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MONDEGO BASIN AND ESTUARY MANAGEMENT:
The role of ecosystem services (e)valuation for human wellbeing endeavour
Universidade de Coimbra

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The role of ecosystem services (e)valuation for human wellbeing endeavour

Doctoral dissertation in the scientific area of Biology (specialty Ecology), supervised by Professor João Carlos Marques and co-supervised by Professor Tiago Domingos, presented to the Department of Life Sciences of the Faculty of Sciences and Technology of the University of Coimbra.

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'...now I know that it is a rising, and not a setting, sun...'

(Benjamin Franklin)

CONTENTS

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… e apenas para finalizar …
'…Aqueles que passam por nós, não vão sós, não nos deixam sós. Deixam um pouco de si, levam um pouco de nós…' (Saint-Exupéry)
… a todos um sincero obrigado!

ABSTRACT

To assume that human wellbeing relies on the services provided by well-functioning ecosystems implies also to admit that changes in the ecological integrity and condition of an ecosystem can have direct and/or indirect impacts upon such wellbeing. The present thesis tried to explore these relationships selecting as case study an Atlantic estuarine ecosystem, the Mondego estuary. Three main objectives were pursued:

The first objective was to identify the most important factors influencing the endeavour of improving estuarine water quality, namely taking into account the ecosystem surrounding activities.

The second objective was to explore and illustrate the potential relationships between biodiversity and human wellbeing, looking at the responses of estuarine dynamics to disturbances, with emphasis on stability as a system's property.

Finally, the third objective was to outline a possible 'path to follow' to integrate ecological, economic and social aspects in estuarine ecosystems management, taking as example the selected case study.

To achieve the first objective, variations in the Mondego estuary water ecological quality status over a 16 years' period were analysed and interpreted, taking into account the way natural and anthropogenic pressures have driven the ecosystem, and the how the observed impacts influenced the uses attribute to it by society. This information was then used to perform a survey to local (estuarine area), regional (Lower Mondego area), basin (the whole Mondego catchment), and national populations, involving a total 1139 inquiries.
Results illustrated that both use and non-use values are reflected in respondents´ willingness to pay (WTP) to improve water quality, showing that: (1) Respondents presented a very high interest in seeing improvements implemented, but only around half were really willing to pay for it; (2) There was a strong relation between socio-economic respondents´ profile (e.g. income, education or number of household members) and WTP for environmental quality improvements; (3) The distance-decay effect and usage of the ecosystem had a significant influence on respondents´ WTP; (4) There was a substantial positive social awareness for environmental issues (e.g. anthropogenic drivers and pollution causes) and economic values (both marketable and non-marketable) attached to ecosystem services of aquatic ecosystems.
Water resources were therefore viewed as a functional component linking several interconnected ecosystems within the Mondego catchment area. Outcomes highlighted respondents' recognition that ecosystems assets are interrelated and that there is an overlap between functions (biodiversity indicators and system integrity) and benefits (activities enrolment) obtained from it. As a whole, the Mondego Estuary case study suggests that to guarantee a sustainable use/conservation of ecosystems and to improve the efficiency and straightforwardness of management actions, issues as population socio-economic profiles, distance from the resource, or even protest answers must be taken into consideration by water authorities.

To pursue the second objective a conceptual framework was developed, interrelating biodiversity indicators and human well-being via ecosystem functioning and ecosystem services provision assessments.

Two steps were considered. In a first step, the Mondego Estuary stability was examined by measuring temporal stability (TS), and inquiring if this ecosystem's property could be related with species number (S) and abundance (N), different habitats within the ecosystem, disturbances effects, or even estuarine services provision. TS maximum values were achieved at an intermediate diversity range and TS increased with species abundance, suggesting that TS might be useful to address sustainable management of estuarine systems.

In a second step, to integrate the outcomes of TS analysis, a conceptual framework was proposed to assess the links between biodiversity and human well-being in a spatial and temporal explicit pattern. This framework relies on three consecutive steps and discriminates biodiversity structural components (expressed by 7 indicators), ecosystem functioning (expressed by 2 proxies), ecosystem stability (expressed by 1 proxy), and services provision (estimated by 6 indicators). Main outcomes highlighted that linear relationships between biodiversity, ecosystem functioning, and services provision are unlikely to occur in estuarine systems. Instead, cumulative and complex relations are observed between factors at both temporal and spatial scales.

Regarding the third and last objective, an integrative analysis of current ecological and socio-economic approaches was carried out, taking into account their inherent advantages and constrains. The goal was to identify a possible integrative ecosystem approach (EA) to be adopted, susceptible of allowing decision-makers to perform balanced and sustainable management decisions. There are several socio-economic methods that can be applied to value ecosystem services, but some difficulties may be found when integrating the information provided by these methods with biological information (usually expressed by values of ecological indicators) into the decision framework. The habitat level is proposed as a possible adequate scale to deal with such integration of information, because ecological functioning (natural compartment), human activities (economic compartment) and ecosystem services to society (socio-ecological compartment) can be chosen and measured. Suitable integral indicators or indices matching EA, and thus covering both ecological and socio-economic aspects are nevertheless required.

Socio-ecological integration can further be facilitated by the use of approaches such as Ecological Network Analysis (ENA), Driver-Pressure-State-Impact-Response (DPSIR) and economic valuation methods. The objective is to develop a 'tailor made' monitoring programme covering ecological and socio-economic aspects in sufficient detail. Such programme will be essential to guarantee the desired and required progress in valuing, weighing, and deciding upon how to proceed to attain sustainable development in practice.

RESUMO

Ao assumirmos que o bem-estar humano depende dos serviços proporcionados pelo funcionamento dos ecossistemas estamos, implicitamente, a admitir que alterações na condição e integridade ecológica dos mesmos pode ter um impacto, directo e/ou indirecto, sobre esse bem-estar. Esta tese tenta precisamente explorar esta dicotomia entre ecossistemas e bem-estar humano, selecionando como caso de estudo o estuário do Mondego. Para tal, foram identificados três objectivos gerais, que, quando definidos em conjunto, permitirão adoptar uma abordagem mais sustentável para o uso e usufruto dos recursos naturais:

O primeiro objectivo foi identificar os principais factores que influenciam a qualidade da água estuarina, incluindo nesta análise as actividades que decorrem em áreas circundantes e que podem influenciar a condição ecológica do sistema.
O segundo objectivo foi explorar as potenciais relações entre biodiversidade e bem-estar humano, dando especial enfâse às respostas dadas pelo sistema quando confrontado com perturbações, evidenciando a estabilidade do sistema como uma propriedade fundamental.
Finalmente, o terceiro objectivo foi delinear um possível 'caminho a seguir', que permita integrar aspectos ecológicos, económicos e sociais na gestão de sistemas estuarinos.

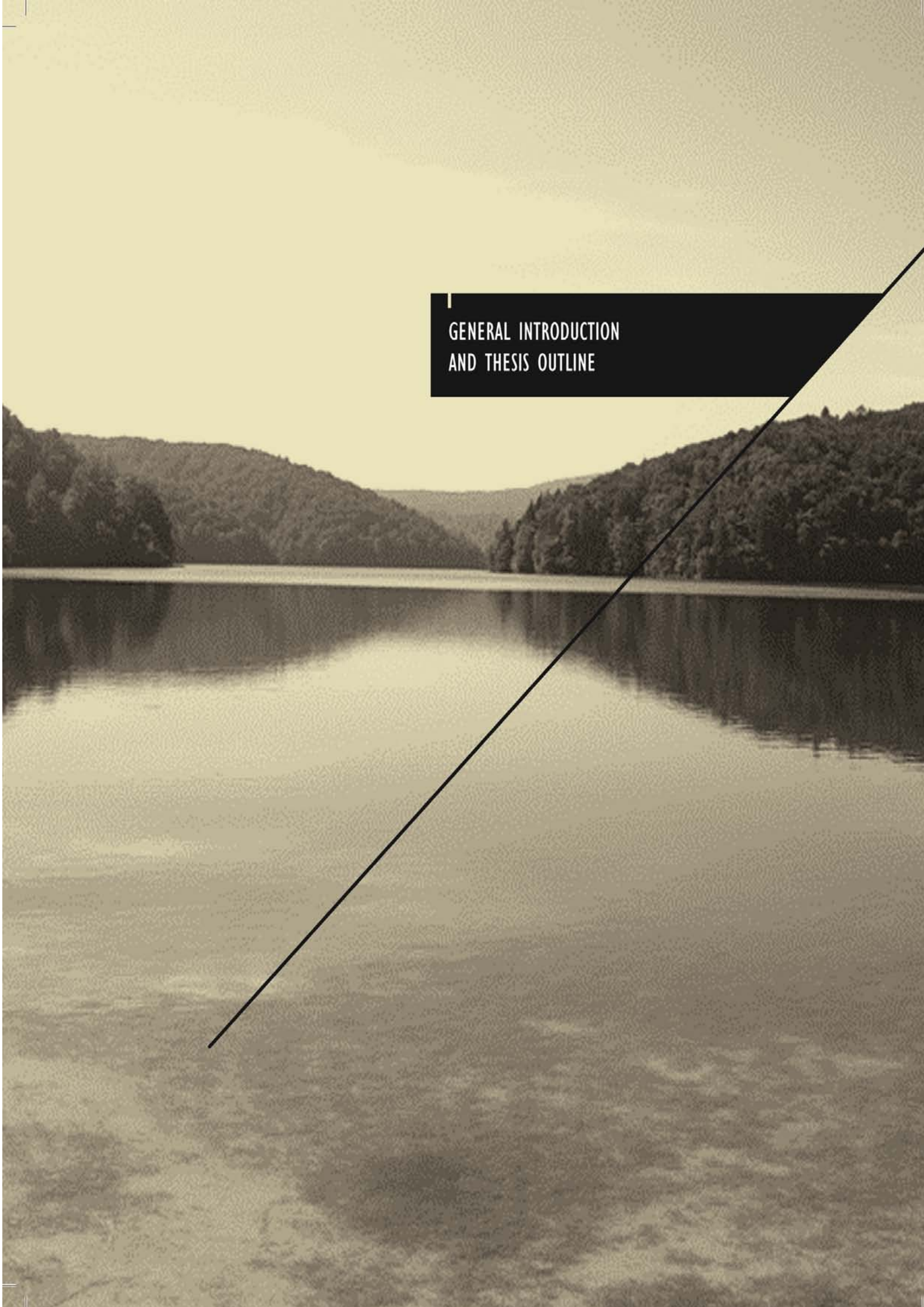
No quadro do primeiro objectivo, foram analisadas as variações no estado de qualidade ecológica do estuário do Mondego ao longo de 16 anos. Para tal teve-se em consideração a forma como as pressões naturais e antropogénicas moldavam o ecossistema e como os impactos daí resultantes influenciavam os usos do sistema pela sociedade em geral. Esta informação foi posteriormente integrada num inquérito, com o objectivo de determinar o valor que as populações locais (área circundante ao estuário), regionais (zona do Baixo Mondego), da bacia (compreendendo toda a bacia do Mondego) e nacionais (Portugal como um todo) atribuíam a uma melhoria do sistema. Tal implicou a realização de 1139 inquéritos. Os resultados demonstraram que a disponibilidade para pagar ("Willingness To Pay" - WTP) com o fim de melhorar a qualidade da água do sistema incorpora valores de uso e valores de não-uso, realçando que: (1) Os inquiridos mostraram um grande interesse em verem as medidas com vista a melhorias implementadas, embora apenas cerca de metade estivesse de facto disposta a pagar por isso; (2) Foi verificada uma forte relação entre a realidade socio-económica dos inquiridos (por exemplo, rendimento, escolaridade ou agregado familiar) e a disponibilidade para pagar pelas melhorias ambientais; (3) O factor 'distância' e o uso do sistema tiveram um efeito preponderante na disponibilidade para pagar (WTP) dos inquiridos; (4) Os inquiridos revelaram uma elevada consciencialização, quer para questões ambientais (reflectido no interesse em ver medidas implementadas que promovam melhorias ambientais), quer para reconhecer o valor económico dos serviços prestados pelo sistema (sejam eles transacionáveis, ou não, em mercados). Os recursos aquáticos foram assim reconhecidos pelos inquiridos como um componente essencial para o bom funcionamento do sistema, permitindo a inter-conectividade de vários ecossistemas dentro da bacia do Mondego. Através dos resultados obtidos foi possível verificar que existia um reconhecimento, por parte dos inquiridos, da inter-relação entre componentes dos ecossistemas e, também, que existe por vezes uma sobreposição entre o que são funções (indicadores de biodiversidade e integridade) e benefícios obtidos (actividades nas áreas circundantes). Desta forma, em tomadas de decisão, tendo em conta os

resultados obtidos a partir do caso de estudo do estuário do Mondego, torna-se imprescindível considerar factores como a realidade socioeconómica das populações que dependem (directa ou indirectamente) do ecossistema, a distância ao recurso e as respostas de protesto. Tal será de facto indispensável no sentido de garantir um uso/conservação sustentável dos recursos naturais e promover a eficiência de acções de gestão.

No quadro do segundo objectivo, foi adoptada uma abordagem conceptual, relacionando indicadores de biodiversidade e bem-estar humano, medidas através do funcionamento dos ecossistemas e da provisão de serviços. Para tal, foram consideradas duas etapas. Numa primeira etapa, foi examinada a estabilidade temporal ("Temporal Stability" - TS) do estuário do Mondego, com o objectivo de determinar se esta propriedade do sistema poderia estar relacionada com o número de espécies (S) e abundâncias (N), diferentes habitats dentro do sistema, efeitos de perturbações, ou até mesmo com a provisão de serviços. Os valores máximos de TS foram obtidos para uma gama intermédia dos valores de diversidade. Paralelamente verificou-se que a TS aumentava com o aumento da abundância das espécies, sugerindo que esta propriedade poderá ser útil para avaliar a gestão sustentável dos sistemas estuarinos. Numa segunda etapa, de forma a integrar os resultados da análise da estabilidade, foi proposta uma abordagem conceptual que pretendia avaliar os elos entre biodiversidade e bem-estar humano, numa perspectiva espácio-temporal. Esta abordagem é composta por três passos consecutivos e discrimina os componentes estruturais da biodiversidade (dados por 7 indicadores), do funcionamento dos ecossistemas (expresso por 2 indicadores), da estabilidade (dada por 1 indicador) e da provisão de serviços (estimada por 6 indicadores). Os principais resultados demonstraram que é improvável a ocorrência de relações lineares entre biodiversidade, funcionamento e provisão de serviços em sistemas estuarinos. Pelo contrário, verificam-se relações cumulativas e complexas entre factores e escalas (temporal e espacial).

No que respeita ao terceiro e último objectivo, foi realizada uma análise integradora de abordagens ecológicas e socioeconómicas, tendo em conta as suas vantagens e desvantagens inerentes. O objectivo consistia em identificar uma possível abordagem ecossistémica ("Ecosystem Approach" - EA) que pudesse ser adoptada em estudos integradores de ecossistemas, de forma a permitir a tomada de decisões sustentáveis e equilibradas. Existem vários métodos socioeconómicos que podem ser aplicados para atribuir um valor monetário aos serviços de ecossistema. No entanto, as dificuldades surgem quando se tenta integrar esta informação com a informação biológica (geralmente expressa por indicadores) numa tomada de decisão. O nível 'habitat' é proposto como a (possível) escala adequada para promover a integração de informação, uma vez que a esta escala podem ser escolhidos e medidos: 1) O funcionamento (compartimento natural), 2) Actividades humanas (compartimento socioeconómico) e 3) Serviços fornecidos pelo ecossistema à sociedade (compartimento socio-ecológico). No entanto, são ainda necessários indicadores ou índices integradores que vão de encontro às necessidades da EA, envolvendo quer os aspectos ecológicos quer os socioeconómicos.
A integração socioecológica pode ainda ser facilitada através do uso de abordagens como Ecological Network Analysis (ENA), Drivers-Pressões-Estado-Impactos-Respostas (DPSIR) e métodos de valoração económicos. O objectivo é então desenvolver um programa de monitorização 'feito à medida', integrando aspectos ecológicos e socio-económicos detalhados. Este tipo de

programas são essenciais para garantir o desejado e necessário progresso em valorizar, avaliar e decidir sobre qual a melhor forma de atingir um desenvolvimento sustentável dos recursos naturais.



GENERAL INTRODUCTION AND THESIS OUTLINE

'Ecology is a new fusion point for all sciences… The emergence of ecology has placed the economic biologist in a peculiar dilemma: with one hand he points out the accumulated findings of his search for utility, or lack of utility, in this or that species; with the other he lifts the veil from a biota so complex, so conditioned by interwoven co-operations and competitions, that no man can say where utility begins or ends.'

(Aldo Leopold, 1939)

General Introduction

1. Capturing ecosystems dynamics, complexity and resilience: which challenges to management?

According to the Millennium Ecosystem Assessment (hereafter MEA) findings, wetland ecosystems (including lakes, rivers, marshes and coastal regions with a depth of 6 meters at low tide) are estimated to cover more than 1,280 million hectares worldwide, although this figure may be underestimated. However, more than 50% of some types of wetlands (in parts of North America, Europe, Australia and New Zealand) were destroyed during the 20th century and others in many parts of the world were degraded (MEA, 2005). This situation has led to on-going efforts to preserve and restore wetland ecosystems, aiming at reverting this progressive loss and overexploitation of natural resources.

In this regard the first step consists in establishing the conceptual framework underlying any action, clearly defining the asset under evaluation, stating scales of action and key-determinant conditions of ecosystems. By definition, an ecosystem is '*a dynamic complex of plants, animals and micro-organisms communities and their non-living environment interacting as a functional unit*' (MEA, 2005). In this sense, the structure and functioning of an ecosystem is sustained by synergistic feedbacks between organisms and their environment (Costanza et al., 2001), determining its properties and setting limits to the types of processes occurring there (Mace and Bateman, 2011). However, in practical terms, due to their inherent heterogeneity and complexity, ecosystems are usually defined by the scope of the function, process or problem being studied (Mace and Bateman, 2011). Due to the complexity intrinsic to these interactions, as the physical, chemical and biological assets change, so will the processes underpinned by those components and, consequently, the functions and services delivered (Mace and Bateman, 2011). This problematic becomes even more pertinent when, allied to the persistent anthropogenic pressures, natural stressors are added, such as extreme events originated for instance from climatic changes. Moreover, ecosystem responses to environmental change may quite commonly be non-linear, difficult to predict or even irreversible (de Jonge, 2007; Carpenter et al., 2009). Therefore, it becomes essential to understand the basis of complexity in order to ensure the effectiveness of response actions.

The inherent ability of ecosystems to balance their internal functioning, when faced with perturbations, is due to their resilience (Holling, 1986; Gunderson and Holling, 2002). Resilience is the ability of a social-ecological system to undergo, absorb and respond to change and disturbance, while maintaining its functions and controls (Carpenter et al., 2001). Therefore, an ecosystem's resilience to changes may have a significant effect on important services provision. An illustrative example of this complexity (and of how physical, chemical and biological assets underpin ecosystem functioning) is for example the reduction, or even disappearance, of macrophytes or seagrasses caused possibly by competition with green macroalgae. These situations are often caused by nutrient enrichment of the water column, caused by anthropogenic activities, combined with high water residence times and good light conditions. The replacement of macrophytes or seagrasses habitats and communities can lead to changes in ecosystem functions and trophic structure (e.g. uncoupling of biogeochemical sedimentary cycles or even changes in the abundance of benthic fauna) (Valiela et al., 1997). Therefore, the main challenge for management is to strengthen the robustness and resilience of these systems and preserve their ability to provide ecosystem services for generations to come (Levin and Lubchenco, 2008).

2. What are and how to (e)valuate ecosystem services?

Definition and Identification

The term 'ecosystem services' was first introduced by Ehrlich et al. in the 1980's (Mooney and Ehrlich, 1997) since then the concept has suffered some evolutions. According to Daily (1997) '*ecosystem services are the conditions and processes through which natural ecosystems and the species that make them up sustain and fulfil human life. They maintain biodiversity and the production of ecosystem goods, such as seafood, forage, timber, biomass fuels, natural fiber and many pharmaceuticals, industrial products and their precursors*'. Following de Groot et al. (2002) ecosystem goods and biodiversity are an output of the inherent functioning of systems. However, there is no single way of describing ecosystem services. Further developments from Daily's definition involved, for example, the clear distinction between ecosystem goods and services, where goods were the 'materials produced' obtained from natural systems for human use (de Groot et al., 2002; Beaumont et al., 2007); proposals considering that ecosystem services '*are the aspects of ecosystems utilised (actively or passively) to produce human wellbeing*' (Boyd and Banzhaf, 2007; Fisher et al., 2009); or even definitions relying on the assumption that ecosystem services are 'the link between ecosystems and things that humans benefit from, not the benefits themselves' (Luisetti et al., 2010) where ecological phenomena encompasses both ecosystem structure and ecosystem processes/functions (Atkins et al., 2011; Mace and Bateman, 2011). Similarly, several proposals have been put forward to classify ecosystem services (e.g. de Groot et al., 2002; MEA, 2005; Beaumont et al., 2007; Luisetti et al., 2010; Atkins et al., 2011). Despite methodological criticisms (e.g. Wallace (2007, 2008) argues that MEA confuses 'ends' with 'means'), the MEA framework is the most widely recognised approach. This framework identifies four categories of ecosystem services: provisioning (e.g. food production), regulating (e.g. water purification), cultural (e.g. opportunities for recreation) and supporting (e.g. nutrient cycling) (MEA, 2005) and has demonstrated to be an efficient tool for both science and policy development (Mace and Bateman, 2011).

Common to all these definitions and approaches is the fact that they all rely and focus on human interests as a central issue, prevailing above other interests (e.g. ecological). In this thesis we assume that ecosystem services can be defined as the functions of ecosystems with value for human well-being (Fisher et al., 2009), i.e., we take an anthropocentric perspective. Thus, the concept of ecosystem services establishes a relationship between ecosystem service suppliers (the producers) and beneficiaries (the demanders) (Fisher and Turner, 2008). The framework adopted follows generally the methods and tools used in the MEA, while also incorporating more recent developments, especially for the valuation of ecosystem services to try to avoid some of the bias usually attributed to these methods (e.g. double counting issues) (Fisher and Turner, 2008; Turner, 2010).

Quantification

Ecosystems deliver services of great value to human society (Pearce and Moran, 1994; Costanza et al., 1997; Daily, 1997). However, continuous exploitation of natural resources, sometimes above their sustainability thresholds (e.g. intensive land and water use/contamination or extraction of natural resources) has led to a worldwide degradation of biodiversity assets and their associated services (Hooper et al., 2005). Approximately 60% of the ecosystem services evaluated in the MEA are being degraded or used unsustainably (MEA, 2005). Moreover, future scenarios, considering several indicators such as species extinctions, changes in species abundance, habitat loss and distribution shifts, seem to indicate that biodiversity will continue to decline over the 21st century (Pereira et al., 2010). As ecosystem services are not fully captured in markets or adequately valued in monetary terms, they are often taken for granted and are not fully included in policy decisions (Costanza et al., 1997; de Groot et al., 2002). Therefore, according to Fisher et al. (2009), the effective use of the ecosystem services concept in decision-making processes demands a clear understanding of the concept (definition and characteristics) and an appropriate way of dealing with it (classifications).

The crucial objective is then to assess pressures and interplays between human uses and natural resources, in order to inform accurately decision-makers and guide management and policy actions that will determine the future sustainability of systems. A more recent line has suggested the use of the ecosystem services approach as an intermediate link between natural ecosystem and human well-being (e.g. Perrings et al., 2009; Haines-Young and Potschin, 2010). For that several components must be considered: the inventory of provided services, their qualitative review, their quantitative assessment, and finally their monetarisation (Figure 1).

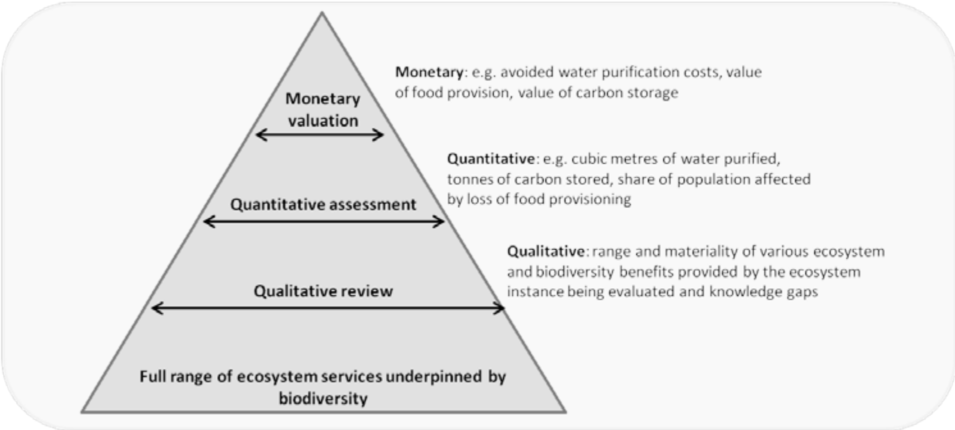


Figure 1. Example of ecosystem services valuation framework (ten Brink, 2008; *cit. in* TEEB, 2008).

It is therefore important not to limit assessments to monetary values, but also to include qualitative analysis and physical indicators (TEEB, 2008), in order to have a full picture of the system and allow that accurate 'fit-for-purpose' measures are developed and applied. Ecosystem services quantification demands a clear integration of the ecological (links between biodiversity assets, functions, intermediate and final services) and the economic perspectives (monetarization and ranking of ecosystem services). Figure 2 summarises this integration taking as example a coastal ecosystem.

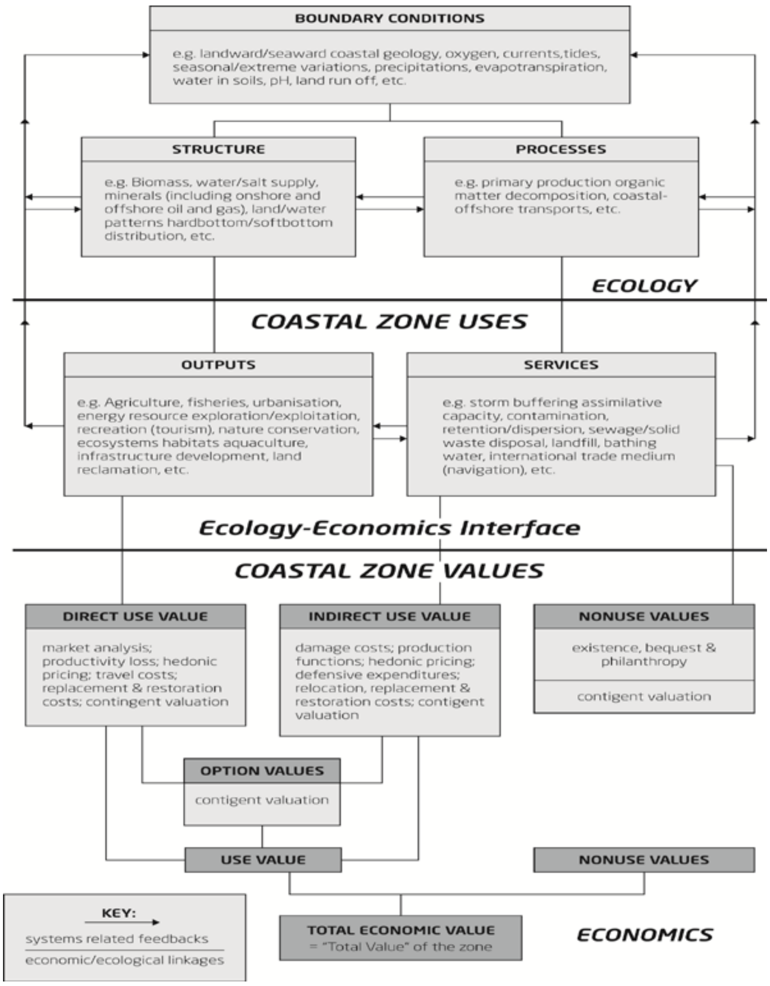


Figure 2. Multidisciplinary integration of functions, uses, and values when addressing the management of a coastal ecosystem (adapted from Turner, 2000).

Valuation

According to Turner (2000), at the core of this integrative framework is the concept of functional value diversity, which links ecosystem assets, processes and functions with outputs of ecosystem services, to which afterwards may be assigned monetary and/or other values. Therefore, the clear separation between ecosystem processes, intermediate services and final ecosystem services is necessary to avoid double counting when valuing the benefits derived from ecosystems (Fisher and Turner, 2008).

To the services provided by ecosystems is attached a value, either direct or indirect, attributed by society, through individuals' preferences. According to Barbier (2007), regardless of whether or not there is a market for ecosystems services, their social value must equal the discounted net present value (NPV) of ecosystem flows. However, the central issue is that ecosystems, and associated services, are generally intermediate inputs to the goods and services that enter in individuals' final demand (Barbier et al., 2009). In this context, the several components of the Total Economic Value approach (TEV) have been used to address ecosystems valuation (Pearce, 1993; Turner, 1999). The TEV approach comprehends the 'use' and 'non-use' values of a system. The use values involve the benefits deriving from consumptive or non-consumptive use by individuals and include three sub-classes of values (direct, indirect and option/ quasi-option), while non-use values comprise benefits from consumptive or non-consumptive use by others (e.g. preservation for future generations) (Barbier et al., 2009). There are various methods that can be used for valuing the services derived from ecosystems (e.g. Birol et al., 2006; Beaumont et al., 2007; Turner et al., 2010), which can be applied depending on the service and situation considered (Figure 3).

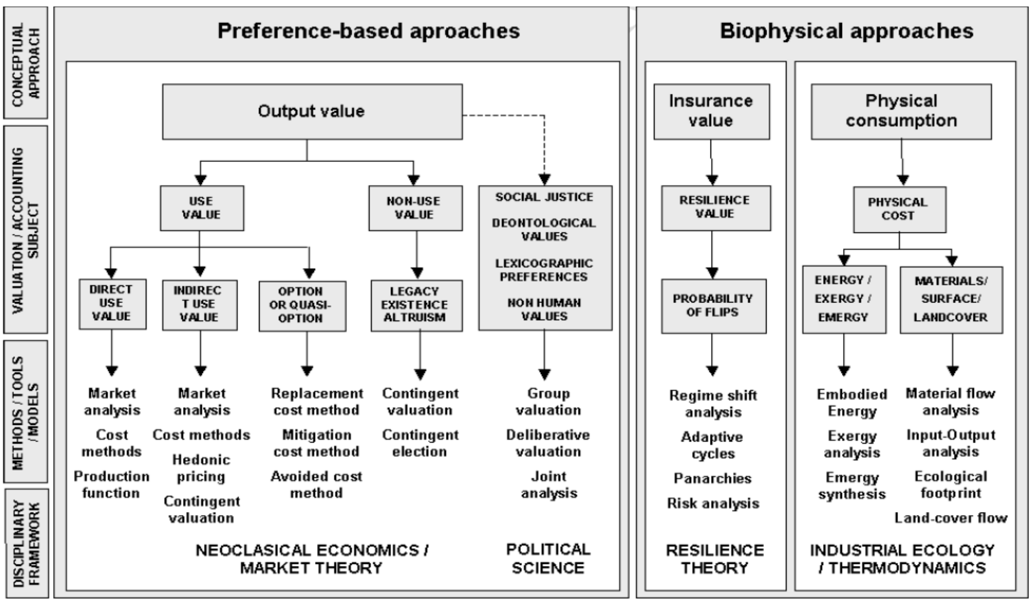


Figure 3: Total Economic Value components (Use and Non-Use values) and main economic methods possible to be used to assess an ecosystem value (after Gómez-Baggethun and de Groot, 2007).

When there is a market for a service, the market price can be used as an estimate of its social worth. However, for many services markets do not actually exist, services being provided at a null price, not reflecting their true social worth, or even sometimes the applied market price does not reflect society's true valuation (Barbier, 2007), both leading to market failure situations (Dasgupta, 2000). This is especially true for those environmental services fitting in the non-use (e.g. existence for future generations) or even in some indirect (e.g. climate regulation or biological productivity) and option (e.g. bequest or biodiversity insurance) values categories. In such cases, decision-makers try to correct the market failure by creating a market-like situation (Kumar, 2005), though the use of methods as the Travel cost, Hedonic price, Production function, Sated preferences methods, Replacement cost, among others (for more details see Freeman, 2003; Pagiola et al., 2004; Barbier et al., 2009). All methods present inherent advantages and constraints. However, among these, the stated preference methods (including, e.g., Contingent Valuation Method or Choice Experiment) are widely used to elicit non-use values, covering a wide range of benefits, including the existence or bequest values that individuals attach to ensuring that a well-functioning system will be preserved for future generations (Barbier, 2007).

3. Biodiversity, ecosystem functioning and services output: which role to social-ecological systems?

Biodiversity plays a key role on ecosystem services provision, since (Mace and Bateman, 2010):

a) Biological composition of ecosystems, measured by its biodiversity, is fundamental for the ecosystem processes that underpin ecosystem service delivery (Díaz et al., 2006), acting as an insurance value of the system (more diversity buffers systems against change; Hooper et al., 2005) and offering more options for the future (Yachi and Loreau, 1999);
b) Genetic and species biological diversity may directly supply some goods;
c) Many components of biodiversity are valued by people for altruistic reasons (e.g. appreciation of wildlife, contribution to spiritual or educational reasons and recreational experiences).

However, it is important to highlight that, biodiversity, *per se*, is not a service (Haines-Young and Potschin, 2010), although biodiversity conservation might be. Thus, when addressing natural resources management, the overarching question is 'how to deal with the relation between environmental quality/biodiversity assets and the provision of services?' Most of this discussion regards the links between biodiversity assets and ecosystem functioning and stability (that can be used as a proxy for the supporting services class from the MEA, or intermediate services). Several works have been conducted addressing this issue (e.g. Pimm, 1984; Schwartz et al., 2000; Loreau et al., 2001; Tilman et al., 2005; Balvanera et al., 2006), although controversy still remains. Some studies claim a positive relation between biodiversity and ecosystem functioning (e.g. Tilman et al., 2005; Balvanera et al., 2006), while other works state that it is difficult to establish a direct mechanism and quantification of this linkage (e.g. Schwartz et al., 2000; Nunes and van den Bergh, 2001; Worm et al., 2006). Nevertheless, common to all, biodiversity importance is '*argued to lie in its role in preserving ecosystem resilience, by underwriting the provision of key ecosystem functions over a range of environmental conditions*' (Perrings et al., 1995).

To address these complex relations, the Convention for Biological Diversity (2004) asks for a clear integration, under the Ecosystem Approach (EA), of all services provided to people, by biodiversity and ecosystems, into a holistic framework. Assuming the EA holistic perspective (Maltby, 2006; de Jonge, 2007), ecosystem services can act as the link between natural assets and human benefits. This approach defends an integration of the ecological, economic and socio-cultural perspectives when valuing a system (de Groot et al., 2002; Farber et al., 2002; MEA, 2005), providing a methodological framework for the integration of ecosystems management (de Jonge, 2007). In fact, ecosystem services clearly have an ecological and a socio-economic aspect, and it is their interdependencies that need to be clarified (Mace and Bateman, 2010). In this sense, ecosystem services research fits well with Berkes and Folke (1998) theory of social-ecological systems (SES), used to investigate the stability, resilience and vulnerability of ecosystems. The authors defined SES as an ecosystem '*that does explicitly include humans or, more specifically, the social system*' (Berkes and Folke, 1998).

4. How applicable are these concepts to estuarine ecosystems?

Estuaries are considered among the most valuable and productive ecosystems around the world. Costanza et al. (1997) estimated that these wetlands had an overall value of 22,832 \$ $ha^{-1}yr^{-1}$, although more recent studies conclude that this value may be much higher (Jørgensen, 2010). However, many of the world's largest urban areas (22 of the 32 largest cities) are located around estuaries (Ross, 1995) and globally around 71% of the world's coastal population is concentrated within 50 km of an estuary (Agardy et al., 2005). In Portugal, coastal areas account for only 8% of the continental area, although 76% of the population is concentrated in these areas (OECD, 2011), with a density which is twice as high as the average for continental Portugal (244.2 to 112.4 inhabitants'/km^{-2}; Pinho, 2007; OECD, 2011). Situations like the ones described may create massive pressures on natural resources (due to activities' expansion, development, nutrients inputs, among others), impacting around 90% of previously important species and destroying roughly 65% of seagrass and wetland habitats, while degrading water quality and accelerating species invasion in estuaries (Lotze et al., 2006; O'Higgins et al., 2010).

In this context the question that emerges is 'How to translate all the previous concepts into practice?' In implementing those concepts, taking estuarine ecosystems as real examples, we adopted a nine steps approach, which seeks to describe and detail, in a sequential way, the main key-issues behind ecosystems sustainable balance (Figure 4). The adopted framework aimed at structuring and evaluating current environmental problems, while simultaneously addressing existing research questions. In a time where the depletion of natural resources is attaining dramatic levels, there is also a growing recognition of the utmost importance of preserving and conserving these same assets on which human well-being relies (MEA, 2005).

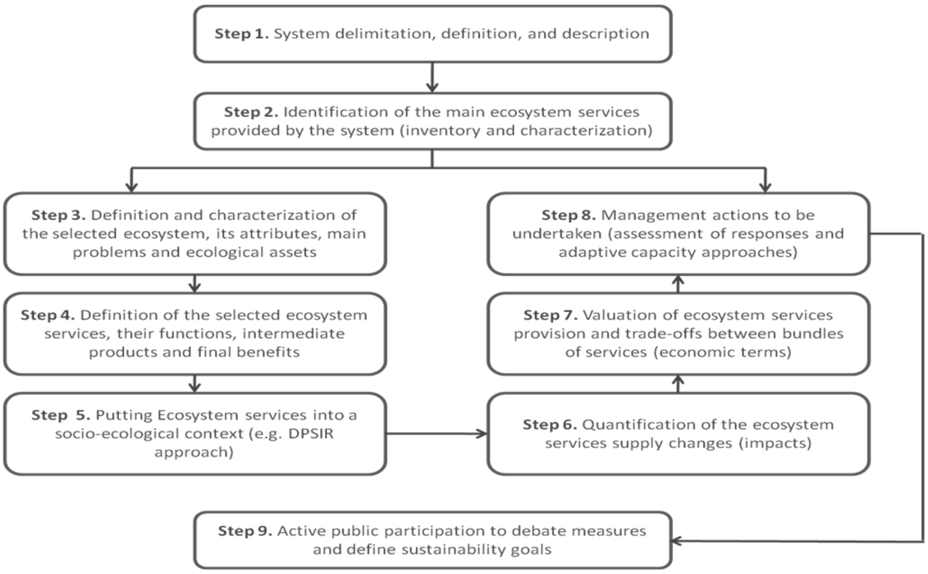


Figure 4: Stepwise implementation for the adopted conceptual framework used to assess estuarine ecosystem services.

