7.5		10	
	TN	g kg^{-1} dw	0		1.0	**1.01**	2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0	**351**	400		500		600		700	
Scenario8	OM	%	0	**2.78**	3.5		5.0		7.5		10	
	TN	g kg^{-1} dw	0	**0.86**	1.0		2.0		3.0		4.0	
	TP	mg kg^{-1} dw	0	**325**	400		500		600		700	

3.2.3 Fish biomass

Figure 6.8 presents the results of final fish biomass for the standard simulation and for each scenario analysed with the model. Comparing to the standard simulation, final fish biomass more than doubled in Scenario1, whereas in Scenario2 there was a small increase, ca. 10%, in fish production. On the contrary, increasing water exchange rates by 25% (Scenario3) had a negative impact on final fish production, since this variable decreased 8% relatively to the standard simulation. A similar result was observed when reducing feeding rates (Scenario4) since fish production decreased 45%, whereas the opposite scenario (Scenario5) yielded a considerable increase (≈ 66%) in the amount of fish produced. A substantial increment (≈ 234 kg) in final fish biomass was also observed when increasing food absorption efficiency (Scenario7), while for Scenario8, fish production increased slightly, ca. 2%, when compared to the standard simulation. The only scenario that did not affect fish biomass, neither positively or negatively was Scenario6, i.e. the decrease of P content in feeds (Figure 6.8).

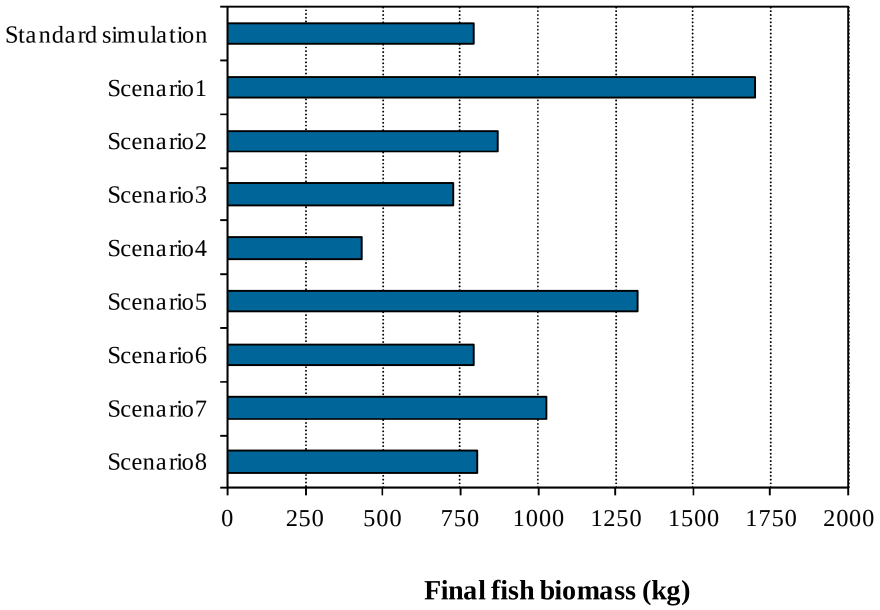


Figure 6.8 – Final fish biomass (kg) in a white seabream production pond, for the standard simulation and for each of the management scenarios analysed in the present work.

3.2.4 Environmental impacts

The total amount of N and P discharged over a white seabream production cycle (651 days), for the different scenarios evaluated in this study, is presented in Table 6.10. The highest nutrient discharges (ca. 249 kg of N and 37 kg of P) were recorded for Scenario1 (i.e. increase stocking density). Opposite scenarios, such as Scenario2 (i.e. decrease water exchange rate) and Scenario3 (i.e. increase water exchange rate), had opposite effects on nutrient discharges. For instance, while Scenario2 yielded a 3% increase in N and P discharges in comparison to the standard simulation, Scenario3 produced a 2% and 7% decrease respectively, in the amount of N and P released into the environment (Table 6.10). A similar situation was observed for Scenario4 and Scenario5. Decreasing feeding rates resulted in a decrease of N and P discharges relatively to the standard simulation, respectively of 6 and 1 kg of N and P, whereas increasing the amount of food supplied (Scenario5) increased nutrient loads to the environment. As would be expected, Scenario6 (i.e. decrease of P content in feeds) resulted in the lowest P discharges, however this represented only a 4% decrease in comparison with the standard simulation. On the other hand, increasing the absorption efficiency of fish food led to the lowest values for N discharges (ca. 224 kg). As regards Scenario8, results were similar to the standard simulation - ca. 237 and 35 kg of respectively, N and P, discharged from the white seabream pond during the production cycle.

3.2.5 Scenario classification

The results of the Analytical Hierarchical Process (AHP) used to identify the management options that maximize fish production while improving the quality of the pond environment and minimizing the environmental impacts of semi-intensive aquaculture, are presented in Table 6.11. The outcome of the AHP strongly depended on the criteria used (A_1 or A_2). When assuming that the three decision elements are equally important for fish farming, the best management option is the decrease of feeding rates (Scenario4), even though final fish biomass decreased 45% compared to the standard simulation (Figure 6.8). Scenario1 (i.e. increase stocking density) and Scenario3 (i.e. increase water exchange rate) ranked respectively, in second and third place according to criteria A_1. On the other hand, if A_2 is applied, i.e. if fish production is the most important indicator, the highest score is that of Scenario1, since it substantially increased (by 906 kg) final fish production (Figure 6.8). Ranking on second and third place, in this case, were respectively, Scenario4 and Scenario7 (i.e. increase food absorption efficiency). Independently from the criteria used, the worst management option was the decrease of water exchange rates (Scenario2).

Table 6.10 – Total nitrogen (TN) and phosphorus (TP) discharges (kg) during a white seabream production cycle. Values in bold correspond to the best scenarios.

	TN	TP
	kg	kg
Standard Simulation	236.2	34.7
Scenario1	249.2	37.1
Scenario2	243.2	35.9
Scenario3	232.2	34.0
Scenario4	229.9	33.6
Scenario5	245.9	36.5
Scenario6	236.2	**33.4**
Scenario7	**224.2**	34.2
Scenario8	236.5	34.8

Table 6.11 – Analytical Hierarchical Approach for each scenario analysed in the present work, considering equal weights (A_1) and different weights (A_2 – higher weight for the production indicator) for the indicators. Values in bold correspond to the best scenarios.

	A_1	A_2
Standard simulation	0.061	0.057
Scenario1	0.143	**0.217**
Scenario2	0.043	0.051
Scenario3	0.123	0.096
Scenario4	**0.233**	0.165
Scenario5	0.084	0.122
Scenario6	0.115	0.102
Scenario7	0.120	0.125
Scenario8	0.077	0.066

4 Discussion

4.1 Model performance

The coupling between the biogeochemical model and the white seabream DEB model developed in the previous chapters recreated reasonably well the dynamics of fishponds. Nevertheless, some water column variables were occasionally over- or underestimated. For example, particulate matter (TPM_w and POM_w) was typically overestimated during autumn periods (Figure 6.2). One possible explanation for these results is that, conversely to the model that assumed constant settling velocities for these compounds, under field conditions, TPM_w and POM_w sedimentation rates probably changed over time, as a result of specific gravities for the different types of biogenic material (e.g. fish waste or senescent algae) in the pond (Avnimelech and Kochba 1999; Jiménez-Montealegre et al. 2002b; Magill et al. 2006).

The model also underestimated the $NH_4^+{}_w$ summer peaks (Figure 6.3) as described for the pond without fish (cf. Chapter 4). These findings are presumably related to sediment resuspension, which was not calculated by the model. Besides wind-driven water turbulence and benthic fauna activity, the disturbance of bottom sediments due to fish foraging activity or mechanical aeration, has been reported to promote NH_4^+ desorption from sediment particles, subsequently leading to an increase in water column NH_4^+ concentrations (Riise and Roos 1997; Hargreaves 1998; Chakrabarty and Das 2007). This process may be particularly relevant during summer months as a result of intense biological activity and the recurrent use of aeration to overcome low oxygen availability in fishponds (Riise and Roos 1997; Hargreaves 1998; Chakrabarty and Das 2007). Another hypothesis for lower $NH_4^+{}_w$ predictions during summer months is the decline of primary producers (Hargreaves 1998; Hargreaves 2006). As macroalgae were not common in the production pond, as opposed to the control pond (Serpa, *unpublished results*), the crash of phytoplankton communities as a result of photoinhibition during periods of higher temperatures, could explain the increase of $NH_4^+{}_w$ availability in the water column (Krom and Neori 1989; Hargreaves 1998; Hargreaves 2006). However, as chlorophyll *a* concentrations registered the highest values (3.5 to 6.4 $\mu g\ L^{-1}$) during summer months (Serpa, *unpublished results*) this hypothesis may be disregarded.

Discrepancies between predicted and measured DON_w and DOP_w values were observed over the production cycle, but the majority of data points were well reproduced by the model (Figure 6.3). Some unaccounted sources or sinks might explain these results. When developing the biogeochemical model (cf. Chapter 4) it was assumed that, supply water and POM hydrolysis were the main sources of dissolved organic N and P compounds to the system whereas most losses occurred via mineralization processes and outflowing water. However, in aquatic systems, DON_w and DOP_w may be released from phytoplankton, algae and protists (Berman and Bronk 2003; Sañudo-Whilhemy 2006) and exploited by the biota, either directly via enzymatic hydrolysis or after bacterial degradation (Berman and Bronk 2003; Sañudo-Whilhemy 2006). The omission of biotic interactions with DON_w and DOP_w pools is likely to have compromised the model ability to simulate the dynamics of dissolved organic N and P compounds in the fishpond.

The underestimation of dissolved oxygen (DO_w) values during spring and summer months (Figure 6.3), suggests the existence of an additional source of DO in fishponds. According to different authors (Culberson and Piedrahita 1996; Hargreaves 1998; Mwegoha et al. 2010), a combination of natural processes like, phytoplankton photosynthetic activity and water reaeration due to turbulent atmospheric conditions, as well as artificial processes like, mechanical aeration and oxygenation, might explain DO_w fluctuations over time. As organic matter biodegradation is one of the most important sinks of DO_w in fishponds (Holmer et al. 2002; Mwegoha et al. 2010), one might conjecture that mineralization rates in field conditions were lower than those predicted by the model. However, as in the sensitivity analysis performed to the biogeochemical model (cf. Chapter 4), water column DO concentrations were practically unaffected by parameters related to mineralization processes, it is unlikely that this process might have accounted for model DO_w underestimation during spring and summer months.

As regards sediment variables, model performance was fairly good for the solid fraction (Figures 6.6; Table 6.6), however, porewater variables were not so well simulated (Figures 6.4 – 6.5; Table 6.5). When comparing model results for the white seabream and the control pond (cf. Chapter 4), it becomes evident that there are common limitations regarding porewater compounds, which may indicate that relevant biogeochemical processes were not included in the model or were not well reproduced. Unlike in the control pond (cf. Chapter 4), the model consistently overestimated porewater ammonium ($NH_4^+{}_s$) in the fishpond, except in June 2004 (Figure 6.4), when this variable was underestimated in both ponds (cf. Chapter 4). The overestimation of $NH_4^+{}_s$ concentrations could be related to benthic oxygen diffusion because in the present model the DO diffusion coefficient (*DsDO*), which substantially increases $NH_4^+{}_s$ production (cf. Chapter 4), was increased to account for the effects of bioturbation in fishpond sediments (cf. Section 2.2). Despite these results, the calibrated value was the one that ensured the best model fit to observed $NH_4^+{}_s$ and NO_{xs} concentrations, since both variables are strongly affected by *DsDO* as a result of coupled nitrification-denitrification process (cf. Chapter 4). Therefore, further research on modelling bioturbation processes is highly desirable to improve model performance.

Inconsistencies between modelled and measured NO_{xs} concentrations after June 2004 (Figure 6.4) were found in the white seabream pond as well as in the pond without fish (cf. Chapter 4). A hypothesized in Chapter 4, the low oxygen predictions (<0.1 mg L^{-1}; Figure 6.9), resulting from macrofauna exclusion from benthic DO dynamics, might have restricted nitrification processes (cf. Chapter 4). Other possibility, proposed by Kittiwanich et al. (2007), is that nitrification could have occurred in anoxic conditions, using oxidizing agents such as manganese and iron oxides by heterotrophic bacteria.

The predicted variation pattern of porewater phosphate in the white seabream pond (Figure 6.5) was also comparable to the pond without fish (cf. Chapter 4), which suggests a common cause in the overestimation of $HPO_4^{2-}{}_s$ concentrations during the first year of the experiment. Similar to what has been reported for the Ria Formosa lagoon (Falcão et al. 2006; Serpa et al. 2007a), oxidized sediment conditions under low fish biomasses (Serpa et al. 2007b) most likely promoted P retention onto iron oxides (van Raaphorst and Kloosterhuis 1994; Slomp et al. 1998; Falcão et al. 2006; Serpa et al. 2007a, b), whereas the model underestimated adsorption rates as a result of low oxygen availability (Figure 6.9).

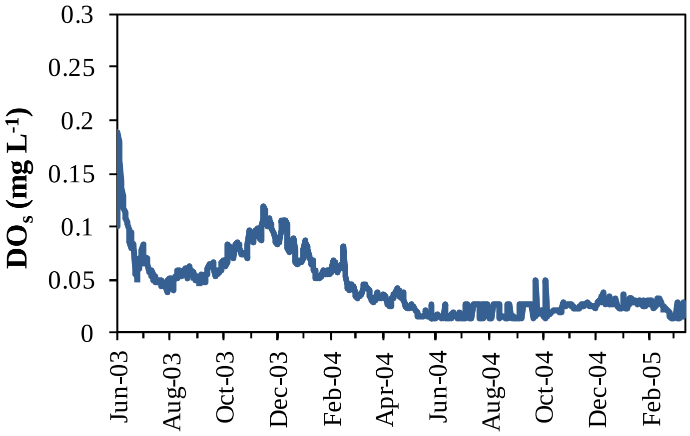


Figure 6.9 – Predicted dissolved oxygen (DO$_s$) porewater concentrations in a white seabream production pond.

Model limitations regarding porewater DON$_s$ and DOP$_s$ concentrations (Figures 6.4 and 6.5), are indicative that further combined experimental and modelling studies are needed to guarantee accurate predictions in fishponds. Mineralization processes, in particular, should be investigated at shorter time scales to clearly identify the effects of abiotic (e.g. temperature and DO availability) and biotic (e.g. benthic activity) factors on DON$_s$ and DOP$_s$ dynamics. Studies focusing on bioturbation processes would also be extremely relevant, since benthic fauna activity has been reported to affect DON$_s$ and DOP$_s$ pools, either by interfering in mineralization and diffusion processes (Burdige and Zheng 1998) or due to animal excretion (Burdige and Zheng 1998).

4.2 Nutrient budgets

Based on model outputs, nutrient budgets (Figures 6.10 and 6.11) were constructed in order to i) evaluate the efficiency of food utilization, ii) define practical ways to improve pond water and sediment quality and, iii) quantify nutrient loss to the environment. This information can enhance the understanding and awareness of fish farmers regarding nutrient management (Papatryphon et al. 2005), and may improve farmers skills to effectively reduce the environmental impacts of farming activity (Briggs and Funge-Smith 1994; Gross et al. 2000).