5. Objectives and thesis outline

When conducting environmental (e)valuation studies it becomes essential to establish an accurate understanding of the ecological, economic and social consequences of biodiversity loss and change. This thesis presents the assessment of estuarine services and goods provision, attempting to integrate not only biological and functional data, but also social and economic information on the Mondego catchment area and, more specifically, on the estuarine part of the system. To achieve this, several ecosystem components and processes were taken into account and discussed. The final outcome tries to cover eight main questions:

1) Which ecosystem services, and at what rates, are provided by estuarine ecosystems? (Chapter 1)
2) What are the main drivers (D) and pressures (P) driving the system status (S), the main impacts (I) on human well-being and the possible responses (R) that may be given? (Chapter 2)
3) Is the DPSIR framework a valid tool for estuarine assessment and management? (Chapter 2)
4) What is the economic value attributed (WTP) to water resources from the Mondego catchment area by society? (Chapter 3)
5) How can the information achieved from contingent valuation surveys help the design of management strategies and policies for the sustainable use of river catchments/estuarine ecosystems? (Chapter 3)
6) Estuarine complexity and services provision: How does system stability change with distinct structural proxies' variations? (Chapter 4)

7) How to assess the link between estuarine components (biodiversity assets), processes (ecosystem functioning), uses (ecosystem services supply), and benefits (human well-being)? (Chapter 5)
8) Managing ecosystems: how to ensure the accurate spatial and temporal integration of ecological, economic and social demands? (Chapter 6)

This thesis aimed at synthesising current understanding on the ecosystem services approach with regard to 1) the assessment and protection of estuarine ecosystem structure and functioning; 2) the influence of the adverse effects of human activities on natural resources in terms of structure and function; and 3) how this information could be used as an added value for decision-making (Figure 5).

This general introduction deals with the concepts and methodologies underlying the ecosystem services approach and their relations to estuarine dynamic functioning. The next six chapters explore the eight main questions. A final discussion examines the use and misuse of the ecosystem services framework as part of decision support systems for estuarine systems, trying to provide a general integration of the different chapters.

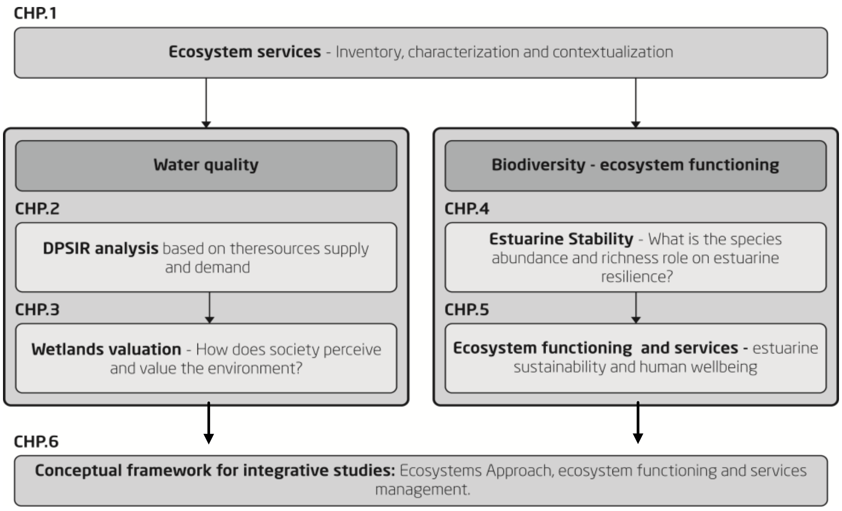


Figure 5: Thesis conceptual structure.

The six core chapters of the thesis are organised as follows:

Chapter 1 – In this chapter, the concept of estuarine ecosystem functioning/health is introduced, delineating the conceptual and theoretical framework of our understanding of estuarine dynamics, under the ecosystem services approach. A preliminary assessment of the services provided by the Mondego Basin is carried out, having as bottom-line the system's ecological quality. The ecological, economic, and societal relations of the Mondego estuarine services are analysed, and an inventory of the main ecosystem services provided by the Mondego system is provided. Three services of this inventory were considered has having a prominent role in the ecosystem (food production, recreation and water quality maintenance), and therefore their conditions and trends were determined. In this analysis special attention was given to the scale dependence effect, and the assessment was analysed at three progressively smaller scales: Mondego Basin, Lower Mondego, and Mondego Estuary. The next steps involved the assessment of services interdependence and an ecological assessment regarding water quality and ecological conditions of the system. In general, from 1992 to 2006, there was an increase in recreation activities and water uses and a simultaneous decrease in services such as food production, where a strong interdependence among services was verified. In the light of these findings, ecological quality improvement is reflected in both local communities' diversity and water quality. Despite the attempt to value the main services, the Mondego basin full value could not be calculated. Uncertainties and shortcomings regarding the reliability of this kind of assessment for implementation on estuarine ecosystems are discussed. After this exercise, two main branches were identified as priority to cover in depth the capacity of the system to contribute to human wellbeing:

1) Water quality improvements (chapters 2 and 3); and
2) Biodiversity-ecosystem functioning (chapters 4 and 5).

Chapter 2 – This chapter explores the overall economic efficacy of competing uses of estuarine resources by integrating ecological value, water uses and ecosystem services into the DPSIR conceptual framework as an added value for policy making and management. The complex interactions between the socioeconomic system and the ecosystem (as part of the 'integral system' as suggested by de Jonge, 2007) require generic but still 'tailor made' techniques to quantify all relevant variables and to provide an integral view of the system's status. One of the few techniques that can assist in structuring such complex data in an integrative way is the Drivers-Pressures-Status-Impacts-Responses (DPSIR) approach. Support and regulatory services (such as water supply and water quality) are essential to sustain crucial ecosystem processes and functions while the water required for human activities (water demand) is an essential system service. With the help of DPSIR, the main changes in the Mondego Estuary ecosystem were outlined, and causes and effects described. Within the Mondego Basin region the main water consumers are agriculture, industry, and households. Baseline scenarios predict an increase in water usage, mainly by the touristic service sector. Our analysis illustrates that pressures caused by human population growth and related activities gradually increased over the studied period. Land-use patterns, diversion of freshwater flows, water pollution and morphological interventions directly caused physical, chemical, and biological modification and degradation. Consequently, this led to negative ecological and socio-economic impacts, such as eutrophication. The scenarios suggest an increased pressure based on an expected 8% annual population growth and an average annual decreased pressure of 5.2% due to the current reduction in agriculture. The results show that understanding the water use-related complex and intricate trade-offs among ecological, social, and economic goals is fundamental in designing and implementing management policies and ecosystems restoration schemes.

Chapter 3 – In the line of the previous chapter, the next step was to evaluate the awareness of individuals through a contingent valuation (CV) study, estimating the benefits of water quality improvements in the Mondego Basin watershed, and to examine how information from CV surveys may help the design of efficient and effective management policies. Two CV surveys (one considering the whole Mondego basin and another considering just the estuarine part) were undertaken. Respondents' perceptions and willingness to pay (WTP) regarding water quality improvements and an ecotourism facility were evaluated. This valuation approach was tested along four geographical sampling levels, to infer the distance-decay effect of respondents (distance from their residence) and the goods elicited. Estimates indicate a WTP around 30€/year per household to achieve a very good water quality status and an ecotourism centre development, with somewhat lower values to achieve good (around 10€/year) or very good (approximately 20€/year) water quality levels. Our findings identify that both use and non-use values are reflected in respondents´ WTP, showing: (1) a strong relation between socio-economic respondents´ profile (e.g. income, education or number of household members) and WTP for environmental quality improvements; (2) that the distance-decay effect and usage of the system had a significant influence on respondents´ WTP; (3) a substantial positive social awareness for environmental issues (e.g. anthropogenic drivers and pollution causes) and economic values (both marketable and non-marketable) attached to ecosystem services of aquatic resources. Results achieved allowed making a set of recommendations for decision-makers towards system conservation and management, acting as an effective instrument to support already existing EU policies, namely the Water Framework Directive (WFD).

Chapter 4 – Stability is thought to rely on the richness (identity) and abundance of the species present in an ecosystem, where higher biodiversity promotes higher stability. Several attempts have been done to test this connection; however, there is still a lack of comprehension regarding the relation between biodiversity and stability. The stability concept is a collective term, defined via three fundamental properties: constancy, resilience and persistence. This manuscript uses theoretical and experimental evidence to explore the effects of biodiversity on estuarine stability, using the temporal stability (TS) measure as a proxy. In 1999 Tilman proposed the use of TS to test the diversity-stability hypothesis in a decade-long grassland experiment. Can TS be useful in estuarine systems? Our approach attempted to analyze estuarine stability from complementary perspectives by allowing the measurement of stability change (i) depending on species number and abundance; (ii) according to the different habitats of the same system; and (iii) disturbances influence. The question that imposed next was if this system property could be related with estuarine services provision. From this study, the main outcomes were that different TS values were found for the same abundance (N) values and the same was observed for species richness (S); TS maximum values were achieved at an intermediate diversity range; and TS increased with species abundance. In general, our results suggest that temporal stability might be useful to address sustainable management of estuarine systems.

Chapter 5 – Assuming that human well-being relies on the services provided by well-functioning ecosystems, changes in the ecological functioning of a system can have direct and indirect effects on human welfare. Intensive land use and tourism have expanded in recent decades along coastal ecosystems, together with increasing demands for water, food and energy; all of these factors intensify the exploitation of natural resources. Nevertheless, many of the interrelations between ecosystem functioning and the provision of ecosystem services still require quantification in estuarine ecosystems. A conceptual framework to assess such links in a spatially and temporally explicit pattern is proposed. This framework relies on three consecutive steps and discriminates among biodiversity structural components (measured by 7 indicators), ecosystem functioning (represented by 2 proxies) and stability (calculated by 1 proxy) and the services provided by the ecosystem (estimated by 6 indicators).

Abiotic factors and natural disturbances were found to have a direct effect on biodiversity, ecosystem functioning and the provision of ecosystem services. The observed changes in the species composition of communities had a positive effect on the ecosystem's productivity and stability. Moreover, the observed changes in the estuarine ecosystem services (ES) provision are likely to come from changing structural and abiotic factors and from the loss or decline of locally abundant species. This study also indicates that linear relationships between biodiversity, ecosystem functioning and services provision are unlikely to occur in estuarine systems. Instead, cumulative and complex relations are observed between factors on both temporal and spatial scales. In this context, the results suggested several additional conclusions: 1) these interactions need to be incorporated into decision-making processes aimed at the conservative management of systems; 2) the institutional use of previous research results must be part of the design and implementation of sustainable management activities, integrating biodiversity indicators and ecosystem services that are important for human well-being; and 3) more integrative tools/studies are required to account for the interactions of estuarine ecosystems with surrounding socio-economic activities. Therefore, when performing integrated assessments of ecosystem dynamics, it becomes essential to consider not only the effects of biodiversity and ecosystem functioning on services provision but also the effects that human well-being and ES provision may have on estuarine biodiversity and ecosystem functioning.

Chapter 6 – A conclusive synthesis of the insights on ecosystem management is presented, aiming to integrate ecological, economic and social concepts for the sustainable use of transitional/coastal systems in general, and estuarine environments in particular. If we as scientists cannot decide upon what research, monitoring and technical tools should be used as a basis for policy making and management, then politicians and other decision makers will continue to follow the line of 'weak' sustainability (applying monetary substitution rules to natural capital) instead of 'strong' sustainability (applying alternative rules like the precautionary principle). Suitable integral indicators or indices covering ecological as well as socio-economic aspects are thus required. There is, however, a clear friction between what can be delivered in terms of useful '(integral) indicators' and what decision makers require us to deliver in terms of 'simple, cheap, easy to understand' while the real situation is extremely complex. This social, economic and ecological complexity has been an important impediment to the required technical co-operation between decision makers and the natural and social scientists since the publication of the Brundtland report. Given the panarchic character of natural systems, realistic base environmental indicators should be anchored on a thorough examination of the functioning and the structure of ecosystems and related integrated indicators instead of the use of dynamical models deficient in reducing the uncertainty as to future system behaviour, or selecting for 'cute and cuddling' icons of any ecosystem without knowing what they ecologically represent. The connection of the social and the ecological aspects in an integrated approach is thus pivotal to making sustainability as starting point a ‘reality’. To arrive at the required integration we propose that decision makers should stop asking for ‘simple’ environmental indicators and accept the complex reality that our environment represents. To achieve this we propose that they should buttress to make the Odum food web ideas functional by the application of ecological network analysis (ENA) at a scale where socio-economic and ecological information can be integrated, which is the ‘habitat’ level. At the habitat level ecological functioning (natural compartment), human activities (economic compartment) and ecosystem functions to humans (socio-ecological compartment) can be designated and measured. This process can further be facilitated by the use of the Driver-Pressure-State-Impact-Response (DPSIR) approach. To facilitate the weighing and decision support systems’ process we propose to apply multi-criteria techniques to integrate all the information. In practice the adaptive management process might be a suitable option. As a consequence it is crucial to define what to investigate, what to monitor and how this subsequently can be related to the relevant sectors of the economic part of the integral system to realize sustainability in line with the Brundtland Commission’s view.

CHAPTER ONE

Assessing estuarine quality under the ecosystem services scope: Ecological and socioeconomic aspects

Abstract

An increasing need for integrative assessments that measure the contributions of the environment to human welfare has recently been recognised. In the present study, a preliminary assessment of the services provided by the Mondego Basin in terms of system ecological quality was carried out. The ecological, economic, and societal relations of the Mondego estuarine services were analysed. An inventory of the main ecosystem services provided by the Mondego system was performed. The conditions and trends of the main services (food production, recreation and water quality maintenance) were determined, and the scale dependence of this assessment was interpolated on three different scales: Mondego Basin, Lower Mondego, and Mondego Estuary. The interdependence among services was quantified; an ecological assessment regarding water quality and ecological conditions was performed, and a preliminary valuation of the food production, recreation and tourism in the region was undertaken. In the study system, from 1992 to 2006, there was an increase in recreation activities and water uses and a simultaneous decrease in services such as food production (i.e., strong interdependence among services). Ecological quality improvement is reflected in both local communities' diversity and water quality. The market prices method was used to estimate the values for the three services considered; however, the Mondego catchment's full value cannot be calculated without estimating the real wetlands value because these are prone to underestimation. Uncertainties and shortcomings regarding the reliability of this kind of assessment for implementation on estuarine ecosystems are discussed.

Keywords:
| Ecosystem services
| Ecological assessment
| Ecosystem services interdependences
| Economic assessment
| Mondego River Basin

1. Introduction

Most of the ecosystem analysis approaches adopt an anthropogenic perspective where the stocks of natural assets found within the system are connected to the flow of services that provide benefits to human society (Costanza et al., 1997; Turner et al., 2000a; de Groot et al., 2002; Farber et al., 2002; MEA, 2005). By definition, ecosystem goods (e.g. food) and services (e.g. water purification), hereafter ecosystem services, represent the benefits human populations derive directly or indirectly from ecosystem functions (Costanza et al., 1997). As such, ecosystem services are generated by ecosystem functions, which in turn are underpinned by biophysical structures and processes inherent to the system (de Groot et al., 2010a). These concepts have been used by the Millennium Ecosystem Assessment framework (MEA, 2005) to represent the flow of benefits to society generated by natural ecosystems and their consequences for human well-being. According to this framework, four categories of services were established (MEA, 2005): provisioning (products obtained directly from the ecosystems); regulating (benefits obtained from the regulation of ecosystem processes); cultural (nonmaterial benefits people obtain from ecosystems through cognitive development and aesthetic experiences, for example); and supporting (those benefits that are necessary for the production of all other ecosystem services). Through the integration of the ecosystem's inherent processes, the associated biodiversity and its sustainable use, the ecosystem approach focuses on conserving natural systems for their inherent value and for human well-being (Vitousek et al., 1997; Nunes and van den Bergh, 2001; de Groot et al., 2010a). As so, it is necessary to identify all costs and benefits to the ecosystem of different human activities in order to protect the system's biodiversity and promote its sustainable use (Nunes and van den Bergh, 2001; Folke et al., 2004). This may be achieved through the identification of the impacts of human activities and through the quantification of their consequences for the supply of ecosystem services. When valuing a system, the intent is to provide a value for a specific asset, and to trace its condition and importance over time. This includes not only the ecosystem services that have a market price (e.g. agriculture products) but also services that currently have no market prices (e.g. disturbance regulation). According to Beaumont et al. (2006), there are two distinct approaches to working with the ecosystem services concept. Among economists, economic valuation methods prevail, which focus on the exchange values of ecosystem services (based on consumer preferences and cost-benefit analyses). On the other hand, there are ecological valuation methods that employ a more sustainability-oriented perspective, which are mainly advocated by natural scientists and ecologists, that derive ecological prices (measurement of biophysical units, rather than social or economic in nature) for ecosystem services via a cost-of-production approach (i.e., by modelling the interrelations between the biotic and abiotic components of a system). According to Costanza et al. (1997), both fields take into account concerns regarding the scale of the economy, distribution and efficient allocation of resources. Two scale-related problems are encountered when assessing ecosystem services (Heal and Kristrom, 2005): (i) the scale at which certain functions become important is not always the same and (ii) problems may arise when integrating and aggregating information at multiple scales where interrelations and feedback loops may operate at scales above the level being assessed. According to Limburg et al. (2002), scaling rules that try to describe the provision and delivery of ecosystem services have yet to be quantified and defined.

Using the Mondego estuary as a case study, we conduct a preliminary assessment of the services provided by the Mondego Basin under the constraint of the present limited data availability. Moreover, the aim of this work was to analyse the ecological/ economic/societal costs and benefits of estuarine ecosystem services, giving special attention to their quality status, along the lines of the ecological sustainability trigon approach proposed by Marques et al. (2009). Specifically, we aimed to accomplish the following: (i) provide a comprehensive inventory of the ecosystem services provided by the Mondego system; (ii) determine the conditions and trends of the main ecosystem services; (iii) interpolate the scale dependence of such assessments using three different spatial scales: the Mondego Basin, the Lower Mondego Valley, and the Mondego Estuary; (iv) estimate the interdependence among services; (v) perform an ecological assessment regarding water quality and ecological conditions; and (vi) perform a preliminary valuation of the food production, recreation and tourism in the region.

2. Methodology

A generic framework has been implemented, assuming several steps in the ecosystem services valuation, together with the methodological tools necessary to provide a more comprehensive assessment. First, the characterisation of the system’s condition and main uses was undertaken at different spatial scales. Second, an inventory of the main services provided by the Mondego Basin was carried out. It is important to note that biodiversity, despite not being considered an ecosystem service, was included in the assessment because it is assumed to play a necessary role in all of the considered services by promoting the correct performance of all ecosystem functions (Marques et al., 2009). Based on this inventory, the conditions and trends of the main considered ecosystem services were assessed and the interdependencies among them evaluated. The next step involved an ecological assessment of the main services and their relations with the biodiversity assets. Finally, the ecological valuation perspective was combined with a range of economic valuation methods (i.e., the economic perspective) in an attempt to provide a preliminary system valuation.

2.1. Spatial scales of the system and scale dependence

The Mondego Basin is located in the centre of Portugal and covers a 6,670 km^2 catchment area with highly diverse characteristics in terms of hydrology, land use, and topography. It presents a peculiar and unique functional structure ranging from mountainous areas to a large alluvial plain discharging into the Atlantic Ocean (Marques et al., 2003), supporting a population currently estimated at 885,561 inhabitants (2006 data). The system can be divided into three main regions (PBH Mondego, 2001; Figure 1):

i) Upper Mondego: basin area located in the 'Serra da Estrela' mountain range at the river headwaters, where it travels through glacial valleys;
ii) Middle Mondego: basin area between the base of the 'Serra da Estrela' and the city of Coimbra, where the river passes through deep valleys; and
iii) Lower Mondego: the final part of the river course, consisting of open valleys and plains, including the Mondego Estuary ecosystem and a thick dune belt along the coastline.

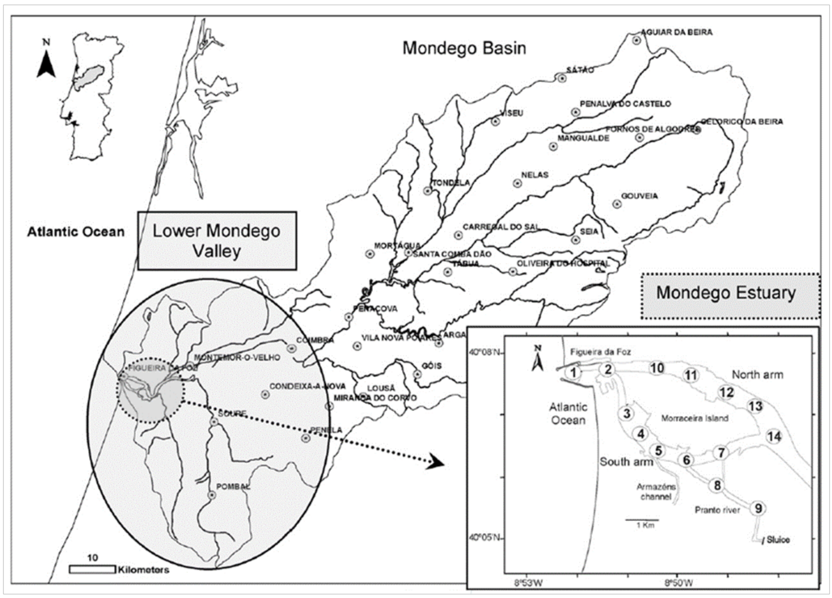


Figure 1: The three different study site scales: Mondego Basin; Lower Mondego Valley; and the Mondego Estuary (with subtidal sampling stations).

Overall, the Mondego Basin has a high natural variability in environmental and social conditions (Table 1). In each of the three main regions, the secondary and tertiary sectors are well represented in the economic activities. However, in the Lower Mondego, a strong pressure is also evident from the abundant agriculture fields (primary sector), as well as from Figueira da Foz harbour. In summary, we may say that at the Basin level industrial activities related to wood extraction (due to the vast forest area), together with the glass, ornamental resources and beverage industries, dominate the economic activities taking place in the system. More specifically, in the Lower Mondego region (near the coastal area), the paper industry and aquaculture play the largest economic roles. The fibre and leather industries have the dominant position among the economic activities in the Upper Mondego area (PBH Mondego, 2001). These variations influence the system's management, water uses, and land occupation rates. Under the Millennium Ecosystem Assessment (MEA) scope, three main areas at different scales were considered for this study (Figure 1): the Mondego Basin (basin scale), the Lower Mondego Valley (regional scale), and the Mondego Estuary (local scale). The scale dependency was mostly examined to infer the effects of upstream activities on local estuarine resources.

To this end, several parameters were taken into account, including not only socioeconomic factors such as demographic pressures or activities around the basin but also ecological factors such as nutrient sources.

Table 1: Main characteristics of the Mondego Basin, Lower Mondego and Mondego Estuary study sites (2006 data).

	Area (km)	Population (nº of ind.)	Small basins included	Land Use (ha)					
				RAN	REN	Urban	Industrial	Urban parks	Tourism
Portugal	92 391	10 599 095	-	x	X	481 082	75 151	37 837	18 707
Mondego Basin	6 645	885 561	9	27 983	466 482	77 560	9 965	3 154	1 209
Lower Mondego	250	334 161	3	x	X	23 078	3 098	1 404	724
Mondego Estuary	7.2	63 372	1	x	22 738	2 537	1 171	165	380

Sources: INE – Instituto Nacional de Estatística (National Institute of Statistics); INAG – Instituto da Água (Water Institute); RAN= Reserva Agrícola Nacional (National Agricultural Reserve); REN= Reserva Ecológica Nacional (National Ecological Reserve); x – data not available.

2.2. Comprehensive inventory of ecosystem services

The goal of the ecosystem services inventory was to provide a set of alternative ways to valuate estuarine services and thus provide insight into the economic perspective within the ecosystem approach. Several services provided by wetlands ecosystems have been identified (Costanza et al., 1997; Acharya, 2000; Atkins and Burdon, 2006). From this available set of services we considered two main factors that determine the Mondego Estuary services: the importance of its natural resources stock to local populations (i.e. estimation of their dependency upon the system) and the ecological importance of the system to the intrinsic biodiversity and human well-being (Figure 2).

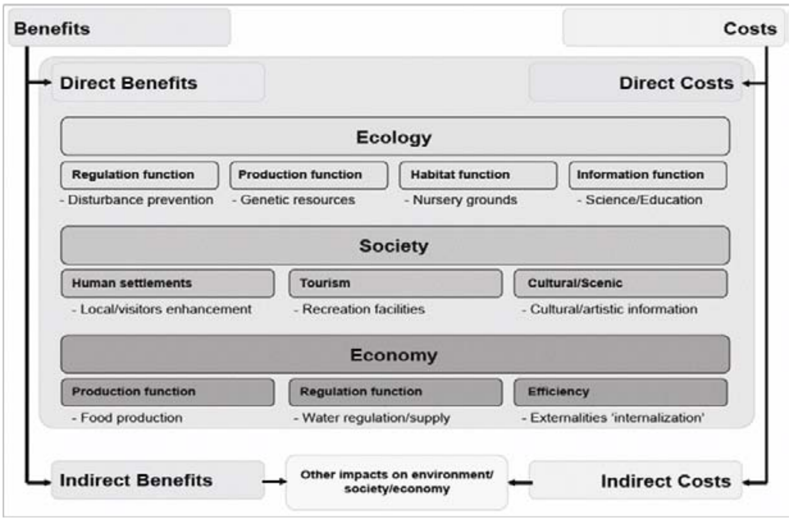


Figure 2: Ecological/Societal/Economic costs and benefits of estuarine services with particular emphasis on the system's quality status (de Jonge and Pinto, 2009, personal communication).

For the inventory assessment, several methods were considered, taking into account their specificities, to evaluate ecosystem services. In the total economic value (TEV) approach, ecosystem services can be divided into use values and non-use values (Loomis et al., 2000; Turner et al., 2000a; de Groot et al., 2002; Young, 2005a; Atkins and Burdon, 2006). Wetlands' non-use values (i.e., existence or bequest values) may be estimated through the use of contingent valuation methods (CVM). Within the wetlands' use values three main categories can be identified: direct use values, which include services such as food production and tourism; indirect use values, e.g. recreation or aesthetic values; and option values, where benefits come from ensuring that a resource will be available for future use. Usually, the direct use values can be calculated through methods such as market analysis prices (MP), productivity loss (PL), hedonic pricing (HP), travel cost (TC), replacement/restoration costs (RRC), or even CVM. The indirect use values can be estimated through such methods as the damage costs (DC), production function (PF), HP, RRC, or CVM. The option values can also be assessed through the use of the CVM technique. According to Seyam et al. (2001), the use value derived from a certain production or information function is calculated as the product of the marginal value of the function and the area of the wetland that contributes to the function. The assumption is that there is a linear relationship between the area of the system that contributes to a certain function and the use value delivered by that function.

2.3. Conditions and trends of main ecosystem services

For the conditions assessments, the evaluation was performed based on the analysis of secondary data, i.e. data collected from the literature (available statistics and studies) and preferably published by governmental institutes (National Institute of Statistics–INE; www.ine.pt), for the period of 1992–2006. For the trends analysis, we considered the evolution of the main services in the region between 1992 and 2006. The objective was to measure the productivity changes of an environmental resource to determine the actual benefit obtained so that it can be related to measures of human well-being.

2.4. Estimation of interdependencies between services

Comprehensive examples were supplied at both the regional and local scales. At the regional scale, the interaction and overlap between agricultural activities, water quality supply and biodiversity was considered, and at the local scale the interdependence between the four main assets (food production, recreation, water quality and biodiversity) was integrated (Table 2). Although this selection may seem limited, it was done for two main reasons: (i) these services have a greater economic or social importance for the region; and (ii) there were more available data for these services. The present assessment can thus act as a basis for more detailed and inclusive studies.

Table 2. Ecosystem services analysed and data availability for the three spatial scales.

	Ecosystem services	Mondego Basin	Lower Mondego	Mondego Estuary
Food Production	Salt			x
	Aquaculture			x
	Agriculture		x*	
	Fisheries			x
Recreation	Salt-works visits			x
Tourism	Tourists	x	x	x
	Establishments	x	x	x
Water	Quality			x
	Availability	x	x	x
	Effluents	x	x	x
	Treated	x	x	x
Biodiversity				x

* for the total central region.

To estimate the food production, several items were considered: agriculture, fisheries, aquaculture and salt production. For the Basin comparative approach, only agricultural production was considered. Unfortunately, agricultural data were only available for the central region of Portugal, which encompasses not only the Mondego Basin but also other locations. Nevertheless, it was taken as indicative of the trends and conditions of the area. The remaining items (fisheries, aquaculture, and salt production) were analysed only at the scale of the Mondego Estuary. For the touristic activity, the number of tourists and touristic capacity, proxied by the number of hotel beds, were used as indicators. The recreational activities occurring in the system were also considered. Here, the term ‘recreation’ refers to the benefits people enjoy from taking part in activities in the natural environment. Given the data problems, the result cannot be interpreted as a valid description of the area under investigation, but should rather be seen as a methodological exercise, an example of integrating different services into a valuation exercise. The indicators (agricultural production, fisheries captures, number of tourists, effluents produced, organic matter content, nutrient concentrations in the water column and ecological conditions) were selected as indicative measures through which it was possible to observe the conditions, trends and changes in the services under study. A Pearson correlation was performed to quantify the relations between services.

2.5. Ecological assessment

A large amount of information regarding the Mondego Basin’s physical structure and functioning is available in the literature (e.g. Marques et al., 1997, 2003, 2007; Graça et al., 2002; Feio et al., 2007; Flindt et al., 1997). Most studies have focused on the macroinvertebrate communities’ biotic integrity as well as on the water quality status, mainly in the scope of the Water Framework Directive (WFD) implementation (EC, 2000). To estimate biodiversity and water quality in the Mondego Estuary (local scale), the chosen dataset was provided by a programme monitoring the estuarine subtidal soft bottom communities. These data characterised the local system with regard to species composition/abundance and water and sediment physicochemical parameters. Samplings were carried out at 14 stations along the two estuarine arms during spring in 1990, 1992, 1998, 2000 and every year from 2002 to 2006: Euhaline estuarine sand (stations 1, 2 and 10), North Arm Polyhaline sand (stations 11 to 14), South Arm Polyhaline sand (stations 3 and 4), South Arm Polyhaline muddy sand (stations 5 to 9) (Teixeira et al., 2009) (Figure 1). For the biodiversity analysis, a 1 mm mesh screen was used to sieve the samples and the collected organisms were identified and counted. To estimate the ecological condition, the Benthic Assessment Tool (BAT) was applied (Teixeira et al., 2009) based on the ecological quality of benthic macroinvertebrates and following the reference conditions proposed for Portuguese transitional water bodies (Teixeira et al., 2009) (Table 3A). The water quality in the estuary was characterised by the concentrations of dissolved nutrients (nitrate-nitrogen, nitrite-nitrogen and phosphorus) in surface and bottom water samples (Strickland and Parsons, 1972; APHA, 1980). The assessment of nitrite+nitrate (mmol l^{-1}) and phosphate (mmol l^{-1}) levels followed the EEA proposal (EEA, 1999) (Table 3B) prepared by the European Topic Centre on Inland Waters (ETC/IW) for transition, coastal and marine waters. This methodology does not take into account the salinity gradient typical of transition systems, but in the absence of a better set of tools, we decided to use it to assess water quality in the Mondego Estuary (Figure 1).

Table 3. Reference conditions for benthic quality assessment (A) and water quality status (B).

A. High Statuses for the Margalef, Shannon-Wiener and AMBI indices used in BAT to assess the different estuarine stretches of Portuguese transitional water bodies (after Teixeira et al., 2009)

	Euhaline	Polyhaline Sand	Polyhaline Muddy
Margalef	5.0	4.0	3.0
Shannon-Wiener (bits/ind)	4.1	4.0	3.8
AMBI	0.8	1 – 1.5	2.4

B. European Environmental Agency criteria for assessing nutrient levels in transition, coastal and marine waters (EEA, 1999)

Quality Status	$NO_2^- + NO_3^-$ (µmol L^{-1})	PO_4^- (µmol L^{-1})
Good	<6,5	<0,5
Fair	6,5-9	0,5-0,7
Poor	9-16	0,7-1,1
Bad	>16	>1,1

2.6. Economic assessment

For the economic and social assessments, evaluations were performed based on the analyses of secondary data, i.e. data collected from the literature (available statistics and studies) and preferably published by governmental institutes (e.g. INE data) for the period of 1992–2006. In a preliminary step, an overview of the Basin conditions was assembled, integrating both social (total population and population density) and socioeconomic factors such as the Human Development Index (HDI; United Nations Development Programme, UNDP data), which is considered as a valuable indicator for population welfare development (Ambuj and Najam, 1998; Hoagland and Jin, 2008) and the population distribution by economic sector (primary, secondary and tertiary). In this particular case study, the market prices method (MP) was employed. This approach was used for services such as food production (agriculture, fisheries and salt outputs), tourism activities (occupation rates given by number of beds) and recreational purposes (salt-works visiting). The MP method uses market transactions as an indicator of value based on the fact that the value that people attribute to a commodity is reflected in its price. Through the estimation of market prices in relation to total production, it was possible to obtain the demand curves for the considered services. These functions provide the consumer surplus in a perfectly competitive market (Barbier et al., 1997; Lambert, 2003; Tietenberg, 2003). To obtain these curves, time-series data (1992–2006) on the quantities demanded at different prices were considered. On the other hand, for outdoor recreational uses of the Mondego Estuary, and more specifically visits to the traditional salt-works, the fees paid by visitors were used as indicator for the service value (data source: Sinergiae Ambiente), although it does not provide valid information regarding the service value. This approach relies on the premise that the time and expenses that people invest to visit an ecosystem represent the value that they give to the service enjoyed. However, to achieve a meaningful estimate the travel time (as opportunity cost) and the travel expenditures should also be included.

3. Results

3.1. Spatial scales of the system and scale dependence

The economic, social and ecological profiles of the Mondego's catchment area are shown in Table 1, including an overview of its main characteristics, proportions and land use distributions at the three assessed scales in 2006. Additionally, to obtain a general profile of the region for the full study period, the total population data (total number of individuals) and population density (inhabitants per km^{-2}) variations were analysed (Figures 3A and 3B) and compared with the HDI value for Portugal, an indicator of individuals' purchasing power and standard of living (Figure 3C). The distribution of the population among economic sectors (i.e. primary, secondary and tertiary) was considered to account for the importance of local resources for human well-being and quality of life. Included in the primary sector were those activities implying the direct use of natural resources, such as agriculture or fisheries. Extractive and transforming industries were included in the secondary sector and services provided by society to the local population (e.g. banks, transportation) were included in the tertiary sector (Figures 3D–F). Although the population distributions for the Mondego Basin as a whole and the Lower Mondego specifically do not show a constant pattern, it was possible to see that it was registered a decrease for both primary and secondary sectors, contrasting with the strong tertiary sector increase on the estuarine region.

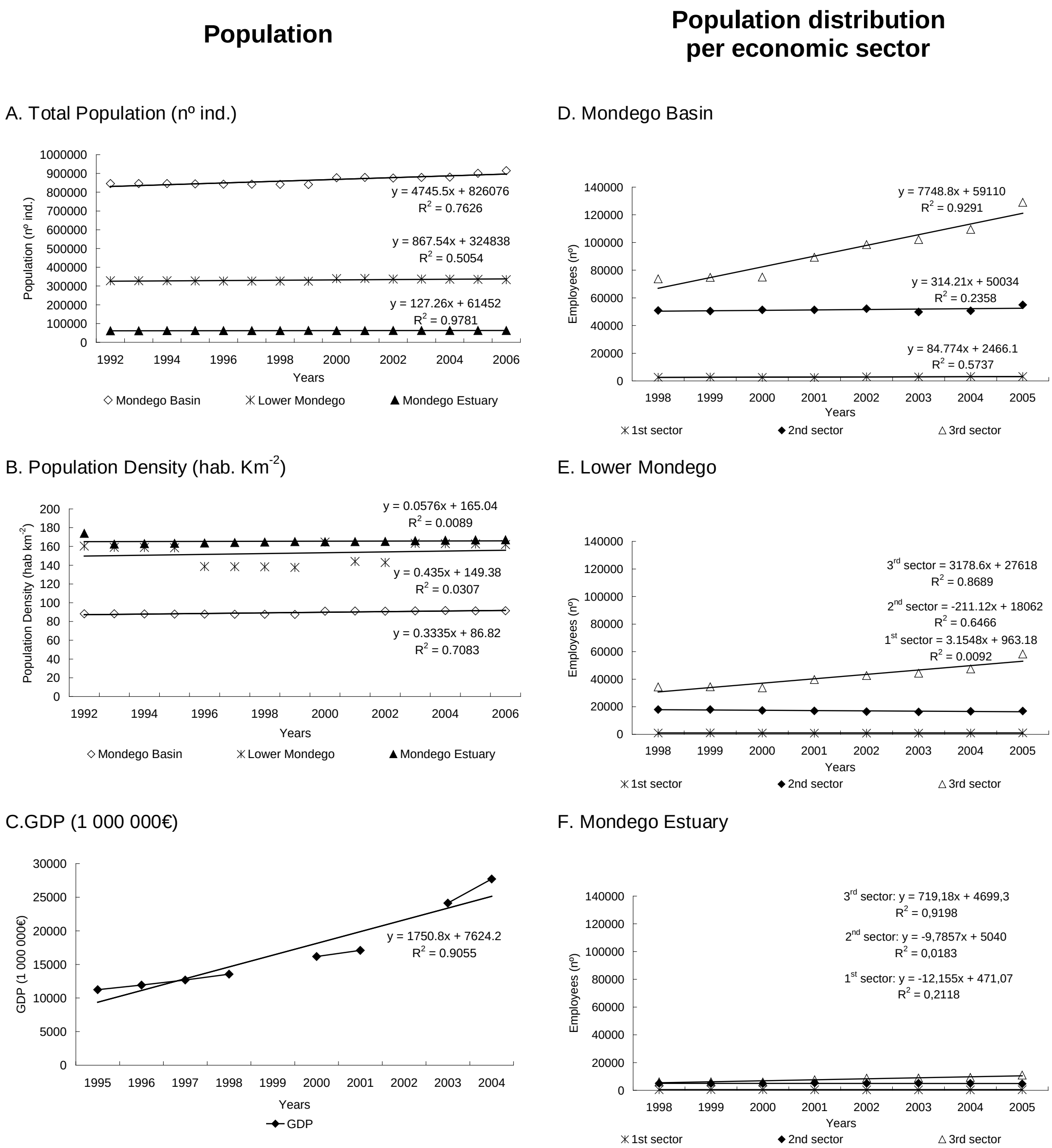


Figure 3: Population data and socio-economic characteristics of the study area from 1992 to 2006 in the three scales under study (Mondego Basin, Lower Mondego and Mondego Estuary): A. Population data (total number of individuals); B. Population density (hab. km^{-2}); C. HDI. Population distribution per economic sector (from 1998 to 2005): D. Mondego basin; E. Lower Mondego; and F. Mondego Estuary (solid line = trend).

In general, the total number of individuals depending on the Mondego catchment area has been increasing, which is reflected in the increasing population density along the basin. This increase is especially significant in the Mondego Estuary region. Concomitantly, the GDP, one of the components of the HDI measure, increased by 25% from 1995 to 2007; it now, with an employment base of 163,395 individuals (2004 data), represents 19.2% of the Portuguese GDP.

3.2. Comprehensive inventory of ecosystem services

An inclusive set of ecosystem services provided by the Mondego's catchment area was assessed (Table 4), based on the system knowledge and literature review. Within the provision category, services such as food production, raw materials, renewable energy and even ornamental resources could be identified. Such services as aesthetic resources, tourism and recreation activities, cognitive values, cultural heritage and non-use values were found within the cultural category. Within the regulation category, we were able to identify such services as gas and climate regulation, disturbance regulation, carbon sequestration, bioremediation, soil erosion prevention, nursery grounds, habitat provision for certain species, nutrient cycling and water supply and water quality assurance. This kind of approach will enable decision makers to consider several alternatives for management based on several available parameters (e.g. uncertainty of results or even reliability of estimates) (Table 4).

Table 4. Inventory of ecosystem services in the Mondego estuary following the MEA classification (MEA, 2003), the valuation method mostly used for each service, its estimated value and level of reliability, uncertainty of results and the level of its impact on biodiversity.

Ecosystem service category	Ecosystem service	Valuation method	Estimated value (1000 €)	Value reliability	Results uncertainty	Impact on biodiversity
Provision services	Food production	MP*; PL	Fisheries: 7,078 – 14,831 Agriculture: 897 – 1,217	Underestimated	High	High
	Raw materials	MP	-	-	-	-
	Renewable energy	CVM	-	-	-	-
	Ornamental resources	MP	-	-	-	-
Cultural services	Aesthetic resources	CVM; BT; HP	insufficient data	-	-	-
	Tourism	MP*; CVM; TC	8,102 – 12,821	Underestimated	Medium	High
	Recreation activities	CVM; MP*; TC	1.5 – 2.2	Underestimated	Very High	-
	Cognitive values	CVM	-	-	-	-
	Cultural heritage	CVM; HP	-	-	-	-
	Non-use values	CVM	-	-	-	-
Regulation services	Gas & climate control	PF	-	-	-	-
	Disturbance regulation	PF; AC; RRC; DC	-	-	-	-
	Carbon sequestration	PF	-	-	-	-
	Bioremediation	RRC; PF	-	-	-	-
	Soil erosion prevention	AC; RRC; DC	-	-	-	-
	Nurseries	PF; PL; AC; RRC; DC	-	-	-	-
	Habitat provision	PF ; AC; RRC; DC	-	-	-	-
	Nutrient cycling	PF	insufficient data	-	-	High
	Water supply	MP	insufficient data	-	-	High
	Water quality	CVM; AC; RRC; DC	insufficient data	-	-	High

Note: MP - market prices method; PL - productivity loss; AC - avoided cost; TC - travel cost; RRC – replacement & restoration costs; HP - hedonic pricing; CVM - contingent valuation method; DC – damage costs; PF – production function; *used in this study

3.3. Conditions and trends of main ecosystem services

A more detailed evaluation was provided for three services: food production, water quality and recreation, as well as for their relation with the biodiversity assets, as a demonstrative example of how assessments can be conducted when evaluating services trade-offs. For this evaluation, only the Mondego Estuary scale was taken into account (Table 4).

3.3.1. Food production

Agricultural productivity was considered as a measure of the system's capacity to support and maintain agricultural activities (Figure 4). Among the seven main crops grown in the region (i.e. potatoes, rice, maize, rye, beans, apples and peaches), there were overall significant decreasing trends in the area, production and productivity of cropland. The main crops' production decreased from 946,298 tonnes in 1992 to 383,165 tonnes in 2006. Data on fish catch and prices for the Mondego Estuary (Figueira da Foz harbour) were used to assess the fisheries' production value. A gradual reduction in the total fish catches (from 16,358 tonnes in 1992 to 11,008 tonnes in 2006) was observed along with an increase in fish prices in agreement with the worldwide trends among fisheries (MEA, 2005). The continuous abandonment of commercial fishing is reflected in a significant reduction in the number of fishing boats observed from 1994 to 2006 (Figure 4D).

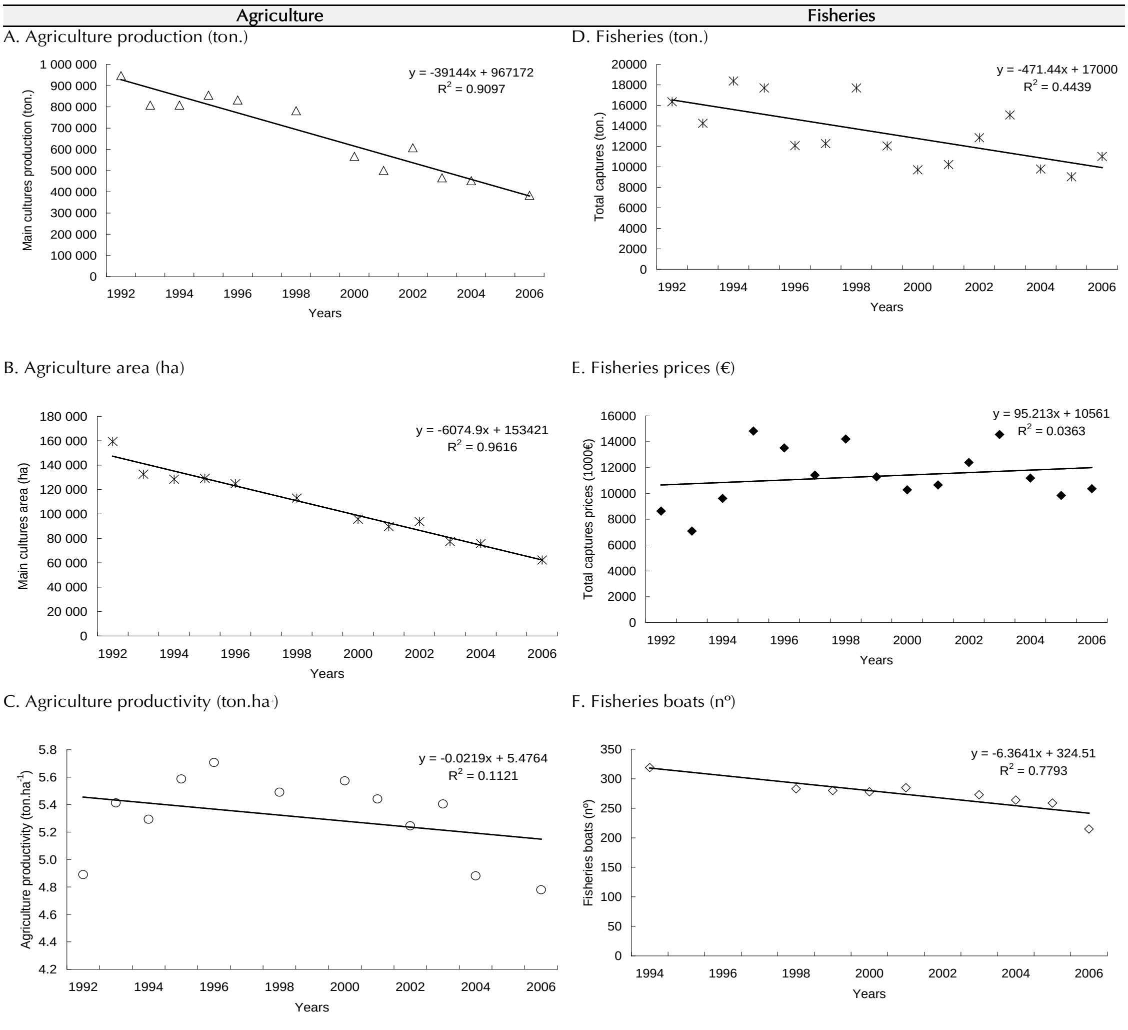


Figure 4. Food production estimation for the Centre Region from 1992 to 2006: A. Agriculture production (tonnes); B. Agriculture production prices (1000 €); C. Agriculture productivity (tonnes.ha^{-1}); D. Fisheries catches (tonnes); E. Fisheries catch prices (1000 €); and F. Fisheries boats (nº) from 1994 to 2006 (solid line = trend).

Along with the significant progressive decline and abandonment of production units, salt production has been declining as well, although this decrease is less marked (Figure 5). On the other hand, since the 1980s, some of the inactive salt-works have been reoriented to fish farming, mainly for intensive production of local species like the gilt-head bream (*Sparus aurata*) and the sea bass (*Dicentrarchus labrax*). Despite the increasing area devoted to fish farming in the estuary, the same trend was not followed by production. In fact, the total aquaculture production in 2003 was 200 tonnes year^{-1}, while 10 years earlier each of the companies involved in such activity produced approximately 120 tonnes year^{-1}.

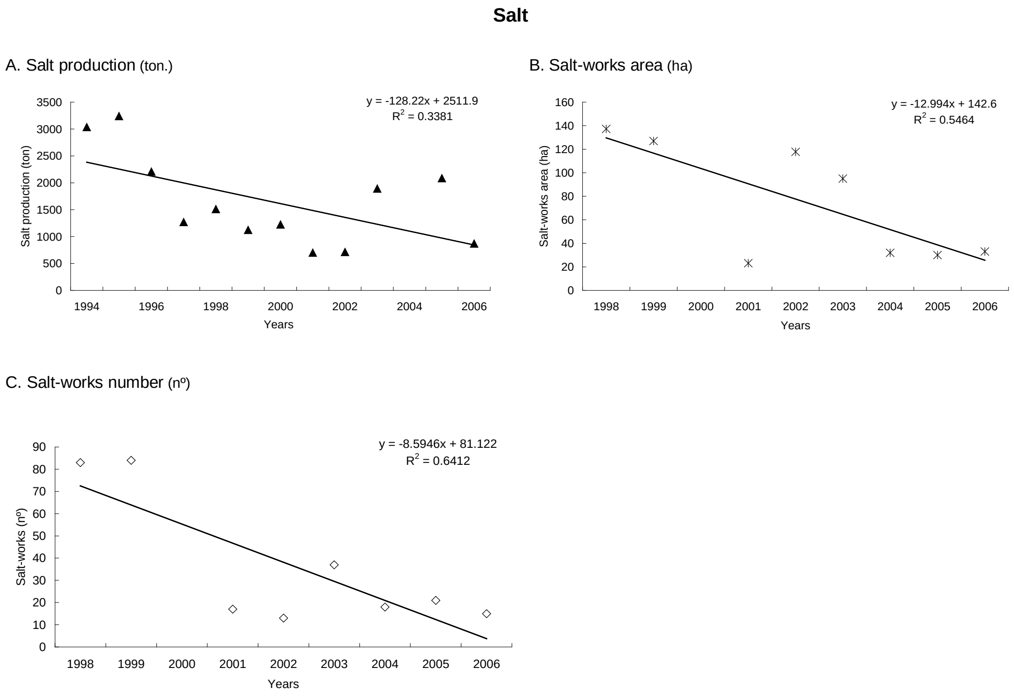


Figure 5. Food production estimation in the Mondego Estuary: A. Salt production (tonnes) from 1994 to 2006; B. Salt-works area (ha) from 1998 to 2006; and C. Salt-works number (nº) from 1998 to 2006 (solid line = trend).

3.3.2. Recreation

The touristic activity in the Mondego catchment area is socially and economically significant (Figure 6). Figure 6A shows a significant progressive increase in the number of tourists visiting the study area, reflected also in the number of facilities provided for tourists. It is also important to consider seasonal impacts, mainly during the summer period (July–September) at the scale of the Mondego Estuary, when the touristic activity reaches a peak (percentage of tourists increase: Mondego Basin: 20.4%; Lower Mondego: 38.3%; Mondego Estuary: 47%; 2006 data).

Visitors attracted to the sites derive benefits from visiting traditional salt-works (Table 5). The area also offers a wide range of opportunities for leisure, bird watching and even a museum. This particular ecosystem represents one of the key green spaces for outdoor activities in the region, contributing to 75% of visitors that are interested in ecotourism activities.

Table 5. Profits from recreational activities (2008 data from Sinergiae Ambiente).

Visitors	%	total number	minimum price	maximum price	Total prices (€)	
Adults	58	81	12	17	967.44	1370.54
Teenagers	22	31	10	15	305.8	458.7
Children	3	4	3.5	3.5	14.595	14.595
Seniors	17	24	10	15	236.3	354.45
Total	100	139			1,524.135	2,198.285

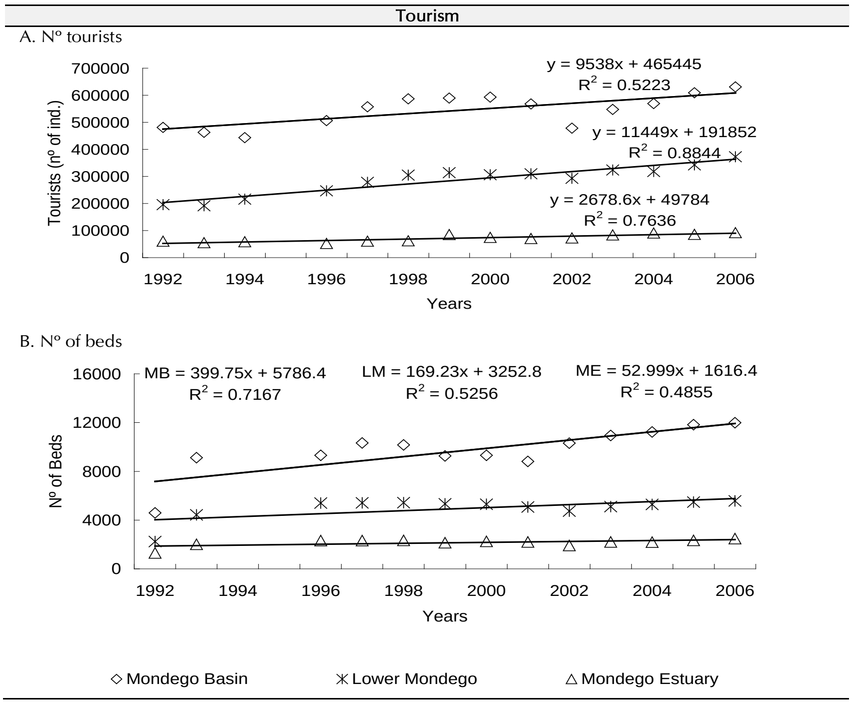


Figure 6. Touristic activities in the Mondego study area at the three studied scales (Mondego Basin, Lower Mondego and Mondego Estuary) from 1992 to 2006: A. Number of tourists; and B. Number of hotel beds (solid line= trend).

3.3.3. Water resources

Water resources are presented at the different scales in terms of water usage and volume of effluents produced by human activities (Figure 7). Increasing water usage was registered, this being particularly significant for the basin and Lower Mondego regions. Regarding effluent production, there was an increasing trend at all of the considered scales.

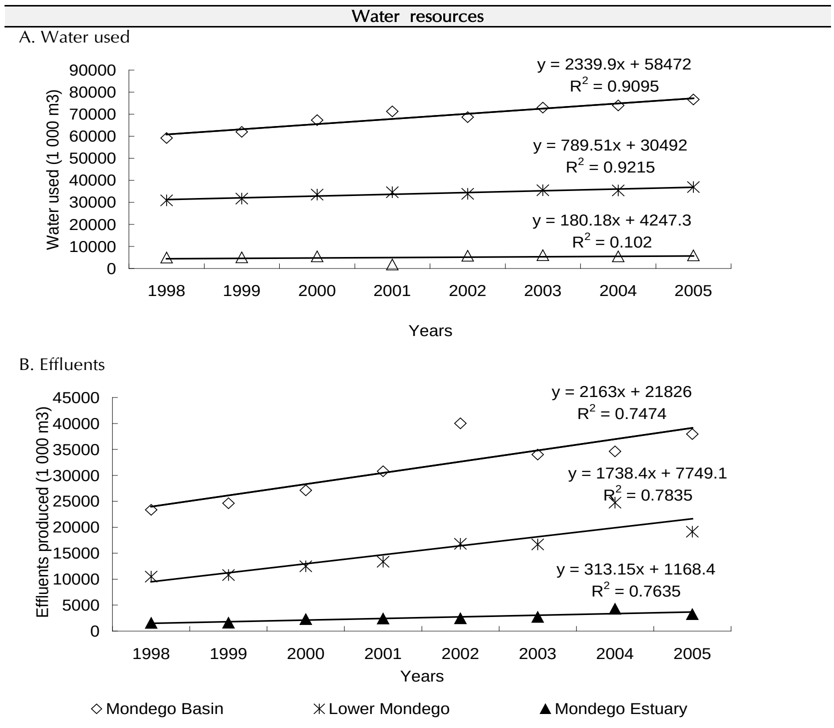


Figure 7. Water resources statistics for the Mondego study area, comparing the three studied scales (Basin, Lower and Mondego Estuary) from 1998 to 2005: A. Water uses (1,000 m^3); B. Effluents produced (1,000 m^3) (solid line = trend).

Due to a lack of data regarding the entire basin, the water quality was assessed only for the Mondego Estuary area. Table 6 gives the results obtained by applying the EEA classification criteria with respect to the nitrite+nitrate and phosphate parameters. A progressive decline in the classification criteria was apparent, with consistently higher levels of nitrite+nitrate in surface waters.

Table 6. EEA classification with respect to the nitrate+nitrite and phosphate water concentrations (surface and bottom), as well as BAT assessment of Ecological Quality Status (EQS) based on macrofaunal communities during spring months (April to June) from 1990 to 2006 in four estuarine areas (E – Euhaline estuarine; PNA – Polyhaline North Arm; PSSA – Polyhaline Sand South Arm; PMSA – Polyhaline Muddy South Arm). EEA classifications - Red: Bad; Yellow: Poor; Green: Fair, Blue: Good. EQS classifications: Orange: Poor; Yellow: Moderate; Green: Good; Blue: Excellent.

	NO3+NO2								PO4								Benthic Macrofauna (EQS)			
	SURFACE WATER				BOTTOM WATER				SURFACE WATER				BOTTOM WATER							
	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA
90					1.5	5.3	5.9	4.9					0.96	0.59	0.92	1.19	M	G	G	M
91																				
92					5.9	15.1	16.0	11.0					7.21	12.47	15.52	11.75	M	M	G	G
93																				
94																				
95																				
96																				
97																				
98					15.9	31.1	16.6	22.6					0.55	0.57	0.24	0.66	G	G	P	M
99																				
00					1.5	23.1	18.4	16.5					1.02	2.25	2.08	2.15	G	M	G	G
01																				
02					12.7	13.8	8.9	8.2					3.20	3.70	3.71	3.46	G	G	M	G
03	10.2	18.2	5.7	21.6	4.5	8.2	4.4	13.6	0.96	1.39	0.71	1.80	0.79	1.03	0.67	1.70	M	M	M	G
04	5.8	11.6	3.8	14.6	1.8	7.1	2.9	13.2	0.68	0.96	0.64	2.13	0.39	1.11	0.51	1.85	G	G	M	G
05	5.9	8.1	5.9	11.5	6.6	7.3	5.7	12.2	0.55	0.88	0.72	2.00	0.49	0.80	0.71	1.89	G	G	G	G
06	17.3	20.4	12.1	20.0	12.8	10.5	13.7	19.9	1.18	1.18	0.87	1.99	0.58	0.78	0.78	1.94	G	M	M	H

3.4. Estimation of interdependence between services

The Pearson correlation analysis (Table 7) revealed a strong relation between population metrics and several activities taking place in the area. Moreover, there is also a negative relation between tourism activities (expressed by the number of visitors to the region) and activities in the primary sector, like agricultural production.

Table 7. Correlation analysis between environmental pressures versus biodiversity assets.

	total population	population density	agriculture production	fisheries captures	salt production	nº tourists	effluents produced	water used	O_2 (mg/l)	NO_3 (mg/l)	NO_2 (mg/l)	PO_4 (mg/l)	%OM	BAT E	BAT PSM	BAT PMS	BAT PS
total population		0.9989	-0.9086			0.9210								0.8985			
population density			-0.9261			0.939								0.9069			
agriculture production				0.8562		-0.9722								-0.8699			
fisheries captures															-0.8277		
salt production																	
nº tourists														0.8223			
effluents produced																	
water used										-0.8739							
O_2 (mg/l)											0.8356						
NO_3 (mg/l)																	
NO_2 (mg/l)																	
PO_4 (mg/l)																	
%OM																	-0.9236
BAT_E																0.9590	
BAT_PSM																	
BAT_PMS																	
BAT_PS																	

3.5. Ecological assessment

Due to a lack of data regarding the larger scales, only the Mondego Estuary was analysed in the biodiversity assessment (Table 6). The North arm (euhaline estuarine and polyhaline sand north arm) presented a strong biodiversity decline in the 1992 followed by some recovery. From 1998 onwards, the estuarine mouth and north arm showed significant improvement from moderate to good Ecological Quality Status (EQS). The south arm also presented a significant decline in biodiversity until 1998. In 1998, following the implementation of several experimental mitigation measures (Teixeira et al., 2009), the system’s biodiversity began to show signs of improvement. As a whole, a gradual enhancement of the system's ecological condition has been taking place. Based on the Pearson correlation analysis performed (Table 7), the impact of ecosystem services in the area on biodiversity assets was assessed (Table 4). This analysis shows that services as tourism activities and agriculture have strong negative impacts on the system, while factors like salt-works and water uses make only minor contributions.

3.6. Economic assessment

An overall estimation of the ecological and social importance of the considered services was performed in light of the benefits obtained. Our findings (Figures 4–6) roughly suggest that the estuarine ecosystem makes a significant contribution to society, especially in terms of food production (e.g. fisheries, 7078–14,831,000s) and tourism activities (8102–12,821,000s) (Table 4). Nevertheless, it should be recalled that this is only a demonstrative application of the methods available, and do not intent to assess the entire basin value. From the estimation of demand curves (Figure 8), it was possible to see that for any given service reduction, the value for the more elastic uses (agriculture and fisheries) was less than the value for the more inelastic uses (e.g. tourism, although this service also exhibited elastic behaviour).

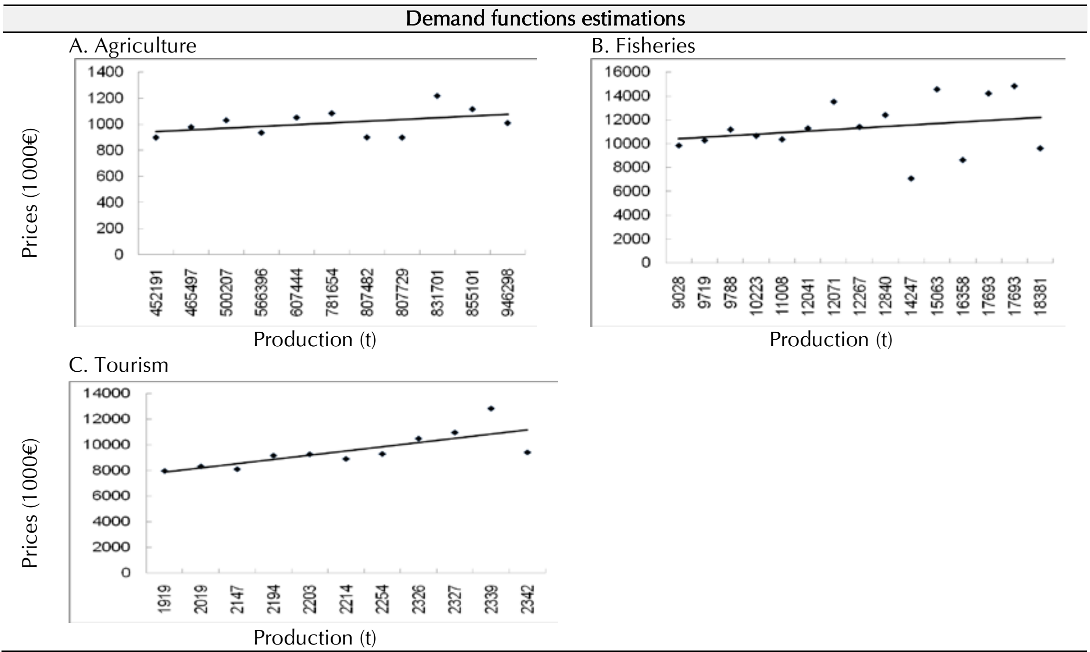


Figure 8. Services supply (quantity) versus market offer, based on market prices, of the study area from 1992 to 2005: A. Agriculture; B. Fisheries; and C. Tourism (solid line = trend).

4. Discussion

4.1. Spatial scales of the system and scale dependence

The scale effect in this study is particularly highlighted when integrating the biodiversity results (local scale) with Lower Mondego agricultural production (regional scale). The upstream activities may influence local biodiversity, having an evident scale effect. In 1993, the South arm exhibited strong eutrophication symptoms, leading to a severe reduction of local biodiversity. It appears that agricultural activities on the Lower Mondego regional scale, mostly due to the release of nutrient-enriched waters from fields, were partly responsible for the eutrophication symptoms observed in the estuary. In the face of this problem, two major mitigation measures were undertaken in 1997/98: (i) the agriculture fields' runoff was diverted into the north arm; and (ii) the connection between the two estuarine arms was improved (Marques et al., 2003). As a consequence, the local estuarine biodiversity started to improve (Marques et al., 2003). On the other hand, when considering the trade-off between food production and ecosystem assets such as biodiversity or ecosystem integrity, it is important to keep in mind that food production is economically crucial in the Lower Mondego River Valley area. Measures that might be undertaken to solve any environmental problem must also take into account the socioeconomic reality of the region. It can be assumed that the highly structured and man-modified environment provides suitable conditions for the achievement of a balanced interaction between services and assets, even at different scales. This study revealed the importance of scale assessments when quantifying areas of concern for provision of ecosystem services and its relation to human well-being, which is in accordance with previous assessments (e.g. Jaarsveld et al., 2005; Barbier et al., 2008).

4.2. Comprehensive inventory and conditions/trends of main ecosystem services

The use of a comprehensive approach (e.g. inventory) to evaluating significant ecological, social, and economic costs and benefits facilitates the work of decision makers regarding the implementation of management and conservation strategies (Figure 2). The findings give insight into how aggregating all of the services into one value could be achieved. Therefore, when implementing a management program, several processes and characteristics of the system have to be taken into account. As demonstrated in Figure 2, both direct (e.g. habitat maintenance, tourism facilities or preservation and functioning of regulatory processes) and indirect (e.g. environmental/societal outcomes) benefits and costs have to be considered, assessed and valuated to ensure an accurate and precise choice of policies. In the Mondego River Basin, population pressure has triggered changes in water use. Shipping, fishing, agriculture and recreation were the most important uses reported. Across the entire Basin, there was a positive trend among all of the economic sectors considered; nevertheless, through a scale refinement, it was possible to recognise a negative trend for the secondary sector at the other two spatial scales (Lower Mondego region and Mondego Estuary). In the estuary, there was also a decrease in the activity of the primary sector, reflecting the abandonment of activities such as agriculture and commercial fishing combined with an important augmentation of provision of services, mainly in the forms of tourism and recreational activities. Regarding the water resources, the variables showed an increasing trend across the three assessed scales. Not surprisingly, these variables followed the population data tendency. Nevertheless, industrial water use and water extraction for domestic usage and irrigation also appear to play an important role at each of the three scales analysed in the system. Land use and water resources are obviously linked. The impact of land use and its different practices and intensities on water quantity or quality can be substantial. Currently, the maintenance of water quality seems to require the most attention as it influences, to a large extent, the courses of all of the other variables. For instance, the decline in fish farming production appears to be mainly related to decreasing water quality. As the population increases, a consequent increment of activities drives water use enhancement and increased effluent production. Moreover, the higher levels of nitrates and nitrites on surface waters than in bottom waters suggest that the main source of these nutrients is upstream of the study area. The system's nutrient enrichment and the subsequent eutrophication effect leading to a decrease in water dissolved oxygen is one of the possible factors affecting the production conditions on the system. Overall, human activities cause a sequence of environmental damages and stresses that may alter the ecosystem's natural processes and thus alter its equilibrium. Based on this specific assessment, several factors were identified as promoters of changes, such as high nutrient concentrations, land occupation rates and habitat maintenance.

4.3. Estimation of interdependence between services

On top of the environmental challenges, social, cultural and economic problems overlap. Activities are never isolated or result from cause-effect linear relations; they interact and compete for area. They sum up effects and produce a complex network of interrelations, which becomes even more difficult to analyse than each ecosystem service is alone (Figure 9).

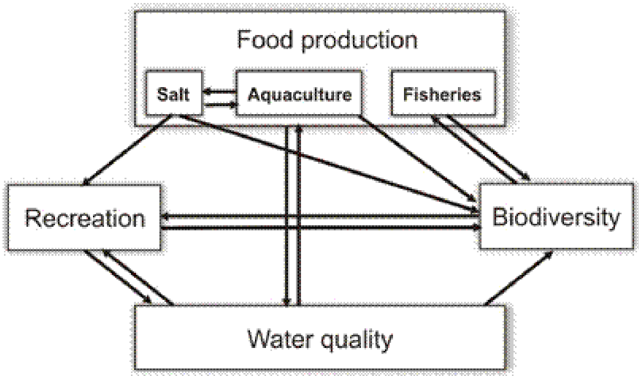


Figure 9. Inter-relations between the different services in the Mondego Estuary.

A typical example of such interactions between different activities can be seen in the Mondego Estuary (Table 7). The selection of only three services for this approach may reduce the validity of valuations for decision-making; nevertheless, the intent was to compile an overview of some of the region's services and examine the interrelations among them. We hope this will provide a basis for further insightful valuations. The increasing nutrient concentration in the water, essentially due to agricultural runoff, influences aquaculture production and affects the aquatic communities' diversity. Impoverished benthic communities, which serve as food for many fish species, might eventually cause a decrease in fish production. In general, due to this intrinsic and complex network of interrelations and interdependencies, any measure undertaken to improve one ecosystem service in isolation will directly or indirectly have repercussions on the others, as also demonstrated by, for example, Acharya (2000), Atkins and Burdon (2006) and Young et al. (2006).

4.4. Ecological assessment

Despite its decline due to the progressive abandonment of production units (implying a drop in salt-related income as well as a drawback form any bird species and, consequently, a decline in biodiversity), the salt-works possess a strong social and cultural inherent value and serve as important stopover and refuge areas for birds migrating along the north western coast of Portugal. These areas are particularly important for waders, especially such species as the Pied Avocet (*Recurvirostra avosetta*) and Greater Flamingo (*Phoenicopterus rubber*). During the breeding season, the Mondego Estuary is regionally important for species such as the Black-winged Stilt (*Himantopus himantopus*) and Little Tern (*Sterna albifrons*). Portuguese salt-pans are usually regarded as mainly providing supplementary feeding over high water on the assumption that this habitat is less suitable than the mudflats for most waders (Lopes et al., 2001; Múrias et al., 2002). A decline of the macrobenthic estuarine biodiversity, usually associated with a degradation of the water quality status, can lead to a reduction of opportunities for these birds to feed. This situation may decrease the ecological and economic potential of this region. Furthermore, the diminishing or absence of certain species may also influence all of the trophic relations in the area while altering prey-predator relations. Moreover, this system was considered as a Ramsar site (Ramsar site 1617) and, consequently, its eventual loss may be detrimental to many species. The traditional salt-works systems, such as the one in the Mondego Estuary, due to its unique characteristics, tend to attract visitors from far away. In theory, if they are near urban areas and are heavily used, such destinations will present higher average values than most similar ecosystems (Gibbons, 1986).

In this particular situation, a complex network of interrelations is verified that may not be well described by the regression reported here. The full value of this particular ecosystem has to take into account several parameters: the recreational value (environmental asset values depend on both the unit value of a user-day and how many consumers visit the site); the food production (including salt and fish values); the biodiversity maintenance value (including the entire trophic network that depends on the system); and also the existence value of the ecosystem. Despite the trend analysis, the market economy fails to regulate the pollution flow to the environment, which is known as a negative externality or negative consequences of human activity. As such, pollution’s social costs (i.e. the lost income due to the considerable loss of bird nidification (or nesting) spots or migratory routes across the Mondego Estuary as a consequence of habitat loss, in this case salt-pans) maybe greater than the private cost (i.e. no private expenses are associated with the contamination of water by aquaculture). Others in society have to pay the economic price of these environmental impacts that they may not have caused (Figure 2). Nevertheless, to achieve a precise impact measure, a survey must be conducted in the area to evaluate the value given by the local people to the system’s natural features in order to protect and preserve it.

4.5. Economic assessment

In addition to the spatial or geographical scales, the valuation and assessment of services are also affected by the temporal scale of analysis. Uncertainty is inherent in the valuation process, not only due to limited data availability but also due to constraints in evaluating services’ impacts. As such, the values considered in this study should be considered with caution as they are mainly approximations (Table 4). Because demand usually becomes more inelastic as supply is reduced (Gibbons, 1986), the marginal values calculated may underestimate the real services' values (Table 4). Additionally, the price data used in the demand studies are rather limited, which may also contribute to our results. In addition, the values considered are averages rather than marginal values of services, which influence the final outcome. Another point to be taken into account is that both agriculture and fisheries (Figures 8A and 8B, respectively) are essential goods, while tourism activities may be considered as secondary goods. This can then influence consumers' preferences and needs when making a choice to consume a good, especially if the aggregate economic value is made the basis of conservation decisions. Nevertheless, to accurately inform policy makers and design better management options, the uncertainties that result from these attempts at valuation of services must be incorporated into decision-making processes (Costanza, 1993). One way to deal with this is through the application of the Precautionary Principle (Andorno, 2004; Myers and Raffensperrger, 2005), which assumes sequential management implementations where decisions are made cautiously until the evidence becomes more sound (Perrings, 1991; Costanza, 1993). Considering the prices charged for the opportunity to engage in wetlands recreation, such as visiting the traditional salt-works, the minimum value attributed to this opportunity can be equated with its price to estimate the consumer demand function (Gibbons, 1986). For the Mondego Estuary, it was not possible to obtain a time-series analysis for this particular service, mostly because these activities (and the charges associated with them) have recently begun in the area. A drawback of this kind of analysis is that it mainly focuses on the value of a recreational activity in Euros per day or per visitor. If a total site value is estimated, it can be ascribed to various constituents of the site, such as the water itself, the aesthetics of the site as a whole, the associated biodiversity or even the available facilities. This analysis may be the starting point for a more exhaustive valuation. Based on a user-day estimation for the Mondego Estuary, it becomes possible to relate visitation rates and flow levels in the future either through direct methods (such as the MP used here) or through stated-preference methods (such as the TC or CV methods).

Although monetary values may not translate into cognitive benefits, non-use values or supporting services, these services must not be ignored as they are thought to play a significant role in maintaining human well-being (Marques et al., 2009). Economic valuation cannot place a value on species survival or on the ecosystem's functional and ecological role, except from the human perspective (MEA, 2005). Nevertheless, it must be highlighted that even when benefit revenues are not the primary objective of wetland exploitation and conversion, activities such as agriculture, aquaculture, and urban and industrial facility expansion are normally considered important for the economic development and social growth of the region (Figure 2). Moreover, this study shows the applicability as well as the hurdles to valuation and the limited contributions of it, given a non-exhaustive data availability, effective measures leading to biodiversity improvement have been taken in the past without valuation, and a price tag while supporting the argument would add limited new information. It would further require an accurate valuation of biodiversity assets, which implies an enormous task with uncertain outcome. This fact has been leading to the gradual shift of biodiversity indicators development from assessing the state to monitoring pressures (e.g. Levrel et al., 2010).

5. General conclusions

The Mondego catchment's full value cannot be calculated without taking into account all of the direct and indirect uses' values or without estimating the real wetlands' value; focusing solely on market prices may lead one to underestimate the real value of the region. With this study it was attempted to exemplify how services values may be estimated, but not the value of the catchment as a whole. Also, assessing the system value would require aggregation over time including the inherent depreciation rates of flows and outputs. Our preliminary values may provide a starting point for a more exhaustive and detailed valuation of the Mondego wetlands. Nevertheless, regarding the Mondego River Basin, some general conclusions may be drawn:

1. Three services were identified in this assessment: food production, water quality and recreation. An increase of services such as recreation activities and water uses has been verified, while there has been a decrease in services like food production;

2. Some promoters have been modifying ecosystems' structures and functions (e.g. population and nutrients concentrations), where we can observe a progressive loss of natural and agricultural lands to development and services provision (e.g. tourism);

3. There is a strong influence of upstream activities on local assets, and so, when performing an integrative analysis, it is extremely important to include the scale effect;

4. The ecological quality of the system has improved, as reflected in both local communities' diversity and water quality;

5. There is a clear need to evaluate the links between land uses, water quality/quantity and biodiversity to achieve good resource management;

6. More insightful uncertainties and valuation studies are needed to clarify the links that leads to changes in ecosystem services supply to ensure adequate management strategies.

Water management plays a crucial role in the provision and delivery of all services considered here. Therefore, it becomes crucial to simultaneously achieve economic efficiency, environmental protection and sustainability within a system. Along with water management and protection, an accurate biodiversity asset evaluation is needed to better understand what ecosystem services that are essential for human populations' wellbeing can be supplied. Both water management and protection and accurate biodiversity asset evaluation are fundamental to ecologically sustainable social and economic growth and development (Figure 2).