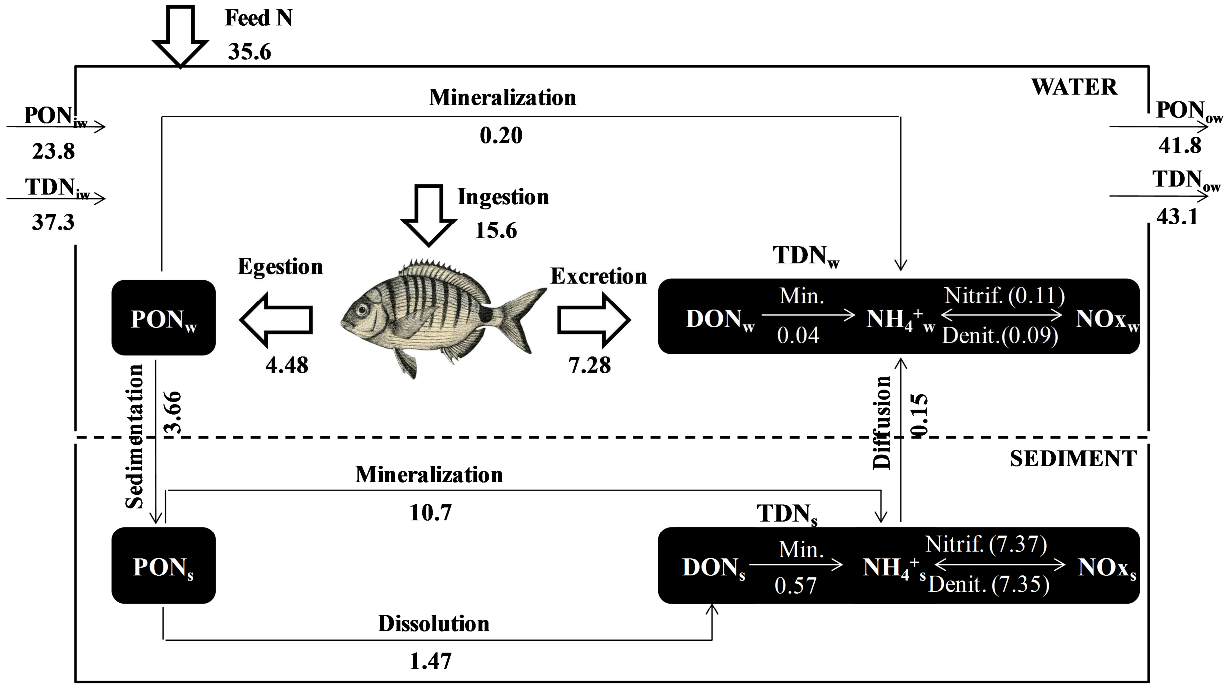


Figure 6.10 – Average nitrogen fluxes (µM d^{-1}) in a semi-intensive white seabream production pond. Abbreviattions: PON$_{iw}$ - particulate organic nitrogen in inflowing water; TDN$_{iw}$ - total dissolved nitrogen in inflowing water; PON$_{ow}$ - particulate organic nitrogen in outflowing water; TDN$_{ow}$ - total dissolved nitrogen in outflowing water; Min. - mineralization; Nitrif. – nitrification and Denit. - denitrification.

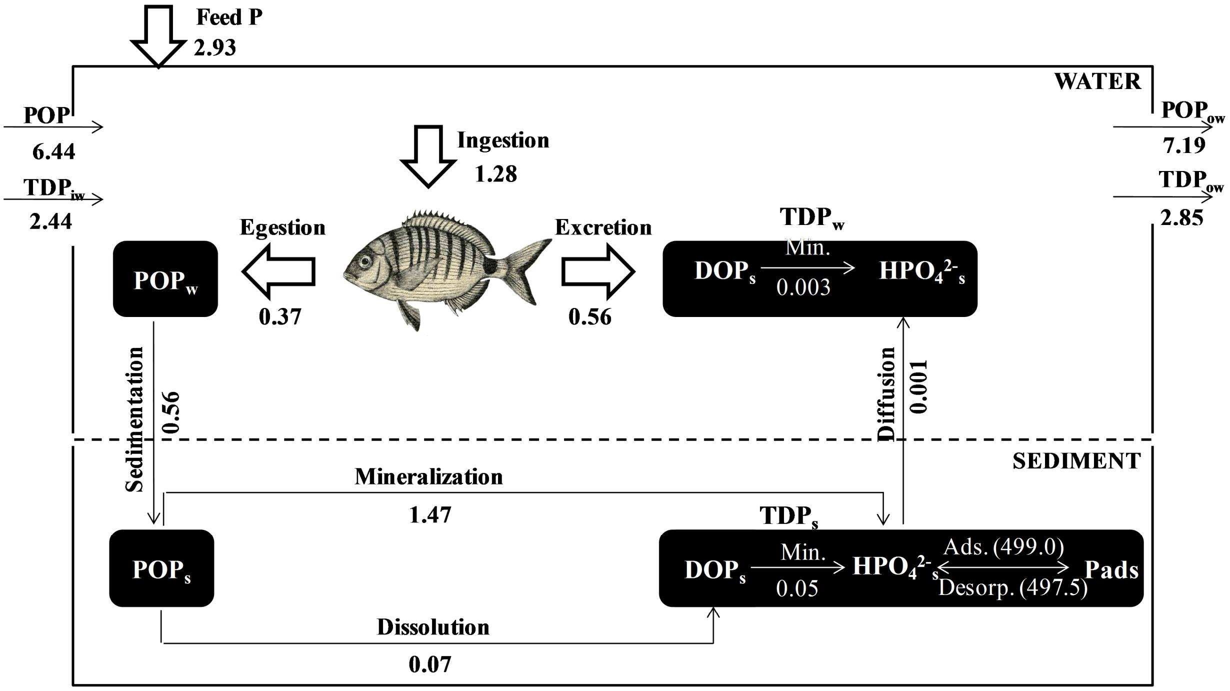


Figure 6.11 – Average phosphorus fluxes (µM d^{-1}) for a semi-intensive white seabream production pond. Abbreviattions: POP$_{iw}$ - particulate organic phosphorus in inflowing water; TDP$_{iw}$ - total dissolved phosphorus in inflowing water; POP$_{ow}$ - particulate organic phosphorus in outflowing water; TDP$_{ow}$ - total dissolved phosphorus in outflowing water; Min. - mineralization; Ads. - adsorption and Desorp. - desorption.

According to the budgets, most of the N (61.1 µM N d^{-1} = 63.2%) and P (8.9 µM P d^{-1} = 75.2%) entering the production pond was supplied by inflowing water, possibly as a result of the high water exchange rates in the system (0.007 to 0.027 m^3 s^{-1}). Even though information on nutrient budgets for semi-intensive ponds is scarce, estimates on the amount of N and P supplied by inlet water (Figures 6.10 and 6.11) were considerably higher than those reported for other semi-intensive fish (Krom et al. 1985b) and shrimp ponds (Casillas-Hernández et al. 2006) with lower (1.5 to 13 times) water exchange rates. According to the aforementioned authors, feeds constituted the major input of N (72 to 92%) and P (53 to 91%) to production ponds, unlike in the present study in which it represented only 36.8% (35.6 µM N d^{-1}) and 24.8% (2.93 µM P d^{-1}) of respectively, N and P external inputs. In fact, feeds contribution to pond nutrient availability seems closely linked to the type of production system since in extensive shrimp ponds (i.e. with no food addition), water exchange was the major pathway for N inputs to the system (Alongi et al. 2000), whereas in intensive systems, food N and P inputs superimpose nutrient inputs via inlet water (Krom and Neori 1989; Briggs and Funge-Smith 1994; Funge-Smith and Briggs 1998; Jackson et al. 2003; Thakur and Lin 2003).

Of the total amount of food supplied, only 43.7% (15.6 µM N d^{-1} and 1.28 µM P d^{-1}) was effectively ingested by white seabream (Figures 6.10 and 6.11), whereas the remaining was integrated in water column PON_w and POP_w pools. From the fraction of food eaten, 28.8% (4.48 µM N d^{-1} and 0.37 µM P d^{-1}) ended up as feces and 45.4% (7.28 µM N d^{-1} and 0.56 µM P d^{-1}) was voided as soluble N and P compounds. As a result of high metabolic wastes, only 24.5% and 27.3% of ingested N and P, respectively, was retained by fish, similarly to what was referred by other authors (Krom et al. 1985b; Krom and Neori 1989; Islam 2005) for aquaculture systems cultivating another Sparidae - the gilthead seabream. Low food assimilation rates together with high food wastes most likely accounted for the extremely high food conversion rate (i.e. the amount in kg, of fish biomass produced per kg of food supplied value) in the production pond (3.73), limiting the productivity of semi-intensive ponds. Enhanced food formulas are therefore highly desirable in order to ensure the economic viability of white seabream production in these systems.

Particulate organic matter resulting from fish activity (i.e. fish feces and uneaten food) or entering the system via inlet water, played a significant role in nutrient budgets (Figures 6.10 and

6.11), has pointed out by several authors (Hargreaves 1998; Holmer et al. 2002; Islam 2005). Particulate organic nitrogen (PON_s) and phosphorus (POP_s) accumulated in pond bottom were recycled (12.2 µM N d^{-1} and 1.54 µM P d^{-1}) as a result of physical, chemical and biological processes occurring in the benthic compartment (Hargreaves 1998; Thakur and Lin 2003). From regenerated N compounds, 0.15 µM N d^{-1} was transported to the water column by diffusion, whereas the remaining was mostly retained as a result of coupled nitrification/denitrification processes (on average, 7.36 µM N d^{-1}). The contribution of benthic fluxes for water column P was extremely low (0.001 µM P d^{-1}), suggesting that most P was retained in sediments for example because of coupled adsorption/desorption processes in organically-richer sediments (Falcão et al. 2006; Serpa et al. 2007a, b). Other studies have also reported sediments as a major sink of nutrients in pond systems (Briggs and Funge-Smith 1994; Funge-Smith and Briggs 1998; Thakur and Lin 2003; Casillas-Hernández et al. 2006), which reinforces the importance of proper sediment treatment between production cycles.

As a result of the low sediment contribution for total dissolved nitrogen (TDN_w) and phosphorus (TDP_w) concentrations most dissolved N (37.3 µM N d^{-1}) and P (2.44 µM P d^{-1}) compounds in fishpond water were supplied by inflowing water (Figure 6.10 and 6.11), in agreement with what was estimated in a previous study (cf. Chapter 3.2). Similarly, the major output of TDN_w and TDP_w (43.1 µM N d^{-1} and 2.85 µM P d^{-1}) occurred via outflowing water (Figures 6.10 and 6.11). If in semi-open systems with high water exchange rates, outlet water is the most important sink for water column nutrients (Krom et al. 1985b; Alongi et al. 2000; Thakur and Lin 2003; d’Orbcastel et al. 2008), in culture systems with low water exchange, losses through the sediment are more important than by pond effluents due to the fast organic matter accumulation in bottom sediments (Briggs and Funge-Smith 1994). Therefore, the optimization of water exchange rates seems crucial for reducing the environmental impacts of this activity while maintaining a good pond environment. Taking into account that fish excretion has an important impact on TDN_w and TDP_w concentrations, since it accounts for 16% (7.28 µM N d^{-1}) and 19% (0.56 µM P d^{-1}) of respectively, TDN_w and TDP_w inputs to the pond (Figures 6.10 and 6.11), water exchange rates should be defined as a function of fish biomass. In fact, the effects of fish excretion on pond water quality seems to superimpose those of of water column processes (e.g. organic matter mineralization) on nutrient dynamics since the contribution of these processes for TDN_w and TDP_w availability in pond water was practically irrelevant (< 1%), whereas in a pond without fish (cf. Chapter 4), organic matter mineralization represented 27% and 6% of respectively, N and P inputs to the system.

4.3 Scenario analysis

Finding management options that maximize fish production without deteriorating the pond environment is a complex task because higher fish biomasses usually aggravate water column and sediment conditions, as a result of the higher amounts of uneaten food and fish wastes in the production ponds (Lin and Yi 2003; Viadero Jr. 2005). Ultimately, this will increase the environmental impacts on the surrounding environment (Bergheim and Brinker 2003), compromising the sustainability of aquaculture operations.

Even though the water quality parameters for the different scenarios (Table 6.8) were within the range of standard values reported for coastal aquaculture systems worldwide (Boyd 2003; Hussenot 2003; Sumagaysay-Chavoso et al. 2004; Viadero Jr. 2005; Ferreira et al. 2010; Lefrançois et al. 2010), global nutrient discharges (Table 6.10) were considerably higher than the values referred for other fish farms, particularly in what concerns TN and TP loadings per tonne of fish produced (Tovar et al. 2000a; Bergheim and Brinker 2003; Boyd et al. 2007; d’Orbcastel et al. 2008). These results are most likely related to the longer duration of the white seabream production cycle (651 days) and the higher water exchange rates in the ponds.

Different priority criteria (Saaty 1980) were defined to identify best and worst scenarios for white seabream production in semi-intensive systems (cf. Section 2.6.1). From the economic point of view, i.e. from the fish farmer’s perspective, the priority goal is usually to maximize fish production (A_2) in order to maximize profits. In this situation, the best option seems to be the increase of standard stocking densities (Table 6.11), because the increase in N and P loadings was small when compared to the increase in fish yields (Table 6.10 and Figure 6.8). Proving the environmental viability of this management option is the fact that when attributing the same “weight” to the three decision criteria in the AHP approach (cf. Section 2.6.1 - A_1), this scenario ranked in second place (Table 6.11). On the other hand, higher stocking densities imply an increase in food inputs, which will lead to a considerable increase in production costs (Rana et al. 2009) because feeds represent 25 to 45% of the overall costs of semi-intensive Mediterranean aquaculture (Stirling Institute of Aquaculture 2004).

Another option for optimizing white seabream production would be the decrease of feeding rates (Table 6.11), since there was an improvement in the ecological status of pond water and sediment (Tables 6.8 and 6.9) and a reduction of total N and P discharges (Table 6.10). Being one of the best scenarios from the ecological viewpoint clearly influenced the score of this scenario in the AHP approach, since it ranked in first and second places in respectively, the A_1 and A_2 criteria (Table 6.11). Nevertheless, decreasing feeding rates presents some drawbacks because total fish biomass decreased 45% compared to the standard simulation (Figure 6.8). On the other hand, other authors have suggested the reduction of feeding rates as a best management practice, due to the faster growth and better food conversion ratios of cultivated species as well as the improvement in pond water quality and less amount of wastes produced (Boyd 2003; Yokoyama et al. 2009).

Among the potential solutions for an effective pond management, one that seems highly viable is the improvement of food absorption efficiency (Scenario7). As a newly cultivated species, the white seabream is usually fed with rations optimised for gilthead seabream (Cejas et al. 2004; Sá et al. 2006), which compromises its growth and subsequently, final fish production. Besides increasing white seabream production by 234 kg relatively to the standard simulation (Figure 6.8), the improvement of food absorption efficiency ensured sound environmental conditions in the fishpond (Tables 6.8 and 6.9), while contributing to a 5% decrease in total N discharges (Table 6.10). According to Gross et al. (2000), improved food digestibility also increases the proportion of nutrient recovered in fish and reduces losses through excretion, fulfilling the objectives of this work. On the other hand, high quality feeds may imply an increase in prices (Sorgal 2011), which would most certainly be compensated by the improvement in the FCRs (Gross et al. 2000; Boyd 2003; Islam 2005).

The other scenarios evaluated in the present work, Scenario3, 5, 6 and 8, all ranked above the standard simulation (Table 6.11), suggesting that these management options might also improve semi-intensive fish farming, whether from an economic, an environmental perspective or both. In the case of an increase in water exchange rates (Scenario3), there was a positive effect on the quality of the pond environment (Tables 6.8 and 6.9) and on effluent discharges (Table 6.10), but a negative impact on fish production (Figure 6.8), which translates into a reduction in profits. Similar results were reported by Avnimelech et al. (1994) in ponds operated with high water exchange rate, because a large fraction of the food and other particulate organic matter is drained out of the system. Other authors have also recommended the increase of water exchange as an effective strategy for reducing the concentrations of potentially toxic compounds like ammonia and nitrites in the production ponds (Burford and Lorenzen 2004) and for maximizing fish production since it will allow higher stocking densities in the production ponds (Burford and Lorenzen 2004). However, the increase in costs related to water pumping, i.e. electricity and maintenance costs, which represent 15 to 20% of overall production costs (SEACASE 2009), might compromise this alternative. The reverse situation occurred when feeding rates were doubled (Scenario5). Despite producing a substantial increase (66%) in final fish biomass (Figure 6.8), this was the worst scenario for pond water and sediment quality (Tables 6.8 and 6.9) and also for effluent quality (Table 6.10), which suggests that this is not be a good option for white seabream production. In fact, in other studies it has been pointed out that feeds should be applied conservatively to avoid overfeeding and to ensure that as much food is consumed as possible (Boyd 2003). The decrease of P content in feeds (Scenario6) did not substantially lowered P discharges into the environment (Table 6.10), as described by other authors (Ferreira et al. 2010) and had no effect on fish biomass (Figure 6.8), which lead us to conclude that this option does not substantially improve production practices. Likewise, the results of increasing food pellets floatability (Scenario8) were very similar to the standard simulation from both the ecological and the economic point of view. Nevertheless, a slight improvement in the pond environmental status was observed (Tables 6.8 and 6.9). From the management options analysed in the present work, the only one that was worse than the standard simulation was the decrease of water exchange rates, probably because the increase in fish yields did not compensated for the impoverishment of pond water and sediment quality and the increase in nutrient discharges. These results contradict the suggestion of other authors that water exchange reduction may be a best management practice because it minimizes effluent discharges (Boyd 2003; Boyd et al. 2007).

As production costs (e.g. food, labour, maintenance costs, etc.) were not included in the AHP, according to the decision criteria used in this study, the best options for optimizing white seabream production in semi-intensive systems are, to duplicate the standard stocking density (from 1.2 kg m^{-3} to 2.5 kg m^{-3}, of final fish biomass) and to develop specific feeds for this species, in order to guarantee a higher absorption efficiency (≈ 80%) than the current rations, which were optimized for gilthead seabream. The information provided herein may provide some guidelines to the aquaculture sector, for developing an Environmentally-Friendly-Allowing-Maximum Production protocol for semi-intensive systems.

5 Conclusions

The ecological model developed in this work reproduced fishpond dynamics reasonably well. Nevertheless, model performance would benefit from improvements on the biogeochemical model, namely As regards the simulation of benthic primary producers and of macrofauna bioturbation effects on nutrient dynamics, as well as from the linkage of the fish model with Fry´s classification of limiting and lethal environmental factors (van der Veer et al. 2009). A model-based nutrient budget revealed that most nutrients available in pond water were supplied by inflowing water whereas major losses occurred via outlet water, which suggests that an efficient pond management relies on optimized water exchange rates. As almost half of the food supplied was not eaten by fish, low food conversion rates were observed for white seabream. The scenarios analysed by the model and classified according to the Analytical Hierarchical Process (a decision support methodology), pointed out that white seabream production may be significantly improved by doubling standard stocking densities and increasing food assimilation efficiency.With slight modifications, the model developed herein can be applied to other semi-intensive monoculture systems as well as to polyculture and Integrated Multi-Trophic Aquaculture (IMTA) systems, constituting a valuable tool for the sustainable management of pond aquaculture.

Acknowledgments

This research was funded by a PhD fellowship from Fundação para a Ciência e Tecnologia (SFRH/BD/27643/2006). Special thanks to Doutora Manuela Falcão for her valuable comments on the manuscript.

Chapter 7

General Conclusions and Future Perspectives

1 General Conclusions

The future of aquaculture seems to rely on balancing its social and economic benefits with its environmental impacts (Grigorakis and Rigos 2011). From the review work presented in Chapter 2, it became evident that an ecosystemic approach should be developed towards sustainable aquaculture. This requires that the relevant stakeholders (e.g. decision makers, ecosystem managers and farmers) have sufficient quantitative information to take informed decisions (Nobre et al. 2010). Likewise, at the farm level, management requires a wide range of skills and knowledge, particularly in what concerns the culture environment itself (Culberson and Piedrahita 1996). This knowledge may be obtained by an adaptative as well as by a modelling approach (Crawford 2003). In the present work, the combination of both approaches provided crucial information for managing the production of white seabream (*Diplodus sargus*) – a new species in Mediterranean aquaculture – in semi-intensive earth ponds.

The study of the physical, chemical and biological processes in white seabream ponds over a production cycle (cf. Chapter 3), revealed that despite the strong correlation between fish activity and bottom sediment enrichment, impacts on the benthic environment (e.g. higher nutrient availability in porewater and intense microphytobenthos production) were only noticeable when fish biomass was above 0.5 kg m^{-3} and the feeding rate was higher than 5 kg d^{-1} (cf. Chapter 3.1). These results indicate that environmental parameters should be carefully monitored from this point on to avoid deterioration of the pond quality status. This experimental work also provided some guidelines for the optimization of white seabream production in semi-intensive systems. Given that the quality of fishpond sediments was comparable to that of the adjacent coastal lagoon and fish survival rate was high (94%), one may conclude that the assayed farming conditions caused no environmental constraints within production ponds (cf. Chapter 3.1). Another proof of the sustainability of assayed farming conditions is the composition profile of pond water, which was similar to inflowing water and to the water from a pond without fish (cf. Chapter 3.2). According to the results of this study, the optimization of water exchange rates is crucial to ensure good water quality within the production ponds, since inflowing and effluent waters were respectively, the main source and sink of dissolved nitrogen and phosphorus in the water column (cf. Chapter 3.2). Sediments also played an important role on pond water quality. As organic matter accumulated in bottom sediments, the benthic fluxes, particularly biologically-mediated fluxes, became a major source of nutrients to the water (Chapter 3.2), which emphasizes the importance of sediment treatment between production cycles, to avoid the deterioration of fishpond environment (Hargreaves 1998).

Further insight into fishpond dynamics was gained by using an ecological model that integrates the physical, chemical and biological processes in these systems. The modelling strategy followed in this work, which consisted in separately developing and calibrating a biogeochemical and a biological model before its coupling, was particularly helpful to investigate the dynamics of pond variables and processes as well as their interactions (cf. Chapter 4). For example, in the sensitivity analysis carried out to the biogeochemical model, it was possible to identify the effects of oxygen-dependent processes on porewater inorganic nutrient variables, and clearly understand how oxygen availability determines inorganic nutrient concentrations in pond sediments (cf. Chapter 4). This analysis also revealed how pond structural features (e.g. pond depth and volume) and operational parameters (e.g. water exchange rates) affect the water and sediment quality of semi-intensive production systems, which is key information for dimensioning new aquaculture units and managing the existing ones. The biogeochemical model developed herein was calibrated for an earth pond without fish, and can be applied to other aquatic systems, such as coastal lagoons and wastewater treatment ponds (cf. Chapter 4).

The validation of the fish Dynamic Energy Budget (DEB) model for white seabream and gilthead seabream (*Sparus aurata*) – a traditionally cultivated species in Mediterranean aquaculture – demonstrates its suitability for simulating finfish growth in a dynamic environment (cf. Chapter 5). One of the advantages of the biological model developed in this study is that it can be used even when there is lack of information for estimating all model parameters. Running the model on its "Individual Based Model" (IBM) mode, in which a population of *n* fishes is simulated each with its specific parameter set assigned randomly, allows a quick model calibration and the selection of the best parameters set describing fish growth. On the other hand, the "state variable" mode is more suitable in a complex ecosystem model, since an average fish is simulated with a particular parameter set, reducing the computational overhead. As the DEB theory is based on physiological principles (Kooijman 2000), this modelling tool can be used to investigate how biological processes (e.g. food ingestion and absorption) affect fish performance and to explain growth differences between species, which is valuable information for maximizing the growth of cultivated fish and thereby the performance of production systems. In this study, a comparison of species-specific model parameters for the two Sparidae species, revealed that the white seabream lower growth rates are presumably linked to a higher energy demand for body maintenance. The lower food absorption efficiency might also explain part of the interspecies growth variability. Taking into account that these seabreams, with different feeding strategies, are currently being fed with the same diet, these findings point out that an improvement in white seabream food formulations is required to increase its production.

The coupled model developed in this study was able to successfully simulate the majority of water column and sediment variables in the fish pond as well as white seabream growth over a production cycle, which makes it a suitable tool for pond management (cf. Chapter 6). The model-based nutrient budget constructed in this study was particularly useful for improving the knowledge on the culture environment, since it provided valuable information on nutrient dynamics (Papatryphon et al. 2005). Noteworthy, is the fact that almost half of the food supplied is not eaten by fish, which justifies the low food conversion rates (3.7) for this species. Possible management options to increase the efficiency of food utilization include the reduction of water exchange rates or the improvement of pellets stability since food is mainly loss through outflowing water or as a result of pellets sedimentation. By providing information on the dynamics of particulate and dissolved nutrient forms in fishponds, this type of budgets may be also used to define practical ways to improve pond water and sediment quality. For instance, the quantification of nutrient fluxes through inflowing and effluent waters may be used for optimizing water exchange rates. As the biogeochemistry of earth ponds is substantially influenced by fish activity when compared to a pond without fish (cf. Chapter 4), estimates on fish contribution to dissolved (through excretion) and particulate (through egestion and uneaten food) nutrient availability, may help defining optimum stocking densities for semi-intensive production systems.

Aside from insights on pond nutrient dynamics, the coupled biogeochemical-biological model presented herein is a valuable tool for optimizing fish production, since it can be used to test the effects of different management scenarios on the quality of the pond environment as well as on fish yields and effluent discharges (cf. Chapter 6). The combination of a modelling approach with a decision support system (Analytical Hierarchical Process) is an efficient methodology for identifying the best management options for semi-intensive fish farming. Based on the indicators used in this study, the best solutions for maximizing white seabream production while maintaining a good pond environment and minimizing the impacts on the adjacent coastal system are to double standard stocking densities and improve food absorption efficiency. Despite its longer production cycle, if there is an investment in the development of white seabream feeds, the cultivation of this species in semi-intensive systems seems not only viable but also promising due to its high market value and highly appreciated flesh (Cejas et al. 2004; Sá et al. 2006, 2007).

2 Future Perspectives

In spite of the general good model fit to observed data, further experimental and modelling work is needed to improve model performance. Future work, include model coupling to a hydrodynamic model, to simulate the effects of sediment resuspension (Peterson et al. 2000), caused by bottom currents, aerators or bioturbation, on pond biogeochemistry. Further studies on the interactions between the biota, namely benthic primary producers and macrofauna, and organic matter and nutrient cycles, could also improve model performance. Likewise, the linkage of the fish model with Fry´s classification of limiting and lethal environmental factors (van der Veer et al. 2009), would help predict critical situations for fish survival, turning the model into a more powerful tool for pond management. It would also be interesting to apply the coupled model to production systems with different functioning, such as polyculture systems or other monoculture systems with different farming conditions and fish species, to test for its applicability.

By being able to estimate the composition and quantity of fish farm wastes, the model developed in this work may be used for defining waste reduction measures or treatment methods for pond aquaculture (Bergheim and Brinker 2003; d'Orbcastel et al. 2008). As an example, the model may be used for dimensioning Integrated Multi-Trophic Aquaculture (IMTA) systems, a biological method for recycling farm wastes within the system itself that has been widely referred as a sustainable option for the development of pond aquaculture, due to its environmental and socio-economic benefits (Neori et al. 2004; Buschmann et al., 2009; Troell et al., 2009). Future possible model applications may include its combination with a Decision Support System (DSS) that includes economic descriptors related to production costs, e.g. food, labour and maintenance costs, or costs related to waste discharges and water use, to effectively assess the costs and benefits of different management scenarios on aquaculture production (Ernst et al. 2000; Pereira et al. 2006).

Aside from the model potential, this work also provides valuable information for defining guidelines on environmental standards (e.g. Average Recommended Values) for coastal and brackish waters used for fish farming, similarly to what is already defined in the Portuguese legislation for shellfish waters and for freshwaters used for Salmonidae culture (Decreto-Lei nº 236/98). As the quality of fishpond water determines the quality of farm effluents, this may be an indirect way of regulating this activity within the scope of the Water Framework Directive (Bergheim and Brinker 2003; d'Orbcastel et al. 2008; Moran and Dann 2008), since regulations on effluent standards are unlikely to be applied to semi-intensive fish farm units as is done for intensive farms in many European countries (Bergheim and Brinker 2003; d'Orbcastel et al. 2008).

Chapter 8

References

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

Agnetis A., Basosi R., Caballero K., Casini M., Chesi G., Ciaschetti G., Detti P., Federici M., Focardi S., Franchi E., Garulli A., Mocenni C. (2006). Development of a Decision Support System for the management of Southern European lagoons WP8 Final Report (DITTY Project) EVK3-CT-2002-00084. Available at, http://www.dittyproject.org/(accessed on November 2008).

Aksnes A., Izquierdo M. S., Robaina L., Vergara J. M., Montero D. (1997). Influence of fish meal quality and feed pellet on growth, feed efficiency and muscle composition in gilthead seabream (*Sparus aurata).* Aquaculture 153, 251–261.

Aller R. C., Aller J. Y. (1992). Meiofauna and solute transport in marine muds. Limnology and Oceanography 37 (5), 1018–1033.

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

Alongi D. M., Johnston D. J., Xuan T. T. (2000). Carbon and nitrogen budgets in shrimp ponds of extensive mixed shrimp-mangrove forestry farms in the Mekong delta, Vietnam. Aquaculture Research 31, 387–399.

Alongi D. M., Tirendi F., Trott L. A. (1999). Rates and pathways of benthic mineralization in extensive shrimp ponds of the Mekong delta, Vietnam. Aquaculture 175, 269–292.

Alunno-Bruscia M., van der Veer H., Kooijman S. A. L. M. (2009). The AquaDEB project (phase I): Analysing the physiological flexibility of aquatic species and connecting physiological diversity to ecological and evolutionary processes by using Dynamic Energy Budgets. Journal of Sea Research 62 (2-3), 43–48.

Andersen J. M. (1976). An ignition method for determination of total phosphorus in lake sediments. Water Resources 10, 329–331.

Anderson M. R., Cranford P., McKindsey C. W., Strain P., Hargrave B. T., Li W. K. W., Harrison W. G. (2006). Cumulative and far-field fish habitat effects. Canadian Science Advisory Secretariat, Research Document 2006/037, 30 pp.

Aníbal J. (1998). Impacte da macroepifauna sobre as macroalgas Ulvales (Chlorophyta) na Ria Formosa. Tese de Mestrado, Universidade de Coimbra, 73 pp.

Anschutz P., Sundby B., Lefrançois L., Luther III G. W., Mucci A. (2000). Interactions between metal oxides and species of nitrogen and iodine in bioturbated marine sediments. Geochimica et Cosmochimica Acta 64, 2751–2763.

Argue B. J., Arce S. M., Lotz J. M., Moss S. M. (2002). Selective breeding of Pacific White Shrimp (*Litopenaeus vannamei*) for growth and resistance to Taura Syndrome Virus. Aquaculture 204 (3), 447–60.

Ashley P. J. (2007). Fish welfare: Current issues in aquaculture. Applied Animal Behaviour Science 104, 199–235.

Asmus R. M., Sprung M., Asmus H. (2000). Nutrient fluxes in intertidal communities of a south European lagoon (Ria Formosa) – similarities and differences with a northern Wadden Sea bay (Syl-R∅m∅ bay). Hydrobiologia 436, 217–235.

Augustine S. L., Gagnaire B., Adam-Guillermin C., Kooijman S.A.L.M. (2011). Consequences of exposure to depleted uranium on developmental energetic of zebrafish, *Danio rerio*. In: Abstracts Book of the 2nd International Symposium on Dynamic Energy Budget Theory, Portugal, 13-15 April 2011, 32 pp.