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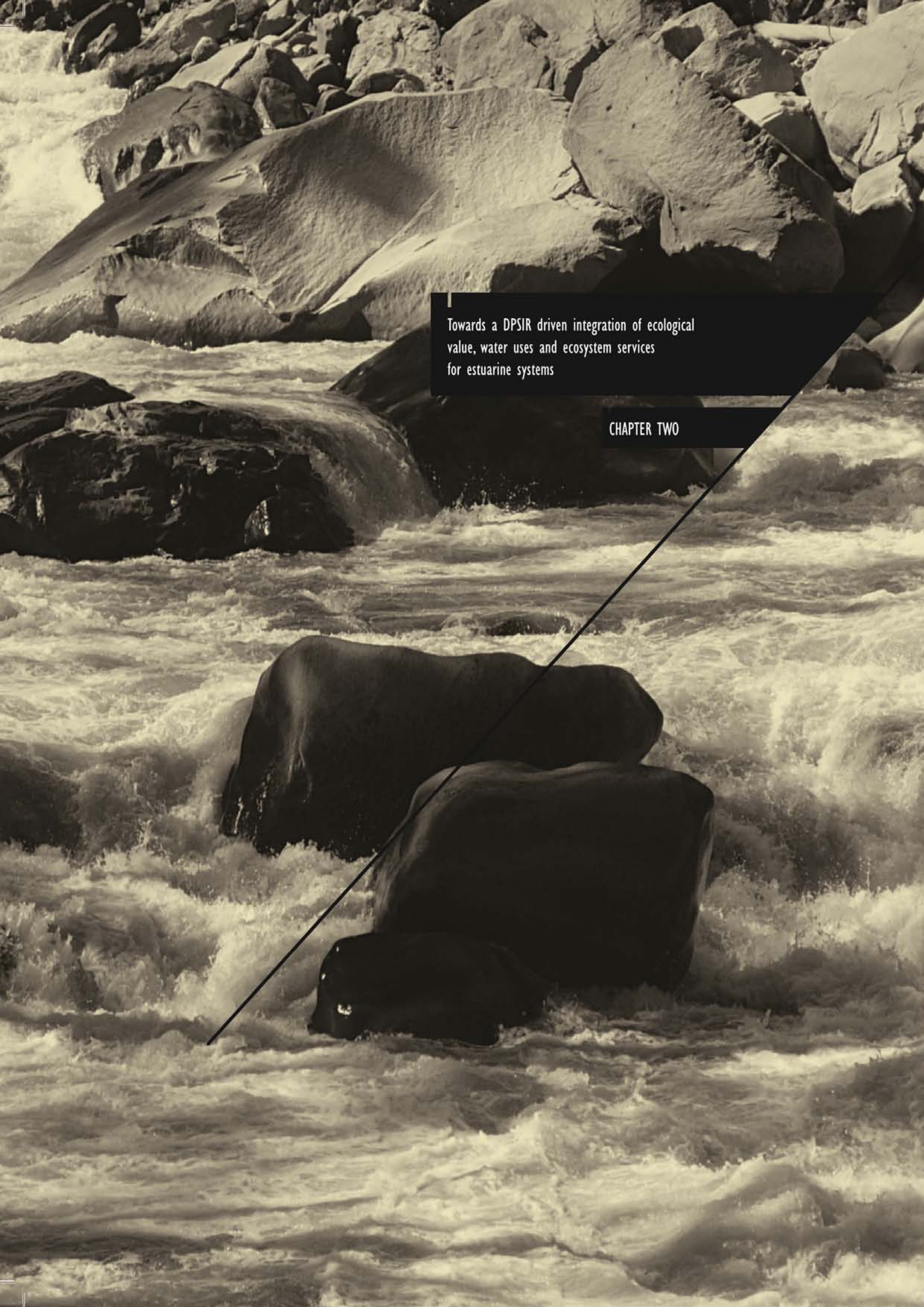

Towards a DPSIR driven integration of ecological value, water uses and ecosystem services for estuarine systems

CHAPTER TWO

Abstract

The aim of this paper is to indicate the overall economic efficacy of competing uses of estuarine resources by integrating ecological value, water uses and ecosystem services into the DPSIR conceptual framework as an added value for policy making and management. The complex interactions between the socioeconomic system and the ecosystem (as part of the 'integral system') require generic but still 'tailor made' techniques to quantify all relevant variables and to provide an integral view of the system's status. One of the few techniques that can assist in structuring such complex data in an integrative way is the Drivers-Pressures-Status-Impacts-Responses (DPSIR) approach. Support and regulatory services (such as water supply and water quality) are essential to sustain crucial ecosystem processes and functions while the water required for human activities (water demand) is an essential system service. With the help of DPSIR, the main changes in the Mondego Estuary ecosystem (Portugal) were outlined, used as an illustrative example, and causes and effects described. Within the Mondego Estuary region the main water consumers are agriculture, industry, and households. Baseline scenarios predict an increase in water usage by mainly the touristic service sector. Our analysis illustrates that pressures from human population growth and related activities gradually increased over the studied period. Land-use patterns, diversion of freshwater flows, water pollution and morphological interventions directly caused physical, chemical, and biological modification and degradation. This consequently led to negative ecological and socio-economic impacts, such as eutrophication. The scenarios suggest an increased pressure based on an expected 8% annual population growth and an average annual decreased pressure of 5.2% *per* annum due to the current reduction in agriculture. The results show that understanding the water use-related complex and intricate trade-offs among ecological, social, and economic goals is fundamental in designing and implementing management policies and ecosystems restoration schemes.

Keywords:
| DPSIR framework
| Ecosystem services
| Water uses and management

1. Introduction

Every ecosystem provides essential services and goods, contributing to the satisfaction of human needs and changes in well-being and delivers irreplaceable support functions on which human life relies (Costanza et al., 1997; Boyd and Banzhaf, 2007). According to the Millennium Ecosystem Assessment (MEA) classification (MEA, 2005) four categories of services can be established: provisioning (products obtained directly from the ecosystems); regulating (benefits obtained from the regulation of ecosystem processes); cultural (nonmaterial benefits people obtain from ecosystems through cognitive development and aesthetic experiences, for example); and supporting (benefits necessary for the production of all other ecosystem services). From these, water resources can be considered as a cross-sectoral issue, being found within several of the mentioned categories. For example, the water cycle may be considered within the regulation category (e.g. biochemical cycles that are fundamental to all living organisms and ecosystem functions, such as chemical, element or nutrient (re)-cycling) (Hawkins, 2003), while water supply for human consumption and for economic activities may follow within the provisioning category. Despite their inherent importance, economic values attributed to water resources depend both on consumer preference and on the perception of possible changes in well-being through ecosystem impacts (Turner et al., 2000b; Chen et al., 2009). Although the MEA framework may be considered as the most widely recognised approach, numerous attempts to develop concepts and classifications have, however, been conducted since 2005 (e.g. Beaumont et al., 2007; Fisher et al., 2009; de Groot et al., 2010b; Atkins et al., 2011). Some achievements are, for example, the clear separation among ecosystem processes, functions, services and benefits to societies, creating a sequence from fundamental services (defined by the environment-organisms relationships), via final services (biotic processes ruling the biology-biology and biology-environment interactions), to the societal benefits (benefits of the ecosystem for humans well-being) (Luisetti et al., 2010; Atkins et al., 2011).

Due to typical aspects of coastal areas such as the high population densities and its increasing socio-economic demands, estuarine ecosystems are progressively subjected to anthropogenic exploitation and disturbance. For water, and for the service categories where it is included, one of the direct consequences of anthropogenic presence may be environmental degradation. The ecological status and assessment of water bodies is thus one of the most prominent environmental, social and economic concerns (Marques et al., 2009; Pinto et al., 2010). Therefore, the accurate characterization of the main pressures and impacts from human activities are required for any economic analysis on water uses and services when performing environmental assessments. From an ecological point of view, the Drivers-Pressures-Status-Impacts-Responses framework (hereafter DPSIR) (OECD, 1993) is considered an insightful framework for integrating quantitative and qualitative ecosystem/socio-economic interactions (Turner et al., 2000b; Elliott, 2002; Borja et al., 2006; Marques et al., 2009). Hence, this approach allows for the assessment of the link between the ecological characterization of ecosystems (wetland functions) and their economic valuation (uses) in a system. This framework is particularly pertinent when used in parallel with scenarios' development, highlighting the potential impact of current socioeconomic developments while assessing current trends in water status (Trombino et al., 2007). It is widely accepted that estuaries are among the most productive and valuable natural systems around the world (Costanza et al., 1997; Jørgensen, 2010). Due to this, the management of estuarine environmental quality should be focussed on the sustainability of human activities in coastal zones. One possible measure of the coastal system's condition is its capacity to sustain human uses. Decisions that may influence wetland resources should consider the full range of benefits and values provided by wetland ecosystem services (Birol et al., 2006; Trush et al., 2009). The combination of composite or integrative approaches (e.g. DPSIR) with social-ecological system analysis may provide a robust approach for implementing and monitoring mitigation strategies to reduce system degradation (Karageorgis et al., 2005). Therefore, wetlands vulnerability analysis should involve water quantity and environmental quality as well as water supply-and-demand requirements.

Based on the above, the DPSIR framework was used as an integrative tool to combine the qualitative/quantitative ecosystem and socio-economic interactions, while assessing the link between wetlands functions and their uses. This analysis allowed the development of scenarios for the area contributing to implementing mitigation strategies meant to reduce estuarine deterioration, under the Water Framework Directive goal (to achieve Good Ecological Quality Status by 2015) (IMPRESS, 2002). Moreover, they serve the improvement of monitoring schemes. The main objective of this study was to demonstrate the overall economic efficacy of competing uses of estuarine resources by integrating ecological values, water uses and ecosystem services into the DPSIR conceptual framework as an added value for policy making and management.

2. Methodology

2.1. The study-site

2.1.1. Mondego Basin: physical characteristics

The Mondego Basin, located in central Portugal, has a catchment area of approximately 6670 km^2, and is highly diverse in topography, hydrology and land use. Its functional structure ranges from mountainous areas to a large alluvial plain discharging into the Atlantic Ocean (Marques et al., 1997; Graça and Coimbra, 1998). The Lower Mondego region (with a total area of 250 km^2) connects the mountain river with the ocean and consists of open valleys and plains. This region also includes the Mondego Estuary (7.2 km^2; Figure 1). The main focus of this study is the Mondego Estuary as part of the Mondego River basin.

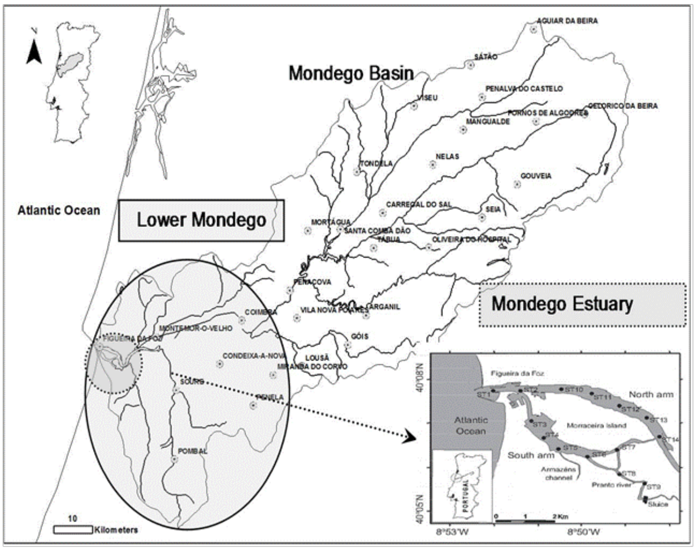

Figure 1. Study-site location and the three considered scales: Mondego Estuary, Lower Mondego, and Mondego Basin.

2.1.2. Socio-economic characteristics and interactions

The Mondego River basin provides a high variability in environmental and social conditions. Over half a million people live and work within the Mondego floodplain. The area around the Mondego Estuary is more densely populated (167 inhabitants km^{-2}) than the rest of the basin (circa 90 inhabitants km^{-2}). The area covers a wide range of uses, such as intensive agriculture and industry. Consequently, the water flowing into the estuary has been loaded with nutrients and polluting compounds for already decades. In the Lower Mondego, strong pressures are caused by the primary economic sector (15,000 ha of highly productive agriculture of mainly rice) and by harbour-related activities in Figueira da Foz. Secondary and tertiary economic sectors are well represented among the total economic activities of the entire basin. There are a number of relevant existing impacts due to human activities and engineering (Pinto et al., 2010). Engineering activities, like the Serra da Estrela hydroelectric system (360 GWh annual production) and the occurrence of some dams have changed the hydrological conditions. The latter system was built to prevent the area from flooding and for irrigating the Lower Mondego region (Lima and Lima, 2002).

2.1.3. The Mondego Estuary ecosystem

The Mondego is a warm-temperate shallow and turbid tidal flat estuary with a mean tidal range of 3 m and strong tidal currents. The estuary is divided into two arms by Murraceira Island 7 km upstream from the tidal inlet of the estuary (Figure 1). The two arms have very distinct physical and chemical characteristics that influence the local ecological conditions. The North arm is relatively deep and constitutes the main navigation channel. The South arm is relatively shallow and is characterised by large intertidal flats during low tide (75% of total area). Land use (mostly fish farming and agricultural areas) has changed the morphology, and consequently the hydraulics and related ecological conditions, of the South arm. Between 1992 and 1998, sediment accumulation at the divergence of the two arms blocked water circulation and changed the southern sub-system almost into a coastal lagoon (Neto et al., 2008), because the fresh river water was mainly discharged via the northern arm. During this period, water circulation in the southern arm was mostly driven by the tides and the freshwater input from the Pranto River tributary of which the discharge is strongly influenced by water extraction for rice agriculture (Flindt et al., 1997; Marques et al., 2003). This freshwater input into the system created an a-typical water regime while at the same time serving as an important nutrient source to the system. The result was blooms of green macroalgae (Martins et al., 2001, 2007; Patrício et al., 2007) and a concomitant decrease of *Zostera noltii* meadows (previously described as a highly productive system) (Marques et al., 1997; Patrício et al., 2004). From 1998 onwards, experimental mitigation measures were implemented (Neto et al., 2008; Lillebø et al., 2007; Patrício et al., 2009): 1) the seagrass meadows were protected from human physical disturbance; 2) there was a reduction of nutrient loadings into the South arm (e.g. mean N/P changed from 39.8 to 13.2; diversion of the nutrient enriched freshwater to the North arm by another sluice located more upstream); 3) public awareness programs on the importance of intertidal seagrass to ecosystem health and economic activities were implemented; and 4) improvement of the hydraulic regime by enlarging the connection between the two arms (reduction of water residence time in the South arm from 9 to 6 days, using the methodology described in Braunschweig et al., 2003). The performed hydro-morphological changes reduced the probability of eutrophication symptoms and other problems associated with water pollution in the South arm. Based on the estuarine salinity gradient and subtidal soft bottom habitat characteristics, the Mondego's lower estuary can be divided into four ecological areas (Figure 1): euhaline estuarine sand (stations 1, 2 and 10), North arm polyhaline sand (stations 11 to 14), South arm polyhaline sand (stations 3 and 4), and South arm polyhaline muddy sand (stations 5 to 9) (Teixeira et al., 2008).

2.2. Adopted framework

The DPSIR framework was used as an analytical tool to trace changes in the transitional wetlands structure and function over time in relation to human uses. The main driving forces were identified and their impacts on the system functioning evaluated (Figure 2).

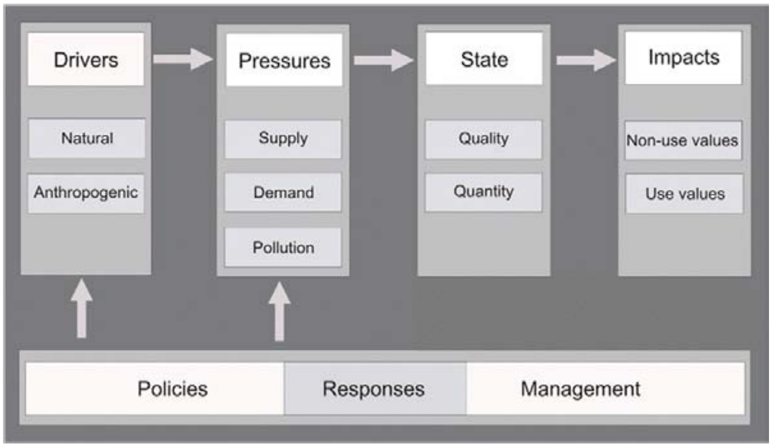

Figure 2. DPSIR approach applied for the Mondego Estuary: identification of (natural and anthropogenic) Drivers, the main Pressures occurring on the supply, demand and pollution of aquatic resources. This allowed for a qualitative and quantitative Status evaluation, and for measuring the Impacts on the use and non-use values of the system. The societal Responses meant to improve the system should take into account both implemented policies and management actions taken (past and future).

At this stage, also the scale issue was considered by drivers and pressures trend analysis carried out at successively higher geographic scales: Mondego Estuary, Lower Mondego, and Mondego Basin (Figure 1). This approach was used to assess water condition and status in the most seaward part of the Mondego River and to make inferences about the effects of upstream activities on the estuarine region. Due to data availability constrains the 1994 (initial condition) - 2006/07 (final condition) time period was selected for the socio-economic quantification, and the years 1990-2006 for natural drivers and ecosystem status evaluation.

2.2.1. Inventory and description of drivers

Identified drivers were divided into two broad categories (IMPRESS, 2002): ‘natural’ (which can be assessed, but not controlled), and ‘anthropogenic’ (human driven changes that can be assessed and controlled) (Figure 2).

a) Natural drivers

Included in this category were species invasions that occurred in the study area and that may have interfered with the system integrity, and extreme events occurring on the system. The assessment on exotic species was mainly based on existing secondary data (e.g. Anastácio and Marques, 1996). For the period 1990-2006, extreme events (like dry years) were defined based on mean annual and mean summer temperatures (ºC) and based on the total annual precipitation (mm). The required data were obtained from national institutes as INAG (Water Institute; www.snirh.pt) and IM (Meteorology Institute; www.meteo.pt).

b) Anthropogenic drivers

The anthropogenic drivers were divided into four sub-classes: social, economic, morphological, and ecological. The social drivers were evaluated by population factors (total number, population density, and household numbers) and urban land occupation rates. These can be used as a useful proxy for the use of freshwater. To do this, human use-related relative changes (expressed in percentages) were weighted against the changes in percentages of the same factor but at other spatial scales (Mondego Estuary, Lower Mondego, and Mondego Basin). The data were then compared with the multi-scaled GDP (Gross Domestic Product) per capita, which can be used as a rough indicator for the standard of living and can be calculated at multiple spatial scales. Other important social drivers that determine ecosystem orientation are the policies and institutional directives implemented by administrative organizations responsible for managing the system. Within this category, several scales can be considered: the local level (PDM, 'Plano Director Municipal'), where municipal institutions drive system characteristics by engaging local development and growth; the regional level; the national level; and finally, the European Union level.

Six main activities were considered in the economic drivers' assessment: agriculture, fisheries, salt production, industry, the commercial harbour, and tourism. Three main parameters were assessed for each activity: 1) the number of production units (total enterprises dedicated to goods production); 2) the output (total production and/or profits); and 3) employment. Dredging activities to maintain the fairway (total annual volume) and main physical barriers were considered to be morphological drivers.

Among the ecological drivers, we have estimated total water extraction and the quantity of effluents produced, drained and treated, over the years. The information for these sub-classes was obtained from national institutions such as INE (National Statistics Institute, www.ine.pt), IPTM ('Instituto Portuário e Transporte Marítimo'), and IPIMAR (National Institute of Biological Resources), consolidated in Pinto et al. (2010).

2.2.2. Evaluation of environmental pressures

A pressure is defined as the direct and quantifiable effect of a driver to the system (e.g. an effect that causes a change in flow or a change in water chemistry) (IMPRESS, 2002). There are several classes of pressures: pressures source type (point and diffuse source pollution), water extraction/regulation, biological resources, hydro-morphological alterations and other anthropogenic impacts (e.g. port maintenance), and land-use patterns (the percentage area dedicated to/in use by each main activity) (IMPRESS, 2002; Borja et al., 2006). These pressures, alone or in combination, may cause (future) perturbations to the system, potentially leading to the failure of meeting the environmental objectives established by the WFD (Good Ecological Quality Status by 2015) (IMPRESS, 2002). The identified pressures for the Mondego system were allocated to 3 main classes (Figure 2) to facilitate the identification of the components that need to be managed for meeting 'sustainable use' of the natural resources: 1) Pressures on water supply (including a water services inventory); 2) Pressures on water demand (including a water uses assessment); and 3) Pressures on water quality. Also the defined main indicators were related to each activity to quantify the several pressures.

We evaluated several items and related socio-economic factors to the environmental assessment. A crucial distinction had to be made between water services and water uses. According to IMPRESS (2002), water services are intermediate between the natural environment and the water use. Stakeholders are an important component in the valuation process and must be considered. Their preferences for how water should be managed determine the planning and decision process.

Therefore, two inventories were performed: 1) water services (all services of extraction, storage, treatment and distribution of surface water or groundwater, including wastewater collection and treatment facilities); and 2) water uses (divided into industrial, agricultural and household). To infer the importance of water to the local human population and to the ecological maintenance, a supply and demand rate analysis of water uses was performed for the main consumers: domestic, agricultural, industry, and tourism. The analysis of agricultural use was based on irrigation for the basin (101,395 ha) and on the irrigation facility constructed to improve water efficiency in the Lower Mondego ('Aproveitamento Hidroagrícola do Baixo Mondego', or AHBM). The general data used for this section were obtained from INE (www.ine.pt), while the data on quantitative water extraction and its use for agricultural irrigation were obtained from INSAAR (http://insaar.inag.pt).

2.2.3. Ecosystem status assessment

Regarding the status of the system, two components were considered: the quality and quantity of water resources (Figure 2):

a) Water quality

In 1990, 1992, 1998 and from 2002 to 2006 sampling was carried out at 14 stations along the seaward part of the estuary which includes all ecological areas (Figure 1). The sampling areas were assumed to deliver information on two general indicators: water quality and ecological state. Water quality was analysed based on spring concentrations of dissolved nutrients (nitrate, nitrite and phosphorus, in mmol l^{-1}). Quantifications were made from surface and bottom-water samples in the main channels. Water nitrate (NO_3) and nitrite (NO_2) concentrations were analysed according to standard methods described in Strickland and Parsons (1972). Ammonium (NH_4) and phosphate (PO_4) concentrations were analysed following the Limnologisk Metodik (1992). Nutrient sources were determined following EEA guidelines ('European Environmental Agency') (EEA, 1999) prepared by the European Topic Centre on Inland Waters (ETC/IW). The subtidal soft-bottom benthic macroinvertebrate community was used as a proxy for the ecological status of the entire ecosystem. The focus relied on the community's responses to pressures on the environment. Benthic communities were chosen as indicators because biological communities are a product of their environment, and also because different benthic organisms have demonstrated to have different habitat preferences and pollution tolerance levels (Pinto et al., 2009). On top of that, the response of the biological communities of the Mondego Estuary to environmental stress have been studied for the past 25 years (Marques et al., 1997; Patrício et al., 2009), allowing inferences to be made on long-term responses to environmental change. The ecological status of the system was assessed using an integrative environmental index (BAT, Benthic Assessment Tool) already applied to the Mondego Estuary before (Teixeira et al., 2008, 2009). To evaluate the status of the ecosystem the BAT integrates three widely used but different metrics, 1) the Shannon-Wiener Index (H') (Shannon and Weaver, 1963), 2) the Margalef Index (d) (Margalef, 1968), and 3) the AMBI (Marine Biotic Index) (Borja et al., 2000). Following the WFD requirements (EC, 2000), the overall index values range from 0 (bad ecological quality) to 1 (high ecological quality). The ecological meaning of the values was subsequently expressed in one of 5 defined quality classes (Bad, Poor, Moderate, Good, and High).

b) Water quantity

Water quantity analysis focused mainly on the available aquatic resources versus their usage in the system over the years (water supply/demand relation). To undertake this analysis secondary data obtained from INE and INAG institutions (e.g. water volumes consumed by economic or domestic activities) were used.

2.2.4. Assessment of impacts

The effects that 'direct' and 'indirect changes' of the ecosystem status may have on human well-being were also considered. The system wide total economic value used includes two main value classes: the use and non-use values. The use values class represents the direct use values, derived from both extractive and non-extractive human activities within the ecosystem (e.g. fisheries); the indirect use values, represented by services provided by the system (e.g. recreation); and the option values, where humans maintain the option to use the system in the future. Finally, the non-use values are associated with the inherent value of the system (e.g. bequest value) (Table 1).

Table 1. Impacts assessment: Water resources total economic value (TEV): water use (direct, indirect and option) values, and non-use values (existence values) of different services/resources (adapted from Atkins et al., 2007); and most used valuation methods to determine water resources values (adapted from Young, 2005).

Values			Services/Resources	Method for valuation
Total Economic Value (TEV)	Water Use values	Direct Use value	Recreation	TC; BT
			Commercial fishing	MP*; PF
			Agriculture	MP; PF
			Households	MP*; DC; CM
			Industry	MP*; PF
			Salt-works	MP; PF; TC
			Urban supply	MP
			Drinking purposes	MP; DC
			Biodiversity uses	MP; RRC; BT
			Energy production (e.g. wind, wave, tidal, and thermal power)	MP; CM
			Wastewater assimilation	DC; MP*; BT
			Research/Education	CVM; BT
			Tourism/Ecotourism	TC*; DC; BT
		Indirect Use value	Aesthetic value	TC; DC; BT
			Recreational fishing	TC; CVM; DC; BT
			Human health	DC; BT
			Tourism/Ecotourism	TC; CVM; DC; BT
			Recreation	TC; CVM; DC; BT
			Biodiversity assets	CVM; RRC; BT
			Wastewater assimilation	DC; RRC; BT
			Landscape maintenance	CVM; BT
			Research/Education	CVM; BT
		Option value	Future uses as per direct and indirect use values	DC; CVM; RRC, BT
	Water Non-Use values	Existence value	Estuarine zone as an object of intrinsic value, as a gift to others, and as a responsibility (stewardship)	HP; TC; CVM; RRC; BT

Note: MP: market prices method; PF: productivity function; BT: benefit transfer; TC: travel cost; RRC: replacement and restoration costs; HP: hedonic pricing; CVM: contingent valuation method; DC: damage costs; CM: choice modelling; *used in this study.

As so, two main categories were considered (Figure 2):

a) Impacts upon use values

Two main parameters were considered for the valuation of water use values: the taxes (benefits) paid by some sectors, and the costs of treating water resources. These measures provide an overall estimation of the values that the local population pays for water services. Following Henriques and West (2000), the following parameters were taken into account in evaluating the costs and benefits of water resources: the benefits obtained (e.g. market price values) and the cost values (e.g. residual water taxes implemented). Environmental costs indicate the damage that water use imposes (e.g. reduction in the ecological quality of aquatic ecosystems) (WATECO, 2003). Likewise, resource costs represent foregone opportunities, which other uses suffer due to the depletion of the resource beyond its natural rate of recharge or recovery. Data on the investments, costs, and income of water supply and wastewater drainage and treatment were obtained from INE and were used as a proxy to estimate the water (financial) costs for 2006. To evaluate the urban water supply and its environmental costs, the figures of the investments in wastewater treatment service were used. The benefits were measured as infrastructure taxes (that internalise the indirect benefits of hydraulic infrastructure) and water extraction taxes (that provide price discrimination through net benefits for users, water availability, and user efficiency) (Henriques and West, 2000). These benefits were calculated by (Henriques and West, 2000)

$$T = A \times K_1$$

where T is the tax value; A is the water volume extracted (m^3); and K_1 is the final value of the water extraction per m^3.

Overall costs were calculated with residual water taxes (that claim to calculate the marginal cost of pollutant reduction and the adequate treatment level in wastewater treatment plants). This is given by (Henriques and West, 2000)

$$T = \sum_{i=1}^{n} p_i \times K_{4i}$$

where T is the tax price; p_i the annual rejected water volume because of any pollutant i; and K_{4i} the treatment cost.

b) Impacts upon non-use values (environmental integrity)

Focussing on water quality impacts evaluation, the main processes and their effects (direct or indirect) on the system status and on human well-being were qualitatively analysed. Using a continuous water quality/quantity analysis, a Spearman correlation was performed to test the relationship between main criteria selected from the estuarine DPSIR approach (e.g. tourism data or urban occupation) and the nutrient concentrations in the water column.

2.2.5. Socio-economic responses

A ‘response’ by society or policymakers is the result of an undesired impact and can affect any part of the chain between drivers and impacts (Turner et al., 2000b). Two main factors were considered (Figure 2):

a) Policies and directives

An assessment was performed of the main actions applied to the system in the past and the simulated system responses to the occurring environmental alterations. These results were transposed to the current problems in the system. Due to its mandatory nature, special attention was given to the WFD. Under the goal of achieving a good ecological status by 2015, the WFD demands the assessment of future trends in environmental conditions that should be performed by developing baseline scenarios which examine the consequences of current trends in population, economy, technology and human behaviour (EC, 2000; Trombino et al., 2007). Data from 2006 were chosen for baseline scenario development, due to its wide-ranging availability (covering all required fields). The baseline scenario analysis was conducted using the average difference between the main drivers and pressures in 1994 and in 2006, assuming that these ratios would not change until 2015.

b) Management actions

A conceptual overview of the main water services and uses, as well as their major trends over time, was set up to establish baseline scenarios for economic analysis. This approach allowed a full integration of ecological potential, natural variability and functioning while providing a comprehensive analysis of the water system and its importance to the local population. The ultimate aims were to contribute to knowledge of the system’s resilience and to make inferences for management-related issues that were or could be implemented on the system.

3. Results

3.1. Inventory and description of drivers

Table 2 shows the main considered drivers acting upon the system and the selected indicators used to quantify the pressures acting on the estuarine resources. The quantification of changes occurring on the system was given by the difference between the indicator value for the initial condition (1994 year) and for the final year (2006/07), allowing the calculation of the relative difference in the indicator value over the 14 years’ timeframe.

Table 2. Main drivers, selected indicators, dissimilitude values and % year between 1994 and 2007, for the different spatial scales (MB- Mondego Basin; CR- Central Region; LM- Lower Mondego; ME- Mondego Estuary).

	Drivers	Selected indicator	Region	Years		Dissimilitude (%)	%/year
Natural	NATURAL						
	Invasive species		MB, LM, ME				
	Extreme events						
	Floods	Precipitation (mm)	ME				
	Drought	Annual temperatures (°C)	ME				
Anthropogenic	SOCIAL						
				1994	2007		
	Population	Total population (nº)	MB	846000	879570	4.0	0.28
			LM	327770	332355	1.4	0.10
			ME	61830	63229	2.3	0.16
		Population density (hab/km^2)	MB	88.1	90.3	2.5	0.18
			LM	158.9	161.1	1.4	0.10
			ME	163.0	166.8	2.4	0.17
		Households number (nº)	MB	373796*	394549	5.6	0.40
			LM	115206*	122743	6.5	0.46
			ME	22976*	24548	6.8	0.48
				2002	2005		
		GDP per capita	MB	62.1	67.5	8.7	2.18
			LM	100.3	103.4	3.1	0.78
			ME	95.2	99.3	4.3	1.10
	Policies & institutional directives	Taxes and incentives; Municipal and oriented directives; Market trends					
	ECONOMIC						
				1999	2007		
	Agriculture	Employment (nº)	CR	330955	114528	- 65.4	- 7.27
		Explorations (nº)	CR	128119	96253	- 24.9	- 2.77
		Output (t)	CR	925227	875781	- 5.3	- 0.59
				1994	2007		
	Fisheries	Employment (nº)	ME	737	560	- 24.0	- 1.7
		Fishing boats (nº)	ME	319	211	- 33.9	- 2.4
		Output (t)	ME	12071	11008	- 8.8	- 0.6
				1998	2006		
	Salt-works	Area (ha)	ME	137	33	- 75.9	- 8.4
		Units (nº)	ME	83	15	- 81.9	- 9.1
		Production (t/year)	ME	1511	870	- 42.4	- 4.7
				1998	2006		
	Industry	Enterprises (nº)	MB	94630	87472	- 7.6	- 0.84
			LM	38158	38282	0.3	0.03
			ME	7625	6714	- 11.9	- 1.32
		Employment (nº)	MB	128472	236645	84.2	9.36
			LM	53391	95695	79.2	8.8
			ME	11500	19895	73.0	8.1
		Output (1 000 €)	MB	9038000	16209328	79.3	8.8
			LM	4166000	6530833	56.8	6.3
			ME	1029000	2041247	98.4	10.9
				2003	2007		
	Commercial harbour	Traffic (entrance nº)**	ME	269	363	34.9	7.0
				1998	2007		
	Tourism	Lodging capacity (nº)	MB	10485	11325	8.0	0.8
			LM	5426	3199	- 41.0	- 4.1
			ME	2339	1499	- 35.9	- 3.6
		Output (1 000 €)	MB	22052	36341	64.8	6.5
			LM	11704	21427	83.1	8.3
			ME	3271	6849	109.4	10.9
	MORPHOLOGICAL						
	Channel modification	Dredging volumes	ME	1 x 106 m^3.yr^{-1}			
	Physical barriers	Dams; Capacity/location	LM				
	ECOLOGICAL						
				1998	2006		
	Water extraction	Water volume (Hm^3)	MB	59,207	64,353	8.7	1.0
			LM	30,894	32,905	6.5	0.7
			ME	4,910	4,877	- 0.7	- 0.1
	Wastewater	Drainage (Hm^3)	MB	23,371	31,219	33.6	3.7
			LM	10,508	15,880	51.1	5.7
			ME	1,586	2,620	65.2	7.2
		Treated (Hm^3)	MB	18,573	19,539	5.2	0.6
			LM	8,146	6,084	- 25.3	- 2.8
			ME	0,539	2,474	359.0	39.9

Notes: * 2001 data; ** IPTM data (http://www.imarpor.pt)

a) Natural drivers

Within this category we have included invasive species that have caused substantial hazards in the system (Table 2). At the basin scale, the silver wattle (*Acacia dealbata*) threatens natural riparian vegetative communities (Costa et al., 2001). At the Lower Mondego scale, more specifically at the rice-fields, the crayfish *Procambarus clarki* was accidentally introduced in the early 1990s and severely damaged drainage systems and rice crops by their digging (Anastácio and Marques, 1996). The introduced mosquitofish (*Gambusia holbrooki*) feeds on eggs of economically desirable fishes, as well as on rare indigenous species (Mieiro et al., 2001). In the Lower Mondego and upstream areas, the introduced Asian clam (*Corbicula fluminea*) has a dispersion potential that may lead to changes in food webs, biofouling problems and competition with local species (Sousa et al., 2007; Miehls et al., 2009). The other natural driver considered were the extreme events occurring on the system (Table 2). Mean annual temperatures varied between 14.4 and 16.6 ºC, while mean summer temperatures ranged from 14.6 to 17.8 ºC, the warmer years being 1998, 2003 and 2005 (Figure 3, dark-Gray bar), along with low annual precipitations (characteristic for drought years). 1995, 1997 and 2000 were flood years with annual precipitations that ranged from1122 to 1378 mm (Figure 3, light-Gray bar).

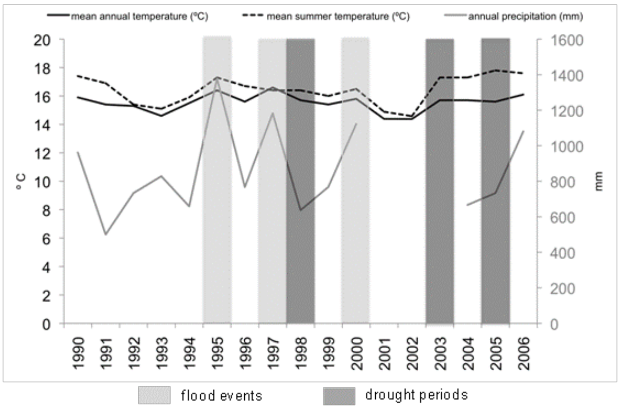

Figure 3. A. Mean (annual and summer) temperatures (ºC); and B. Mean annual precipitation (mm). The light-gray columns represent years with flood events; the dark-gray columns represent years with drought periods.

b) Anthropogenic drivers

Within the anthropogenic drivers, the following four subcategories were analysed:

a. Social drivers

There was an increase (2.3%) in the number of residents in the Mondego Estuary between 1994 and 2007 (Table 2). Although the Lower Mondego region growth rate was twofold that of the other regions, it still was half of the Basins' growth rate. Population density and household drivers (2.4% and 6.8%, respectively) showed the same pattern. These rates were below average across all scales assessed. Nonetheless, GDP (Gross Domestic Product) at the basin level was higher (8.7%) than at the other scales. Several actions have been implemented for local economic and social development. These implementations may have influenced the ecology of the ecosystem. Specifically, these were 1) reinforcement of the urban system; 2) diversification of tourism, with emphasis on ecotourism, and 3) reorganisation of rural areas.

b. Economic drivers

There was a reduction in the total number of explorations in all activities (by number of enterprises, number of explorations, lodging capacity, fishing boats and port entrances). This reduction was concomitant a reduction of the total economic production in primary sector activities. Nevertheless, the total output (by profits and employment rates) of the secondary and tertiary sectors appeared highly profitable (Table 2).

c. Morphological drivers

Through the years, several construction or engineering projects have been completed, such as the creation of artificial river banks, to mitigate the effect of floods, especially in the northern arm. Recurrent dredging and sand mining increase and maintain the channel depth (8-9 m depth below Chart Datum) to facilitate ship access to the commercial harbour (Cunha and Dinis, 2002), especially during winter when strong sedimentation may occur (1-4 cm year^{-1}) (Rocha and Freitas, 1998).

d. Ecological drivers

Water extraction and wastewater management were also considered. Although total wastewater drainage has increased over the years (Table 2) there was a slight decrease in water extraction.

3.2. Evaluation of environmental pressures

In order to evaluate the environmental pressures occurring on the Mondego Estuary, we have attempted to provide an explicit link between driving forces and pressures (measured through specific indicators such as nutrient concentrations or effluents produced) on the supply, demand and quality of water resources (Figure 2). Each of these pressures may have a positive or negative effect on the water resources condition, which consequently is going to determine the water status (either concerning quality or quantity).

3.2.1. Pressures on water supply

The Mondego Basin has highly diversified water services that are extremely important to the economic development of the region, such as water supply to households and urban systems or agricultural irrigation in the Lower Mondego region. In addition to this direct consumption, a wide range of services and goods depends, either directly (e.g. recreational activities) or indirectly (e.g. maintenance of biodiversity), on the aquatic system quality. The total annual rate of water extraction has increased from 1998 to 2006 in the Lower Mondego (30,894 to 32,905 Hm^3), while there was a slight decrease in the Mondego Estuary (4,910 to 4,877 Hm^3) (Table 2). Most of the water supply at the Lower Mondego region comes from surface sources, although there was a slight reduction from the surface sources contribution in 2006 (1998: 91.3%; 2006: 74.8%). This may have been a consequence of the AHBM water management project for efficient use of water resources. Moreover, in the Mondego Estuary, most of the water comes from underground sources (1998 and 2006: 60.7%) (INE data).

3.2.2. Pressures on water demand

In the Mondego system the main water users (in terms of total water volume) are population, agriculture, industry and tourism. Table 3 presents the water demand estimations by the main water consumers of this region.

Table 3. Water and wastewater prices, in 2006, by sector (Domestic, Agriculture, Industrial/Commercial/Tourism and Others): A. Water consumption and wastewater treatment prices for the different spatial scales (Mondego Basin, Lower Mondego and Mondego Estuary) (€/1000m^3) (source: AHBM-'Aproveitamento Hidroagrícola do Baixo Mondego'); B. Investments, costs and income of water supply (Hm^3) and wastewater treatment services (1000 €) by management operators, in Lower Mondego (INE data).

A. Consumption and wastewater treatment prices

Water consumption

	Sector	Mean supply tariff (€)	Mean water prices (€)	Water volume used (Hm^3)	Mean water extraction prices (€/Hm^3)
Mondego Basin	Domestic	146.40	0.615985	43 264	26 796
	Agriculture total	-	-	466 213	-
	Agriculture AHBM	-	-	-	-
	Industrial/ Commercial/ Tourism	167.34	1.2296	47 740	587 179
	Others	125.52	0.6104	0.113	194
Lower Mondego	Domestic	146.40	0.615985	19 742	12 307
	Agriculture total	-	-	241 233	-
	Agriculture AHBM	-	-	212 250	-
	Industrial/ Commercial/ Tourism	167.34	1.2296	36 377	447 459
	Others	125.52	0.6104	-	-
Mondego Estuary	Domestic	146.40	0.615985	4 677	3 028
	Agriculture total	-	-	43 310	-
	Agriculture AHBM	-	-	26 481	-
	Industrial/ Commercial/ Tourism	167.34	1.2296	35 763	439 910
	Others	125.52	0.6104	-	-

Wastewater treatment

Spatial scale	Mean wastewater supply tariff* (€)	Wastewater volume discharged (Hm^3)	Mean wastewater services prices (€/Hm^3)
Mondego Basin	6.14	499 423	3 068 625
Lower Mondego	6.14	227 857	1 400 031
Mondego Estuary	6.14	47 732	293 282

B. Investments, costs and income of water supply and wastewater treatment services (1 000€)

Water supply

	Investments (1 000 €)	Costs (1 000 €)			Revenue (1 000 €)		
		Total	General	Management & exploration	Total	Tariff	Other
Lower Mondego	12 651	18 741	3 869	14 872	24 234	22 578	1 657

Drainage and wastewater treatment service

	Investments (1 000 €)	Costs (1 000 €)			Revenue (1 000 €)		
		Total	General	Management & exploration	Total	Tariff	Other
Lower Mondego	21 822	13 963	2 390	11 573	8 582	6 942	1 640

Note:* Mean national values; - no data available.

Agricultural fields, by far the biggest demanders on water, are divided between rice (45%), corn (51%), and other crops (4%) together covering 12,546 ha (Costa et al., 2001). The agricultural sector has a number of potential impacts on water quality, not only as a water extractor, but also as a source of diffuse pollution (mainly nitrogen and phosphorous compounds, and pesticides). Per capita the household water consumption is one of the main drivers in the Mondego region (Mondego Basin: 42 m^3; Lower Mondego: 65 m^3; Mondego Estuary: 57.5 m^3) (INE data). With a population of 63,372 in 2006, the Mondego Estuary has a total household consumption of 3,643 Hm^3 (corresponding to 16.7% of the Lower Mondego water consumption and 9.8% of that of the Mondego Basin). Growth in households has increased over the study years and is expected to further increase the domestic demand for water. Around 97% of the Mondego Estuary population is served by municipal water supplies and roughly 87% benefit from drainage and treatment wastewater facilities (2006 data, INE data source). These values are higher than regional numbers (98%; 78%). Most households are connected to the main sewage systems ('Inventário Nacional de Sistemas de Abastecimento de Água e de Águas Residuais'-INSAAR data). Wastewaters suffer several treatments along the several considered scales (Table 4A).

Table 4. A. Volumes of treated waste water effluents (m^3), treatment type (total, primary, secondary, tertiary, and unspecified) and treatment points, in 2006 (INSAAR based data); and B. Mean effluents types and discharges (cumulative values among spatial scales; tons $year^{-1}$): Biochemical Oxygen Demand (BOD); Chemical Oxygen Demand (COD); Total Suspended Solids (TSS) at the different spatial scales (Mondego Basin, Lower Mondego and Mondego Estuary).

A. Effluents treatment types, volumes treated (m^3) and treatment points

Treatment type	Basin			Lower			Estuary	
	Volume	%	Treatment points	Volume	%	Treatment points	Volume	Treatment points
Total	15312102	-	676	5130214	-	74	2276908	8
Primary	3952632	25.8	573	429575	8.4	26	-	-
Secondary	4104338	26.8	68	1139900	22.2	16	-	-
Tertiary	506964	3.3	6	92141	1.8	4	-	-
Unspecified	678169	4.4	120	3468598	67.8	28	2276908	8

B. Mean effluents types and discharges ($tons.year^{-1}$)

	BOD			COD			TSS			Nitrate+Nitrite			Phosphates		
	Basin	Lower	Estuary	Basin	Lower	Estuary	Basin	Lower	Estuary	Basin	Lower	Estuary	Basin	Lower	Estuary
Domestic	15 235	6 347	1 439	34 279	14 281	3 238	22 852	9 521	2 159	2 031	846	192	381	158	36
Industry	10 537	9 054	8 107	29 529	26 635	24 761	5 675	4 188	3 550	-	-	-	-	-	-
Others point sources	1 399	0.4	0.4	3 443	1	0.9	3 000	1.3	0.7	209	0.1	0.1	70	0.0	0.0
Diffuse pollution	-	-	-	-	-	-	-	-	-	2 254	85	2.4	158	5.9	0.2
Total	27 171	15 401	9 547	67 251	40 917	28 000	31 527	13 710	5 710	4 494	931	194	609	164	36

Note: - no data available.

3.2.3. Pressures on water quality

A full inventory of pressures on the system was performed. This was based on official reports and local knowledge of system functioning, relating the 5 identified source types with pressures on water resources (demand, supply and quality). Change in land cover or land use, through land reclamation and use intensification, directly impacted water quality. These activities include land alterations to accommodate households, industries or infrastructures. The main consequences of these land modifications are soil sealing, decreasing soil permeability, and consequently higher peak runoff levels. The opposite effect (less pressure on water resources) is caused by the salt pans areas. This activity, due to its traditional way of salt-extraction, represents a ‘less negative’ pressure on the system. Despite this activity represents a direct human action and occupation of natural ecosystems, it is less invasive for the system quality than most of the other activities present on the system, and so may provide a positive signal on biodiversity and water quality.

Along with these factors there are also the natural characteristics of the system. Recorded sedimentation in the estuarine area has influenced the estuarine water circulation and has increased the water residence time (Marques et al., 2003). Although the level of water extraction has not changed over the years, wastewater drainage and treatment volumes have experienced substantial increases. Based on an approximately constant pattern of effluent emissions, wastewater treatment and population behaviour, the wastewater discharges into the (hydrologic) estuarine system were estimated (Table 4B; after Costa et al., 2001). The high contribution of domestic and urban effluents to water pollution is reflected by BOD (Biochemical Oxygen Demand), COD (Chemical Oxygen Demand), and nutrient inputs (dissolved nitrogen and phosphorous compounds). Industrial effluents strongly impact the water quality of the Lower Mondego and the Mondego Estuary (Table 4B). From data on total wastewater and treated volumes at different scales, it was verified that most effluents undergo secondary treatment. It was, however, not possible to reliably estimate discharges of all compounds at the scale of the Mondego Estuary.

3.3. Assessing the ecosystem status

a) Water quality

Two parameters were used to assess the status of the Mondego Estuary: 1) the water nutrient condition, and 2) the ecological quality. During almost the entire year, the estuary presented water stratification. Surface waters had consistently higher levels of NO_3-NO_2 and lower quality (worse classifications) than bottom waters (Table 5), suggesting that the main nutrient sources lie upstream the Mondego Estuary terminus. The South arm, a polyhaline muddy sand area, also had worse classifications for NO_3-NO_2 and PO_4 than the North arm. In this southern inland area, the water residence time was longer and agricultural runoff pressure was stronger than in the North arm. Runoff was extremely variable in the Mondego Basin, both intra-annually (winter months with high runoff) and inter-annually (e.g. 2000, a flood year, or 2005, a dry year, INAG data). Mean annual rainfall was 1,136 mm, with 720 mm going to evaporation and approximately 400 mm to runoff. Stations with higher water circulation generally had higher quality classifications, which allowed us to infer dilution and runoff capacities. Years with higher precipitation (Figure 3) showed lower water quality (Table 5), reflecting higher levels of pollutant runoff. In general, we observed poor classifications from 1992 to 2002, which then improved in the period 2003 to 2005 and worsening in the spring of 2006. Episodes of contamination (e.g. 2006) by paralytic shellfish poisoning have been caused by harmful algal blooms, which then led to stopping the shellfish fishery in the Mondego Estuary (IPIMAR data). Regarding the ecological conditions of the system in 1992, the ecological quality of the North arm strongly declined (Table 5). The ecological quality of the South arm declined until 1998. After 1998, following several experimental mitigation measures, the ecological quality status of the entire system began to show signs of improvement. On the whole, a gradual enhancement of the system has taken place (Teixeira et al., 2009).

Table 5. European Environment Agency (EEA) classification with respect to the Nitrate+Nitrite (NO_3+NO_2) and Phosphate (PO_4) water concentrations, in µmol l^{-1} (surface and bottom), as well as BAT assessment of Ecological Quality Status (EQS) based on macrofaunal communities, during spring months (April to June) from 1990 to 2006 in four estuarine areas (E–Euhaline estuarine; PNA–Polyhaline North Arm; PSSA–Polyhaline sand South Arm; PMSA–Polyhaline muddy South Arm). EEA classification: black: Bad; medium-gray: Poor; dotted light-gray: Fair; light-gray: Good. EQS classification: dotted medium-gray: Poor; medium-gray: Moderate; dotted light-gray: Good; light-gray: High.

	NO3+NO2							
	SURFACE WATER				BOTTOM WATER			
	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA
90					1.5	5.3	5.9	4.9
91								
92					5.9	15.1	16.0	11.0
93								
94								
95								
96								
97								
98					15.9	31.1	16.6	22.6
99								
00					1.5	23.1	18.4	16.5
01								
02					12.7	13.8	8.9	8.2
03	10.2	18.2	5.7	21.6	4.5	8.2	4.4	13.6
04	5.8	11.6	3.8	14.6	1.8	7.1	2.9	13.2
05	5.9	8.1	5.9	11.5	6.6	7.3	5.7	12.2
06	17.3	20.4	12.1	20.0	12.8	10.5	13.7	19.9

	PO4							
	SURFACE WATER				BOTTOM WATER			
	E	PNA	PSSA	PMSA	E	PNA	PSSA	PMSA
90					0.96	0.59	0.92	1.19
91								
92					7.21	12.47	15.52	11.75
93								
94								
95								
96								
97								
98					0.55	0.57	0.24	0.66
99								
00					1.02	2.25	2.08	2.15
01								
02					3.20	3.70	3.71	3.46
03	0.96	1.39	0.71	1.80	0.79	1.03	0.67	1.70
04	0.68	0.96	0.64	2.13	0.39	1.11	0.51	1.85
05	0.55	0.88	0.72	2.00	0.49	0.80	0.71	1.89
06	1.18	1.18	0.87	1.99	0.58	0.78	0.78	1.94

	Benthic Macrofauna (EQS)			
	E	PNA	PSSA	PMSA
90	M	G	G	M
91				
92	M	M	G	G
93				
94				
95				
96				
97				
98	G	G	P	M
99				
00	G	M	G	G
01				
02	G	G	M	G
03	M	M	M	G
04	G	G	M	G
05	G	G	G	G
06	G	M	M	H

b) Water quantity

The comparison between water usage (Table 3) and water availability (e.g. runoff parameters) enables to demonstrate that the actual volume of water provided by the river basin was sufficient to cover all the regional needs. These observations allowed for a dependency estimation that water quality and quantity have on the several pressures occurring on the system. It is indeed possible to demonstrate that factors such as floods and dry conditions may cause a high dependence on both water quality and quantity, while other factors (e.g. urban discharges) may have an opposite effect on both assets (e.g. high for quality and low for quantity).

3.4. Impacts assessment

3.4.1. Impacts upon use values (direct and indirect use values)

At all scales, economic sectors were the main water consumers (Table 3A). The current average selling price is approximately 1.23 €/m^3 for the domestic and industrial sectors (Table 3A), representing an average increase of 3.6% from 1999 to 2006. The total income of water supply activities exceeds the costs of extracting it (Table 3B). The quantification of the benefits related to irrigation constructions for agricultural use, was based on available irrigation networks, like the one in the Lower Mondego region (AHBM). This hydraulic structure has a maximum storage capacity of 500 Hm^3 and supplies water for several purposes (40% industry; 52% agricultural irrigation; 8% public consumption). It was built for a total of 675,000€ (1997 prices) (Costa et al., 2001). When only the financial costs and revenues achieved from drainage and wastewater treatment are considered then the costs by far outweigh the revenues for Portugal as a whole as well as the Lower Mondego (Table 3B). However, this estimate does not take into account, for example, the non-use values. Additionally, fisheries or recreational activities were also not considered in the cost recovery analysis because these activities do not involve extraction, regardless of the contaminants and pollutants that they may release into the water column (negative externalities).

3.4.2. Impacts upon non-use values (environmental integrity)

Along with the use values, the non-use values (essential to calculate the Total Economic Value) must be included in the analysis. However, due to data constraints it was not possible to calculate the values for all water-related assets. A Spearman correlation analysis was performed to test (at a significance level of $p<0.05$) the impacts between identified pressures and environmental assets (mostly nutrient concentration and physical parameters). The analysis crossed environmental assets (nutrients concentrations in the water column and water extraction) and drivers acting on the system (total population, urban land occupation, effluents produced, agricultural area and production, fisheries production and number of boats dedicated, industrial land area, commercial harbour entrances and dredging activities, salt-works area and production, tourists numbers). It was possible to demonstrate that effluents production (from the several sources) and the reduction of activities belonging to the primary sector had a significant influence on water status (at $p<0.05$). From this analysis, we could see important trade-offs and competing forces among activities in the estuarine area, especially between primary sector activities (such as fisheries or agriculture) and tertiary sector activities and social indicators (population indicators, tourist numbers, and effluent levels). A significant role of wetlands in highly nutrient-loaded agricultural catchments was also inferred. The existent positive relationship between organic matter percentage and agricultural production in estuarine areas, suggests that the parameter 'agriculture' may significantly impact water quality.

3.5. Socio-economic Responses

3.5.1. Policies and directives

The societal response emphasizes existing policies or programmes to reduce pressures and negative impacts acting on the ecosystem. A progressive increase in social drivers occurred during the study years concomitantly with a decrease in most economic drivers, especially those related to the primary sector (Table 6, 'Trends' column). Due to uncertainty in the estimated area of land use, changes in the area dedicated to these activities must be treated with caution. The available data reflect the estimated changes in land-use patterns for ecosystem services (baseline scenarios). The selected indicators showed that agricultural area occupies the largest portion of the estuarine area (Table 6). This area has, however, decreased at an average rate of 5% per year (Table 6, 'Trends' column). In contrast, the urban area has increased at an average rate of almost 8% per year (Table 6, 'Trends' column). Assuming these trends for 2015, it is possible to see that special attention has to be given to activities and pressures coming from the social drivers, water uses, and activities as tourism or even commercial harbour (Table 6, '2015 scenario' column).

Table 6. Baseline 2015 scenario for the Mondego Estuary region, following the 2006 observed data and posterior trends (GDP: Gross Domestic Product) considering selected indicators of natural and anthropogenic (social, economic and ecological) drivers.

Drivers		Selected indicator	1994 data	2006 data	Trends (%/year)	2015 scenario
Natural	Natural					
	Invasive species[3]					
	Extreme events[3]					
	Floods					
	Drought					
Anthropogenic	Social					
	Population	Area (ha)	661.7*	1 773.9**	7.9	6 110.4
		Total population (nº)	61 830	63 372	0.16	64 386
		Population density (hab/Km^2)	163	167	0.17	170.04
		Households number (nº)	22 976	42 685	0.48	44 734
		GDP *per capita*	95.2	99.3	1.10	110.2
	Economic					
	Agriculture	Area (ha)[1]	272 107.7*	124 917.2**	-5.2	12 491.7
		Employment (nº)[2]	330 955	114 528	- 7.27	31 266
		Explorations (nº)[2]	128 119	96 253	- 2.77	69 591
		Output (t)[2]	925 227	875 781	- 0.59	824 110
	Industry	Enterprises (nº)	7 625	6 714	- 1.32	5 828
		Employment (nº)	11 500	19 895	8.1	36 009.9
		Output (1 000 €)	1 029 000	2 041 247	10.9	4 266 206.2
	Salt-works	Area (ha)	137	33	- 8.4	5.28
		Units (nº)	83	15	- 9.1	1.35
		Production (t/year)	1 511	870	- 4.7	461.1
	Tourism	Lodging capacity (nº)	2 339	1 499	- 3.6	959
		Output (1 000 €)	3 271	6 849 000	10.9	14 314 410
	Fisheries	Employment (nº)	737	560	- 1.7	465
		Fishing boats (nº)	319	211	- 2.4	160
	Commercial harbour	Traffic (entrance nº)	269	363	7.0	617
	Ecological					
	Water extraction	Water volume (Hm^3)	4 910	4 877	- 0.1	4 828
	Wastewater	Drainage (Hm^3)	1 586	2 620	7.2	4 506
		Treated (Hm^3)	0.539	2 474	39.9	12 345

Note: [1] for the Mondego Estuary only; [2] trends for the Centre region; *1990 data; **2000 data.

3.5.2. Management actions

When considering the scenarios trends and expected values for 2015 (e.g. increase of wastewater volumes), and combining it with the system knowledge, efforts were dedicated to better understand the water dynamics and related ecosystem functioning, with the intention of preparing management responses. After several events leading to the degradation of the Mondego Estuary ecological condition (e.g. eutrophication in 1993), in combination with the freshwater flow interruption into the estuary South arm, responses were targeted to prevent further environmental problems, rather than being conventionally, reactive. The restoration of the *Z. noltii* habitat was one example: in 1986, the estuary had 15 ha of highly productive meadows, but by 1997-1998, this area was reduced to 0.02 ha. In 2005, after the implementation of several mitigation measures (1998), the *Z. noltii* area had recovered to about 4.2 ha (Patrício et al., 2009). Efforts have been dedicated to protect and restore this area, preventing future situations that could contribute to its further degradation or loss.

4. Discussion

4.1. Was DPSIR an appropriate tool to discern ecosystem-socioeconomic interactions?

While analysing the integral system by the DPSIR framework for the Mondego Estuary, a clear picture arose: there are no linear relationships or direct cause-and-effect patterns among drivers, impacts, and status; the interactions among them are complex and at least cumulative (Figure 4).

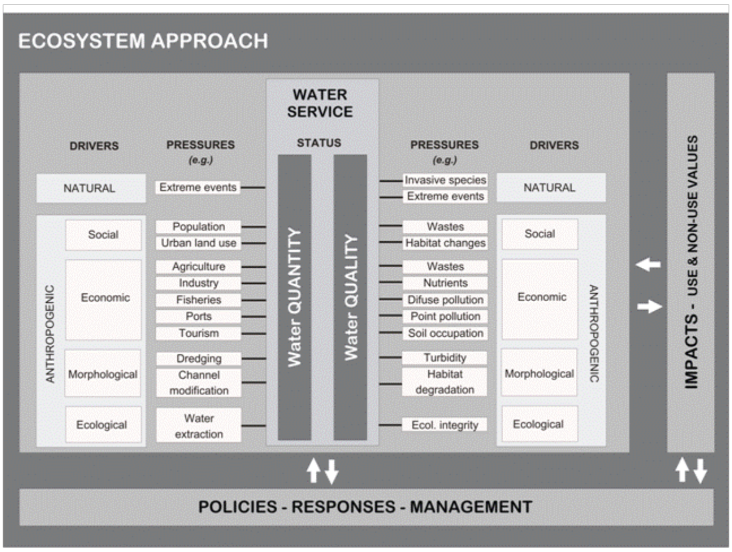

Figure 4. The drivers and pressures acting upon bio-physical processes can lead to changes in the systems' functions and so change the outputs quality and quantity, resulting in a wide range of environmental impacts. Two main types of drivers were considered, Natural and Anthropogenic. Within the Anthropogenic Driver 4 sub-classes were distinguished (Social, Economic, Morphological, and Ecological). Some examples of pressures are illustrated in the diagram, showing the connection and influence upon the water service status (both quality and quantity). All these variations in the water service status will be reflected in the use and non-use values of the system. Responses allow integrating all these measures and interests into the management action more suitable for the estuarine system.

The relationships, moreover, occur in addition to the natural variation (de Jonge et al., 2003; de Jonge, 2007). Three main sources of impact are acting upon the entire system. First, at the whole estuary level, diffuse pollution, is a major concern. Our case study clearly showed that the discharges might pollute, may be contaminate, the water courses and influence the estuarine water quality (illustrated by the significant Spearman relation between the produced volumes of urban wastes and the water quality). Inflows of water, nutrients and sediments from surrounding fields and activities greatly influence the overall water condition. Therefore, diffuse pollution, arising from catchment wide highly dispersed land use activities, can collectively have substantial impact on the ecosystem (WATECO, 2003). Discharges of contaminants and nutrients not only degrade the system (with an increase in primary production, higher oxygen demands and high organic matter contents) but also may lead to human health problems, such as contamination of (consumable and valued) bivalves. Second, at the North arm sub-system level, the physical interventions (sediment dredging and sand mining) are of particular importance. A mean annual dredging of about $1x10^6$ m^3yr^{-1} is considered necessary to maintain the Figueira da Foz harbour (Cunha and Dinis, 2002). Moreover, several barriers along the Mondego mainstream protect towns along the banks from floods; these include dams that control water runoff or excessive silting. Third, at the South arm sub-system level, the nutrient discharges may have important effects. Indeed, Baeta et al. (2009a;b) using stable-isotope analysis of the intertidal community has shown that the sources of nitrogen in this arm are coming from human activities. In fact, $\delta^{15}N$ values ranging from +10 to +20‰ in primary producers strongly indicate anthropogenic sources, whereas nitrate derived from atmospheric deposition produces values smaller than 6‰ (Kendall, 1998). Aiming to integrate impacts and possible solutions, the DPSIR framework can be an effective tool to communicate complex interrelations occurring on a system (Turner et al., 2000b). By its implementation we were able to identify the main relevant variables that can determine the systems' functioning and resilience ability. Nevertheless, and although it may be seen as simple to implement, this framework has revealed two main drawbacks: a) the weighting of pressures is difficult to estimate, not only due to multiple relations among factors, but also because it is difficult to determine which contribution may lead to what pressure (e.g. difficult to determine what nutrient percentage comes from agriculture fields, or from aquaculture, etc.); b) the cumulative interactions among the DPSIR categories are not fully taken into account (it is considered for the responses category, but there is not a clear relation among the other variables). Another criticism pointed out to the DPSIR approach is the absence of an explicit stakeholder role on the process, they may participate but the engagement cannot be described satisfactorily (Bruins and Heberling, 2005). In our study, the inclusion of the water uses and services approach within this framework was an attempt to suppress this drawback.

4.2. Is it possible to link wetlands functions and their economic valuation?

Water pollution from households and economic activities can cause severe degradation of water quality and can lead to significant changes in ecosystem structure and functioning and thus also functions. These changes and impacts ultimately reduce the overall ecosystem services. When the demand for certain services increases, human actions are often accompanied by the modification of ecosystems to increase their provisioning capacity. Although, in contrast to other estuaries worldwide with denser population and industry (e.g. Hu et al., 2001; Boyes and Elliott, 2006), the Mondego has been considered as a medium-sized estuary, nevertheless we recognized a strong social and economic functional dependence upon the estuarine region. The main aim of the valuation was then to indicate the overall economic efficacy of competing uses of the ecosystem resources.

In general, water consumption increased with population and GDP. From 1994 to 2007, and following national and international trends, there was a decrease in the commercial fish catches parallel to a decrease in the number of fishing boats (MEA, 2005). This represents a 24% decrease of people employed in the fisheries sector. Between 2003 and 2007, the ship traffic in the harbour increased by almost 35%, mostly for transport purposes, which, despite the decreases in the fisheries sector, led to higher pressure intensity to the system over the years. These intrinsic trades-offs among activities create several pressures and impacts on natural resources. As environmental assets in general and water resources in particular are becoming increasingly precious goods, there is a need for further regulation to avoid market failures. Likewise, the increasing watershed and aquatic activities taking place along the estuary lead to higher water extractions, and possible higher water stresses. The drivers' analysis showed that concomitant with population growth there was also an increase in economic activities related to the secondary and tertiary economic sectors. This was clearly visible in the increasing employment rates for the industrial sector (despite a decrease in the number of tourists and industrial facilities) and in overall profits (109.4% and 98.4%, respectively). This implies an increasing demand for food, water supply, water usage, and wastewater discharges. A total wastewater drainage increase over the years was in fact observed already as a direct result of an increase in the social and economic drivers. Contrary to this there was a slight decrease in water extraction, which may be explained by the decreasing importance of mainly agriculture activities (primary sector) at all the considered scales. A key approach for preserving the wetlands is to maintain the quantity and quality of the water on which they depend. The observed pressures on water quality and estuarine resources mainly result from the choices society makes to economically develop and to conserve the watershed. Thus, the valuation of resources involves identification of the changes in economic costs and benefits due to changes in environmental assets. The flow of costs and benefits over time is used to determine the asset value of the resource. Water services can enter into an individual's utility function directly through consumption, indirectly through the household production function, or as factor inputs in production (Aylward, 2002). According to Henriques and West (2000), the total cost estimate must be based on the rigorous calculation of water services, environmental costs, and scarcity costs. From this analysis, it was only possible to calculate the water services. Nevertheless, there is much uncertainty regarding environmental costs and scarcity (Birol et al., 2006), which contributes to the inefficient use of water resources (Henriques and West, 2000). Still, the main objective of water valuation initiatives is to guarantee that water users and the general population are involved in conservation, either through a water use fee or by taking direct action to reduce pressures on water resources. Despite the controversy, it can be postulated that through the attribution of an economic value to water resources could be an effective way to protect and manage it (Young, 2005b; Birol et al., 2006). Here the goal was not to attribute a total value to these resources because the intrinsic behaviour of biotic and abiotic components of water is too complex and intricate to be measured relying solely on a couple of ecosystem attributes. It is required to include in the future also other system components like e.g. fishes, plants, phytoplankton, etcetera.

4.3. Management concerns and recommendations

This study identified two main management concerns regarding the estuary:

1. An increase of economic activities that relies on a good system quality

- *Tourism/ecotourism activities*: A strong seasonal variation in local population densities is observed. During summer months (June to September) a 47% increase in the number of tourists occurs (INE data). This leads to extra system pressures, not only due to the extra loads of pollutants, but also due to physical perturbations of the system. Although there is no shortage of water for any type of use in the Mondego catchment area, water quality is a matter of great concern, not only because of the population growth, but also because of the increase in activities which depend on aquatic ecosystems. Ecotourism projects for example have been developed for Murraceira Island (mostly bird-watching and traditional salt-extraction tours). These activities claim to promote economic and social development while also maintaining the ecological processes, the species structure, and the system functions to society.

2. The water quality improvement or/and maintenance

- *Habitat condition*: A good system condition supports its optimal functioning and species structure. Based on the developed scenarios, the present impact on water quality can be expected to decrease, mostly due to management as human response actions, while changes in water allocation are more unpredictable. For example, the water turnover increase combined with a reduction in the nutrients discharges, mainly in the South arm sub-system, has created conditions for a better performance of the system. Furthermore, it was also verified that the decrease in agriculture might be compensated for by an increase in industrial activity and household consumption (both in water consumption as in effluents produced). With the agriculture activities reduction, it was expected to observe a reduction in the nutrient concentrations in the water column. However this was not observed, which reinforces the importance of activities trade-offs (e.g. agriculture reduction but industry and households increment) and even of the inherent ecosystem properties (nutrients remain in the sediments). River hydrology, water quality, and ecosystem integrity determine the catchment habitats', productivity, and supply. Any management action must take this reality into account.

- *Activities trade-offs*: Salt production can be regarded as a 'less aggressive' system pressure than others because it maintains the diversity of the waders that use the Mondego Estuary during breeding and migration (Lopes et al., 2000; Pinto et al., 2010). From the considered activities occurring, the salt pans offer the more stable situation when considering the human-ecosystem interface. This traditional activity is decreasing, as reflected in the area used and annual salt production. Nonetheless, this activity has a strong relationship with local fish farming, because areas for aquaculture are often converted salt-works. This physical replacement is a concern from the biodiversity perspective, because it implies extra, untreated organic loads into the estuary. Agricultural practices are also a major concern for ecosystem management. Intensive agriculture has strong environmental impacts through high fertiliser quantities, usually nitrogen compounds and phosphate, which promote rapid development of opportunistic macroalgae (Rocha and Freitas, 1998). This situation can lead to significant impacts on the native biodiversity because it reduces the natural barriers and intrinsic resilience of the system (Tilman, 1996; Rocha and Freitas, 1998).

- *Production of goods/human welfare*: A poor status of the water quality is going to influence and determine local/ traditional activities, that although may not substantially contribute to the national economy, may have a strong local importance (e.g. manual and commercial cockle harvesting). The decrease in production of these species (mostly *Cerastoderma edule*) affects the socio-economic conditions of a part of the local population, since this activity is a highly valued good.

In general we observed that ecologically the system has recovered over the years. However, in more detail, it is possible to see that the overall situation of nutrient concentrations has only slightly improved. This places the system into an elusive situation, where if by chance the mitigation measures do not work as predicted, it may return to the degraded initial condition. The implemented mitigation measures solved existing eutrophication symptoms, mainly by changing the water circulation inside the southern arm which resulted in a reduced water residence time. There were no changes observed into the agriculture practices, and nutrients remained at similar concentrations in the water column. Those solutions do not offer a final solution to the existing problems. Therefore, assuming that the goal is the 2015 established scenario (achievement of a Good Ecological Quality Status), and from the observed trends for both drivers/pressures and status evolution obtained from the DPSIR analysis, alternative recommendations of actions are required. Examples are:

1. Creation of buffer zones for the extraction of the nutrients added by mainly inadequate agriculture practices. Former agriculture fields could simply be used as water pathways where nutrients could sink/be removed from the water before it is discharged in the estuary. The application of the EcoWin2000 model for the Tagus estuarine system, for example, demonstrated that the nitrogen removal by salt-marshes (with a total area of approximately 8.2×10^6 m^2) is equivalent to the loadings from about 400,000 people (20% of total population in the estuarine surroundings) (Simas and Ferreira, 2007). This sort of option may contribute to reducing costs from wastewater treatment plants functioning;

2. Higher control and management of surrounding activities. To ensure that impacts and effects (e.g. nutrient loads) on the system remain within tolerable limits, the application of pro-environmental taxes to economic activities or even to the population involved in the production of effluents could be considered;

3. Promotion of pro-environmental activities that, although still using the system and occupying its areas, could have a lower environmental impact on the overall system quality (e.g. ecotourism activities or certifying of salt production companies), instead of invasive activities. A possible activity could be, for example, the cultivation of bivalves. Assuming that the capacity of the bivalves to extract nutrients from the water column is sufficient (Ferreira et al., 2007), a more intensive production of these assets would imply an added socio-economic value.

To adequately manage the ecosystem, policymakers should have to consider the conservation of the system assets, along with its sustainable use. Socio-economic and environmental compromises need to be made, aiming at system preservation along with efficient supply of services. The development of scenarios might be another useful tool for water resource management in order to: achieve efficient supply and allocation of resources while guaranteeing their rational use; promote sustainable exploration of existing resources; and minimise the direct and indirect costs associated with its use and conservation, while assuring overall economic development (Manoli et al., 2005). Changes in land use, future development and urbanisation pressures, and the increased use of water, as shown by the baseline scenario, may result in threats to the estuarine environment. However, it should be highlighted that external factors, such as the current economic crisis, were not considered in the scenario development and that this may have influenced the final outcome as well. It is important to regulate new pressures to ensure that no deterioration in status occurs, while minimizing present pressures. Continuous monitoring and controlling of pollution within aquatic ecosystems is essential to facilitate the selection of the most reliable and efficient methodology. The overall societal response to water pollution and nutrient enrichment is characterised by several conflicting factions, ranging from farmers to public drinking-water constituents, and from water-related recreationalists to environmentalists. Among this wide range of interests, policymakers at different administrative levels should try to find a compromise between both private and public goods for an efficient and consensual water management program. Understanding the complex and intricate trade-offs among ecological, social, and economic goals is fundamental in designing and implementing management policies and for ecosystems restoring (Marques et al., 2009).

5. General conclusions

Analysis of the relations between the main services and uses provided by estuarine systems assessed that the factor water quality improvement was most strongly influencing estuarine functioning within the related ecological and socio-economic spheres. The results illustrate that the integration of information by applying the DPSIR approach provides a common framework of analysis that benefits 1. better management actions, 2. Weighting activities trade-offs (e.g. salt pans and aquaculture), 3. Societal actions to be taken, 4. gap analysis studies, through the identification of the major driving forces acting on systems, 5. ways to deal with it, and 6. execution of monitoring to follow it.

The application of the DPSIR approach identified two main future research topics 1) The need for a quantitative ecological approach (e.g. catchment model) including the determination of flows and values of each service in addition to the present qualitative one (which factors and how they interact); and 2) The need for integrating more components of the integral system than we did so far (e.g. fishes, geographical, cultural values) and that strongly contribute to the estuarine functioning.

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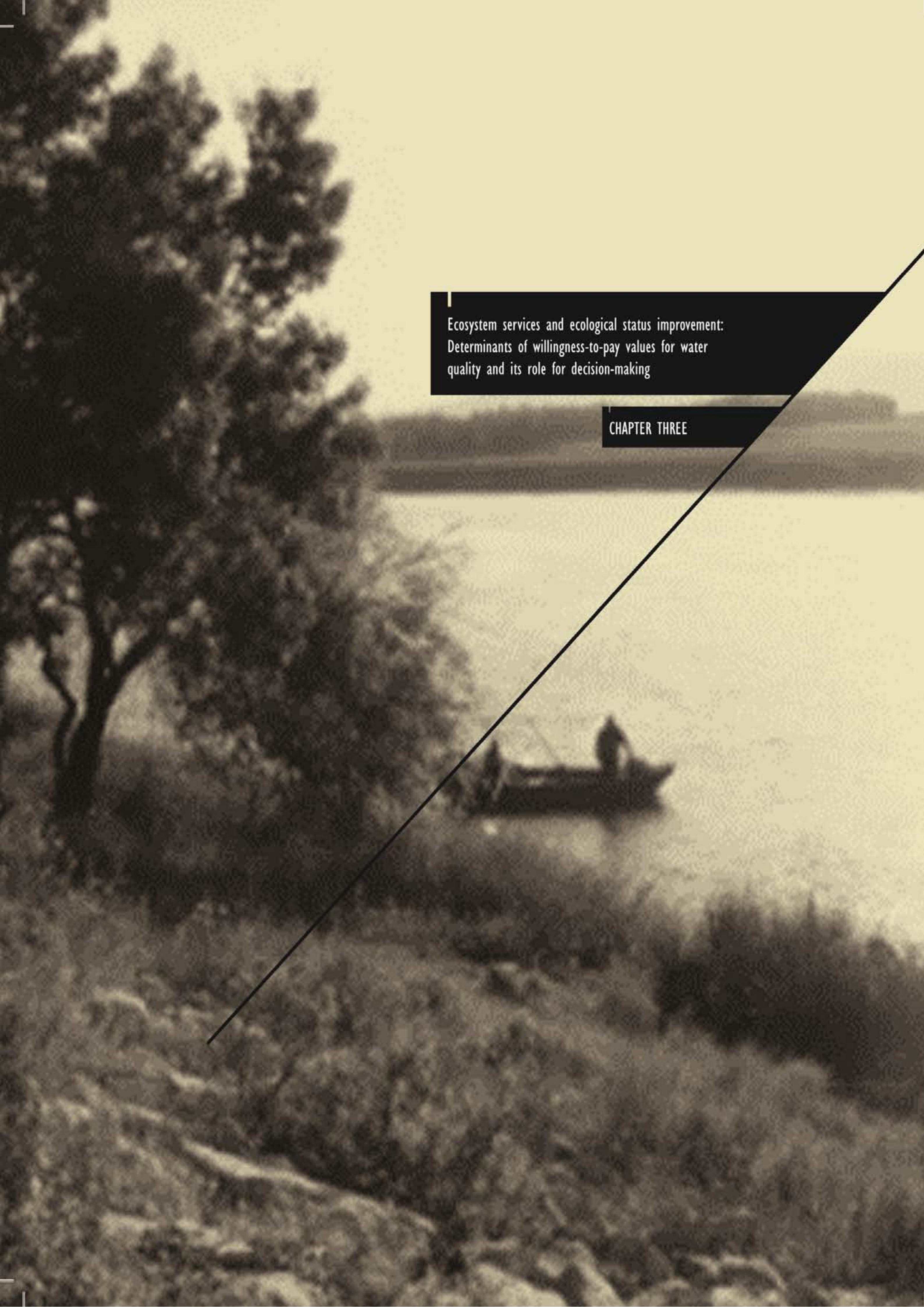

Ecosystem services and ecological status improvement: Determinants of willingness-to-pay values for water quality and its role for decision-making

CHAPTER THREE

Abstract

Due to the increasing demands and pressures on water resources, there is a growing need for efficient and effective implementation of management policies, as highlighted by the European Water Framework Directive (WFD). The purpose of this study is to evaluate the awareness of individuals through a contingent valuation (CV) study, estimating the benefits of water quality improvements in a watershed (Mondego Basin, Portugal) and to examine how information from CV surveys may help the design of management strategies and policies. Two CV surveys (one considering the whole Mondego Basin and another considering just the estuarine part) were undertaken and respondents' perceptions and willingness to pay (WTP) regarding water quality improvements and an ecotourism facility were evaluated. This valuation approach was tested along four geographical sampling levels, to infer the distance-decay effect on respondents (distance from their residence) and the goods elicited. Estimates indicate a WTP around 30€/year per household to achieve a very good water quality status and an ecotourism centre development, with somewhat lower values to achieve good (around 10€/year) or very good (circa 20€/year) water quality levels. Our findings identify that both use and non-use values are reflected in respondents´ WTP, showing: (1) a strong relation between socio-economic respondents´ profile (e.g. income, education or number of household members) and WTP for environmental quality improvements; (2) that the distance-decay effect and usage of the system had a significant influence on respondents´ WTP; (3) a substantial positive social awareness for environmental issues (e.g. anthropogenic drivers and pollution causes) and economic values (both marketable and non-marketable) attached to ecosystem services of aquatic resources. From the achieved results, a set of recommendations are made for decision-makers towards system conservation and management, acting as an effective instrument to already existing EU policies, namely WFD.