Austoni M., Viaroli P., Giordani G., Zaldívar J. M. (2004). Intercomparison among the test sites of the DITTY Project using the IFREMER classification scheme for coastal lagoons. Institute for Environment and Sustainability, Island and Waters Unit. Institute for Environment and Sustainability, Inland and Marine Waters Unit, EUR 21286 EN, Ispra, Italy, 52 pp.

Avnimelech Y. (1999). Carbon/nitrogen ratio as a control element in aquaculture systems. Aquaculture 176, 227–235.

Avnimelech Y., Kochba M. (1999). Sedimentation and Resuspension in Earthen Fish Ponds. Journal of the World Aquaculture Society 30 (4), 401–409.

Avnimelech Y., Kochba M., Diab S. (1994). Development of controlled intensive aquaculture systems with a limited water exchange and adjusted carbon to nitrogen ratio. Israeli Journal of Aquaculture-Bamidgeh 46 (3), 119–131.

Avnimelech Y., Mazes N., Weber B. (1992). Effects of aeration and mixing on nitrogen and organic matter transformations in simulated fish ponds. Aquacultural Engineering 11, 157–169.

Bacher C., Duarte P., Ferreira J. G., Héral M., Raillard O. (1998). Assessment and comparison of the Marennes-Oléron Bay (France) and Carlingford Lough (Ireland) Carrying Capacity with ecosystem models. Aquatic Ecology 31, 379–394.

Banas D., Masson G., Leglize L., Usseglio-Polatera P., Boyd C. E. (2008). Assessment of sediment concentration and nutrient loads in effluents drained from extensively managed fishponds in France. Environmental Pollution 152, 679–685.

Barazi-Yeroulanos L. (2010). Regional synthesis of the Mediterranean marine finfish aquaculture sector and development of a strategy for marketing and promotion of Mediterranean aquaculture. GFCM Studies and Reviews No. 88, FAO, Rome, Italy, 218 pp.

Barbato F., Izzo G., Meloni F., Savarino R. (1996). Preliminary results on the estimation of effluents from seabream (S*parus aurata*) rearing in floating cages in a Sardinian lagoon. Cogresso della Societa Italiana di Biologia Marina, Sciacca (Italy), 22-27 May 1995.

Bargelloni L., Alarcon J.A., Alvarez M.C., Penzo E., Magoulas A., Palma J., Patarnello T. (2005). The Atlantic–Mediterranean transition: Discordant genetic patterns in two seabream species, *Diplodus puntazzo* (Cetti) and *Diplodus sargus* (L.). Molecular Phylogenetics and Evolution 36, 523–535.

Başçınar N., Çakmak E., Çavdar Y., Aksungur N. (2007). The Effect of Feeding Frequency on Growth Performance and Feed Conversion Rate of Black Sea Trout (*Salmo trutta labrax* Pallas, 1811). Turkish Journal of Fisheries and Aquatic Sciences 7, 13–17.

Bayne B. L. (1993). Feeding physiology of bivalves: time-dependence and compensation for changes in food availability. In: Dame R. F. (ed.) Bivalve filter-feeders in estuarine and coastal ecosystems. NATO ASI Series vol. G33, Springer-Verlag, Berlin, Germany, pp. 1–24.

Belias C., Dessanakis M., Scoullos M. (2007). Study of N, P and Si fluxes between fish farm sediment and seawater. Results of simulation experiments employing a benthic chamber under various redox conditions. Marine Chemistry 103, 266–275.

Bergheim A., Brinker A. (2003). Effluent treatment for flow through systems and European Environmental Regulations. Aquacultural Engineering 27, 61–77.

Berman T., Bronk D. A. (2003). Dissolved organic nitrogen: a dynamic participant in aquatic ecosystems. Aquatic Microbial Ecology 31, 279–305.

Berner R. A. (1980). Early diagenesis - A theorical approach. Series in Geochemistry. Princeton University Press, New Jersey, United States of America, 256 pp.

Beveridge M. C. M., Phillips M. J., Clarke R. (1991). A quantitative and qualitative assessment of wastes from aquatic animal production. In: Brune D. E., Tomasso J. R. (eds.) Advances in World Aquaculture (Vol III), World Aquaculture Society, Baton Rouge, Los Angeles, United States of America, pp. 506–533.

Biao X., Kaijin Y. (2007). Shrimp farming in China: Operating characteristics, environmental impact and perspectives. Ocean & Coastal Management 50, 538–550.

Biao X., Zhuhong D., Xiaorong W. (2004). Impact of the intensive shrimp farming on the water quality of the adjacent coastal creeks from Eastern China. Marine Pollution Bulletin 48, 543–553.

Billen G., Lancelot C. (1988). Modelling benthic nitrogen cycling in temperate coastal ecosystems. In: Blackburn T. H., Sorensen J. (Eds.) Nitrogen cycling in coastal marine environments. John Wiley & Sons, New York, Unites States of America, pp. 341–378.

Black K. D. (2001). Sustainability of aquaculture. In: Black K. D. (ed.) Environmental Impacts of Aquaculture, Sheffield Academic Press, Sheffield, United Kingdom, pp. 199–212.

Blackburn T. H., Lund B. A., Krom M. D. (1988). C and N mineralization in the sediments of earthen marine fish pond. Marine Ecology Progress Series 44, 221–227.

Boaventura R., Pedro A. M., Coimbra S., Lencastre E. (1997). Trout farm effluents: characterization and impact on the receiving streams. Environmental Pollution 95 (3), 379–387.

Bodiguel X., Maury O., Mellon-Duval C., Roupsard F., Le Guellec A. M., LoizeauV. (2009). A dynamic and mechanistic model of PCB bioaccumulation in the European hake (*Merluccius merluccius*). Journal of Sea Research 62, 124–134.

Booth M. A., Tucker B. J., Allan G. L., Fielder D. S. (2008). Effect of feeding regime and fish size on weight gain, feed intake and gastric evacuation in juvenile Australian snapper *Pagrus auratus*. Aquaculture 282, 104–110.

Bosma R. H., Verdegem M. C. J. (2011). Sustainable aquaculture in ponds: Principles, practices and limits. Livestock Science 139, 58–68.

Boudreau B. P. (1997). Diagenetic models and their implementation: modelling transport and reactions in aquatic sediments. Springer-Verlag, New York, United states of America, 414 pp.

Boyd C. E. (1990). Water Quality in Ponds for Aquaculture. Alabama Agricultural Experiment Station, Auburn University, AL, United States of America.

Boyd C. E. (1995a). Chemistry and efficacy of amendments used to treat water and soil quality imbalance in shrimp ponds. In: Browdy C. L., Hopkins J. S. (ed.) Swimming through troubled water, Proceedings of the special session on shrimp farming, Aquaculture’95, World Aquaculture Society, Baton Rouge, Louisiana, United States of America, pp. 183–199.

Boyd C. E. (1995b). Bottom soils, Sediment and Pond Aquaculture. Chapman and Hall, New York, United States of America, 348 pp.

Boyd C. E. (2003). Guidelines for aquaculture effluent management at the farm-level. Aquaculture 226, 101–112.

Boyd C. E., Clay W. (1998). Shrimp Aquaculture and the Environment. Science America 58, 59–65.

Boyd C., Corpron K., Bernard E., Pengsang P. (2006). Estimates of Bottom Soil and Effluent Load of Phosphorus at a Semi-intensive Marine Shrimp Farm. Journal of the World Aquaculture Society 37 (1), 41–47.

Boyd C. E., Massaut L. (1999). Risks associated with the use of chemicals in pond aquaculture. Aquacultural Engineering 20, 113-132.

Boyd C. E., Tucker C. S. (1998). Pond aquaculture water quality management. Kluwer Academic Publishers, Dordrecht, The Netherlands, 720 pp.

Boyd C. E., Tucker C., Mcnevin A., Bostick K., Clay J. (2007). Indicators of Resource Use Efficiency and Environmental Performance in Fish and Crustacean Aquaculture. Reviews in Fisheries Science 15, 27–360.

Boyd C. E., Woods C. W., Thunjai T. (2002). Aquaculture Pond Bottom Soil Quality Management. Pond Dynamics/Aquaculture Collaborative Research Support Program. Oregon State University, Corvallis, United States of America, 41 pp.

Brafield A. E., Llewellyn M. J. (1982). Animal energetics. Blackie, Glasgow, United Kingdom, 168 pp.

Brambilla F., Lalumera G., Terova G., Crosa G., Saroglia M. (2007). Inflow and outflow water quality control in aquaculture systems: a case study. Aquaculture Research 38, 1654–1663.

Briggs M. R. P., Funge-Smith S. J. (1994). A nutrient budget of some intensive marine shrimp ponds in Thailand. Aquaculture and Fisheries Management 25, 789–811.

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

Brummet R. E. (1999). Integrated aquaculture in sub-saharan Africa. Environment, Development and Sustainability 1, 315–321.

Bunting S. W., Shpigel M. (2009). Evaluating the economic potential of horizontally integrated land-based marine aquaculture. Aquaculture 294 (1-2), 43–53.

Burdige D. J. (2006). Geochemistry of Marine Sediments. Princeton University Press, Princeton, United States of America, 609 pp.

Burdige D. J., Alperin M. J., Homstead J, Martens C. S. (1992). The role of benthic fluxes of dissolved organic carbon in oceanic and sedimentary carbon cycling. Geophysical Research Letters 19, 1851–1854.

Burdige D. J., Zheng S. (1998). The biogeochemical cycling of dissolved organic nitrogen in estuarine sediments. Limnology and Oceanography 43 (8), 1796–1813.

Burford M. A., Costanzo S. D., Dennison W. C., Jackson C. J., Jones A. B., McKinnon A. D., Preston N. P., Trott L. A. (2003). A synthesis of dominant ecological processes in intensive shrimp ponds and adjacent coastal environments in NE Australia. Marine Pollution Bulletin 46, 1456–1469.

Burford M. A., Lorenzen K. (2004). Modeling nitrogen dynamics in intensive shrimp ponds: the role of sediment remineralization. Aquaculture 229, 129–145.

Burns L. A. (2000). Exposure analysis modelling system (EXAMS): User manual and system documentation. U.S. Environmental Protection Agency, United States of America, 197 pp.

Buschmann A. H., Cabello F., Young K., Carvajal J., Varela D. A., 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.

Buschmann A. H., López D. A., Medina A. (1996). A Review of the Environmental Effects and Alternative Production Strategies of Marine Aquaculture in Chile. Aquacultural Engineering 15 (6), 397–421.

Byers S. C., Mills E. L., Stewart L. (1978). A comparison of methods for determining organic carbon in marine sediments, with suggestions for a standard method. Hydrobiologia 58, 43–47.

Cancemi G., De Falco G., Pergent G. (2003). Effects of organic matter input from a fish farming facility on a *Posidonia oceanica* meadow. Estuarine, Coastal and Shelf Science 56, 961–968.

Cardoso J., Witte J., van der Veer H. W. (2006). Intra- and interspecies comparison of energy flow in bivalve species in Dutch coastal waters by means of the Dynamic Energy Budget (DEB) theory. Journal of Sea Research 56, 182–197.

Carvalho S., Barata M., Pereira F., Gaspar M. B., Cancela da Fonseca L., Pousão-Ferreira P. (2007). Distribution patterns of macrobenthic species in relation to organic enrichment within aquaculture earthen ponds. Marine Pollution Bulletin 52, 1573–1584.

Carvalho S., Falcão M., Cúrdia J., Moura A., Serpa D., Gaspar M., Dinis M. T., Pousão-Ferreira P., Cancela da Fonseca L. (2009). Benthic dynamics within a land-based semi-intensive aquaculture fish farm: the importance of settlement ponds. Aquaculture International 17, 571–587.

Carver C. E. A., Mallet A. L. (1990). Estimating carrying capacity of a coastal inlet for mussel culture. Aquaculture 88, 39–53.

Casillas-Hernández R., Magallón-Barajas F., Portillo-Clarck G., Páez-Osuna F. (2006). Nutrient mass balances in semi-intensive shrimp ponds from Sonora, Mexico using two feeding strategies: trays and mechanical dispersal. Aquaculture 258, 289–298.

Casillas-Hernández R., Nolasco-Soria H., García-Galano T., Carrillo-Farnes O., Paéz-Osuna F. (2007). Water quality, chemical fluxes and production in semi-intensive Pacific white shrimp

(*Litopenaeus vannamei*) culture ponds utilizing two different feeding strategies. Aquaculture Engineering 36, 105–114.

Cejas J. R., Almansa E., Jéreza S., Bolaños A., Samper M., Lorenzo A. (2004). Lipid and fatty acid composition of muscle and liver from wild and captive mature female broodstocks of white seabream, *Diplodus sargus*. Comparative Biochemistry and Physiology Part B 138, 91–102.

Chakrabarty D., Das S. K. (2007). Bioturbation-induced phosphorous release from an insoluble phosphate source. BioSystems 90, 309–313.

Chang S. C., Jackson M. L. (1957). Fractionation of soil phosphorus. Soil Science 84, 133–144.

Chapelle A. (1995). A preliminary model of nutrient cycling in sediments of a Mediterranean lagoon. Ecological Modelling 80, 131–147.

Chapelle A., Ménesguen A., Deslous-Paoli J., Souchu P., Mazouni N., Vaquer A., Millet B. (2000). Modelling nitrogen, primary production and oxygen in a Mediterranean lagoon. Impact of oysters farming and inputs from the watershed. Ecological Modelling 127, 161–181.

Chávez-Crooker P., Obreque-Contreras J. (2010). Bioremediation of aquaculture wastes. Current opinion in Biotechnology 21 (3), 313–317.

Chen S., Coffin D. E., Malone R. F. (1997). Sludge production and management for recirculating aquacultural systems. Journal of the World Aquaculture Society 28, 303–315.

Choo P.-S. (2001). Environmental effects of warm water culture in ponds/lagoons. In: Black K. D. (ed.) Environmental Impacts of Aquaculture, Sheffield Academic Press, Sheffield, United Kingdom, pp. 76–98.

Costello M. J., Grant A., Davies I. M., Cecchini S., Papoutsoglou S., Quigley D., Saroglia M. (2001). The control of chemicals used in aquaculture in Europe. Journal of Applied Ichthyology 17 (4), 173–180.

Crawford C. (2003). Environmental management of marine aquaculture in Tasmania, Australia. Aquaculture 226, 129–138.

Crawford C. M., Macleod C. K. A., Mitchell I. M. (2003). Effects of shellfish farming on the benthic environment. Aquaculture 224, 117–140.

Cromey C. J., Nickell T. D., Black K. D. (2002). DEPOMOD - modelling the deposition and biological effects of waste solids from marine cage farms. Aquaculture 214, 211–239.

Cuenco M., Stickney R., Grant W. (1985a). Fish bioenergetics and growth in aquaculture ponds: I- Individual fish model development. Ecological Modelling 27 (3-4), 169–190.

Cuenco M., Stickney R., Grant W. (1985b). Fish bioenergetics and growth in aquaculture ponds: II- Effects of interactions among size, temperature, dissolved oxygen, unionized ammonia and food on growth of individual fish. Ecological Modelling 27 (3-4), 191–206.

Cuenco M., Stickney R., Grant W. (1985c). Fish bioenergetics and growth in aquaculture ponds: III- Effects of intraspecific competition, stocking rate, stocking size and feeding rate on fish productivity. Ecological Modelling 28 (1-2), 73–95.

Culberson S., Piedrahita R. (1996). Aquaculture pond ecosystem model: temperature and dissolved oxygen prediction – mechanism and application. Ecological Modelling 89, 231–258.

Dale A. W., Prego R. (2002). Physico-biogeochemical controls on benthic-pelagic coupling of nutrient fluxes and recycling in a coastal upwelling system. Marine Ecology Progress Series 235, 8–15.