Keywords:
| Ecosystem services
| Water quality
| Contingent valuation method
| Distance-decay effect
| Protest effects

1. Introduction

1.1. Water quality and watershed management: the ecosystem services approach

The structure and functioning of an ecosystem is sustained by synergistic feedbacks between organisms and their environment determining ecosystem properties and setting limits to the types of processes occurring there (Mace and Bateman, 2011). Moreover, ecosystem responses to environmental change may quite commonly be non-linear, difficult to predict or even irreversible (de Jonge, 2007; Carpenter et al., 2009). Therefore, it becomes essential to understand the basis of complexity to guarantee the effectiveness of response actions and to ensure the continuation of important services provision. An illustrative example of this complexity (and how physical, chemical and biological assets underpin the ecosystem functioning) is for example the reduction, or even disappearance, of macrophytes or seagrasses caused possibly by competition with green macroalgae, as illustrated in the Mondego Estuary (e.g. Marques et al., 1997, 2003; Patrício et al., 2009). These situations often are produced by nutrient enrichment of the water column, caused by anthropogenic activities (e.g. agriculture fields) combined with high water residence time and good light conditions. The replacement of these habitats and communities can lead to changes in ecosystem functions and trophic structure (e.g. uncoupling of biogeochemical sedimentary cycles or even changes in the abundance of benthic fauna) (Marques et al., 1997; Valiela et al., 1997; Baeta et al., 2009). The ecological functioning of aquatic ecosystems, and more specifically estuaries and river watersheds, can provide important benefits to humans, including socio-economic benefits.

Regarding water quality and ecological functioning evaluation of a system, is necessary to include not only local conditions (including the surrounding human activities), but also the conditions and pressures within its upstream watershed (Pinto et al., 2011). In fact, the watershed has been widely acknowledged to be the appropriate unit of analysis for many water resources planning and management (McKinney et al., 1999; Mirchi et al., 2009). Therefore, an accurate analysis of the biophysical knowledge of a system is essential to allow decision-makers to understand the dynamics and complex interactions between the environment and diverse socio-economic contexts (Elmqvist et al., 2010).

The driving forces underneath environmental degradation can have multiple and synergistic impacts on the system. This kind of situations might create massive pressures upon natural resources (due to activities expansion, development, nutrients inputs, among others), impacting around 90% of previously important species and destroying circa 65% of seagrass and wetland habitats, while degrading water quality and accelerating species invasion in estuaries (Lotze et al., 2006; O'Higgins et al., 2010). Assuming the Ecosystem Approach holistic perspective (Maltby, 2006; de Jonge, 2007; de Jonge et al., in press), ecosystem services can act as the link between natural assets and human benefits. This approach defends an integration of the ecological, economic and socio-cultural perspectives when valuing a system (de Groot et al., 2002; Farber et al., 2002; MEA, 2005), providing a methodological framework for the integration of wetland management (de Jonge, 2007).

It has been argued that the Water Framework Directive (WFD) implementation, besides the benefits related with an integrated approach to water preservation and management, imposes direct costs on society, such as costs of treatment, mitigation and/or compensation (Elliott and de Jonge, 2002). The willingness to pay (WTP) values attributed to a good can be viewed as a monetary value of the benefits that beneficiaries' in a target area derive from environmental and, in particular, water quality improvements or maintenance. A study conducted at the Serpis River Basin (Spain) by Del Saz-Salazar et al. (2009), comparing the economic valuation of the non-market benefits derived from a hypothetical improvement of the water quality with the population WTA compensation if the projected measures were not implemented (therefore comparing WTP and WTA estimates), concluded that, from a social point of view, the WFD implementation seems to be advantageous. Therefore, water quality improvement measures appear to be very desirable for both public bodies involved in water management and for the main beneficiaries of these water policies, reinforcing the Contingent Valuation role into management actions planning and implementation, confirming its importance in assessing non-use environmental values (Birol et al., 2006; Del Saz-Salazar et al., 2009).

In this context, this paper applies the Contingent Valuation method to explore public preferences for water quality improvements. The investigation of these preferences is important once that the development of EU water legislation, like the WFD, is imposing significant costs on society (Elliott and de Jonge, 2002). Therefore, it becomes crucial to estimate the social awareness for water-related environmental problems and the economic importance of water-quality improvements to human well-being. In order to fulfil this goal, the Mondego river catchment area, Portugal, was used as an illustrative exercise and the available ecological knowledge of the area was taken into account to conduct a socio-economic filed survey.

1.2. Contingent Valuation method: pros and cons

For some services, such as water quality, regular markets do not exist or fail to accomplish the service's full value. In this sense, methods have been developed for the valuation of such nonmarket goods using both direct and indirect means of measurement, like the contingent valuation method (hereafter CV). This method is a stated preference technique frequently used to estimate the value of a broad range of ecosystem benefits (Carson, 2000). It has a wide spectrum of action, being capable of measuring both use and non-use values (Mitchell and Carson, 1989; Spurgeon, 1992) and being especially useful to estimate public goods values (Wattage, 2002; Young, 2005), such as water resources and its associated environments (Young, 2005). CV methods have been widely used to examine water quality improvements, in several contexts (e.g. Mitchell and Carson, 1989; Söderqvist, 1996; Gren et al., 2000; Atkins and Burdon, 2006). CV studies allow to estimate respondents' preferences by directly asking how much they would be willing to pay (WTP) or willing to accept (WTA) for the change in the provision of a good or service (Mitchell and Carson, 1989). CV relies on the construction of a hypothetical market for the surveyed good. Thus, the elicited WTP amount is contingent upon the hypothetical market presented to the respondent (Mitchell and Carson, 1989; Wattage, 2002). Hence, through CV surveys implementation is possible to examine irreversible changes on a system provision capacity and to evaluate the direct (e.g. recreational fishing) and indirect use values (e.g. improved water quality), while also promoting the measurement of the associated option use values and non-use values of a system (Birol et al., 2006).

Despite its potential to estimate the value of ecosystem benefits whose value is not revealed in conventional markets, CV is not exempt from problems and has been criticised for its lack of validity and reliability (Kahneman and Knetsch, 1992; Diamond and Hausman, 1994). Among the most common pointed biases are (for more details see Mitchell and Carson, 1989; Bateman and Willis, 1999): (i) the survey design and implementation imprecision's (Young, 2005); (ii) inadequately perceived budget constraints making responses hypothetical (Seaman, 2006); (iii) the possible unreliability and expensiveness of implementations (Venkatachalam, 2004); or even (iv) the embedding effects that can be created (Mitchell and Carson, 1989). The embedding effect has been used to describe how WTP for a specific good may vary "*over a wide range depending on whether the good is assessed on its own or embedded as part of a more inclusive package*" (Kahneman and Knetsch, 1992).

Another issue that requires attention is the potential effect that protest responses may have on the final valuation results. Respondents' refusal to pay (zero bids) for certain assets' conservation or improvement can be expressed in two ways: true zero values or protest responses. True zeros occur when individuals are indifferent to whether the good is provided or not or if the income constraint is binding (Brouwer and Slangen, 1998; García-Llorente et al., 2011). Protest zeros occur when a respondent gives a zero bid response to a question, even though the good may have a positive WTP value for him (Brouwer and Slangen, 1998; Carson, 2000). Despite the difficulty, and lack of precise guidelines to make this distinction (Boyle and Bergstrom, 1999), the way of differentiating a true zero from a protest response is to question the underlying reasons of the respondents' unwillingness to pay, and based on their answers, to decide for each zero whether it is a true economic zero or a protest against the valuation scenario (García-Llorente et al., 2011). It should be highlighted that these responses may have a very strong role in the final WTP estimates (Mitchell and Carson, 1989; Jorgensen et al., 1999), and so it becomes essential to accurately identify protest responses (Dziegielewska and Mendelsohn, 2007; Meyerhoff et al., 2009) and explore how protest responses are motivated (Jorgensen and Syme, 2000; Meyerhoff and Liebe, 2006; García-Llorente et al., 2011).

To minimise these issues, several suggestions regarding the best practice guidelines for the design and implementation of CV studies have been recommended (Blue Ribbon Panel; Arrow et al., 1993). Therefore, due to the inherent properties of the CV methods, several studies have been conducted aiming to contribute to a better management and conservation of aquatic systems (e.g. Loomis, 1998; Birol et al., 2005; Atkins and Burdon, 2006; Atkins et al., 2007), based on the assumption that CV studies can produce estimates reliable enough to be the starting point for a judicial or administrative determination of natural resource damages, including lost passive-use values (Arrow et al., 1993).

1.3. The study-site: the Mondego catchment area

1.3.1. Mondego Basin

The Mondego River, located in central Portugal (39º46' and 40º48'N; 7º14' and 8º52'W; Lima and Lima, 2002), has a catchment area of approximately 6670 km^2, and its basin is highly diverse in topography, hydrology and land use. Its functional structure ranges from mountainous areas to a large alluvial plain discharging into the Atlantic Ocean (Marques et al., 1997; Graça et al., 2002). The Lower Mondego region (with a total area of 250 km^2) connects the mountain river with the ocean and consists of open valleys and plains. This region also includes the Mondego Estuary (Figure 1). The Mondego is the largest Portuguese river which entire watershed is contained in national territory (without transboundary limits or constrains).

Figure 1. Mondego watershed system and the several geographic sampling levels of the study: 1. Mondego Estuary (ME); 2. Lower Mondego (LM); 3. Mondego Basin (MB); 4. Portugal (Pt).

The Mondego Basin has a highly natural variability in environmental and social conditions. The downstream portions of the catchment area are densely populated while the upper and middle basin regions have low to moderate human impacts (Feio et al., 2009). The basin, being densely occupied and used, supporting over half a million inhabitants (Marques et al., 2003; Pinto et al., 2011), covers a wide range of uses, ranging from intensive agriculture to industry. The whole system is characterised by activities belonging to the secondary and tertiary economic sectors, however, in the terminal part of the system (Lower Mondego region) a strong pressure from activities belonging to the primary sector are also felt (Pinto et al., 2010). Other impacts are dams and barrages built along the river course that may influence the environmental quality of running waters (Feio et al., 2009). Among these water uses, some can have a direct effect on water quantity and quality, such as the agriculture fields or industry (both through water extraction and wastewater discharges); while other uses, such as tourism/recreation activities (e.g. sport fishing or canoeing), have an indirect impact on the system quality. Overall, the water quality of the Mondego watershed has been classified as Moderate (INAG 2009 data). Nevertheless, data review from several studies conducted on the system allow saying that water quality has been improving over the years, allied to biodiversity indicators improvement (Marques et al., 1997; Graça et al., 2002; Feio et al., 2007; Neto et al., 2010).

1.3.2. Mondego Estuary

The terminal part of the basin comprehends the Mondego Estuary, with a total area of 7.2 km^2 (Figure 1). In this region, activities belonging to the primary sector (e.g. agriculture and fisheries) and touristic activities play a major role, while supporting higher population densities (around 167 inhabitants km^{-2}) than the rest of the basin (circa 90 inhabitants km^{-2}). Moreover, the water flowing into the estuary has been loaded for decades with nutrients, mainly due to the upstream-downstream effects caused by activities taking place along the river watershed (Pinto et al., 2011). In the Lower Mondego, strong pressures are caused by the primary economic sector (15,000 ha of highly productive agriculture of mainly rice) and by harbour-related activities in Figueira da Foz.

1.4. Study objectives

The main objective of this study was to estimate the social awareness for water-related environmental problems and the economic importance of water quality improvements to human well-being, measured at a local, regional, basin and national scale. Moreover, it was also intended to examine the practicability of environmental valuation (given all its demands, such as, time dispended, monetary and human resources allocation) in ecosystem management contexts. Given this context our study aims at:

- examining the awareness and attitudes of respondents towards the preservation and use of wetlands;
- analysing the willing to pay responses:
  - to assess the motivations behind individual valuation, what will allow to explain differences across individuals and regions;
  - to estimate the WTP of residents in the Mondego Basin and Portugal as a whole for water quality improvements in a national river. In particular, aiming to assess:
    - » whether the value attributed to water quality improvements considering the Mondego Basin is significantly different (higher) from the improvements value given to a small subset of that basin, more specifically the Mondego Estuary (to test the 'embedding' effect of the analysis); and
    - » whether the value of water quality changes significantly varies across different surveying geographic sampling levels (distance-decay analysis);
- analysing the responses of those that refuse to pay, more specifically to estimate the role of the zero bids and protest responses on individual refusals (protest effects analysis);
- investigate the feasibility of applying non-market valuation techniques to estimate the value of water quality improvements; and
- proposing guidelines for structuring and conducting management actions, based on the information gathered by the present study.

2. Methodology

2.1. Contingent Valuation Survey

To fulfil the study's objectives, a field survey was conducted in February and March 2011, after a pre-test in December 2010. The Winter of 2011 was selected to conduct the surveys to avoid the touristic period, since local populations were the main target. The interviewers were six members of the IMAR team that were aware of the dynamics of the system.

The analysis relied on the hypothetical accomplishment of improving the water quality of the Mondego catchment area from moderate (assumed as the current condition) to very good status, together with developing recreational facilities. The underlying idea was to create a hypothetical market where the good in question (in this case water quality improvement) could be traded. In this sense, at four geographic sampling levels, it was directly asked to local populations to express their WTP for a hypothetical change in the level of provision of the good.

Different survey modes were applied at the four geographical levels. At the Mondego Estuary (ME), Lower Mondego (LM) and Mondego Basin (MB) geographical levels were carried out in-person interviews. At the national geographical level (Portugal), mail was the instrument chosen, due to time and human resources constraints. Both methods have pros and cons, the mail method gives time to think and reduce the pressure on respondents, while the in-person method may be better suited for recalling past situations (Mannesto and Loomis, 1991; Marta-Pedroso et al., 2007). In general, the authors conclude that both methods provide similar results. In this work, it was assumed that the use of different methods could cause bias on the achieved outcomes, and the mail surveys were mainly used to provide an indicative value of the system in a national context.

The questionnaire was divided into three main sub-sections: (i) socio-economic data, (ii) awareness and uses of natural areas, and (iii) valuation of water quality improvements. The survey was based on a survey previously implemented in Denmark (Hasler et al., 2009), that was adapted for the Mondego catchment area. Both introductory and follow-up questions (sections i and ii) helped to reveal respondents' relations and attitudinal behaviours to the good in question and provided input to identify the driving social variables underneath their relation with the environmental resources (Figure 2). Besides socio-economic information, respondents were asked about their recreational habits (how often they used natural areas for recreation and which activities they usually undertook), their familiarity with the system (never heard, knows but has never went there, use it often), ecosystem activities/uses (taxes for recreational activities, e.g. marina, fishing, etc.) and awareness (estimation of the perception level to pollution, identification of main pollution sources and impacts of activities).

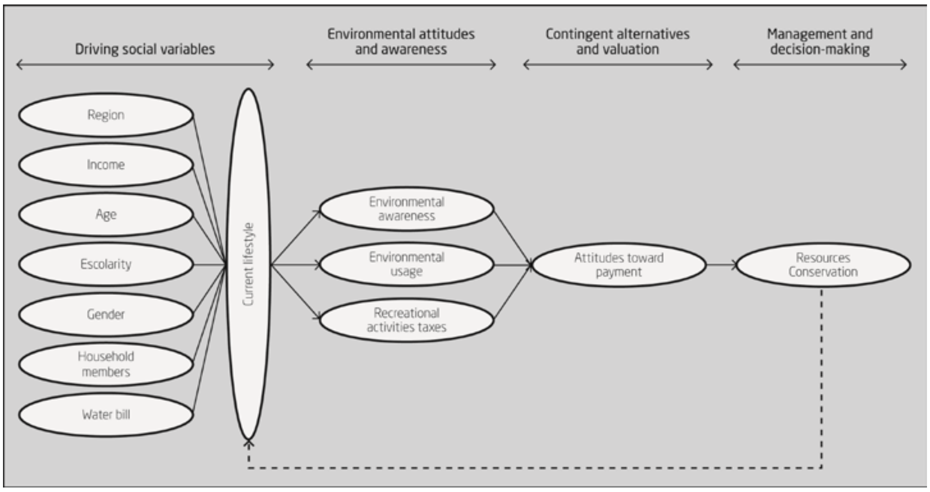

Figure 2. Relation among factors (driving social variables) conditioning environmental awareness, and the respondents attitudes toward payment under the contingent alternatives presented, aiming at the resources conservation.

To test the distance decay effect in the economic value of water quality improvement in the Mondego catchment area, a contingent valuation survey of randomly selected households at four nested geographic sampling levels was carried out. The considered geographic sampling levels were the Mondego Estuary (ME) (including 1, out of 1, municipality), Lower Mondego (LM) (comprehending 7, out of 8, municipalities), Mondego Basin (MB) (integrating 26, out of 36, municipalities), and Portugal (Pt) (incorporating 87, out of 279, municipalities) (Figure 1). Although randomly obtained, the achieved household sample was not exactly representative of the socio-economic characteristics of the populations. To test if the geographic area of the system to be improved may influence the respondents' perception and attitudes, two questionnaires were conducted: one focusing on the entire Mondego catchment area, and another one only focusing in the terminal part of the system (corresponding to the Mondego Estuary). A total of 1259 households were sampled, with half of the respondents being asked to state their preferences regarding the Mondego basin (hereafter BV) and the other half regarding the estuary (hereafter EV). These two questionnaires were conducted at the four geographical levels. To test the potential effect that geographical scaling might have on responses, and to test if differences observed between WTP values (and availability to contribute) were significant between samples (H_0: WTP values or availability to contribute equal for the 4 geographical scales), Chi-square tests and one-way ANOVA were performed, using the SPSS 19.0 software package. For the WTP analysis and to accurately estimate the distance decay effect for the system improvement, the four scales were assumed as Mondego Estuary (ME_{km}): 0 to 21 km; Lower Mondego (LM_{km}): 22 to 40 km; Mondego Basin (MB_{km}): 41 to 234 km; Portugal (Pt_{km}): area outside the Mondego Basin.

2.2. Contingent Valuation Alternatives

Regarding the contingent valuation alternatives, the system's Current Condition was defined as a Moderate water quality status (*sensu* WFD), with good conditions for fishing and boating, although with restricted access and surrounded by agriculture fields. Based on this, three alternatives were given to respondents:

Alternative 1 (A1): the water quality of the system improved a little (Good status *sensu* WFD), with very good conditions for fishing, boating and bathing, with good access (e.g. roads or tracks to reach the wetlands), and surrounded by green areas (acting as buffer zones);

Alternative 2 (A2): the water quality of the system improved substantially (High status *sensu* WFD), with excellent conditions for fishing, boating and bathing, with good access and surrounded by green areas (acting as buffer zones);

Alternative 3 (A3): the water quality of the system improved substantially (High status *sensu* WFD), with excellent conditions for fishing, boating and bathing, with good access and surrounded by green areas. A3, being similar to A2 in environmental terms, included the extra measure of creating an ecotourism centre, and other related infrastructures, that would increment tourist visits.

The elicitation process started with a question to determine whether the respondents were willing to pay to improve water quality on the system (Figure 3). Using a dichotomous choice format, an improvement alternative was presented to respondents and they were asked to attribute a value to it (see Bishop and Heberlein, 1979). An increase in the annual water bill was used as the payment vehicle for both surveys (annual tax added to the water bill), assuming categories of taxes that ranged from 0.50 to >200€. The respondents accepted or rejected the alternative, and those expressing a positive WTP were faced with another valuation question (concerning the following improvement alternative). At the end respondents' answered a de-briefing question to evaluate the underlying reasons for their decision (Figure 3). Those respondents expressing to be unwilling to pay were asked to enumerate the main reasons for not contributing, in order to allow us to identify and distinguish protest from true zero votes.

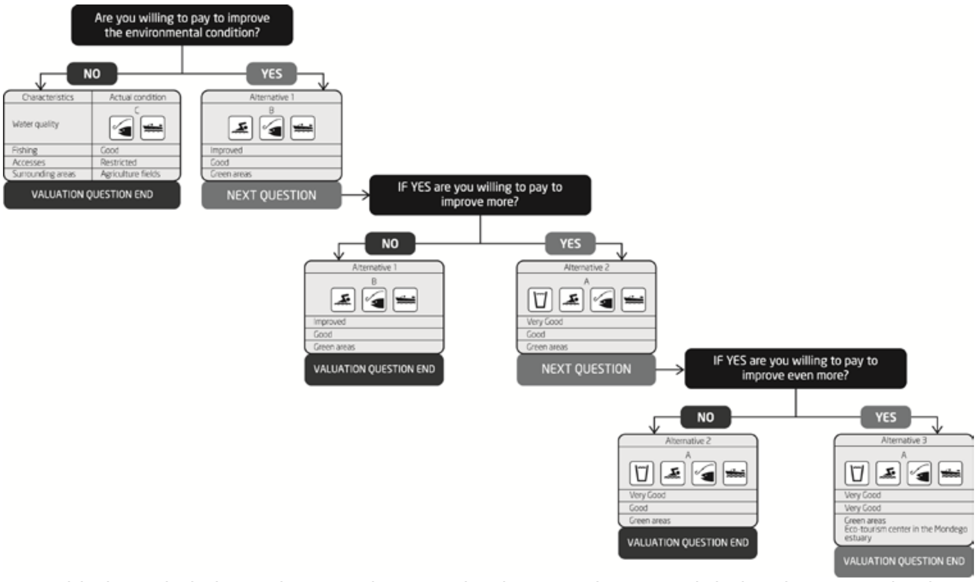

Figure 3. Double-bounded alternatives contingent valuation questions, used during the survey implementation (images were adapted from Del Saz-Salazar et al., 2009).

Since the values attributed might vary over respondents, the population mean WTP was estimated (Hanemann, 1984) for the four geographic sampling levels. For this, the average point of the considered payment card categories was considered. Even though some authors consider that the dichotomous choices format may induce some bias on the analysis (e.g. Mitchell and Carson, 1989), it is recommended by the NOAA experts panel (Arrow et al., 1993) and frequently used (e.g. Hasler et al., 2009), because it is argued that this format is closer to the economic decisions made by individuals in everyday life (Hanemann and Kanninen, 1999).

3. Results

3.1. Respondents' socio-economic profile

From 859 attempts, a total of 832 in-person interviews were conducted during the survey, corresponding to a response ratio of 96.9%. Regarding the mail survey (at the Portugal level), a lower response ratio (76.8%) was obtained, where 307 surveys were successfully returned out of 400 attempts (Figure 4).

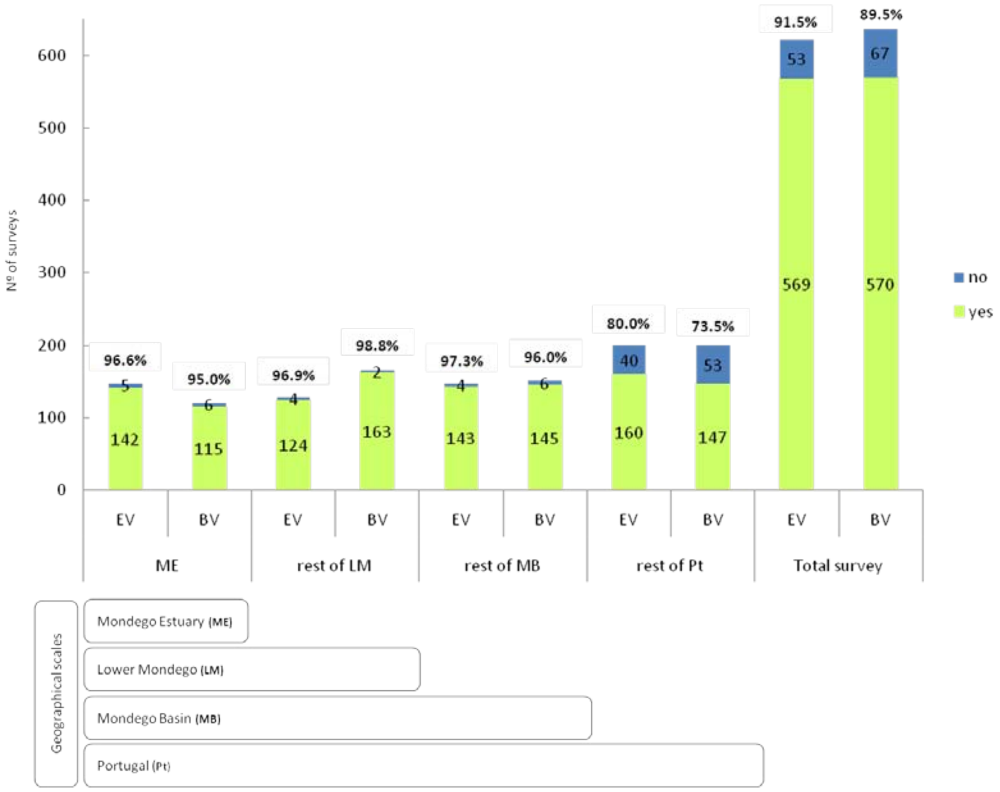

Figure 4. Number of surveys conducted per geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), including respondents that agreed to participate in the survey (YES) and those not willing to participate (NO). Percentage values represent the responses ratios for the several geographic levels.

Table 1 reports the basic socio-economic characteristics of the respondents (for each survey version and geographic sampling level). Based on respondents' answers, different respondents' profiles could be created and linked with the contingent valuation responses (Figure 2). In general, considering gender and age, the surveyed population characteristics are similar among geographic sampling levels, especially when considering the BV questionnaires where the majority of the respondents were in their mid-thirties to mid-forties. Both versions were overrepresented by females (in the range 50-54%), except for the ME level (males: 53.5%), which is also in accordance with the population composition of those geographical level (INE data; Table 1). However, differences among geographic levels were noted for variables as education level (majority of respondent's education level for LM and Pt geographic levels was higher than for the ME and MB levels, that were dominated by the secondary school level) and monthly income (e.g. the ME geographic level had the lower income level, with 44% of the surveyed persons receiving less than 500€/month). Nevertheless, the respondents' monthly income (less than 500€/month to more than 3000€/month) and education level (primary school to university) were distributed across all categories, showing that the survey sample covered all educational/economic backgrounds (Table 1). Comparing these sample characteristics with population composition at those geographic levels is possible to see that some differences were achieved, where the sampled population presented a higher average education level and lower incomes than the official averages observed for these communities (Table 1).

Table 1. Respondents socio-economic profile (%), at the several geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), for both versions (Estuary and Basin surveys).

		Scales								INE offical data (%)				
		Estuary version (%)				Basin version (%)								
variables		ME	LM	MB	Pt	ME	LM	MB	Pt		ME	LM	MB	Pt
N (valid)		142	266	409	569	115	278	423	570		-	-	-	-
Gender										2009 data				
	male	53.5	50.4	47.2	45.7	40.0	42.1	43.0	41.8		47.7	47.5	47.9	48.4
	female	46.5	49.6	52.8	54.3	60.0	57.9	57.0	58.2		52.3	52.5	52.1	51.6
Age										2009 data				
	<30	11.3	18.1	16.6	21.1	18.3	16.2	15.6	21.6	<25	23.3	23.1	23.9	26.0
	31 - 40	11.3	15.4	20.5	23.0	25.2	28.8	26.5	25.6	25-64	56.2	56.2	54.7	55.9
	41 - 50	19.0	21.1	22.5	20.7	14.8	17.3	19.6	18.9	>65	20.7	20.7	21.3	18.1
	51 - 60	24.6	20.3	19.1	17.6	15.7	19.1	18.2	17.5		-	-	-	-
	61 - 70	21.8	15.0	13.0	10.9	20.0	12.2	13.0	11.8		-	-	-	-
	71 - 80	9.2	7.5	6.6	4.9	5.2	4.0	4.5	3.5		-	-	-	-
	>81	2.8	2.6	1.7	1.2	0.9	2.5	2.4	1.8		-	-	-	-
	not answering	0	0	0	0.5	0	0	0.2	0.4		-	-	-	-
Households														
	1	16.2	13.9	11.7	13.4	7.8	13.3	9.9	11.1		-	-	-	-
	2	33.8	33.8	32.8	29.3	34.8	30.6	33.1	32.8		-	-	-	-
	3	24.6	28.2	28.9	27.9	20.9	25.2	27.4	25.1		-	-	-	-
	4	19.0	16.5	19.3	21.6	23.5	22.7	21.3	22.5		-	-	-	-
	5	4.9	6.4	5.6	5.4	6.1	4.0	4.0	4.0		-	-	-	-
	6	0	0	1.0	1.3	1.7	1.4	2.1	2.3		-	-	-	-
	not answering	1.4	1.1	0.7	1.1	5.2	2.9	2.1	2.3		-	-	-	-
Education level										2001 data				
	primary school	28.9	22.2	20.5	17.2	22.6	12.9	15.6	13.5		41.5	40.4	44.2	42.6
	basic school	21.1	19.2	22.0	17.2	25.2	17.6	20.8	17.9		28.0	26.0	25.6	27.6
	secondary school	31.0	32.0	34.7	30.2	34.8	33.1	35.2	31.9		12.2	12.8	10.4	11.8
	university	17.6	25.6	21.8	34.4	16.5	36.0	27.7	36.1		6.2	8.9	6.4	6.6
	other	1.4	1.1	0.7	0.7	0.9	0.4	0.5	0.4		-	-	-	-
	not answering	0	0	0.2	0.2	0	0	0.2	0.2		-	-	-	-
Income*										2009 data (€) [1]				
	<500€	45.1	37.2	38.1	30.2	44.3	30.6	33.6	28.6					25.8
	501 - 1000€	27.5	34.6	36.7	38.8	37.4	37.1	38.5	38.2					51.4
	1001 - 2000€	18.3	21.1	18.6	23.6	12.2	21.2	16.3	22.1					19.6
	2001 - 3000€	3.5	4.1	2.7	3.5	2.6	6.8	5.4	6.0					3.2**
	>3001€	2.1	1.1	0.7	0.7	1.7	2.5	1.9	1.9					
	not answering	3.5	1.9	3.2	3.2	1.7	1.8	4.3	3.2					

* Income is assumed as the individual monthly salary; **values corresponding to incomes superior to 2000€; [1] data obtained from Ministry of Solidarity and Social Security.

3.2. Awareness of respondents towards the preservation and uses of wetlands

As mentioned previously, the survey design facilitated the testing of awareness and usage levels of respondents in relation to the study site.

When inquired about the system status, the majority of the surveyed population (for both EV and BV) considered that the system was polluted, although with a generalised sense that improvements over time had been registered (Table 2A). Factors such as age, education level, natural uses of the system and municipality had a significant effect on the system status perception.

In general, people older than 70 (for the EV) and younger people (30 to 40s' for the BV) tend to consider the system as not that polluted (although the last ones believe it is getting worse), while people with higher education levels, using it fewer times for recreational purposes (e.g. once/twice a month), or living closer to the system tend to consider the system polluted. In fact, the variable 'distance to the system' had a strong influence on the answers (e.g. for Portugal level the 'don't know' responses increased around 20 to 25%, compared to the other surveyed geographic levels). This fact indicated that the proximity to the asset under evaluation was determinant to the resource valorisation (geographic level effect: X^2_{EV}=69.5, df=15; p=0.000; X^2_{BV}=98.6, df=15; p=0.000). Another evidence of the distance decay effect was the knowledge on local restrictions of bivalves capture due to pollution effects: moving from the ME to wider geographic levels there was a gradual decreasing trend of awareness of this restriction (from EV: 76 to 46% and BV: 82 to 45%; Table 2A) (geographic level effect: X^2_{EV}=118.9, df=9; p=0.000; X^2_{BV}=137.2, df=9; p=0.000), reflecting the importance that this resource may have to local populations.

Concerning the main pollution sources, respondents considered that the effluents coming from industrial plants were the main impact causers, followed by agriculture and residential areas (Table 2A). The main identified effects of those activities were environmental consequences such as mud, smell or low diversity (Table 2B). Pollutants as pesticides and fertilizers coming from the agriculture fields, and industrial and domestic effluents were considered, by the majority of the respondents, as very important factors impacting the system, while activities as the fishing port, marina for recreational boats, river margin erosion and aquaculture activities were looked at as not that important for the system conservation and ecological status (Table 2B). Additionally, a strong relation between education level and environmental perception of pollution was also observed (geographic level effect: X^2_{EV}=43.7, df=25; p=0.012; X^2_{BV}=44.8, df=25; p=0.009).

Table 2. Awareness level of sampled communities: A. Awareness for the system actual condition, trends over time, restriction on its uses and main pollution sources; B. Pollution consequences and importance of pollutants, at the several geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), for both versions (Estuary and Basin surveys).

	Scales							
	Estuary version				Basin version			
	ME	LM	MB	Pt	ME	LM	MB	Pt
A. Awareness of system condition								
N	142	266	409	569	115	278	423	583
The Mondego system is... (%)								
...very polluted	15.5	15	13.9	12.3	12.2	12.2	13.5	11.2
...polluted	38.7	39.1	37.7	34.8	41.7	39.2	37.8	37.9
...not that polluted	31.7	30.1	28.4	25.1	33.9	37.8	33.3	28.4
...not polluted	3.5	3.4	3.2	2.6	3.5	3.6	3.8	3
don't know	10.6	12.4	16.9	25.2	8.7	7.2	11.5	19.5
it has been... (%)								
...improving	45.1	43.4	39.6	37.2	13,3	43,3	40	37,2
...worsening	30.3	32.8	32.4	32.3	38.9	36	35.7	36.7
...same	11.3	7.5	5.9	4.2	42.5	4.4	4.3	3.5
don't know/not answering	13.4	16.2	22.1	26.3	5.3	16.4	20	22.5
Have you ever heard of any restriction to the bivalves or fishes catches								
yes	76.1	64.7	55	46.4	81.7	64.7	52.7	44.9
no	19	30.1	38.1	41.7	15.7	27.7	37.4	39.6
don't know/not answering	5.1	5.2	6.9	12	2.6	7.6	9.9	15.5
Main source of pollution (%)								
agriculture fields	44.4	45.5	39.4	35.7	42.6	49.3	44.2	38.9
industrial areas	50.7	51.1	57	58.3	71.3	69.1	70.7	67.4
domestic effluents	36.6	39.5	38.4	36.2	32.2	35.6	38.3	35.3
aquaculture	2.8	3.8	2.7	3.9	3.5	5.4	4	4
recreational/tourism activiti	7	6	7.3	9.5	2.6	3.6	5	6.8
don't know	7.7	7.5	7.6	13.7	22.6	4.3	5.9	11.6
B. Pollution consequences and importance of pollutants								
Consequences of that pollution felt (%)								
muddy	60.6	56.2	57.2	48.9	63.7	59.3	57.6	50.3
smells	60.6	55.5	56.5	52.3	77	57.1	56	50.6
not proper to be used	48.6	46.8	47.9	46.3	63.7	54.9	52.9	49.4
low diversity	-	-	-	-	60.2	49.1	48.1	42.2
presents a different colour	51.4	48.3	51.4	47.5	53.1	52	52.1	47.6
Pollutants classification (%)								
pesticides and fertilizers								1.2
not answering	-	-	0.2	0.4	-	1.4	1.2	1.4
not important	7	6	4.6	3.5	1.7	2.2	1.4	13.3
moderate	14.1	15.4	16.6	16	12.2	12.2	15.4	28.9
important	33.8	28.6	28.4	27.9	32.2	28.4	30.3	44.6
very important	39.4	45.9	44.7	40.2	47	51.8	46.6	10.5
don't know	5.6	4.1	5.4	12.1	7	4	5.2	1.9
industrial effluents								
not answering	0.7	0.8	0.7	0.7	0.9	3.2	2.4	3.2
not important	12	9	6.4	4.9	6.1	4.7	3.5	5.1
moderate	13.4	13.9	12.2	10.5	2.6	5.8	6.4	24
important	33.8	28.2	28.6	25.4	21.7	23.4	24.3	54.4
very important	35.2	43.6	46.9	46	61.7	51.2	57	11.4
don't know	4.9	4.5	5.1	12.5	7	5.8	6.4	1.6
domestic effluents								
not answering	-	0.8	0.7	0.7	-	2.2	1.7	6.8
not important	9.9	9	7.6	7.5	10.4	7.6	6.9	19.5
moderate	20.4	20.3	23.7	22.8	19.1	23	20.8	30
important	39.4	35.7	32.3	29.3	30.4	32.7	32.4	30.7
very important	26.8	30.5	30.1	27	33.9	29.5	32.6	11.4
don't know	3.5	3.8	5.6	12.6	6.1	5	5.7	3.2
fishing port impacts								
not answering	0.7	1.1	1	1.1	-	4	3.1	18.9
not important	26.8	25.2	22.2	18.6	28.7	21.6	23.2	24.4
moderate	23.9	23.7	25.2	24.2	23.5	26.3	24.8	19.3
important	21.8	18.4	16.1	17.9	23.5	20.5	18	11.2
very important	14.8	13.9	11.7	10.9	13.9	12.9	11.3	23
don't know	12	17.7	23.7	27.4	10.4	14.7	19.6	2.8
recreational boats impacts								
not answering	2.1	1.9	1.5	1.4	-	3.2	2.6	20.4
not important	37.3	30.5	27.1	22.6	31.3	24.8	24.6	31.9
moderate	30.3	29.3	30.3	28.1	31.3	34.2	32.2	16.5
important	16.2	15.8	14.7	16.7	21.7	16.9	16.1	7.7
very important	4.2	7.5	6.4	6.7	7.8	9	7.8	20.7
don't know	9.9	15	20	24.6	7.8	11.9	16.8	2.5
river margin erosion								
not answering	5.6	4.1	3.7	3.3	-	2.9	2.4	24
not important	38.7	32.7	29.8	25.3	40	27	27.4	23.5
moderate	15.5	18.8	20.3	22.5	22.6	24.1	24.3	18.4
important	13.4	17.3	14.4	12.8	15.7	22.7	18.9	6.7
very important	5.6	6	6.4	6.5	6.1	6.8	7.6	24.9
don't know	21.1	21.1	25.4	29.6	15.7	16.5	19.4	4.9
aquaculture effluents								
not answering	4.9	3.8	3.7	3.3	0.9	5.4	5.4	4.9
not important	35.2	29.3	29.1	22.6	25.2	19.1	21.7	18.6
moderate	19	19.2	17.6	20.5	29.6	28.1	25.5	25.4
important	12	12	9.8	11.4	13.9	13.7	11.8	12.3
very important	3.5	4.9	4.4	5.1	4.3	8.3	6.6	6.3
don't know	25.4	30.8	35.5	37	26.1	25.5	28.8	32.5

Table 3. Uses of natural areas in general and river/estuarine areas, most common activities undertaken along a river bed or estuarine area, and licenses pay to use the Mondego aquatic system, at the several geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), for both versions (Estuary and Basin surveys).