Dalsgaard T., Krause-Jensen D. (2006). Monitoring nutrient release from fish farms with macroalgal and phytoplankton bioassays. Aquaculture 256, 302–310.

Dalvi R. S., Pal A. K., Tiwari L. R., Das T., Baruah K. (2009). Thermal tolerance and oxygen consumption rates of the catfish *Horabagrus brachysoma* (Günther) acclimated to different temperatures. Aquaculture 295 (1-2), 116–119.

Dame R. F., Prins T. C. (1998). Bivalve carrying capacity in coastal ecosystems. Aquatic Ecology 31, 409–421.

Das T., Pal A. K., Chakraborty S. K., Manush S.M., Sahua N. P., Mukherjee S. C. (2005). Thermal tolerance, growth and oxygen consumption of *Labeo rohita* fry (Hamilton, 1822) acclimated to four temperatures. Journal of Thermal Biology 30, 378–383.

Deb A. (1998). Fake blue revolution: environmental and socio-economic impacts of shrimp culture in the coastal areas of Bangladesh. Ocean & Coastal Management 41, 63–88.

Decreto-Lei n.º 236/98. Diário da República, I Série-A Nº 176, de 1 de Agosto de 1998.

Del Toro-Silva F. M., Miller J. M., Taylor J. C., Ellis T. A. (2008). Influence of oxygen and temperature on growth and metabolic performance of *Paralichthys lethostigma* (Pleuronectiformes: Paralichthyidae). Journal of Experimental Marine Biology and Ecology 358, 113–123.

Dias J., Conceição L. E. C., Ribeiro A. R., Borges P., Valente L. M. P., Dinis M. T. (2009). Practical diet with low fish-derived protein is able to sustain growth performance in gilthead seabream (*Sparus aurata*) during the grow-out phase. Aquaculture 293, 255–262.

d'Orbcastel E. R., Blancheton J.-P., Boujard T., Aubin J., Moutounet Y., Przybyla C., Belaud A. (2008). Comparison of two methods for evaluating waste of a flow through trout farm Aquaculture 274, 72–79.

Drakeford B., Pascoe S. (2008). Substitutability of fishmeal and fish oil in diets for salmon and trout: a meta-analysis. Aquaculture Economics & Management 12, 155–175.

Duarte P. (2003). A review of current methods in the estimation of environmental carrying capacity for bivalve culture in Europe. In: Yu H., Bermas N. (ed.) Determining environmental carrying capacity of coastal and marine areas: progress, constraints and future options, PEMSEA Workshop Proceedings No. 11, pp. 37–51.

Duarte P., Azevedo B., Ribeiro C., Pereira A., Falcão M., Serpa D., Bandeira R., Reia J. (2007). Management oriented mathematical modelling of Ria Formosa (South Portugal). Transitional Waters Monographs 1, 13–51.

Duarte P., Fernández-Reiriz M. J., Filgueira R., Labarta U. (2010). Modelling mussel growth in ecosystems with low suspended matter loads. Journal of Sea Research 64 (3), 273–286.

Duarte P., Hawkins A. J. S., Pereira A. (2005). How does estimation of environmental carrying capacity for bivalve culture depend upon spatial and temporal scales? In: Dame R., Olenin S. (ed.) The comparative role of suspension feeders in aquatic systems, NATO ARW in Nida, Lithuania, 3-9 October 2003, Kluwer Scientific Publishers, Dordrecht, The Netherlands, pp. 121–135.

Duarte P., Meneses R., Hawkins A. J. S., Zhu M., Fang J., Grant J. (2003). Mathematical modelling to assess the carrying capacity for multi-species culture within coastal waters. Ecological Modelling 168, 109–143.

Dunham R. A., Majumdar K., Hallerman E., Bartley D., Mair G., Hulata G., Liu Z., Pongthana N., Bakos J., Penman D., Gupta M., Rothlisberg P., Hoerstgen-Schwark G. (2001). Review of the Status of Aquaculture Genetics. In: Subasinghe R., Bueno P., Phillips M. J., Hough C., McGladdery S. E., Arthur J. R. (eds.) Aquaculture in the Third Millennium, Bangkok: NACA; and Rome: FAO.

Edgar G. J., Macleod C. K., Mawbey R. B., Shields D. (2005). Broad-scale effects of marine salmonid aquaculture on macrobenthos and the sediment environment in southeastern Tasmania. Journal of Experimental Marine Biology and Ecology 327, 70–90.

Eichinger M., Loizeau V., Roupsard F., Le Guellec A. M., Bacher C. (2010). Modelling growth and bioaccumulation of Polychlorinated biphenyls in common sole (*Solea solea*). Journal of Sea Research 64 (3), 373–385.

Enell M., Ackerfors H. (1991). Nutrient discharges from aquaculture operations in Nordic countries into adjacent sea areas. ICES report CM. 1991/F: 56, 17 pp.

Ernst D. H., Bolte J. P., Nath S. S. (2000). AquaFarm: simulation and decision support for aquaculture facility design and management planning. Aquacultural Engineering 23, 121–179.

Falcão M. (1997). Dinâmica de Nutrientes na Ria Formosa: efeitos da interacção da laguna com as suas interfaces na reciclagem do azoto, fósforo e sílica. Tese de Doutoramento, Universidade do Algarve, Portugal, 223 pp.

Falcão M., Caetano M., Serpa D., Gaspar M., Vale C. (2006). Effects of infauna harvesting on tidal flats of a coastal lagoon (Ria Formosa, Portugal): Implications on phosphorus dynamics. Marine Environmental Research 61, 136–148.

Falcão M., Vale C. (1998). Sediment-water exchanges of ammonium and phosphate in intertidal and subtidal areas of a mesotidal coastal lagoon (Ria Formosa). Hydrobiologia 373/374, 193–201.

Falcão M., Vale C. (2003). Nutrient dynamics in a coastal lagoon (Ria Formosa, Portugal): The importance of lagoon-sea water exchanges on biological productivity. Ciencias Marinas 29 (4), 425–433.

FAO, Food and Agriculture Organization of the United Nations (2007). The State of World Fisheries and Aquaculture 2006. FAO, Rome, Italy, 162 pp.

FAO, Food and Agriculture Organization of the United Nations (2008). Available at, http://www.fao.org/fishery/statistics/global-production/ (accessed on November 2008).

FAO, Food and Agricultural Organization of the United Nations (2010). The state of fisheries and aquaculture. Fisheries and Aquaculture Department, Rome, Electronic Publishing Policy and Support Branch Communication Division, 197 pp.

FAO, Food and Agricultural Organization of the United nations (2011a). Available at, http://www.fao.org/fishery/statistics/en (accessed on September 2011).

FAO, Food and Agriculture Organization of the United Nations (2011b). Available at: http://www.fao.org/fishery/culturedspecies/Sparus_aurata/en (accessed on March 2011).

FAO, Food and Agriculture Organization of the United Nations (2011c). Available at: http://www.fao.org/fishery/species/2370/en (accessed on March 2011).

Felsing M., Glencrossa B., Telferb T. (2005). Preliminary study on the effects of exclusion of wild fauna from aquaculture cages in a shallow marine environment. Aquaculture 243, 159–174.

Ferreira H., Ribeiro A., Dias J., Yúfera M., Arias A., Falcão M., Serpa D., Aires A., Pousão-Ferreira P., Cunha M. E., Valente L., Dinis M. T., Conceição L. (2010). Sustainable semi-intensive polyculture of seabream and sole in earthen ponds. Aquaculture Europe 35 (3), 17–21.

Ferreira J. G., Duarte P., Ball B. (1998). Trophic capacity of Carlingford Lough for aquaculture - analysis by ecological modelling. Aquatic Ecology 31, 361–379.

Ferreira J. G., Hawkins A. J. S., Monteiro P., Moore H., Service M., Pascoe P. L., Ramos L., Sequeira A. (2007). Integrated assessment of ecosystem-scale carrying capacity in shellfish growing areas. Aquaculture 275, 138–151.

Figueiredo M., Morato T., Barreiros J. P., Afonso P., Santos R. S. (2005). Feeding ecology of the white seabream, *Diplodus sargus*, and the ballan wrasse, *Labrus bergylta*, in the Azores. Fisheries Research 75, 107–119.

Fishbase (2010). Available at, http://www.fishbase.org (accessed on December 2010).

Folke C., Kautsky N., Berg H., Jansson Å., Troell M. (1998). The Ecological Footprint Concept for Sustainable Seafood Production - A Review. Ecological Applications 8 (1), 63–71.

Forja J. M., Blasco J., Gómez-Parra A. (1994). Spatial and seasonal variation of in situ benthic fluxes in the Bay of Cadiz (South-west Spain). Estuarine Coastal and Shelf Science 39, 127–141.

Freitas V., Cardoso J. F. M. F., Santos S., Campos J., Drent J., Saraiva S., Witte J., Kooijman S. A. L. M., van der Veer H. W. (2009). Reconstruction of food conditions for Northeast Atlantic bivalve species based on Dynamic Energy Budgets. Journal of Sea Research 62, 75–82.

Funge-Smith S. J., Briggs M. R. P. (1998). Nutrient budgets in intensive shrimp ponds: implications for sustainability. Aquaculture 164, 117–133.

Funge-Smith S., Phillips M. J. (2001). Aquaculture systems and species. In: Subasinghe R. P., Bueno P., Phillips M. J., Hough C., McGladdery S. E., Arthur J. R. (eds.) Aquaculture in the Third Millennium, Technical Proceedings of the Conference on Aquaculture in the Third Millennium, Bangkok, Thailand, 20-25 February 2000, pp. 129–135.

Furumai H., Kondo T., Ohgaki S. (1989). Phosphorus exchange kinetics and exchangeable phosphorus form in sediments. Water Research 6, 685–691.

Gatlin D. M., Barrows F. T., Brown P., Dabrowski K., Gaylord G. T., Hardy R. W., Herman E., Hu G., Krogdahl Å., Nelson R., Overturf K., Rust M., Sealey W., Skonberg D., Souza E. J., Stone D., Wilson R., Wurtele E. (2007). Expanding the utilization of sustainable plant products in aquafeeds: a review. Aquaculture Research 38, 551–579.

Gazi N. H., Khan S. R., Chakrabarti C. G. (2009). Integration of mussel in fish farm: Mathematical model and analysis. Nonlinear Analysis: Hybrid Systems 3 (1), 74–86.

GESAMP, IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection (2001). Planning and management for sustainable coastal aquaculture development. Reports and Studies, GESAMP, Rome, Italy, 68, 90 pp.

GESAMP, IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP Joint Group of Experts on Scientific Aspects of Marine Environmental Protection (2008). Assessment and communication of environmental risks in coastal aquaculture. Reports and Studies, GESAMP, Rome, Italy, 76, 198 pp.

Gibbs M. T. (2007). Sustainability performance indicators for suspended bivalve aquaculture activities. Ecological Indicators 7, 94–107.

Gibbs M. T. (2009). Implementation barriers to establishing a sustainable coastal aquaculture sector. Marine Policy 33, 83–89.

Gifford S., Dunstan R. H., O'Connor W., Roberts T., Toia R. (2004). Pearl aquaculture-profitable environmental remediation. Science Total Environment 319, 27–37.

Giles H., Pilditch C. A., Bell D. G. (2006). Sedimentation from mussel (*Perna canaliculus*) culture in the Firth of Thames, New Zealand: Impacts on sediment oxygen and nutrient fluxes. Aquaculture 261, 125–140.

Giovannini P., Piedrahita R. (1994). Modeling photosynthetic production optimization for aquaculture ponds. Aquacultural Engineering 13, 83–100.

Golomazou E., Athanassopoulou F., Vagianou S., Sabatakou O., Tsantilas H., Rigos G., Kokkokiris L. (2006). Diseases of White Sea Bream (*Diplodus sargus* L.) Reared in Experimental and Commercial Conditions in Greece. Turkish Journal of Veterinary and Animal Sciences 30, 389–396.

Gowen R. J., Bradbury N. B. (1987). The ecological impact of salmonids farming in coastal waters: a review. Oceanography Marine Biology Annual Review 25, 563–575.

Graca B., Witek Z., Burska D., Bialkowska I., Lukawska-Matuszewska K., Bolalek J. (2006). Pore water phosphate and ammonia below the permanent halocline in the south-eastern Baltic Sea and their benthic fluxes under anoxic conditions. Journal of Marine Systems 63, 141–154.

Grant J., Curran K. J., Guyondet T. L., Tita G., Bacher C., Koutitonski V., Dowd M. (2007). A box-model of carrying capacity for suspended mussel aquaculture in Lagune de la Grande-Entrée, Iles-de-la-Madeleine, Québec. Ecological Modelling 200, 193–206.

Grant J., Bacher C. (2001). A numerical model of flow modification induced by suspended aquaculture in a Chinese Bay. Canadian Journal of Fisheries and Aquatic Sciences 58, 1–9.

Gräslund S., Bengtsson B.-E. (2001). Chemicals and biological products used in south-east Asian shrimp farming, and their potential impact on the environment - a review. The Science of the Total Environment 280, 93–131.

Grasshoff K. (1983). Methods of seawater analysis. Verlag Chemie, New York, United States of America, 317 pp.

Grigorakis K., Rigos G. (2011). Aquaculture effects on environmental and public welfare – The case of Mediterranean mariculture. Chemosphere 85 (6), 899–919.

Gross A., Boyd C. E., Wood C. W. (2000). Nitrogen transformations and balance in channel catfish ponds. Aquacultural Engineering 24, 1–14.

Guinea J., Femandez F. (1997). Effect of feeding frequency, feeding level and temperature on energy metabolism in *Sparus aurata*. Aquaculture 148, 125–142.

Gutiérrez D., Gallardo V. A., Mayor S., Neira C., Vásquez C., Sellanes J., Rivas M., Soto A., Carrasco F., Baltasar M. (2000). Effects of dissolved oxygen and fresh organic matter on the bioturvation potencial of macrofauna in sublittoral sediments of central Chile during 1997/1998 El Niño. Marine Ecology Progress Series 202, 81–99.

Gutierrez-Wing M. T., Malone R. F. (2006). Biological filters in aquaculture: Trends and research directions for freshwater and marine applications. Aquaculture Engineering 34 (3), 163–171.

Gyllenhammar A., Håkanson L. (2005). Environmental consequence analyses of fish farm emissions related to different scales and exemplified by data from the Baltic – a review. Marine Environmental Research 60, 211–243.

Haas H., Van Weering T., Stigter H. (2002). Organic carbon in shelf seas: sinks or sources, processes and products. Continental Shelf Research 22, 691–717.

Hágaret H., Wikfords G. H., Shumway S. E. (2007). Diverse feeding responses of five species of bivalve molluscs when exposed to three species of marine algae. Journal of Shellfish Research 24, 549–559.

Hall P. O. J., Holby O., Kollberg S., Samuelsson M. O. (1992). Chemical fluxes and mass balances in a marine fish cage farm: IV. Nitrogen. Marine Ecology Progress Series 89, 81–91.

Hall P., Hulth S., Hulthe G., Landén A., Tengberg A. (1996). Benthic Nutrient Fluxes on a Basin-wide Scale in the Skagerrak (North-Easten North Sea). Journal of Sea Research 35 (1-3), 123–137.

Halpern B. S.,Walbridge S., Selkoe K. A., Kappel C. V., Micheli F., D'Agrosa C., Bruno J. F., Casey K. S., Ebert C., Fox H. E., Fujita R., Heinemann D., Lenihan H. S., Madin E. M. P., Perry M. T., Selig E. R., Spalding M., Steneck R., Watson R. (2008). A global map of human impact on marine ecosystems. Science 319, 948–952.

Hargreaves J. A. (1997). A simulation model of ammonia dynamics in commercial catfish ponds in the southeastern United States. Aquacultural Engineering 16, 27–43.

Hargreaves J. (1998). Nitrogen biogeochemistry of aquaculture ponds. Aquaculture 166, 181–212.

Hargreaves J. A. (2006). Photosynthetic suspended-growth systems in aquaculture. Aquacultural Engineering 34, 344–363.

Hargreaves J. A., Tucker C. S. (2003). Defining loading limits of static ponds for catfish aquaculture. Aquacultural Engineering 28, 47–63.

Hawkins A. J. S., Bayne B. L., Bougrier S., Héral M., Iglesias J. I. P., Navarro E., Smith R. F. M., Urrutia M. B. (1998). Some general relationships in comparing the feeding physiology of suspension-feeding bivalve molluscs. Journal of Experimental Marine Biology and Ecology 219, 87–103.

Heilskov A. C., Holmer M. (2001). Effects of benthic fauna on organic matter mineralization in fish-farm sediments: importance of size and abundance. ICES Journal of Marine Science 58, 427–434.

Helder W., Andersen F. (1987). An experimental approach to quantify biologically mediated dissolved silicate transport at the sediment-water interface. Marine Ecology Progress Series 39, 305–311.

Henriksen K., Kemp W. M. (1988). Nitrification in estuarine and coastal marine sediments. In: Blackburn T. H., Sorensen J. (eds.) Nitrogen cycling in coastal marine environments, John Wiley & Sons, New York, Unites States of America, pp. 207–249.

Hernández J. M., Gasca-Levya E., Léon C. J., Vergara J. M. (2003). A growth model for gilthead seabream (*Sparus aurata*). Ecological Modelling 165, 265–283.

Holby O., Hall P. O. J. (1991). Chemical flux and mass balances in a marine fish cage farm. II. Phosphorus. Marine Ecology Progress Series 70, 263–272.

Holmer M., Duarte C. M., Heilskov A., Olesen B., Terrados J. (2003). Biogeochemical conditions in sediments enriched by organic matter from net-pen fish farms in the Bolinao area, Philippines. Marine Pollution Bulletin 46, 1470–1479.

Holmer M., Frederiksen M. S. (2007). Stimulation of sulfate reduction rates in Mediterranean fish farm sediments inhabited by the seagrass Posidonia oceanica. Biogeochemistry 85, 169–184.

Holmer M., Heilskov A. C. (2008). Distribution and bioturbation effects of the tropical alpheid shrimp *Alpheus macellarius* in sediments impacted by milkfish farming. Estuarine Coastal and Shelf Science 76, 657–667.

Holmer M., Kristensen E. (1996). Seasonality of sulfate reduction and pore water solutes in a marine fish farm sediment: the importance of temperature and sedimentary organic matter. Biogeochemistry 32 (1), 15–39.

Holmer M., Marbà N., Díaz-Almela E., Duarte C. M., Tsapakis M., Danovaro R. (2007). Sedimentation of organic matter from fish farms in oligotrophic Mediterranean assessed through bulk and stable isotope (δ13C and δ15N) analyses. Aquaculture 262 (2–4), 268–280.

Holmer M., Marbà N., Terrados J., Duarte C. M., Fortes M. D. (2002). Impacts of milkfish (*Chanos chanos*) aquaculture on carbon and nutrient fluxes in the Bolinao area, Philippines. Marine Pollution Bulletin 44 (7), 685–696.

Holmer M., Wildish D., Hargrave B. (2005). Organic enrichment from marine finfish aquaculture and effects on sediment biogeochemical processes. Handbook of Environmental Chemistry 5, 181–206.

Hopkins J., Hamilton II D., Sandier P., Browdy C., Stokes A. (1993). Effect of water exchange rate on production, water quality, effluent characteristics and nitrogen budgets of intensive shrimp ponds. Journal of the World Aquaculture Society 24 (3), 304–320.

Hosomi M., Sudo R. (1982). Release of phosphorous from lake sediments. Environment International 7, 93–98.

Hulata G. (2001). Genetic Manipulations in Aquaculture: A Review of Stock Improvement by Classical and Modern Technologies. Genetica 111 (1-3), 155–73.

Hussenot J. M. E. (2003). Emerging effluent management strategies in marine fish-culture farms located in European coastal wetlands. Aquaculture 226, 113–128.

Inglis G. J., Hayden B. J., Ross A. H. (2000). An overview of factors affecting the carrying capacity of coastal embayments for mussel culture. NIWA, Christchurch, Client Report CHCOO/69, 31 pp.

Ingram B. A. (2009). Culture of juvenile Murray cod, trout cod and Macquarie perch (Percichthyidae) in fertilised earthen ponds. Aquaculture 287, 98–106.

Islam M. S. (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, 48–61.

Islam M. S., Sarker J., Yamamoto T., Wahab A., Tanaka M. (2004). Water and sediment quality, partial mass budget and effluent N loading in coastal brackish water shrimp farms in Bangladesh. Marine Pollution Bulletin 48, 471–485.

Jackson C., Preston N., Thompson P. J., Burford M. (2003). Nitrogen budget and effluent nitrogen components at an intensive shrimp farm. Aquaculture 218, 397–411.

Jambrina M. C., Jimeno R., Silva L., Cañavate J. P. (1995). Estimación del Vertido de Sólidos en Suspensión y Materia Orgánica Particulada en los Efluentes de un Cultivo semiintensivo de Dorada S. aurata L. (1758) en Estanques de Tierra de la Bahía de Cádiz. In: Castelló F., Calderer A. (eds.) V Spanish Congress on Aquaculture, Publicaciones Universidad de Barcelona, Barcelona, Spain, pp. 935–940.

Jamu D. M., Piedrahita R. H. (2001). Ten-year simulations of organic matter concentrations in tropical aquaculture ponds using the multiple pool modeling approach. Aquacultural Engineering 25, 187–201.

Jamu D. M., Piedrahita R. H. (2002). An organic matter and nitrogen dynamics model for the ecological analysis of integrated aquaculture/agriculture systems: 1. model development and calibration. Environmental Modelling & Software 17, 571–582.

Jegatheesan V., Zeng C., Shu L., Manicom C., Steicke C. (2007). Technological advances in aquaculture farms for minimal effluent discharge to oceans. Journal of Cleaner Production 15, 1535–1544.

Jiang W., Gibbs M. T. (2005). Predicting the carrying capacity of bivalve shellfish culture using a steady, linear food web model. Aquaculture 244, 171–185.

Jiménez-Montealegre R., Verdegem M. C. J., van Dam A., Verreth J. A. J. (2002a). Conceptualization and validation of a dynamic model for the simulation of nitrogen transformations and fluxes in fishponds. Ecological Modelling 147, 123–152.

Jiménez-Montealegre R., Verdegem M., Zamora J., Verreth, J. (2002b). Organic matter sedimentation and ressuspension in tilapia (*Oreochromis niloticus*) ponds during a production cycle. Aquacultural Engineering 26, 1–12.

Jones A. B., O’Donohue M. J., Udy J., Dennison W. C. (2001). Assessing Ecological Impacts of Shrimp and Sewage Effluent: Biological Indicators with Standard Water Quality Analyses. Estuarine, Coastal and Shelf Science 52, 91–109.

Jordan T. E., Correl D. L., Miklas J., Weller D. E. (1991). Nutrients and chlorophyll at the interface of a watershed and an estuary. Limnology and Oceanography 36, 251–267.

Jørgensen S. E., Nielsen S., Jørgensen L. (1991). Handbook of Ecological Parameters and Ecotoxicology. Elsevier, Amsterdam, The Netherlands, 1263 pp.

Joyni M. J., Kurup B. M., Avnimelech Y. (2011). Bioturbation as a possible means for increasing production and improving pond soil characteristics in shrimp-fish brackish water ponds. Aquaculture 318, 464–470.

Jusup M., Klanjcek, T., Matsuda, H., Kooijman, S. A. L. M. (2011). A full lifecycle bioenergetic model for bluefin tuna. PLoS One 6 (7), e21903 http://dx.doi.org/10.1371/journal.pone.0021903.

Kaiser M. J. (2001). Ecological effects of shellfish cultivation. In: Black K. D. (ed.) Environmental Impacts of Aquaculture, Sheffield Academic Press, Sheffield, United Kingdom, pp. 51–75.

Kalantzi I., Karakassis I. (2006). Benthic impacts of fish farming: meta-analysis of community and geochemical data. Marine Pollution Bulletin 52, 484–493.

Kapuscinski A. R., Brister D. J. (2001). Genetic impacts of aquaculture. In: Black K. D. (ed.) Environmental Impacts of Aquaculture, Sheffield Academic Press, Sheffield, United Kingdom, pp. 128–153.

Karakassis I., Tsapakis M., Hatziyanni E. (1998). Seasonal variability in sediment profiles beneath fish farm cages in the Mediterranean. Marine Ecology Progress Series 162, 243–252.

Karakassis I., Tsapakis M., Hatziyanni E., Papadopoulou K.-N., Plaiti W. (2000). Impact of cage farming of fish on the seabed in three Mediterranean coastal areas. ICES Journal of Marine Science 57, 1462–1471.

Kaushik S. J., Covés D., Dutto G., Blanc D. (2004). Almost total replacement of fish meal by plant protein sources in the diet of a marine teleost, the European seabass, *Dicentrarchus labrax*. Aquaculture 230, 391–404.

Kautsky N., Rönnbäck P., Tedengren M., Troell M. (2000). Ecosystem perspectives on management of disease in shrimp pond farming. Aquaculture 191, 145–161.

Kawamiya M., Kishi K. J., Yamanaka Y., Suginohara N. (1995). An ecological-physical coupled model applied to Station Papa. Journal of Oceanography 51, 635–664.

Kelly L. A., Elberizon I. R. (2001). Freshwater finfish cage culture. In: Black K. D. (ed.) Environmental Impacts of Aquaculture, Sheffield Academic Press, Sheffield, United Kingdom, pp. 32–50.

Kelly L. A., Stellwagen J., Bergheim A. (1996). Waste loadings from a fresh-water Atlantic Salmon farm in Scotland. Water Resources Bulletin 32, 1017–1025.

Kempf M., Merceron M., Cadour G., Jeanneret H., Mear Y., Miramand P. (2002). Environmental impact of a salmonid farm on a well-flushed marine site: 2. Biosedimentology. Journal of Applied Ichthyology 18, 51–60.

King S., Pushchak R. (2008). Incorporating cumulative effects into environmental assessments of mariculture: Limitations and failures of current sitting methods. Environmental Impact Assessment Review 28 (8), 572–586.

Kittiwanich J., Yamamoto T., Kawaguchi O., Hashimoto T. (2007). Analyses of phosphorus and nitrogen cycling in the estuarine ecosystem of Hiroshima Bay by a pelagic and benthic coupled model. Estuarine, Coastal and Shelf Science 75, 189–204.

Klaoudatos S. D., Klaoudatos D. S., Smith J., Bogdanos K., Papageorgiou E. (2006). Assessment of site specific benthic impact of floating cage farming in the eastern Hios island, Eastern Aegean Sea, Greece. Journal of Experimental Marine Biology and Ecology 338, 96–111.

Kochba M., Diab S., Avnimelech Y. (1994). Modelling of nitrogen transformations in intensively aerated fishponds. Aquaculture 120 (1-2), 95–104.

Kongkeo H. (1997). Comparison of intensive shrimp farming systems in Indonesia, Philippines, Taiwan and Thailand. Aquaculture Research 28, 789–796.

Kooijman S. A. L. M. (2000). Dynamic Energy and Mass Budgets in Biological Systems, Cambridge University Press, Cambridge, United Kingdom, 444 pp.

Kooijman S. A. L. M. (2009). Social interactions can affect feeding behaviour of fish in tanks. Journal of Sea Research 62, 175–178.

Kooijman S. A. L. M. (2010). Dynamic Energy Budget theory for Metabolic Organization, 3rd ed., Cambridge University Press, Cambridge, United Kingdom, 532 pp.

Kristensen E. (1993). Seasonal variations in benthic community metabolism and nitrogen dynamics in a shallow, organic-poor Danish lagoon. Estuarine Coastal and Shelf Science 36, 565–586.

Krom M. D., Ellner S., van Rijn J., Neori A. (1995). Nitrogen and phosphorus cycling and transformations in a prototype ‘non polluting’ integrated mariculture system. Marine Ecology Progress Series 118, 25–36.

Krom M. D., Neori A. (1989). A total nutrient budget for an experimental intensive fishpond with circularly moving seawater. Aquaculture 83 (3-4), 345–358.

Krom M. D., Porter C., Gordin H. (1985a). Description of the water quality conditions in a semi-intensively cultured marine fish pond in Eilat, Israel. Aquaculture 49, 141–157.

Krom M. D., Porter C., Gordin H. (1985b). Nutrient budget of a marine fish pond in Eilat, Israel. Aquaculture 51 (1), 65–80.

Lamoureux J., Tiersch T. R., Hall S. G. (2005). Pond heat and temperature regulation (PHATR): Modeling temperature and energy balances in earthen outdoor aquaculture ponds. Aquacultural Engineering 34 (2), 103–116.

Laws E. A., Archie J. W. (1981). Appropriate use of regression analysis in marine biology. Marine Biology 65, 99–118.

Lefebvre S., Bacher C., Meuret A., Hussenot J. (2001). Modelling approach of nitrogen and phosphorus exchanges at the sediment-water interface of an intensive fishpond system. Aquaculture 195, 279–297.

Lefrançois P., Puigagut J., Chazarenc F., Comeau Y. (2010). Minimizing phosphorus discharge from aquaculture earth ponds by a novel sediment retention system. Aquacultural Engineering 43, 94–100.

Leitão F., Santos M.N., Monteiro C.C. (2007). Contribution of artificial reefs to the diet of the white sea-bream, *Diplodus sargus* (Linnaeus, 1758). ICES Journal of Marine Science 64 (3), 473–478.

Lerat Y., Lasserre P., Corre P. (1990). Seasonal changes in porewater concentrations of nutrients and their diffusive fluxes at the sediment-water interface. Journal of Experimental Marine Biology and Ecology 135, 135–160.

Li Y.-H., Gregory S. (1974). Diffusion of ions in sea water and in deep-sea sediments. Geochimica et Cosmochimica Acta 38, 703–714.

Li L., Yakupitiyage A. (2003). A model for food nutrient dynamics of semi-intensive pond fish culture. Aquacultural Engineering 27, 9–38.

Libralato S., Solidoro C. (2008). A bioenergetic growth model for comparing *Sparus aurata*'s feeding experiments. Ecological modeling 214, 325–337.

Lin C. K., Yi Y. (2003). Minimizing environmental impacts of freshwater aquaculture and reuse of pond effluents and mud. Aquaculture 226, 57–68.

Lotze H. K., Lenihan H. S., Bourque B. J., Bradbury R. H., Cooke R. G., Kay M. C., Kidwell S. M., Kirby M. X., Peterson C. H., Jackson J. B. C. (2006). Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312, 1806–1809.

Loy A., Bertelletti M., Costa C., Ferlin L., Cataudella S. (2001). Shape Changes and Growth Trajectories in the Early Stages of Three Species of the Genus *Diplodus* (Perciformes, Sparidae). Journal of Morphology 250, 24–33.

Lupatsch I. (2003). Factorial approach to determining energy and protein requirements of gilthead seabream (*Sparus aurata*) for optimal efficiency of production, Ph.D. Thesis, University of Bonn, Germany, 110 pp.

Machias A., Karakassis I., Labropoulou M., Somarakis S., Papadopoulou K. N., Papaconstantinou C. (2004). Changes in wild fish assemblages after the establishment of a fish farming zones in an oligotrophic marine ecosystem. Estuarine Coastal and Shelf Science 60, 771–779.

Magill S. H., Thetmeyer H., Cromey C. J. (2006). Settling velocity of faecal pellets of gilthead sea bream (*Sparus aurata* L.) and sea bass (*Dicentrarchus labrax* L.) and sensitivity analysis using measured data in a deposition model. Aquaculture 251, 295–305.