	Scales							
	Estuary version				Basin version			
	ME	LM	MB	Pt	ME	LM	MB	Pt
N	142	266	409	569	115	278	423	583
Use of natural areas (%)								
less once per year	3.5	4.2	4.9	6.7	0.9	1.4	5	6
once per year	11.3	10.6	13.7	13.4	7	8.7	10	12
twice per year	4.3	11.7	17.2	16.9	11.3	14.4	15.9	15.1
4 times a year	12.1	13.6	14.7	15	7.8	11.9	12.3	14.9
12 times a year	16.3	15.8	15.9	15.1	22.6	20.2	20.9	19.2
25 times a year	10.6	11.7	8.8	9.2	7	9.4	7.1	6.9
50 times a year	12.1	12.5	8.6	9	13.9	10.5	9	8.4
100 times a year	9.2	6	4.2	3.5	2.6	4.7	3.1	2.5
200 times a year	2.1	1.5	1	1.4	13	8.7	5.9	4.9
every day	15.6	9.4	7.1	5.8	12.2	6.9	5.2	4.6
never	2.1	2.6	3.4	3.5	1.7	2.5	5	4.4
don't know	0.7	0.4	0.5	0.5		0.8	0.7	1.4
Use of river/estuarine areas (%)								
not answering				0.2	0.9		0.7	0.9
once per year	20.7	18.1	20.7	21.6	14.2	13	15.5	18
twice per year	9.3	10.4	12.9	13.4	4.4	9.3	12.2	12.7
4 times a year	7.9	8.1	9.6	11.1	8.8	11.9	11.7	11.9
12 times a year	14.3	15.8	12.9	12.9	15.9	15.9	15	13.1
25 times a year	4.3	6.5	5.1	6	13.3	10.4	7.7	6.8
50 times a year	11.4	10	7.3	6.7	10.6	8.5	7.5	8.1
100 times a year	6.4	3.8	2.5	2.4	3.5	5.2	3.5	2.9
200 times a year	3.6	2.7	2	2.2	13.3	8.9	6	4.8
every day	14.3	8.5	5.8	4.7	11.5	6.7	5	4.4
summer months	5	11.2	16.4	14.5	3.5	8.1	11.2	11.9
don't know	2.9	5	4.8	4.4		2.2	4	4.4
Activities performed along a river/estuary (%)								
walking. biking....								
not answering	8	6	4.2	3.6	0.9	4.1	3.1	2.4
often	22.6	25.4	24.4	25.2	23	28.1	26.6	25.2
sometimes	40.1	36.5	37.4	37.2	40.7	30	31.5	31.8
rarely	17.5	21.4	24.7	24.9	29.2	29.3	30.4	31.2
never	11.7	10.7	9.4	9.2	6.2	8.5	8.4	9.4
non-motorizes navigation								
not answering	8.8	7.1	4.9	4.3	0.9	4.8	3.6	3.2
often	4.4	2.8	1.8	1.9	1.8	2.6	2.8	2.4
sometimes	8.8	6.3	7	8.4	3.5	4.1	3.6	4.1
rarely	4.4	10.3	9.4	13.1	5.3	14.1	12.3	15.8
never	73.7	73.4	76.9	72.4	88.5	74.4	77.7	74.4
motorized navigation								
not answering	9.5	7.9	5.7	4.9	0.9	4.8	3.6	3.2
often	5.1	3.2	2.6	2.1	1.8	1.1	1.5	1.3
sometimes	4.4	4	3.4	4.3	7.1	3.7	3.1	3
rarely	5.8	5.2	6.5	9.9	2.7	7	6.6	9.8
never	75.2	79.8	81.8	78.9	87.6	83.3	85.2	82.7
baths. swimming								
not answering	9.5	7.1	4.9	4.5	0.9	4.8	3.6	3.2
often	1.5	5.6	6.5	7.5	5.3	3.7	7.7	7.3
sometimes	11.7	15.5	16.4	22.9	12.4	17.4	19.2	21.6
rarely	8	9.5	14	15.7	6.2	13.7	13	16.5
never	69.3	62.3	58.2	49.4	75.2	60.4	56.5	51.3
sport fishing								
not answering	9.5	7.1	4.9	5.2	1.8	5.6	4.1	3.6
often	5.8	5.2	4.9	4.3	4.4	2.6	2.3	2.1
sometimes	8.8	6.7	5.7	6	10.6	7.4	6.4	7.4
rarely	3.6	2.8	4.9	6.2	2.7	8.1	7.2	7.3
never	72.3	78.2	79.5	78.4	80.5	76.3	79.5	79.7
bird/animal watching								
not answering	9.5	8.3	5.7	5		4.1	2.8	3
often	8.8	11.5	10.6	10.3	5.3	8.9	11.5	10.3
sometimes	15.3	16.7	15.8	17.5	24.8	26.3	22.5	21.8
rarely	14.6	17.5	19.2	22.6	7.1	16.3	18.9	21.8
never	51.8	46	48.6	44.6	62.8	44.4	44.2	43
walking the dog								
not answering	11.7	10.3	4	5.8	2.7	7.4	5.6	4.5
often	5.8	6.3	7	7.1	13.3	8.9	9.2	7.9
sometimes	5.1	6.7	5.7	6.5	7.1	7	5.9	7.7
rarely	1.5	2.8	4.2	5	2.7	4.4	4.3	5.8
never	75.9	73.4	75.8	75.4	74.3	72.3	75	74.1
Households licenses (%)								
marina	5	4.4	3.9	2.8	4.3	2.5	2.1	1.8
fishing	11.4	12.3	13	10.7	13.9	11.9	11.1	8.8
canoeing	0.1	0.4	0.2	0.7	0	0.4	0.2	0.5
other	0.8	0.4	0.7	0.9	0	0.7	0.9	0.9
no	82.5	83	84.1	86.5	81.7	85.6	86.1	88.4

Table 3 shows the uses of the Mondego system. The distance from respondents' residence to the system had a significant effect on its usage (people living closer to the system used it more often; Table 4), conditioning also the activities performed (Table 3), like walking/travelling along the margins, non-motorised activities, as canoeing, bathing or swimming, sport fishing and bird/animal watching (Table 5A). Table 4's results reflected that a substantial part of the respondents for the EV survey have never heard about the Mondego Estuary, or never used it before, comparatively to the entire river.

Table 4. Mondego system use by the surveyed population (%) and its relation to the geographical geographic sampling level (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), for both versions (Estuary and Basin surveys), and level of significance across geographic levels (X^2 results).

	Geographical scales	N	Mondego use (%)						Pearson Chi-square		
			NO	YES				not answered	X^2	df	p
			never heard	never went	did not use last year	1/2 times last year	use it often				
Estuary version	ME	142	4.2	4.2	19.7	33.1	32.4	6.3	-	-	-
	LM	266	7.5	11.7	20.3	31.6	25.6	3.4	32.511	5	0.000
	MB	409	12.7	15.9	23.5	27.4	18.3	2.2	92.715	10	0.000
	Pt	569	14.9	22.3	23.6	22.7	13.7	2.8	162.376	15	0.000
Basin version	ME	115	1.7	7.0	20.0	30.4	40.9	0	-	-	-
	LM	278	1.8	5.4	19.8	33.5	38.5	1.1	3.834	5	0.574
	MB	423	1.2	9.9	27.2	31.2	29.8	0.7	63.892	10	0.000
	Pt	570	1.6	14.4	32.3	28.1	22.5	1.2	146.411	15	0.000

Most of the respondents did not pay any type of fee to use natural areas (between 82 and 88%) (Table 3), and the most common license was the one dedicated to recreational fishing (around 10% of the respondents), being directly related to the natural areas usage, especially in people using the system every day to once a month (geographic level effect: X^2_{EV}=34.2, df=22; p=0.047; X^2_{BV}=134.9, df=22; p=0.000), and household members, where smaller households had higher percentages of licenses (geographic level effect: X^2_{BV}=24.0, df=14; p=0.046).

3.3. 'Willing to pay' responses

3.3.1. Motivations behind individual willingness to pay

When considering the receptiveness of populations towards environmental quality, the interest in seeing improvements implementation was very high, ranging between 91 to 97% (Table 6). Moreover, respondents in lower geographic sampling levels (e.g. ME or LM levels) were slightly more concerned in helping to improve the system quality than the respondents of the national level. However, the actual willingness to pay for the suggested alternatives ranged between 49 to 56%. In fact, the refusal rates were quite high (ranging between 44 to 50%, depending on the survey version and alternative considered; Table 6). A strong relation between socio-economic profiles and WTP (collected as an additional tax to be included in the household monthly water bill) for environmental quality improvements was observed, where the covariates were significant and their qualitative impact was as expected. In a general way, variables as 'Income' or 'Education' had a significant and positive effect on the system quality status improvement (and WTP) – individuals having a higher monthly income were willing to pay more for the improvements while individuals with higher education level were usually more available to contribute (Table 7). In fact, both factors had a strong effect in A1 and A2 implementations. In general, the Alternative 2 received, proportionally, higher WTP from all geographic sampling levels and survey versions. Alternative 3 was the one receiving fewer contributions from local populations (Table 6). When comparing both survey versions, was possible to observe that the basin survey has received higher percentage of positive WTP, comparatively to the estuary survey, for all geographic sampling levels, however this difference was not statistically significant (A1: F=0.734, p=0.431; A2: F=5.682, p=0.063; A3: F=1.942, p=0.222). Moreover, a significant relation between non-extractive activities and households' alternatives interest (both implementation and WTP) was also observed (Table 8). The usage and system knowledge had particularly importance for A1 implementation, especially for the geographical levels closer to the estuarine system (EV survey). Activities as walking along the margins, non-motorised activities and swimming were also highly related to respondents´ preferences for the alternatives presented to them, while activities as bird watching and fishing were mainly related to the respondents' WTP (Table 9). About half of the respondents thought the proposal would success in accomplish the goal of increasing the quality status of the system, and an additional 30-40% believed that it may succeed to achieve the improvements (Table 6). Perceptions of success were weakly related with the scope treatments (high rates of people considering that was in fact possible to improve for both EV and BV).

Table 5. Significant relations between A. respondents' recreational activities versus the distance to the resource; and B. Zero WTP bids responses versus socio-economic profiles of respondents (R1_NWTP- 1st invocated reason for 'not to be willing to pay', R2_NWTP- 2nd invocated reason for 'not to be willing to pay'); at the several geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), for both versions (EV and BV).

	EV												BV											
	ME			LM			MB			Pt			ME			LM			MB			Pt		
	χ^2	df	*p*	χ^2	df	*p*	χ^2	df	*p*	χ^2	df	*p*	χ^2	df	*p*	χ^2	df	*p*	χ^2	df	*p*	χ^2	df	*p*
A. Region *vs* recreational activities																								
walking/travelling							18.0	8	0.022	21.3	12	0.046				15.3	4	0.004	21.3	8	0.006	29.0	12	0.004
non-motorised activities				16.9	4	0.002	32.9	8	0.000	52.8	12	0.000				22.6	4	0.000	34.1	8	0.000	48.9	12	0.000
bathing/swimming				17.3	4	0.002	38.0	8	0.000	88.4	12	0.000				25.5	4	0.000	53.2	8	0.000	77.9	12	0.000
sport fishing							23.2	8	0.003	26.3	12	0.018				25.5	4	0.000	53.2	8	0.000	77.9	12	0.000
bird/animal watching							18.2	8	0.019	34.8	12	0.001				35.8	4	0.000	56.3	8	0.000	63.7	12	0.000
B. Zero WTP bids responses *vs* socio-economic profile																								
income vs R1_NWTP													68.9	45	0.013				88.0	65	0.030			
income vs R2_NWTP																96.5	65	0.007	109.5	65	0.000	98.0	70	0.015
municipality vs R1_NWTP										900.5	644	0.000				142.3	88	0.000	313.3	260	0.013	946.7	714	0.000
municipality vs R2_NWTP										716.9	644	0.024				242.8	104	0.000	383.4	260	0.000	105.8	714	0.000
distance vs R1_NWTP							46.9	28	0.014	128.2	42	0.000				21.5	11	0.029	44.4	26	0.014	120.5	42	0.000
distance vs R2_NWTP				24.6	14	0.038	62.8	28	0.000	94.1	42	0.000										81.5	42	0.000
water bill vs R1_NWTP				154.4	108	0.002	217.5	126	0.000	191.2	154	0.022												
household members vs R2_NWTP				125.3	70	0.000	143.5	98	0.002	143.4	98	0.002												
age vs R1_NWTP										148.7	112	0.012							112.3	78	0.007	147.3	112	0.014
education vs R1_NWTP										93.7	70	0.031										72.0	42	0.003
education vs R2_NWTP																						67.7	42	0.007
possibility of improvement vs R1_NWTP													45.3	27	0.015	67.9	33	0.000	75.1	39	0.000	65.6	42	0.011
possibility of improvement vs R2_NWTP													43.5	24	0.009	63.7	39	0.007	54.7	39	0.049			
system use vs R1_NWTP				83.8	60	0.023	98.5	70	0.014	119.0	70	0.002												
system use vs R2_NWTP										99.6	70	0.012										95.4	70	0.023

Table 6. WTP estimation: intention of seeing implemented a plan to improve the system quality (intention to improve), actual acceptance ratios of the improvement alternatives, and probability of improvement (after a management plan implementation), for the several geographic sampling levels considered at the several geographic levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal) and both versions (Estuary and Basin surveys).

Geographical scales		N	Intention to improve (%)			Stated improvements plan voting								Possibility of improvement (%)			
						Against the plan		For the plan									
						Actual condition		Alternative 1		Alternative 2		Alternative 3					
			Yes	No	do not know	N_0	%	N_1	%	N_2	%	N_3	%	for sure	probably	hardly	do not know
Estuary version	ME	142	94.4	0.7	4.9	70	49.3	72	50.7	37	51.4	19	51.4	43.7	38.0	15.5	2.8
	LM	266	94.0	1.9	4.1	121	45.5	145	54.5	86	59.3	44	51.2	44.7	37.6	14.3	3.4
	MB	409	93.9	1.5	4.6	199	48.7	210	51.3	123	58.6	58	47.2	41.8	41.3	14.2	2.7
	Pt	569	90.9	1.6	7.6	278	48.9	290	51.1	171	59.0	85	49.7	38.8	42.4	14.9	3.9
Basin version	ME	115	97.4	2.6	0	58	50.4	57	49.6	33	57.9	14	42.4	40.0	38.3	19.1	2.6
	LM	278	95.3	1.4	3.3	122	43.9	156	56.1	99	63.5	53	53.5	39.9	38.5	18.0	3.6
	MB	423	95.7	1.4	2.9	190	44.9	233	55.1	137	58.8	72	52.6	45.2	36.6	15.6	2.6
	Pt	570	94.7	1.2	4.0	275	48.2	295	51.8	180	61.0	93	51.7	40.4	39.6	15.8	4.2

Table 7. Significant relations between respondents’ education versus the improvement alternatives; and between the mean respondents income versus the mean willing to pay taxes for the different alternatives, at the several geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), for both versions (Estuary and Basin surveys).

	Geographical scales	N	escolarity vs Alternative 1 implementation			escolarity vs Alternative 2 implementation			escolarity vs Alternative 3 implementation			mean income vs mean tax Alternative 1			mean income vs mean tax Alternative 2			mean income vs mean tax Alternative 3		
			Pearson Chi-square			Pearson Chi-square			Pearson Chi-square			Pearson Chi-square			Pearson Chi-square			Pearson Chi-square		
			X^2	df	p	X^2	df	p	X^2	df	p	X^2	df	p	X^2	df	p	X^2	df	p
Estuary version	ME	142	14.720	4	0.005	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	LM	266	27.922	4	0.000	13.489	3	0.004	-	-	-	81.637	32	0.000	59.345	30	0.001	-	-	-
	MB	409	26.781	5	0.000	14.580	3	0.002	-	-	-	108.182	36	0.000	57.944	30	0.002	42.161	24	0.012
	Pt	569	15.600	5	0.008	18.500	4	0.001	-	-	-	131.300	44	0.000	48.100	33	0.043	-	-	-
Basin version	ME	115	-	-	-	-	-	-	-	-	-	59.832	20	0.000	33.298	16	0.007	-	-	-
	LM	278	27.293	4	0.000	-	-	-	10.553	4	0.032	51.544	32	0.016	37.686	24	0.037	-	-	-
	MB	423	35.089	5	0.000	14.934	5	0.011	12.752	4	0.013	61.425	32	0.001	-	-	-	-	-	-
	Pt	570	29.299	5	0.000	13.947	5	0.016	-	-	-	58.087	32	0.003	-	-	-	-	-	-

Note: spaces marked with ‘-‘ represent the non-significant relations.

Table 8. Number of surveys (N, including both respondents that attributed a value to the described improvements and those inability to state a value - ‘do not know’ answers), minimum (Min.) and maximum (Max.) amounts given, mean WTP, the respective standard deviation (SD) and the percentage of values from the mean respondents monthly income for: A. the three considered alternatives and for the four geographical geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal); and B. considering the distance from the respondents’ home to the surveyed ecosystem; for both versions (Estuary and Basin surveys). In this table, the Nvalid represents the total number of surveys of positive (‘YES’) responses.

A. WTP values for the four geographical levels

Survey version	Surveyed geographical levels	Scenario 1						Scenario 2						Scenario 3						Remarks
		N	N_{valid}	WTP			SD	N	N_{valid}	WTP			SD	N	N_{valid}	WTP			SD	
				Min.	Max.	Mean				Min.	Max.	Mean				Min.	Max.	Mean		
Estuary version	Mondego estuary	72	**65**	0.5	145	10	20.7	37	**32**	2.5	65	16.25	15.52	19	**14**	7.5	75	31.61	22.52	
	Lower Mondego	145	**135**	0.5	145	8.22	16.42	86	**78**	2.5	105	16.06	18.99	44	**35**	7.5	145	31.71	29.57	
	Mondego Basin	210	**199**	0.5	145	7.91	15.26	123	**114**	2.5	105	16.21	18.5	58	**48**	7.5	145	30.21	26.3	
	Portugal	290	**274**	0.5	145	8.01	15.58	171	**157**	2.5	115	16.59	20.22	85	**70**	7.5	145	30.18	26.09	mail survey
Basin version	Mondego estuary	57	**55**	2.5	55	9.18	12.1	33	**30**	7.5	65	19.58	16.39	14	**12**	15	85	30.83	21.93	
	Lower Mondego	156	**146**	2.5	125	9.55	15.06	99	**89**	7.5	65	18.46	14.9	53	**44**	15	85	29.09	17.43	
	Mondego Basin	233	**217**	0.5	125	8.54	13.42	137	**122**	2.5	115	17.87	16.38	72	**59**	7.5	145	29.79	22.11	
	Portugal	295	**278**	0.5	125	8.42	12.78	180	**164**	2.5	115	17.35	15.72	93	**79**	7.5	145	29.08	20.94	mail survey

B. WTP values for the four geographical levels considering the distance from respondents' home to the surveyed good

Survey version	Surveyed geographical levels	Scenario 1						Scenario 2						Scenario 3						Remarks
		N	N_{valid}	WTP			SD	N	N_{valid}	WTP			SD	N	N_{valid}	WTP			SD	
				Min.	Max.	Mean				Min.	Max.	Mean				Min.	Max.	Mean		
Estuary version	Mondego estuary	72	**65**	0.5	145	10	20.7	37	**32**	2.5	65	16.25	15.52	19	**14**	7.5	75	31.61	22.52	
	Lower Mondego _km	73	**70**	0.5	55	6.57	10.98	49	**46**	2.5	105	15.92	21.24	25	**21**	7.5	145	31.78	34.01	
	Mondego Basin_km	65	**64**	0.5	95	7.26	12.54	37	**36**	2.5	105	16.53	17.67	14	**13**	7.5	55	26.15	14.42	
	Portugal_km	80	**75**	0.5	105	8.27	16.52	48	**43**	2.5	115	17.62	24.42	27	**22**	7.5	125	30.11	26.21	mail survey
Basin version	Mondego estuary	57	**55**	2.5	55	9.18	12.1	33	**30**	7.5	65	19.58	16.39	14	**12**	15	85	30.83	21.93	
	Lower Mondego _km	99	**91**	2.5	125	9.78	16.99	66	**59**	7.5	65	17.88	14.19	39	**32**	15	75	28.44	15.78	
	Mondego Basin_km	77	**71**	0.5	115	16.29	20.01	38	**33**	2.5	115	16.29	20.01	19	**15**	7.5	145	31.93	32.96	
	Portugal_km	62	**61**	0.5	45	7.99	10.28	43	**42**	2.5	55	15.83	13.72	21	**20**	15	65	27.00	17.35	mail survey

Table 9. Relation between river/estuarine uses (visits per year), activities performed, and Mondego system knowledge to the several alternatives (implementation and WTP) of improvement considered for the watershed, for the four geographical geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), for both versions (Estuary and Basin surveys).

Survey version			EV								BV							
Surveyed geographical scales			ME		LM		MB		Pt		ME		LM		MB		Pt	
N			142		124		143		160		115		163		145		147	
variables			implementation	WTP	implementation	WTP	implementation	WTP	implementation	WTP	implementation	WTP	implementation	WTP	implementation	WTP	implementation	WTP
A1	visits	x^2	18.825		20.297		21.264										23.433	
		df	10		10		10										11	
		p	0.043		0.027		0.019										0.015	
	activities_walking/ travelling/byking	x^2	10.202			62.100	10.759	74.816	9.926	95.347			16.376		18.993	61.553	13.255	53.245
		df	4			36	4	40	4	48			4		4	32	4	32
		p	0.037			0.004	0.029	0.001	0.042	0.000			0.003		0.001	0.001	0.010	0.011
	activities_non-motorised	x^2	12.636	72.996	13.193	78.493	19.018	91.541	15.509	78.912			16.247		13.159		15.189	
		df	4	32	4	36	4	40	4	48			4		4		4	
		p	0.013	0.000	0.01	0.000	0.001	0.000	0.004	0.003			0.003		0.011		0.004	
	activities_motorised	x^2		47.706				60.927		73.445		32.771						
		df		32				40		48		18						
		p		0.037				0.018		0.010		0.018						
	activities_bathing/ swimming	x^2	15.057		17.587	52.581	21.764		23.553						9.784		11.269	
		df	4		4	36	4		4						4		4	
		p	0.005		0.001	0.037	0.000		0.000						0.044		0.024	
	activities_fishing	x^2														62.764		60.434
		df														40		40
		p														0.012		0.022
	activities_bird watching	x^2		39.942		51.181						32.944					10.372	
		df		32		36						18					4	
		p		0.001		0.048						0.017					0.035	
	Mondego use	x^2		91.796		78.704	14.608	98.14	18.949	84.948	19.712		33.618		33.741		57.515	
		df		40		45	5	50	5	60	4		5		5		5	
		p		0.000		0.001	0.012	0.000	0.002	0.019	0.001		0.000		0.000		0.000	
A2	visits	x^2				141.391												
		df				110												
		p				0.023												
	activities_walking/ travelling/byking	x^2					12.256		17.033							53.78		
		df					4		4							36		
		p					0.016		0.002							0.029		
	activities_non-motorised	x^2		53.352		80.493		100.547		83.879								
		df		32		44		44		48								
		p		0.010		0.001		0.000		0.001								
	activities_motorised	x^2				65.249		68.331		88.892								
		df				44		44		48								
		p				0.020		0.011		0.000								
	activities_bathing/ swimming	x^2			11.103		9.553											
		df			4		4											
		p			0.025		0.049											
	activities_fishing	x^2														73.932		85.859
		df														45		45
		p														0.004		0.000
	activities_bird watching	x^2			11.684							29.739						
		df			4							15						
		p			0.02							0.013						
	Mondego use	x^2																
		df																
		p																
A3	visits	x^2																
		df																
		p																
	activities_walking/ travelling/byking	x^2					10.425	53.995	10.979							58.729		
		df					4	36	4							40		
		p					0.034	0.027	0.027							0.028		
	activities_non-motorised	x^2		43.625								19.727						
		df		28								10						
		p		0.030								0.032						
	activities_motorised	x^2																11.087
		df																4
		p																0.026
	activities_fishing	x^2								60.834						63.343		75.052
		df								40						40		40
		p								0.018						0.011		0.001
	activities_walking dog	x^2																67.903
		df																50
		p																0.047
	Mondego use	x^2																
		df																
		p																

The main reasons to pay (Table 10A) ranged between the conservational aspects of the environment (‘want to contribute to the protection of water resources’) and a more ‘service’-vision of the system (‘interested in the advantages that will come from the improvements’), and their associated benefits provision.

Table 10. Reasons invocated by respondents when considering the improvement scenarios participation (values are percentages of the total samples), at the several geographic sampling levels (ME-Mondego Estuary; LM-Lower Mondego; MB-Mondego Basin; Pt-Portugal), for both versions (Estuary and Basin surveys): A. The two more important reasons (1^{st} and 2^{nd} main reasons) that leaded the respondents to ‘being willing to pay’ for the proposed water quality improvements; B. Main reasons invocated by respondents for ‘not being willing to pay’ for the proposed water quality improvements (T0: true zero answers, and P: percentage of protest answers).

			Scales															
			Estuary version (%)								Basin version (%)							
			ME		LM		MB		Pt		ME		LM		MB		Pt	
	A. Reasons to pay		1st	2nd	1st	2nd	1st	2nd	1st	2nd	1st	2nd	1st	2nd	1st	2nd	1st	2nd
1	The improvements are good and will be valorised by me and my		7.1	1.4	6.9	2.1	8.6	4.3	10.7	9.8	7.0	5.2	8.3	7.7	12.0	10.7	12.2	9.2
2	I am interested in these improvements, no matter the costs		12.7	-	8.3	-	6.7	1.4	4.2	4.2	1.8	3.5	4.5	3.2	3.9	2.1	3.4	2.7
3	I am interested in the advantages that all will win with these		33.8	19.7	31.3	22.8	35.4	19.6	35.2	36.8	38.6	21.1	35.9	20.5	33.9	16.3	31.2	17.6
4	Everyone should experience river/estuaries of high quality, no matter		15.5	16.9	15.3	18.8	15.4	17.2	14.0	13.0	14,0	15.8	9.6	21.8	12.9	21.0	11.5	22.0
5	I want to contribute to the protection of the aquatic environment for the		23.9	32.4	30.6	29.2	27.8	31.2	26.6	26.6	36.8	42.1	40.4	35.3	35.6	34.8	36.9	32.5
6	Morally I felt that was the correct answer		2.8	4.2	4.2	6.3	3.3	5.7	4.7	4.2	1.8	3.5	1.3	1.9	1.7	3.1	4.1	3.1
7	I did not understood the question		-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8	Other		2.8	-	2.1	-	1.4	-	2.4	3.0	-	-	-	-	-	12.0	0.3	-
9	Not answering		1.4	25.4	1.3	20.8	1.4	20.6	2.2	2.4	-	8.8	-	9.6	-	-	0.4	12.9
	B. Reasons not to pay		1st	2nd	1st	2nd	1st	2nd	1st	2nd	1st	2nd	1st	2nd	1st	2nd	1st	2nd
1	The increase in the water bill is too high comparatively to the water quality improvements described	P	5.6	4.2	4.9	4.1	4.0	2.5	6.1	1.8	5.2	1.7	4.9	3.3	3.2	3.2	4.0	2.2
2	The river stays very far from my place	T0	-	-	1.6	3.3	2.5	5.0	6.4	7.9	-	-	2.5	1.6	3.2	1.6	9.5	2.5
3	I do not use the Mondego system	T0	4.2	0.8	4.9	1.5	6.0	5.5	6.1	7.1	5.2	10.3	5.7	6.6	6.3	4.7	7.3	8,0
4	The current condition is enough	-	5.6	2.8	4.1	2.5	3.0	2.0	2.5	1.4	-	-	1.6	0.8	1.6	0.5	1.1	0.7
5	I do not believe the water quality improves as the way it was described	P	1.4	1.6	0.8	1.6	0.5	1.5	0.7	1.1	-	-	-	0.8	0.5	1.1	0.7	1.1
6	I would rather pay for other river/estuary	P	-	1.4	-	0.8	0.5	0.5	4.3	2.1	1.7	-	0.8	-	1.1	-	2.5	1.8
7	I would rather use the money in other things	T0	-	1.4	-	3.3	0.5	2.0	0.4	2.1	-	1.7	-	0.8	0.5	1.1	0.4	0.7
8	I cannot afford an extra increase in my water bill	-	14.1	5.6	11.5	4.1	11.0	6.0	9.3	5.4	1.7	5.2	6.6	9.0	6.8	6.8	6.9	5.5
9	The Mondego users should support these increases	P	-	-	0.8	2.5	4.5	3.0	4.6	1.3	5.2	1.7	5.7	2.5	5.3	2.6	4.4	2.2
10	The water companies should support these improvements	P	18.4	18.8	17.2	18.0	13.0	12.0	10.0	11.6	10.3	17.2	10.7	12.3	8.9	14.7	9.8	12.0
11	The government should pay for these improvements	P	14.1	25.4	20.5	21.3	24.5	24.5	20.0	21.5	20.7	25.9	22.1	23.0	28.4	20.0	22.9	19.6
12	The water bills are already too high	P	28.2	15.5	23.1	12.3	19.5	8.5	15.0	8.9	43.1	20.7	27.9	16.4	22.6	14.7	17.1	12.7
13	The question is too hard to answer	-	2.8	2.8	1.6	3.3	1.0	2.0	1.1	1.4	-	-	-	0.8	-	1.1	0.4	0.7
14	Other	-	2.8	-	7.4	2.5	7.0	2.5	8.6	2.1	1.7	-	1.6	0.8	3.2	4.7	6.5	3.6
-	Not answering	-	2.8	19.7	1.6	18.9	2.5	22.5	4.9	24.3	5.2	15.6	9.9	21.3	8.4	23.2	6.5	26.7
	Total True zeros (%)		4.2	2.2	6.5	8.1	9.0	12.5	12.9	17.1	5.2	12.0	8.2	9.0	10.0	7.4	17.2	11.2
	Total Protests (%)		67.7	66.9	67.3	60.6	66.5	52.5	60.7	48.3	86.2	67.2	72.1	58.3	70.0	56.3	61.4	51.6

Note: spaces marked with ‘-‘ represent the reasons given that are not either true zeroes, neither protest responses.

3.3.2. Estimation of WTP for water quality improvement

‘Embedding’ effect

Considering the MB geographic level, the WTP for a good water quality improvement (A1) in the EV and BV surveys was estimated, ranging from 7.91 to 8.54€/year, respectively. With regard to a very good water quality improvement (A2) the WTP ranged between 16.21 and 17.87€/year. Finally, when questioned about ecotourism facilities and to ensure its promotion (A3) the respondents’ WTP ranged between 30.21 and 29.79€/year. The average annual household WTP to improve water quality in the whole basin was found not to be significantly higher than for only part of the system (Mondego Estuary) for all alternatives (A1: $F=2.80$, $p=0.12$; A2: $F=4.80$, $p=0.05$, with a high Fcritic value; A3: $F=0.69$, $p=0.42$), showing that respondents are not sensitive to the geographic level issue (Table 8).

Distance-decay, uses and socio-economic profiles effects on the value of water quality changes

Three factors had a significant role in determining respondents' WTP availability and amount of contributions: the distance from the surveyed good (distance-decay effect), the use of the system and the socio-economic conditions/profiles of the respondents (Table 8). Comparing the respondents' WTP with the respondents' mean monthly income it is possible to see that the ME geographic level for the EV survey (1.22% for A1) and the ME and MB geographic levels for the BV survey (1.20 and 0.98% for A1, respectively) presented the highest WTP/income ratios. The proximity to the resource under valuation played a determinant role when deciding to pay for its conservation, especially for the BV survey where higher percentages of acceptance of A1 were achieved for the LM and BM geographic levels (closer to the river; X^2_{BV}=11.0, df=3, p=0.012); while for the EV, higher percentages for A1 were obtained for ME and LM geographic levels (closer to the estuary; although not statistically significant: X^2_{BV}=4.9, df=3, p=0.182). This observation was also consolidated by the low ratios observed for the Portugal level, in both survey versions (analysing only the mail responses, without integration with catchment responses). In fact, this geographic level presented very low WTP values comparatively to their monthly income (0.68 and 0.77% for A1) (Table 8). This result was also reinforced by the increasing percentages of respondents attesting that they would prefer to pay for another system closer to their residence (Table 10B).

Comparing the achieved values when considering the distance from respondents' homes to the resource (Table 8B) is possible to see that decreasing WTP values were achieved for A1 and A3 alternatives for the Estuary version (the WTP value decreased as the distance increased), while increasing values were achieved for A2. The opposite pattern was observed for the Basin version (WTP for A1 and A3 increased from ME to MB; and A2 WTP values decreased). Moreover, lower values were achieved for A1 implementation for the estuary version (WTP ranged from 6.57 to 10.00€) comparatively to the basin version (WTP ranged from 9.18 to 16.29€).

3.4. Zero responses analysis

High refusal rates were obtained for all geographic sampling levels, survey versions and alternatives considered (Table 6). This fact was particularly evident for the Portugal level (Table 6), with refusal rates around 49% (EV survey) and 48% (BV survey), suggesting that the distance to the system might also have a substantial effect on the respondents' awareness, consolidated by the significant relation between being available to pay and being a Mondego system user (X^2_{EV}=99.6, df=70; p=0.012; X^2_{BV}=95.4, df=70; p=0.023). Moreover, the 'geographic level effect', for the EV survey, had a strong role in determining the reasons not to pay for improvements in the system at least in one of the reasons given, at the LM level (reason not to pay 2: X^2=24.6, df=14; p=0.038), MB level (reason not to pay 1: X^2=46.9, df=28; p=0.014; reason not to pay 2: X^2=62.8, df=28; p=0.000), and Pt level (reason not to pay 1: X^2=128.2, df=42; p=0.000; reason not to pay 2: X^2=94.1, df=42; p=0.000). The same pattern was observed for the BV survey at the LM level (reason not to pay 1: X^2=21.5, df=11; p=0.029), MB level (reason not to pay 1: X^2=44.4, df=26; p=0.014), and Pt level (reason not to pay 1: X^2=120.5, df=42; p=0.000; reason not to pay 2: X^2=81.5, df=42; p=0.000), which suggests that the distance-decay effect may have played a significant role when considering the system valuation.

The main reasons invoked for not being willing to pay are listed in Table 10B. Nevertheless, a distinction has to be made between 'true zero' (T0) and protest (P) responses. In most cases, the zero bids for the good improvement were derived from: 1) the generalised sense that water good quality maintenance and improvement should be of the competent authorities' responsibility (related to respondents with high environmental awareness, education levels, income and distance to the resource); 2) the fact that most of the respondents considered the water bills already too high (highly related to the income, education, geographic sampling levels, age and to the decision of not implementing the suggested improvements), and so, would be not willing to pay any further. Most of the refusals were, in fact, protest answers (Table 6B), wishing to highlight the importance of conservation, but reinforcing that it is a responsibility of the competent authorities.

Therefore, different motives of zero WTP bids responses (essentially protests) were characterised by different socio-economic and environmental variables related to the respondents' realities (Table 5B):

1. Answers were significantly associated with income constraints, municipality, distance to the resource, water bill, and household members. Factors as age and education levels of respondents only had a significant effect when there was a successive integration of information between geographic sampling levels (specifically to the MB and Pt geographic levels);
2. The effects of respondents' profiles on zero bids responses revealed that most of those not willing to contribute to the water quality improvements generally had lower levels of confidence that the system will in fact respond as predicted, stating that it was only probable, not certain, that the system would improve, especially for the BV survey; and
3. The system use and knowledge also had a significant influence on the respondents' attitudes.

4. Discussion

4.1. 'Willing to pay' responses

The relatively lower response ratio achieved for the mail survey, comparatively to the in-person survey, is in accordance with other CV studies, where higher responses rates were achieved for the second method (e.g. Mannesto and Loomis, 1991; Marta-Pedroso et al., 2007). Even though, the high survey responses ratios obtained suggest that most of the respondents were in fact interested in the survey topic.