Maldonado M., Carmona M. C., Echeverría Y., Riesgo A. (2005). The environmental impact of Mediterranean cage fish farms at semiexposed locations: does it need a re-assessment?. Helgoland Marine Research 59, 121–135.

Mazón M. J., Piedecausa M. A., Hernández M. D., García B. G. (2007). Evaluation of environmental nitrogen and phosphorus contributions as a result of intensive ongrowing of common octopus (*Octopus vulgaris*). Aquaculture 266, 226–235.

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.

Meade J. W. (1985). Allowable ammonia for fish culture. Progress in Fish Culture 47, 135–145.

Merino G., Barange M., Mullon C., Rodwell L. (2010). Impacts of global environmental change and aquaculture expansion on marine ecosystems. Global Environmental Change 20, 586–596.

Mesnage V., Ogier S., Bally G., Disnar J.-R., Lottier N., Dedieu K., Rabouille C., Copard Y. (2007). Nutrient dynamics at the sediment–water interface in a Mediterranean lagoon (Thau, France): Influence of biodeposition by shellfish farming activities. Marine Environmental Research 63, 257–277.

Meyer D. I., Brune D. E. (1982). Computer modelling of the diurnal oxygen levels in a stillwater aquaculture pond. Aquacultural Engineering 1, 245–261.

Miranda F. R., Lima R. N., Crisóstomo L. A., Santana M. G. S. (2008). Reuse of inland low-salinity shrimp farm effluent for melon irrigation. Aquacultural Engineering 39, 1–5.

Mischke C. C., Zimba P. V. (2004). Plankton community responses in earthen channel catfish nursery ponds under various fertilization regimes. Aquaculture 233, 219–235.

Mitsch W. J. (1997). Ecological engineering: The roots and rationale of a new paradigm. In: Etnier C., Guterstam B. (eds.) Ecological engineering for waste water treatment, CRC Press, Boca Raton, United States of America, pp. 1–20.

Montoya R. A., Lawrence A. L., Grant W. E., Velasco M. (2000). Simulation of phosphorus dynamics in an intensive shrimp culture system: effects of feed formulations and feeding strategies. Ecological Modelling 129, 131–142.

Moran D., Dann S. (2008). The economic value of water use: Implications for implementing the Water Framework Directive in Scotland. Journal of Environmental Management 87, 484–496.

Moriarty D. J. W. (1997). The role of microorganisms in aquaculture ponds. Aquaculture 151, 333–349.

Moss J., Beauchamp D. A., Cross A. D., Farley E. V., Murphy J. M., Helle J. H., Walker R. V., Myers K. W. (2009). Bioenergetic model estimates of inter annual and spatial patterns in consumption demand and growth potential of juvenile pink salmon (*Oncorhynchus gorbuscha*) in the Gulf of Alaska. Deep Sea Research Part II: Topical Studies in Oceanography 56 (24), 2553–2559.

Muendo P. (2006). The role of fish ponds in the nutrient dynamics of mixed farming systems. PhD Thesis, Wageningen University, The Netherlands, 120 pp.

Muir J. (2005). Managing to harvest? Perspectives on the potential of aquaculture. Philosophical Transactions of the Royal Society 360, 191–218.

Mukherjee B., Mukherjee D., Nivedita M. (2008). Modelling carbon and nutrient cycling in a simulated pond system at Ranchi. Ecological Modelling 213, 437–448.

Mwashote B. M., Jumba I. O. (2002). Quantative aspects of inorganic nutrient fluxes in the Gazi Bay (Kenya): implications for coastal ecosystems. Marine Pollution Bulletin 44, 1194–1205.

Mwegoha W. J. S., Kaseva M. E., Sabai S. M. M. (2010). Mathematical modeling of dissolved oxygen in fish ponds. African Journal of Environmental Science and Technology 4 (9), 625–638.

Nash C. E., Burbridge P. R., Volkman J. K. (2005). Guidelines for Ecological Risk Assessment of Marine Aquaculture. Prepared at the NOAA Fisheries Service Manchester Research Station International Workshop, April 11–14, 2005. U.S. Department of Commerce, National Oceanic and Atmospheric Agency (NOAA), National Marine Fisheries Service, Tech. Memo. NMFS-NWFSC-71, 90 pp.

Naylor R. L., Goldburg R. J., Primavera J. H., Kautsky N., Beveridge M. C. M., Clay J., Folke C., Lubchenco J., Mooney H., Troell M. (2000). Effect of aquaculture on world fish supplies. Nature 405, 1017–1024.

Naylor R., Hindar K., Fleming I. A., Goldburg R., Williams S., Volpe J., Whoriskey F., Eagle J., Kelso D., Mangel M. (2005). Fugitive Salmon: Assessing the Risks of Escaped Fish from Net-Pen Aquaculture. BioScience 55 (5), 427–37.

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.

Neori A., Krom M. D., Cohen I., Gordin H. (1989). Water Quality Conditions and Particulate Chlorophyll a of New Intensive Seawater Fishponds in Eilat, Israel: Daily and Diel Variations. Aquaculture 80, 63–78.

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.

Nhan D. K., Milstein A., Verdegem M. C. J., Verreth J. A.V. (2006). Food inputs, water quality and nutrient accumulation in integrated pond systems: A multivariate approach. Aquaculture 261, 160–173.

Nixon S. W., Pilson, M. E. Q. (1983). Nitrogen in estuarine and coastal marine ecosystems. In: Carpenter E. J., Capone D. G. (ed.) Nitrogen in the Marine Environment, Plenum Press, New York, United States of America, pp. 565–648.

Nizzoli D., Bartoli M., Cooper M., Welsh D. T., Underwood G. J. C., Viaroli P. (2007). Implications for oxygen, nutrient fluxes and denitrification rates during the early stage of sediment colonisation by the polychaete *Nereis* spp. in four estuaries. Estuarine Coastal and Shelf Science 75, 125–134.

NOAA, National Oceanic and Atmospheric Administration (2010). Available at, http://aquaculture.noaa.gov/what/welcome.html (accessed on December 2010).

Nobre A. M., Robertson-Andersson D., Neori A., Sankar K. (2010). Ecological–economic assessment of aquaculture options: Comparison between abalone monoculture and integrated multi-trophic aquaculture of abalone and seaweeds. Aquaculture 306, 116–126.

Nordvarg L., Håkanson L. (2002). Predicting the environmental response of fish farming in coastal areas of the Aland archipelago (Baltic Sea) using management models for coastal water planning. Aquaculture 206, 217–243.

Nowicki B. L., Nixon S. W. (1985). Benthic nutrient remineralization in a coastal lagoon ecosystem. Estuaries 8, 182–190.

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 (1-4), 257–277.

Nwanna L. C. (2003). Risk Management in Aquaculture by Controlled Feeding Regimen Pakistan Journal of Nutrition 2 (6), 324–328.

OAERRE, Oceanographic Applications to Eutrophication in Regions of Restricted Exchange (2001). Available at, http://www.oarre.napier.ac.uk (accessed on May 2001).

Ozorio R., Valente L., Pousão-Ferreira P., Oliva-Teles A. (2006). Growth performance and body composition of white seabream (*Diplodus sargus*) juveniles fed diets with different protein and lipid levels. Aquaculture Research 37, 255–263.

Páez-Osuna F. (2001a). The Environmental Impact of Shrimp Aquaculture: Causes, Effects, and Mitigating Alternatives. Environmental Management 28 (1), 131–140.

Páez-Osuna F. (2001b). The environmental impact of shrimp aquaculture: a global perspective. Environmental Pollution 112, 229–231.

Páez-Osuna F., Guerrero-Galvan S. R., Ruiz-Fernandez A. C. (1998). The environmental impact of shrimp aquaculture and the coastal pollution in Mexico. Marine Pollution Bulletin 36, 65–75.

Páez-Osuna F., Guerrero-Galvan S. R., Ruiz-Fernandez A. C., Espinoza-Angulo R. (1997). Fluxes and mass balances of nutrients in a semi-intensive shrimp farm in North-Western Mexico. Marine Pollution Bulletin 34, 290–297.

Papatryphon E., Petit J., van der Werf H. M. G., Sadasivam K. J., Claver K. (2005). Nutrient-Balance Modeling as a Tool for Environmental Management in Aquaculture: The Case of Trout Farming in France. Environmental Management 35 (2), 161–174.

Parsons T., Maita C., Lally C. (1984). A Manual of Chemical and Biological Methods of Seawater Analysis. Pergamon Press, Oxford, United Kingdom, 173 pp.

Paspatis M., Boujard T., Maragoudaki D., Kentouri M. (2000). European sea bass growth and N and P loss under different feeding practices. Aquaculture 184, 77–88.

Pearson T. H., Black K. D. (2001). The environmental impacts of marine fish cage culture. In: Black K. D. (ed.) Environmental Impacts of Aquaculture, Sheffield Academic Press, Sheffield, United Kingdom, pp. 1–31.

Pecquerie L., Petitgas P., Kooijman S. A. L. M. (2009). Modeling fish growth and reproduction in the context of the Dynamic Energy Budget theory to predict environmental impact on anchovy spawning duration. Journal of Sea Research 62, 93–105.

Pecquerie L., Johnson L. R., Kooijman S. A. L. M., Nisbet R. (2011). Analyzing variations in life-history traits of Pacific salmon in the context Dynamic Energy Budget (DEB) theory. Journal of Sea Research 66 (4), 424–433.

PEMSEA, Partnerships in Environmental Management for the Seas of East Asia (2003). Sustainable Development Strategy for the Seas of East Asia: Regional Implementation of the World Summit on Sustainable Development Requirements for the Coasts and oceans. PEMSEA, Quezon City, Philippines, 111 pp.

Peng J.-F., Wang B.-Z., Song Y.-H., Yuan P. (2007). Modeling N transformation and removal in a duckweed pond: Model development and calibration. Ecological Modelling 206, 147–152.

Pereira A., Duarte P., Norro A. (2006). Different modelling tools of aquatic ecosystems: A proposal for a unified approach. Ecological Informatics 1, 407–421.

Pereira A., Duarte P., Reis L. P. (2007). An Integrated Ecological Modelling and Decision Support Methodology. Proceedings of the 21st European Conference on Modelling and Simulation – ECMS 2007, Prague, 4-6 Jun 2007, pp. 497–502.

Pérez M., García T., Invers O., Ruiz J. M. (2008). Physiological responses of the seagrass *Posidonia oceanica* as indicators of fish farm impact. Marine Pollution Bulletin 56, 869–879.

Pérez M., Rodríguez C., Cejas J. R., Martín M. V., Jerez S., Lorenzo A. (2007). Lipid and fatty acid content in wild white seabream (*Diplodus sargus*) broodstock at different stages of the reproductive cycle. Comparative Biochemistry and Physiology Part B 146, 187–196.

Peterson E. L., Harris J. A., Wadhwa L. C. (2000). CFD modeling pond dynamic processes. Aquacultural Engineering 23, 61–93.

Piedecausa M. A., Aguado-Giménez F., Cerezo-Valverde J., Hernández-Llorente M. D., García-García B. (2010). Simulating the temporal pattern of waste production in farmed gilthead seabream *(Sparus aurata*), European seabass (*Dicentrarchus labrax*) and Atlantic bluefin tuna (*Thunnus thynnus*). Ecological Modelling 221, 634–640.

Piedecausa M. A., Aguado-Giménez F., García-García B., Ballester G., Telfer T. (2009). Settling velocity and total ammonia nitrogen leaching from commercial feed and faecal pellets of gilthead seabream (*Sparus aurata* L. 1758) and seabass (*Dicentrarchus labrax* L. 1758). Aquaculture Research 40 (15), 1703–1714.

Piedrahita R. H. (1988). Introduction to computer modelling of aquaculture pond ecosystems. Aquaculture and Fisheries Management 19, 1–12.

Piedrahita R. H. (1991). Simulation of short-term management actions to prevent oxygen depletion in ponds. Journal of the World Aquaculture Society 22 (3), 157–166.

Piedrahita R. H. (2003). Reducing the potential environmental impact of tank aquaculture effluents through intensification and recirculation. Aquaculture 226, 35–44.

Piedrahita R. H., Brune D. E., Tchnobanoglous G., Orlob G. T. (1984). A general model of the aquaculture pond system. Journal of the World Mariculture Society 14, 355–366.

Pillay T. V. R., Kutty M. N. (2005). Aquaculture: principles and practices. Blackwell Publishing, Oxford, United Kingdom, 624 pp.

Pitta P., Apostolaki E. T., Tsagaraki T., Tsapakis M., Karakassis I. (2006). Fish farming effects on chemical and microbial variables of the water column: a spatio-temporal study along the Mediterranean Sea. Hydrobiologia 563, 99–108.

Primavera J. H. (1998). Tropical shrimp farming and its sustainability. In: DeSilva S. (ed.) Tropical mariculture, Academic Press, London, United Kingdom, pp. 257–289.

Primavera J. H. (2006). Overcoming the impacts of aquaculture on the coastal zone. Ocean & Coastal Management 49, 531–545.

PROMAR (2006). Cooperação Algarve – Andaluzia para a promoção de recursos aquícolas marinhos no litoral Atlântico – Sul), EU Project INTERREG III-A.

Rana K. J., Siriwardena S., Hasan M. R. (2009). Impact of Rising Feed Ingredient Prices on Aquafeeds and Aquaculture Production. FAO Fisheries and Aquaculture Technical Paper 541, Rome, Italy, 63 pp.

Rao A. M. F., Jahnke R. A. (2004). Quantifying porewater exchange across the sediment-water interface in the deep sea with in situ tracer studies. Limnology and Oceanography: Methods 2, 75–90.

Read P., Fernandes T. (2003). Management of environmental impacts of marine aquaculture in Europe. Aquaculture 226, 139–163.

Read P. A., Fernandes T. F., Miller K. L. (2001). The derivation of scientific guidelines for best environmental practice for the monitoring and regulation of marine aquaculture in Europe. Journal of Applied Ichthyology 17 (4), 146–152.

Rees W. E., Wackernagel M. (1994). Ecological footprints and appropriated carrying capacity: measuring the natural capital requirements of the human economy. In: Jansson A. M., Hammer M., Folke C., Costanza R. (eds.) Investing in Natural Capital: the Ecological Economics Approach to Sustainability, Island Press, Washington DC, United States of America, pp. 362–390.

Remen M., Imsland A. K., Stefansson S. O., Jonassen T. M., Foss A. (2008). Interactive effects of ammonia and oxygen on growth and physiological status of juvenile Atlantic cod (*Gadus morhua*). Aquaculture 274, 292–299.

Requena A., Fernández-Borrás J., Planas J. (1997). The effects of a temperature rise on oxygen consumption and energy budget in gilthead seabream. Aquaculture International 5, 415–426.

Riise J. C., Roos N. (1997). Benthic metabolism and the effects of bioturbation in a fertilised polyculture fish pond in northeast Thailand. Aquaculture 150, 45–62.

RIVM (1995). A General Strategy for Integrated Environmental Assessment at the European Environment Agency. European Environment Agency, Copenhagen, Denmark.

Rodríguez-Gallego L., Meerhoff E., Poersch L., Aubriot L., Fagetti C., Vitancurt J., Conde D. (2008). Establishing limits to aquaculture in a protected coastal lagoon: Impact of *Farfantepenaeus paulensis* pens on water quality, sediment and benthic biota. Aquaculture 277, 30–38.

Rowland S. J., Allan G. L., Hollis M., Pontifex T. (1995). Production of the Australian freshwater silver perch, *Bidyanus bidyanus* (Mitchell), at two densities in earthen ponds. Aquaculture 130, 317–328.

Ruardji P., van Raaphorst W. (1995). Benthic nutrient regeneration in the ERSEM ecosystem model of the North Sea. Netherlands Journal of Sea Research 33, 453–483.

Ruiz J. M., Pérez M., Romero J. (2001). Effects of fish farm loadings on seagrass (Posidonia oceanica) distribution, growth and photosynthesis. Marine Pollution Bulletin 42, 749–760.

Ryther J. H., Dunstan W. H. (1971). Nitrogen, phosphorus and eutrophication in the coastal marine environment. Science 171, 1008–1013.

Sá R., Pousão-Ferreira P., Oliva-Teles A. (2006). Effect of dietary protein and lipid levels on growth and feed utilization of white sea bream (*Diplodus sargus*) juveniles. Aquaculture Nutrition 12, 310–321.

Sá R., Pousão-Ferreira P., Oliva-Teles A. (2007). Growth performance and metabolic utilization of diets with different protein: carbohydrate ratios by white sea bream (*Diplodus sargus*, L.) juveniles. Aquaculture Research 38, 100–105.

Sá R., Pousão-Ferreira P., Oliva-Teles A. (2008). Dietary protein requirement of white seabream (*Diplodus sargus*) juveniles. Aquaculture Nutrition 14, 309–317.

Saaty T. L. (1980). The Analytic Hierarchy Process. McGraw-Hill, New York, United States of America, 287 pp.

Sandifer P. A., Hopkins J. S. (1996). Conceptual design of a sustainable pond-based shrimp culture system. Aquaculture Engineering 15, 41–52.

Sandnes J., Forbes T., Hansen R., Sandnes B., Rygg B. (2000). Bioturbation and irrigation in natural sediments, described by animal-community parameters. Marine Ecology Progress Series 197, 169–179.

Sañudo-Wilhemy S. A. (2006). A phosphate alternative. Nature 439, 25–26.

Sarà G. (2007). A meta-analysis on the ecological effects of aquaculture on the water column: Dissolved nutrients. Marine Environmental Research 63, 390–408.

Schwartz M. F., Boyd C. E. (1994). Effluent quality during harvest of Channel Catfish from watershed ponds. Progressive Fish Culturist 56, 25–32.

Schulz H. (2000). Quantification of Early Diagenesis: Dissolved Constituents in Marine Pore Water. In: Schulz H., Zabel M. (eds.) Marine Geochemistry, Springer, New York, United States of America, pp. 75–124.

SEACASE, Sustainable Extensive And semi-intensive Coastal Aquaculture in Southern Europe (2009). Available at, http://www.seacase.org (accessed on December 2009).

SECRU, Scottish Executive Central Research Unit (2002). Review and synthesis of the environmental impacts of aquaculture, The Scottish Association for Marine Science and Napier University, Edinburgh, Scotland, 62 pp.

Serpa D., Duarte P. (2008). Impacts of aquaculture and Mitigation measures. In: Russo R. (eds.) Dynamic Biochemistry, Process Biotechnology and Molecular Biology 2 (Special issue 1), Global Science Books, 1–20.

Serpa D., Falcão M., Duarte P., Cancela da Fonseca L., Vale C. (2007a). Evaluation of ammonium and phosphate release from intertidal and subtidal sediments of a shallow coastal lagoon (Ria Formosa-Portugal): a modelling approach. Biogeochemistry 82, 291–304.

Serpa D., Falcão M., Pousão-Ferreira P., Vicente M., Carvalho S (2007b). Geochemical changes in white seabream (*Diplodus sargus*) earth ponds during a production cycle. Aquaculture Research 38, 1619–1626.

Shin P. K. S., Wu R. S. S. (2003). Estimating the environmental carrying capacity for sustainable marine fish culture: a modelling approach. In: Yu H., Bermas N. (eds.) Determining environmental carrying capacity of coastal and marine areas: progress, constraints and future options, PEMSEA Workshop Proceedings 11, pp. 85–94.

Silvert W., Cromey C. (2001). Modelling impacts. In: Black K. D. (ed.) Environmental Impacts of Aquaculture, Sheffield Academic Press, Sheffield, United Kingdom, pp. 154–181.

Slomp C. (1997). Early diagenesis of phosphorus in continental margin sediments. PhD Thesis, Wageningen Agricultural University, The Netherlands, 178 pp.

Slomp C. P., Malschaert J. F., van Raaphorst W. (1998). The role of absorption in sediment-water exchange of phosphate in North Sea continental margin sediments. Limnology and Oceanography 43 (5), 832–846.

Sorgal, Sociedade de Óleos e Rações (2011). Available at, http://www.sorgal.pt (accessed on March 2011).

Spreij M. (2004). Trends in national aquaculture legislation (part II). FAO Aquaculture Newletter 31, 22–27.

Steeby J., Hargreaves J., Tucker C., Kingsbury S. (2004). Accumulation, organic carbon and dry matter concentration of sediment in commercial channel catfish ponds. Aquacultural Engineering 30, 115–126.

Stirling Institute of Aquaculture (2004). Study of the market for aquaculture produced lubina y dorada species. Report to the European Commission, DG Fisheries.

Subasinghe R. P., Phillips M. J. (2002). Aquatic Animal Health Management: Opportunities and Challenges for Rural, Small-Scale Aquaculture and Enhanced Fisheries Development: Workshop Introductory Remarks. In: Arthur J. R., Phillips M. J., Subasinghe R. P., Reantaso M. B., MacRae I. H. (eds.) Primary Aquatic Animal Health Care in Rural, Small- Scale Aquaculture Development, FAO Fisheries Technical Paper, FAO, Rome, Italy, 406,1–5.

Sugiura S. H., Marchant D. D., Kelsey K., Wiggins T., Ferraris R. P. (2006). Effluent profile of commercially used low-phosphorus fish feeds. Environmental Pollution 140, 95–101.

Sumagaysay-Chavoso N. S., San Diego-McGlone M. L., David L. T. (2004). Environmental capacity of receiving water as basis for regulating intensity of milkfish (*Chanos chanos* Forsskal) culture. Journal of Applied Ichthyology 20, 476–487.

Tacon A. G. J., Forster I. P. (2003). Aquafeeds and the environment: policy implications. Aquaculture 226, 181–189.

Tacon A. G. J., Hasan M. R., Subasinghe R.P. (2006). Use of fishery resources as feed inputs for aquaculture development: trends and policy implications. FAO Fisheries Circular No. 1018, FAO, Rome, Italy, 99 pp.

Teague K. G., Madden C. J., Day J. W. (1988). Sediment-water oxygen and nutrient fluxes in a river-dominated estuary. Estuaries 11, 1–9.

Thakur D. P., Lin C. K. (2003). Water quality and nutrient budget in closed shrimp (*Penaeus monodon*) culture systems. Aquacultural Engineering 27, 159–176.

Timmons M. B., Ebeling J. M., Wheaton F. W., Summerfelt S. T., Vinci B. J. (2001). Recirculating Aquaculture Systems. Cayuga Aqua Ventures, Ithaca, New York, United States of America, 647 pp.

Torres-Beristain B., Verdegem M., Kerepeczki E., Verreth J. (2006). Decomposition of high protein aquaculture feed under variable oxic conditions. Water Research 40, 1341–1350.

Tovar T., Moreno C., Mánuel-Vez M. P., Garcia-Vargas M. (2000a). Environmental Implications of Intensive Marine Aquaculture in Earthen Ponds. Marine Pollution Bulletin 40 (11), 981–988.

Tovar A., Moreno C., Mánuel-Vez M., Garcia-Vargas M. (2000b). Environmental impacts of intensive aquaculture in marine waters. Water Research 34, 334–342.

Troell M. (1997). Aquaculture: Searching for footprints. Samudra, Report No. 17, 26–28.

Troell M., Joyce A., Chopin T., Neori A., Buschmann A. H., Fang J. G. (2009). Ecological engineering in aquaculture - Potential for integrated multi-trophic aquaculture (IMTA) in marine offshore systems. Aquaculture 297 (1–4), 1–9.

Tsagaraki T. M., Petihakis G., Tsiaras K., Triantafyllou G., Tsapakis M., Korres G., Kakagiannis G., Frangoulis C., Karakassis I. (2010). Beyond the cage: Ecosystem modelling for impact evaluation in aquaculture. Ecological Modelling 222 (14), 2512–2523.

UNEP/RIVM (1994). An Overview of Environmental Indicators: State of the art and perspectives. UNEP/EATR.94-01, RIVM/402001001, Nairobi, Kenya.

Valdemarsen T. B., Kristensen E., Holmer M. (2009). Metabolic threshold and sulfide-buffering in diffusion controlled marine sediments impacted by continuous organic enrichment. Biogeochemistry 95 (3), 335–353.

Valiela I., Bowen J. L., York J. K. (2001). Mangrove forests: One of the world’s threatened major tropical environments. BioScience 51, 807–815.

van de Nieuwegiessen P. G., Olwo J., Khong S., Verreth J. A. J., Schrama J. W. (2009). Effects of age and stocking density on the welfare of African catfish, *Clarias gariepinus* Burchell. Aquaculture 288, 69–75.

van der Loeff M., Anderson L. G., Hall P. O., Iverfeldt A., Josefson A., Sundby B., Westerlund S. (1984). The asphyxiation technique: An approach to distinguishing between molecular diffusion and biologically mediated transport at the sediment-water interface. Limnology and Oceanography 29 (4), 675–686.

van der Molen D. T. (1991). A simple dynamic model for the simulation of the release of phosphorus from sediments in shallow eutrophic systems. Water Research 25, 737–744.

van der Veer H. W., Cardoso J. F. M. F., van der Meer J. (2006). The estimation of DEB parameters for various Northeast Atlantic bivalve species. Journal of Sea Research 56, 107–124.

van der Veer H. W., Cardoso J., Peck M., Kooijman S. A. L. M. (2009). Physiological performance of plaice *Pleuronectes platessa* (L.): A comparison of static and dynamic energy budgets. Journal of Sea Research 62 (2-3), 83–92.

van der Veer H. W., Kooijman S. A. L. M., van der Meer J. (2001). Intra- and interspecies comparison of energy flow in North Atlantic flatfish species by means of dynamic energy budgets. Journal of Sea Research 45 (3-4), 303–320.

van der Veer H. W., Kooijman S. A. L. M., van der Meer J. (2003). Body size scaling relationships in flatfish as predicted by Dynamic Energy Budgets (DEB theory): implications for recruitment. Journal of Sea Research 50, 255–270.

van Raaphorst W., Kloosterhuis H. T. (1994). Phosphate sorption in superficial intertidal sediments. Marine Chemistry 48, 1–16.

van Raaphorst W., Kloosterhuis H. T., Berghuis E. M., Gieles A. J. M., Malschaert J. F. P., Van Noort G. J. (1992). Nitrogen cycling in two types of sediments of the southern north sea (frisian frint, broad fourteens): field data and mesocosm results. Netherlands Journal of Sea Research 28 (4), 293–316.

Vassallo P., Doglioli A. M., Rinaldi F., Beiso I. (2006). Determination of physical behaviour of feed pellets in Mediterranean water. Aquaculture Research 37 (2), 119–126.

Verbeke W., Vermeir I., Brunsø K. (2007). Consumer evaluation of fish quality as basis for fish market segmentation. Food Quality and Preference 18 (4), 651–661.

Via J. D., Villani P., Gasteiger E., Niederstätter H. (1998). Oxygen consumption in sea bass fingerling *Dicentrarchus labrax* exposed to acute salinity and temperature changes: metabolic basis for maximum stocking density estimations. Aquaculture 169, 303–313.

Viadero Jr. R. C., Cunningham J. H., Semmens K. J., Tierney A. E. (2005). Effluent and production impacts of flow-through aquaculture operations in West Virginia. Aquacultural Engineering 33, 258–270.

Wallin M., Håkanson L. (1991). Nutrient loading models for estimating the environmental effects of marine fish farms. Marine Aquaculture and Environment. Nordic Council of Ministers, Copenhagen, Denmark, pp. 39–57.

Wang H., Appan A., Gulliver J. S. (2003). Modeling of phosphorus dynamics in aquatic sediments: I - model development. Water Research 37, 3928–3938.

Watson-Capps J., Mann J. (2005). The effects of aquaculture on bottlenose dolphin (*Tursiops* sp.) ranging in Shark Bay, Western Australia. Biological Conservation 124, 519–526.

Wearing S., Neil J. (1999). Ecotourism: Impacts, potentials and possibilities. Butterworth-Heinemann, Oxford, United Kingdom, 163 pp.

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.

Wilson J. G., Brennan M. T. (2004). Spatial and temporal variability in modelled nutrient fluxes from the unpolluted Shannon estuary, Ireland, and the implications for the microphytobenthic productivity. Estuarine Coastal and Shelf Science 60, 193–201.

Wing-Keong N. G., Kim-Sun LU, Roshada H. and Ahyaudin A. (2000). Effects of feeding rate on growth, feed utilization and body composition of a tropical bagrid catfish. Aquaculture International 8, 19–29.

World Bank (2006). Aquaculture: Changing the Face of the Waters Meeting the Promise and Challenge of Sustainable Aquaculture. The World Bank, Washington, United States of America, 138 pp.

Worm B., Barbier E. B., Beaumont N., Duffy J. E., Folke C., Halpern B. S., Jackson J. B. C., Lotze H. K., Micheli F., Palumbi S. R., Sala E., Selkoe K. A., Stachowicz J. J., Watson R. (2006). Impacts of biodiversity loss on ocean ecosystem services. Science 314, 787–790.

Worsfold P. J., Monbeta, P., Tappin A. D., Fitzsimons M. F., Stiles D. A., McKelvie I. D. (2008). Characterisation and quantification of organic phosphorus and organic nitrogen components in aquatic systems: A Review. Analytica Chimica Acta 624, 37–58.

Wu R. S. S. (1995). The environmental impact of marine fish culture: towards a sustainable future. Marine Pollution Bulletin 31, 159–166.

Xinglong J., Boyd C. E. (2006). Relationship between organic carbon concentration and potential pond bottom soil respiration. Aquacultural Engineering 35 (2), 147–151.

Yan J., Ma S. (1991). The function of ecological engineering in environmental conservation with some case studies from China. In: Etnier C., Guterstam B. (Eds.) Ecological engineering for waste water treatment, Proceedings of the International Conference on the Ecological Engineering for Wastewater Treatment, 24–28 March 1991, Gothenburg, Sweden, pp. 80–94.

Yi Y. (1998). A bioenergetics growth model for Nile tilapia (*Oreochromis niloticus*) based on limiting nutrients and fish standing crop in fertilized ponds. Aquacultural Engineering 18, 157–173.

Yokoyama H., Takashi T., Ishihi Y., Abo K. (2009). Effects of restricted feeding on growth of red sea bream and sedimentation of aquaculture wastes. Aquaculture 286, 80–88.

Youngson A. F., Dosdat A., Saroglia M., Jordan W. C. (2001). Genetic interactions between marine finfish species in European aquaculture and wild conspecies. Journal of Applied Ichthyology 17 (4), 153–162.

Zar J. H. (1999). Biostatistical Analysis. Prentice-Hall, New Jersey, United States of America, 663 pp.

Zhou Y., Yang H., Hu H., Liu Y., Mao Y., Zhou H., Xu X., Zhang F. (2006). Bioremediation potential of the macroalga *Gracilaria lemaneiformis* (Rhodophyta) integrated into fed fish culture in coastal waters of north China. Aquaculture 252, 264–276.

Zhou Z., Xie S., Lei W., Zhu X., Yang Y. (2005). A bioenergetic model to estimate feed requirement of gibel carp, *Carassius auratus gibelio*. Aquaculture 248, 287–297.

Zimmer E., Jager T., Ducrot V. (2011). Juvenile food limitation: Ecotoxicologists, be warned! In: Abstracts Book of the 2nd International Symposium on Dynamic Energy Budget Theory, Portugal, 13-15 April 2011, 32 pp.