The ecosystem services approach tries to find linkages between natural resources assets, functioning of ecosystem, ecosystem services, and its benefits to human well-being (de Groot et al., 2002; Birol et al., 2005; Beaumont et al., 2006). However, the relation among these factors is not always straightforward and requires integration of multidisciplinary methods. For example, currently there is some debate of how to relate and quantify the system assets and functioning with the benefits derived from its good ecological status (e.g. Martin-Ortega and Berbel, 2010; Turner et al., 2010). This information is at the basis of an Ecosystem Approach implementation, which provides a methodological framework for the integration of wetland management while meeting economic and sustainable development objectives (as defended by Maltby, 2006; de Jonge, 2007; de Jonge et al., in press). In this sense, people and societal choices are at the centre of management actions. Therefore it becomes crucial to clearly identify the main issues that drive individuals' choices and valuation. The surveyed population showed very high interest and social concern regarding water quality improvements. Interestingly, only around half of respondents were actually willing to pay extra to improve from the current situation to a Good (A1) water status. One of the critics that have been made to the CV method is that the individuals' WTP responses may express 'narrow economic preferences' (Nunes and Schokkaert, 2003). Some works (e.g. Kahneman and Knetsch, 1992; Nunes and Schokkaert, 2003) claim that respondents mostly search for moral satisfaction when answering to CV survey. In this study respondents agreed to improve water quality level stating that they: (R1) were interested in the advantages they would gain from it; (R2) would like to preserve the water resources; and (R3) would like to conserve it, so that everyone could experience ecosystems in good conditions. The reasons invoked suggest that, when contributing, individuals were concerned with the public good conservation (R2), while also interested in the benefits achieved from it (R1 and R3). This interest was then translated into a WTP for those improvements, for the basin as a whole, or for only part of it, up to good/very good levels. According to the attributes used in this study, this situation comprised the levels of water quality where the danger for public health are minimal (e.g. good conditions for bathing or fishing), with a recreational direct use of the system (e.g. boating and canoeing, swimming, wildlife watching), and even ensuring the integrity of the ecosystem and its surroundings (e.g. green areas surrounding the system). Therefore, the CV method proved to be sensitive to the improvements and valuation by different populations in distinct zones. This fact may be related with the geographical heterogeneity of respondents, both in terms of socio-economic profiles or in the direct (non-)use of the system, where the achieved results suggested that WTP could vary between the surveys and geographic levels factors considered.

When considering the mean WTP for the three alternatives, was possible to see that for the basin version there was a gradual increase from alternative 1 to 3 (mean WTP around 10, 20, 30€), while for the estuary version the obtained values were around 10, 15 and 30€, respectively, which induces two main conclusions: (i) A3 (specially the eco-tourism facilities) is highly valued; (ii) when addressing smaller scales proportionally higher values of WTP are obtained for similar environmental improvements. This effect has long been discussed in the CV literature (e.g. Mitchell and Carson, 1989; Carson, 2000; Venkatachalam, 2004), and occurs when the WTP for one good is not significantly different from the WTP for a more inclusive good.

Another key point determining the WTP of respondents to improve the system's quality was related to their knowledge of the system, for direct use, non-use and existence values. According to our results, WTP values slightly decrease as the distance from the resource increases. This finding is also in accordance with other studies that also tested the distance decay effect on asset valuation (e.g. Bateman et al., 2006; Hasler et al., 2009). This information can then be used to estimate the mean WTP of households to protect and improve the system quality, where was possible to see that the mean WTP to move from A1 to A2 was circa €10, while the mean WTP to move from A2 to A3 was around €20 per annum. These outcomes were also reinforced by the WTP/income ratios (despite WTP values are given by household and mean income is given by individual, the WTP/income can be used as a proxy), which allows to highlight the validity of the achieved results. The stated WTP amounts did not exceed 5% of their income, which is a theoretically acceptable value (Monarchova and Gudas, 2009). Since the achieved values are measured for only a sample of the population, to estimate the total value for a resource those values need to be aggregated over large geographic levels, and so the distance decay function plays a crucial role. Nevertheless, this poses the problem of defining the relevant population or the 'geographic extent of the market' (Smith, 1993; Hein et al., 2006), which might limit the applicability of aggregated values to decision-making processes (Garrod and Willis, 1995). In an attempt to overcome the first issue, and knowing that aggregate benefits depend on per-person benefit and on the number of beneficiaries, a value function based on the distance-decay should be applied (Hanley et al., 2003), which theoretically should provide different aggregate values than simply multiplying the mean WTP value by the number of inhabitants in the considered regions (Bateman et al., 2006).

4.2. Zero responses analysis

CV methods are useful in estimating the use and non-use values, since they rely directly on respondents' answers regarding the importance they give to certain ecosystem assets. However, this kind of method can fail to determine the correct economic value of a good if respondents' do not express their true estimation of its value (García-Llorente et al., 2011). This can be especially important in cases were a high number of zero bids responses are recorded. In our case study, as with previous ones (e.g. Meyerhoff and Liebe, 2006), a high percentage of zero bids was observed. Considering the importance of obtaining the most informative possible data in valuation studies (Del Saz-Salazar et al., 2009), motives that determine zero bids, and more specifically protest behaviours, should be explored. In practice, the common way of dealing with protest responses is through its elimination from the sample analysis (Morrison et al., 2000). However, issues have been raised questioning this procedure. For example, Meyerhoff and Liebe (2006) stated that some of those who are willing to pay might also hold protest beliefs, what strongly supports recent evidences claiming that censoring protests answers is unjustified (Jorgensen and Syme, 2000; Barrio and Loureiro, 2010). What emerged from this research was that protest answers provided some clues regarding system status, trends and importance to local and wider populations.

The relation between system use/knowledge had a significant influence on respondents' attitudes. This relation was particularly evident for the Mondego Estuary survey version, which may induce two conclusions: a) the Mondego River is better known than the estuary and so it is easier for respondents to associate a WTP value to the system; or b) the Mondego Estuary survey version only looks at the improvements of a small subset of the whole system, determining the lower availability of respondents to contribute. Our results showed that zero bids responses have a determinant role on the sample's representativeness and on the overall WTP estimates in CV studies. Therefore, the inclusion of protests' beliefs and motivations should be taken into account when determining the system's value, since including only the WTP estimations can lead to a loss of information about the heterogeneity of preferences of populations that may be relevant in management contexts. The way to achieve it could be, for example, through the identification and discrimination of types of protest classes i.e. through the application of econometric models (e.g. Marta-Pedroso et al., 2007; Barrio and Loureiro, 2010).

4.3. Practicability of applying non-market valuation techniques to estimate the value of water quality improvements

Regarding the consistency of the CV method used, some authors (e.g. Mitchell and Carson, 1989) argue that this referendum format tends to lead to 'yea-saying' bias, where respondents tend to agree to contribute even if they are not totally confident regarding the presented alternatives. Nevertheless, according to Mitani and Flores (2010) it is hard to measure and determine the causes of hypothetical bias, as well as its underlying role in stated preference economic analysis, with some authors arguing that hypothetical WTP may exceed the actual value by a factor of two or three (Murphy et al., 2005). Other issue to be taken into account is that this survey results infer mostly the non-market benefits provided by wetlands. Since most of the outputs, functions and services that wetlands generate are not traded in markets, non-market valuation techniques (as the CV) could be used to determine the value of their benefits (Birol et al., 2005). In this context, this study pretended to outline the relation among policies implementation that aim at good ecological conditions of systems (e.g. WFD), ecosystem services approach role to society and to the ecosystem inner functioning, and its measurement (using the CV method). Our findings allow identifying that both use and non-use values are reflected in respondents´ WTP. If a full economic analysis of the system value was required (e.g. cost-effectiveness assessments within the WFD), it should rely on the inclusion of both non-market and market estimates. Therefore, the water quality improvements valuation cannot be confused with the value of the entire ecosystem.

4.4. Guidelines for structuring and conducting management actions

Given the increasing pressures upon water environments in general, through the application of valuation tools and resultant policy measures implementation, a compromise among ecological integrity, with social welfare fairness of local populations and economic activities sustainability may be attained. In this study, water resources were viewed as a functional component that linked several interconnected ecosystems within a catchment area. The outcomes highlighted the respondents' recognition that ecosystems assets are interrelated and that there is an overlap between functions (biodiversity assets and ecosystem integrity) and benefits (activities enrolment) obtained from it. The question that arises next, as identified by Ahearn et al. (2006), is how to include these estimates into policy and management actions, which depends not only on the policy context, but also on the viability of those estimates in that context.

Based on the gathered information, to improve the efficiency and straightforwardness of management actions, some issues must be taken into consideration by water authorities:

- *Socio-economic profiles*: people with higher education attainment and higher income level are more aware of conservational and environmental issues. The significant relation between education and environmental awareness (e.g. pollution level), reinforces the idea that more efforts on environmental education programs regarding conservation policies should be made, especially designed for older people (since a higher percentage of people more than 30 to 40 years old tend to consider the system as 'not that polluted'), with lower education levels and incomes, along with the implementation and instigation of current measures. A higher emphasis on environmental education programs could be one way to highlight the importance of improving and preserving natural resources, aiming at long-term environmental benefits.
- *Geographic sampling level effect*: people are more willing to pay for systems closer to their homes than at a national level. In this sense, this work seems to reinforce sustainability utmost 'Think global, act local' (World Commission on Environment and Development, 1987). Following a successive ladder of integration, it provides an example of how a CV study could address a specific public policy issue, estimating the benefits and importance that natural wetlands may have to local and national populations (Figure 5) and so contributing to a more integrative vision for its management. As so, a successive integration of information, ranging from individuals preferences (awareness and uses), household levels (e.g. through their WTP), to city, regional or even district levels (e.g. aggregate WTP and municipal plans), a national (or even wider) strategy can then be applied and moulded according to the socio-economic profile of the regions where the actions plans have to be implemented. Performing such an integration of information for several parts of a system turns possible to elaborate wider management strategies that meet, for example, the whole basin districts objectives. Once again, and through the elaboration of such an analysis for several systems, is possible then to develop more accurate strategies at a national level, with a full integration of needs and policies.

Figure 5. Successive (hierarchical) levels of information integration that may contribute to a better design of sustainable management strategies, for natural resources conservation.

- *Embedding effect*: when including several ecosystem assets and wide range of habitats, a proper value may not be given to each and similar values may be obtained when performing the same analysis for only a small subset of habitats. Moreover, the level of knowledge of the system by respondents is also an important factor to take into account when designing management actions. In this study, 14.9% of the respondents have never heard about the Mondego Estuary (Portugal level), while only 1.6% have never heard about the Mondego River. From those that have already heard, 45.8% and 46.7% do not use it often, respectively. Managers should be sensitive to this effect.
- *Generalisation of results*: this study highlights the importance of the inclusion of estimates from local and basin levels (populations that rely and use the system), but also of the estimates obtained from national population, including the non-use benefits of those who do not reside in the Mondego catchment area. If only the estimates from populations living in the surroundings of the system were included, higher values could be achieved for the obtained benefits. However, the exclusion of the values from the areas in the nation not under the direct influence of this system may mask the true policy strength of this kind of issues and methods.
- *CV inherent advantages and constraints*: the use of CV in this system allowed to translate respondents' preferences for the proposed changes in monetary terms which is of importance to compare alternative management options as then costs and benefits of such options can be compared on the same basis. Moreover, it allowed to estimate the importance of the system for social wellbeing and its contribution to economic activities, such as the importance given to eco-tourism projects' development or the concern regarding the impacts that surrounding activities may have on local resources quality.
- *Protest effects*: a high number of protest responses were registered despite the high percentage of respondents showing interest in seeing the improvements implemented. The protest responses expressed the conviction that the Government and/or Water authorities should be the ones responsible for 'paying' the water quality improvements, since the respondents considered to be already overloaded in taxes.

5. General conclusions

From this work four main conclusions should be outlined:

a) respondents presented a very high interest in seeing the implementation of improvements, but only around half were really willing to pay;

b) respondents are willing to pay mainly due to their belief that it is actually possible to improve water quality status, with higher WTP rates coming from those living nearby, with higher incomes, higher education levels, and/or being users of the system;

c) most of the zero WTP bids were protests, mainly against the current water bills' value and water authorities' behavior. Nevertheless, these answers should also be included in the ecosystem valuation, since they show that respondents recognize a great deal of complexity within the ecosystem and overlap between functions, quality and benefits;

d) it was possible to infer that the passive use values and indirect values of natural catchments are likely to be quite large, and hence worth considering in program design and implementation. Thus, when calculating aggregate benefit values the ideal would be to include national estimates for the passive use values as well of the area where is located the system under study.

Despite the interesting results, a number of issues still remain open, for example the fact that aggregated values estimation should be done with caution, given that almost half of the respondents voted against the action plan implementation (mainly protests reasons). Also, possible gaps on the survey evidences can be filled through future statistical analysis that will provide further insights on:

1. The role of protest classes to econometric models application to aquatic resources, and aggregate value estimations;
2. The generalization of preferences for water quality improvements by users *vs.* non-users or local *vs.* national populations; and
3. The integration of environmental-economic information into multi-criteria analysis procedures, to better inform managers and decision-makers concerning social welfare changes and priorities.

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Temporal stability in estuarine systems: implications for ecosystem services provision

CHAPTER FOUR

Abstract

Stability is thought to rely on the richness (identity) and abundance of the species present in an ecosystem, where higher biodiversity promotes higher stability. Several attempts have been done to test this connection; however, there is still a lack of comprehension regarding the relation between biodiversity and stability. The stability concept is a collective term, defined via three fundamental properties: constancy, resilience and persistence. This manuscript uses theoretical and experimental evidence to explore the effects of biodiversity on estuarine stability, using the temporal stability (TS) measure as a proxy. In 1999 Tilman proposed the use of TS to test the diversity-stability hypothesis in a decade-long grassland experiment. Can TS be useful in estuarine systems? Our approach attempted to analyze estuarine stability from complementary perspectives by allowing the measurement of stability change (i) depending on species number and abundance; (ii) according to the different habitats of the same system; and (iii) disturbances influence. The question that imposed next was if this system property could be related with estuarine services provision. From this study, the main outcomes were that different TS values were found for the same abundance (N) values and the same was observed for species richness (S); TS maximum values were achieved at an intermediate diversity range; and TS increased with species abundance. In general, our results suggest that temporal stability might be useful to address sustainable management of estuarine systems.

Keywords:
| Biodiversity
| Temporal Stability
| Disturbance
| Habitats
| Ecosystem services
| Estuary

1. Introduction

A long and rather confusing debate has taken place in ecology relating system's biodiversity with several stability properties. Grimm et al. (1992) and Grimm and Wissel (1997) papers presented a methodological approach to address the puzzling of terms and concepts that usually underneath ecological stability. According to Grimm and Wissel (1997), the stability concept is a collective notion or term, which is defined via three fundamental properties: constancy (a system staying essentially unchanged), resilience (the ability of a system to return to the reference or dynamic state after a temporary disturbance) and persistence (the system ability to persist through time). Throughout this manuscript, the term 'stability' is assumed to represent this collective term while emphasizing the close connection between the three fundamental properties *sensu* Grimm and Wissel (1997).

Several attempts have been undertaken to investigate the relationship between biodiversity and the stability properties of an ecosystem, using different proxies, habitats or types and levels of disturbance (e.g. Remane, 1934; MacArthur, 1955; Odum, 1959; May, 1972; Pimm, 1984; Ives et al., 1999; Loreau et al., 2001; Balvanera et al., 2006; Isbell et al., 2009; Valdivia and Molis, 2009; Campbell et al., 2011; Godbold et al., 2011). One of the hypotheses tested states that 'higher biodiversity promotes higher stability' (e.g. MacArthur, 1955; Elton, 1958; Odum, 1959; Margalef, 1969). Several decades after its formulation, there is still a lack of comprehension regarding the relation between biodiversity and stability (e.g. McNaughton, 1977; Hughes and Roughgarden, 1998; Ives et al., 1999; Worm et al., 2006; Ives and Carpenter, 2007; Baraloto et al., 2010). One of the major difficulties relies on the selection and use of tools and measures able to correctly 'quantify' the system stability properties. As a consequence, our ability to perform an accurate and integrative management of our natural assets is reduced. Why? If we are not able to measure the ecosystem capacity to respond to different drivers and pressures, how can we then implement an adaptive management (*sensu* de Jonge et al., in press) able to optimize ecosystem services (functions of ecosystems with value for human wellbeing; see Fisher et al., 2009) provision and ecosystem preservation and functioning? In other words, how can we tangle the implications of stability on ecosystem services provision?

A few studies (e.g. Srivastava and Vellend, 2005; Tilman et al., 2006; Bodin and Wiman, 2007) have tried to assess the connection between ecosystem stability and services provision. Although not exempt of criticism, some studies suggest that ecosystem functions are more stable through time at relatively high levels of biodiversity (e.g. Hooper et al., 2005). It is suggested that the level and stability of ecosystem services, over space and time, tend to improve with increasing biodiversity (Kremen, 2005; Díaz et al. 2007; Winfree and Kremen, 2009; Norris et al., 2010; Haines-Young and Potschin, 2010). Nevertheless, the majority of these studies were done in terrestrial ecosystems (e.g. Kremen et al., 2002; Klein et al., 2003; 2007; Tilman et al., 2005). Attempts to extend the debate on ‘biodiversity *vs.* stability *vs.* ecosystem services provision’ to aquatic systems are rare (e.g. Valdivia and Molis, 2009). Transitional habitats, like estuaries, are particularly challenging for many reasons, from which we highlight three: 1) biological communities are under naturally stressful conditions (Elliott and McLusky, 2002); 2) the biota is under multiple and historical anthropogenic pressures (Wilkinson et al., 2007); 3) the estuarine communities are generally characterised by low number of species and high species abundance (Elliott and Quintino, 2007), although their number is increasing due to invaders (e.g. Nehring, 2006).

Tilman (1999), Lehman and Tilman (2000) and Tilman et al. (2006) proposed the use of ‘temporal stability’ (TS - defined as the ratio of mean abundance to its standard deviation) to test the diversity-stability hypothesis in a decade-long grassland experiment. An interesting question coupled to their idea is: Can TS be useful in estuarine systems?

The main objectives of this study were to test the behaviour of TS in an estuarine system (using macroinvertebrate data) and to evaluate if TS might be used as proxy for estuarine stability. In concrete:

1. Question 1 (Q1): How does TS change with species number (S) and abundance (N)?
2. Question 2 (Q2): Does TS change between different habitats of the same system?
3. Question 3 (Q3): Does TS change with different disturbance levels?

2. Methodology

2.1. Study-site description and sampling procedures

To test TS behaviour, different datasets were used, collected in two Portuguese estuaries: Mondego and Mira (Figure 1). The systems significantly differ in terms of pressures such as land use and occupation. Table 1 presents the main socio-economic features for both systems.

Table 1. Main characteristics of the Mira and Mondego catchment areas, comparatively to Portugal values (data from 2007).

	Area	Population	%	Population density	Land use					
	(Km^2)	(nº of individuals)		(inh/Km^2)	Urban	Others	Industrial	Tourism	RAN	REN
					(ha)					
Portugal	92 090.1	10 126 880	-	115.3	484 877.3	38 197.5	76 784.0	19 070.9	x	x
Mira										
Basin	798	81 759	0.8	25.6	8 170.1	409.2	391.5	26.3	2 310.5	174 290.7
Estuary	5	25 510	0.3	14.8	1 217.2	44.2	20.7	23.8	x	72 510.6
Mondego										
Basin	6 645	707 352	7.0	76.4	61 184.2	2 628.7	7 144.4	1 147.2	18 120.9	322 529.4
Estuary	7.2	63 229	0.6	166.8	2 537.0	165.2	1 170.9	379.7	x	22 738.5

Sources: INE – Statistics Portugal; INAG – Water Institute.
Legend: RAN= National Agricultural Reserve; REN= National Ecological Reserve; x – data not available.

Benthic communities were chosen as proxy for the ecological status of the entire ecosystems for the reason that biological communities are a product of their environment, and also because different benthic organisms have demonstrated to have distinct habitat preferences and pollution tolerance levels (e.g. Gaston et al., 1998; Pinto et al., 2009).

A) Mondego Estuary:

The Mondego Estuary is located on the northwestern coast of Portugal (Figure 1A). This intertidal warm-temperate system is divided, in its terminal part, into two arms (North and South arms) divided by the Murraceira Island. These two arms present very different characteristics being considered as two sub-systems, where the North Arm has been considered as possessing benthic communities' impoverishments relative to the South Arm, mainly due to higher sediment instability (Marques et al., 1993). The North Arm is deeper than the South Arm (5–10m during high tide), presents stronger daily salinity changes (the freshwater flows basically through this arm), and the bottom sediments consist mainly of medium to coarse sand (Marques et al., 1993). This estuarine branch constitutes the principal navigation channel, supporting the harbour and city of Figueira da Foz, and is subject to regular dredging activities. The South Arm is shallower (2–4m during high tide) and consists of mostly sandy to muddy channel beds. Until recently (2006), the upstream connection to the main river course was almost closed by sediment deposition. This constraint forced the water circulation to be mainly dependent on tidal penetration and the freshwater inflow of a tributary, Pranto River, controlled by a sluice (Marques et al., 1993; Patrício et al., 2004). This South Arm sub-system is also characterised by large areas of intertidal mudflats (almost 75% of the area) exposed during low tide (Neto et al., 2008). The combined effect of an increased water residence time (*circa* 9 days) and high nutrient concentrations lead to the occurrence of seasonal blooms of *Ulva* spp. and a concomitant severe reduction of the area occupied by *Zostera noltii* beds, previously the richest habitat in terms of productivity and biodiversity (Marques et al., 1993, 2003; Patrício et al., 2009). Moreover, the entire estuary is characterised for being under permanent anthropogenic pressure (e.g. dredging activities, nutrients runoff).

In this system, two major habitat types were considered: the subtidal and the intertidal.

» Subtidal sampling program: samplings were conducted during the spring of 1998, 2000, 2002, 2005, 2006 and 2007. In 2006, sampling was carried out in the four seasons for comparison with the Mira Estuary. Based on the estuarine salinity gradient and subtidal soft bottom habitat characteristics, the Mondego's lower estuary can be divided into four habitats (Figure 1A): euhaline estuarine sand (EE: stations 1 and 2), North Arm polyhaline sand (PS: stations 12 and 14), South Arm polyhaline sand (PSM: stations 3 and 4), and South Arm polyhaline muddy sand (PMS: stations 6 to 9) (Teixeira et al., 2008). Five replicate samples were randomly collected, at each station, using a van Veen LGM grab, with 0.078m^2 dredging area, immediately sieved through sieve-screen with a 0.5mm mesh, and preserved within a 4% formalin solution. In the laboratory, samples were sorted and identified to the lowest possible category, preferentially to species level. Once identified and counted, biomass was estimated as ash free dry weight (g AFDWm^{-2}) by drying to constant weight at 60ºC and ashing at 450ºC for 8h.

» Intertidal sampling program: sampling was conducted fortnightly (February 1993–June 1994), and monthly (July 1994–December 1995; January 1999–December 2000; January 2001–December 2003; June 2008-July 2009). In all cases, sampling was performed at three sites in the South Arm: within the *Z. noltii* meadow, in an intermediate area and in a bare sediment area (Figure 1). Each time, at each site, six replicate cores were taken. The sediment was washed through a 0.5mm mesh sieve and the biological material preserved in 4% buffered formalin. Samples were identified to species level and subsequently dried at 70ºC for 72h to estimate biomass as dry weight (DW), and ash free dry weight (AFDW) after combusting samples for 8h at 450ºC. For each time period, the weights of all taxa were summed to obtain an annual average standing stock.

B) Mira Estuary:

The Mira system (Figure 1B) is located in the southwest coast of Portugal, and has a flood-tide delta at its entrance with shallow tidal flats that are exposed during low tide (Ferreira et al., 2003). This estuary is a Ria-type estuary (Fairbridge, 1980) bordered by 285 ha of salt-marsh (Costa et al., 2001). The mean depth is about 6 m with a maximum of 13 m (Castro et al., 2002). The mean freshwater inflow was 7.13 m^3s^{-1} but in dry years it can be close to 0 m^3s^{-1} (Costa et al., 2001). The estuary is considered as relatively undisturbed, free from large urban and industrial areas (Castro and Freitas, 2006). Nevertheless, main stress factors can be due to some agriculture and aquaculture farms (Castro and Freitas, 2006). Seasonal impacts due to tourism activities may also occur at the estuarine mouth (Costa et al., 2001).

Samplings of subtidal soft-bottom benthic macroinvertebrate community were done seasonally, in 2006, following identical sampling strategy and methodologies as for the Mondego Estuary subtidal sampling program (the main differences were related to the fact that only three replicate samples were randomly collected, at each station, using a van Veen LGM grab, with 0.05m^2

dredging area). Samples were collected along five estuarine stretches (EE-Euhaline Estuarine; P-Polyhaline; M-Mesotidal; O-Oligohaline; F-Tidal Freshwater) (Figure 1B).

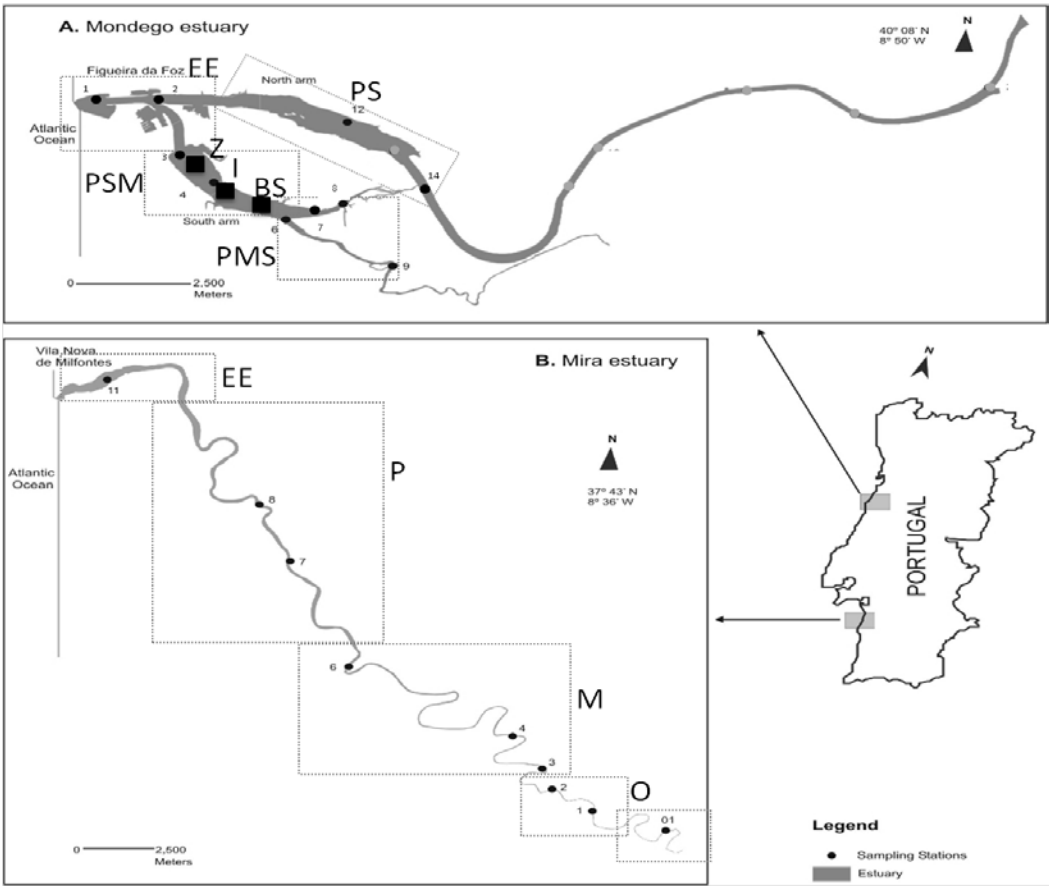

Figure 1. Study-sites. A. Mondego Estuary; B. Mira Estuary. In the figure, black squares are the intertidal sampling areas (Z- *Zostera noltii* meadows; I- Intermediate area; BS- Bare Sediment), and dashed squares the subtidal stretches (EE- Euhaline Estuarine; PS- Polyhaline Sand; PSM- Polyhaline Sand-Mud; PMS- Polyhaline Muddy-Sand; P- Polyhaline; M- Mesotidal; O- Oligohaline; F- tidal freshwater) (figure adapted from Alves et al., 2009).

2.2. Stability measure

Temporal stability has been estimated using the coefficient of variation [CV=100/(standard deviation/mean)] (Tilman 1996), for which smaller values represent greater stability (Tilman, 1999). In this paper, we have tested the usefulness of using the Temporal Stability (TS) variations in biomass, as defined by Tilman (1999) (Eq. 1), as a stability measure in estuarine systems, over a 10 years period (from 1998 to 2007) which is thus comparable to the time period used by Tilman (1999) for their study. The TS of a system is quantified as mean macroinvertebrate biomass ($\overline{b}$, gC m^{-2}) divided by the standard deviation of community biomass production through time (Tilman, 1999; $\sigma_{xi \to xj}$; gC m^{-2}yr^{-1}):

$$TS = \frac{\overline{b}}{\sigma_{xi \to xj}} \quad (1)$$

Temporal Stability (TS) measures the percentage of variation around the mean, where larger values of TS represent greater stability, i.e., lower temporal variation around the mean (Tilman, 1999). If there was no variation at all, temporal stability would be maximal (infinite). When variation is large relatively to the mean, temporal stability is small (near 0) (Lehman and Tilman, 2000). TS measures relative stability for both non-equilibrium and near-equilibrium conditions (Tilman, 1999).

2.3. Questions and datasets

To test the behaviour of TS in an estuarine system and to evaluate if TS might be used as proxy of stability in these systems, we have considered three questions:

Q1: How does TS change with species number (S) and abundance (N)?

Q2: Does TS change between different habitats of the same system?

Q3: Does TS change with different disturbance levels?

In order to answer each question, different datasets were used (Table 2). To test if TS changes were significantly correlated with S and N (Q1) a Pearson correlation analysis was undertaken. To test if TS changed between different habitats of the same system (within intertidal areas or subtidal stretches; Q2) a one-way ANOVA test was performed. To test for significant differences between systems and among seasons for the subtidal communities (Q3), with respect to N, S and TS, separate PERMANOVAs based on Euclidean Distance after data normalization were performed. For the PERMANOVA tests a 2-way design was considered with 'system' and 'season' as fixed factors with 2 and 4 levels, respectively. For the tests, 4999 permutations were used and the 'Permutation of residuals under a reduced model' was the permutation method chosen. A significance level (*p*) of 0.05 was considered for all analyses.

Table 2. Datasets designation, system and different habitats considered, time period and data type used to answer to the identified question (S: species richness; N: species abundances).

Dataset designation	System	Habitat	Time period	Data type	Question
D1	Mondego	Intertidal 3 areas: - *Zostera* meadows - Intermediate - Bare sediment	1993-1995 1999-2003 2008-2009	Macroinvertebrate data (S, N)	Q1, Q2
D2	Mondego	Subtidal (4 salinity stretches)	1998-2007 (spring)	Macroinvertebrate data (S, N)	Q1, Q2
D3	Mondego	Subtidal (4 salinity stretches)	2006 (seasonal)	Macroinvertebrate data (S, N)	Q3
D4	Mira	Subtidal (5 salinity stretches)	2006 (seasonal)	Macroinvertebrate data (S, N)	Q3

3. Results

3.1. How does TS change with species number and abundance (Q1)?

Regardless the habitat, the general picture is that the TS increases with increasing species abundance (N) and thus presenting the asymptotic-type behaviour (Figure 2A), which was also shown by the significant Pearson correlations achieved in most treatments (Table 3). The intertidal bare sediment area presented TS values ranging from 3.63 to 0.72, with a mean TS value of 1.84; while *Z. noltii* meadows showed TS values ranging from 3.52 to 0.51, with a mean TS value of 2.23, presenting consistently higher mean TS values at higher values of species abundance and richness (S), also reflected on the significant Pearson correlations (Table 3). In the *Z. noltii* habitat, higher species abundance are related to higher TS values (Figure 2B), while in the bare sediment habitat lower N implied lower TS values (Figure 2A). Regarding the subtidal habitat, the maximum TS value (TS=3.04) was obtained for the Polyhaline muddy-sand area (PMS). In general, the subtidal samples from the different estuarine stretches exhibited higher values of TS with higher abundance, with a maximum value of TS around 1000 inds/m^2 (Figure 2A). The maximum TS value for all subtidal stretches was found in the range of 18 to 35 species (Figure 2B). Worth noting is that different TS values were found for the same N, and the same observation applied for S.

Table 3. Pearson correlations between TS and species richness (S) and species abundance (N) for the different datasets considered (in gray are highlighted the significant relations).

	r	std error	p
TS x N Mondego subtidal	0.293	0.138	0.165
TS x S Mondego subtidal	0.510	0.130	0.011
TS x N Mondego intertidal	0.454	0.148	0.012
TS x S Mondego intertidal	0.388	0.148	0.034
TS x N Mondego subtidal+intertidal	0.452	0.093	0.000
TS x S Mondego subtidal+intertidal	0.553	0.077	0.000
TS x N Mira season	0.360	0.116	0.024
TS x S Mira season	0.295	0.117	0.068
TS x N Mondego season	0.188	0.134	0.258
TS x S Mondego season	0.484	0.084	0.002
TS x S Mi+Mo season	0.458	0.064	0.000
TS x N Mi+Mo season	0.348	0.078	0.002

3.2. Does TS change between different habitats of the same system (Q2)?

Habitat heterogeneity assessment was done by the analysis of TS behaviour on the intertidal and subtidal habitats (Figures 2A and 2B) of the Mondego Estuary. Graphically the two habitats were clearly separated from each other, with the intertidal samples having higher species abundance than the subtidal samples (Figure 2A). Nevertheless, in both habitats a tendency was found for higher TS values in samples with higher species abundance. Slightly higher TS maximum values were found in the intertidal habitat compared to the subtidal one. In fact, for both habitats significant relations were achieved within species abundances (intertidal: F=6.405; *p*=0.005; subtidal: F=5.323; *p*=0.007). When inspecting the relation between TS and the number of species then the intertidal habitat exhibited TS values moderately higher as compared to the subtidal habitat ($TS_{intertidal}$: 1-2.5; $TS_{subtidal}$: 0.2-1.5), in both habitat types the highest TS values were achieved for the same intermediate range of species richness (between 18 and 35) (Figure 2B). No significant relations were observed for TS and S within the subtidal and intertidal habitats.

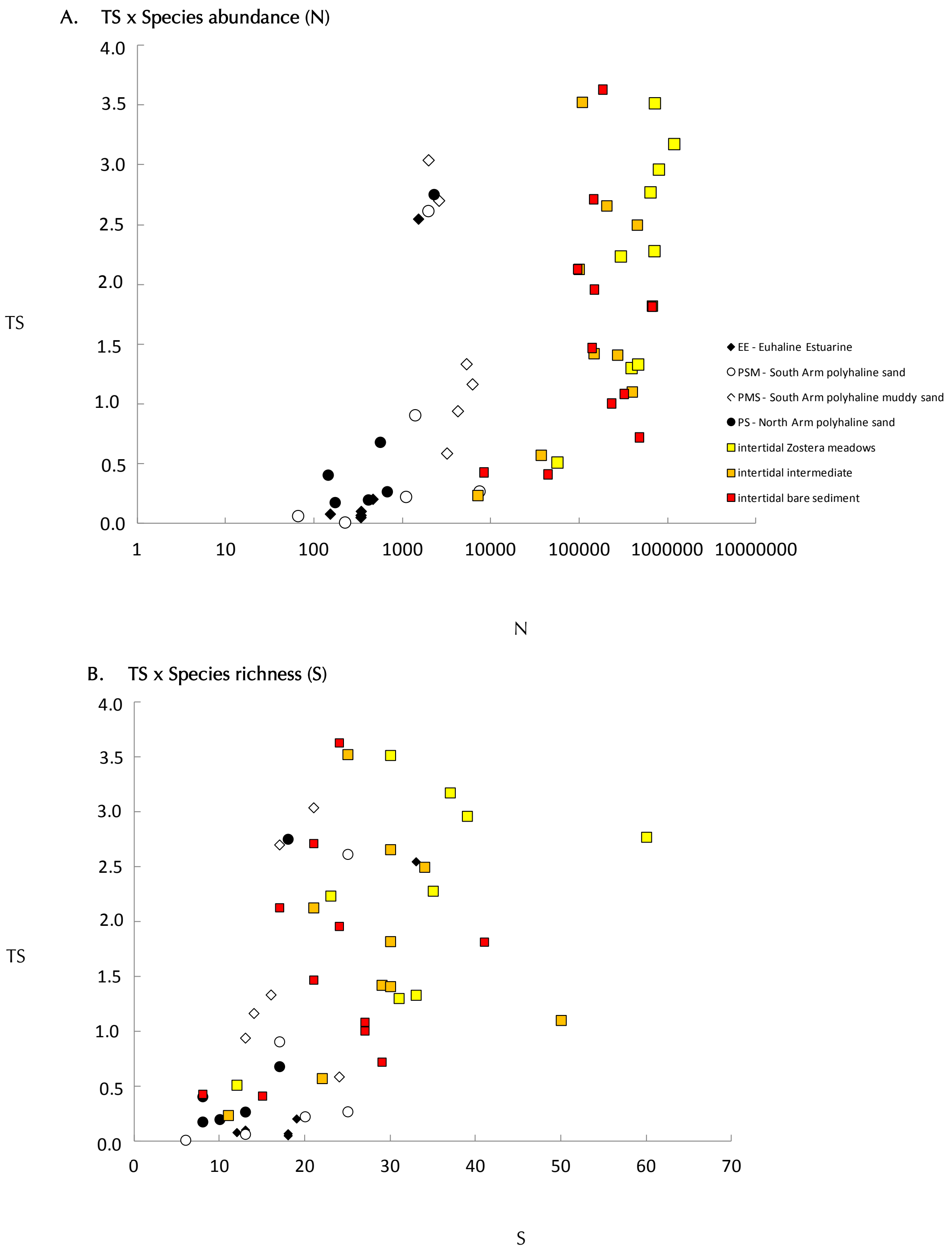

Figure 2. Variation of Temporal Stability (TS) with A. Species Abundance (N); B. Species Richness (S), in the subtidal and intertidal habitats of the Mondego Estuary. Subtidal stretches: EE – Euhaline estuarine; PSM – Polyhaline Sand-mud; PMS – Polyhaline muddy-sand; PS – Polyhaline sand).

3.3. Does TS change with different disturbance levels (Q3)?

When comparing systems with different human-induced impact levels (Mondego: impacted, Mira: well-preserved) distinct patterns were observed (Figure 3A and 3B). In general, for the same species richness (S) value, samples from the Mira Estuary presented higher temporal stability values (TS) than the Mondego Estuary ones (Figure 3A). Overall, more than half of the Mira samples presented TS values between 1.0 and 3.0, while most of the Mondego samples showed values between 0.1 and 0.5. In both estuaries, the maximum TS values were found for the same range of richness (11 to 20 species; Figure 3A) and TS increased with increasing species abundance (asymptotic-type curve) (Figure 3B). Significant differences between systems were evident for TS, N and S metrics (TS: $F=6.8672$; $p=0.0126$; N: $F=14.3800$; $p=0.0002$; S: $F=4.1747$; $p=0.042$).

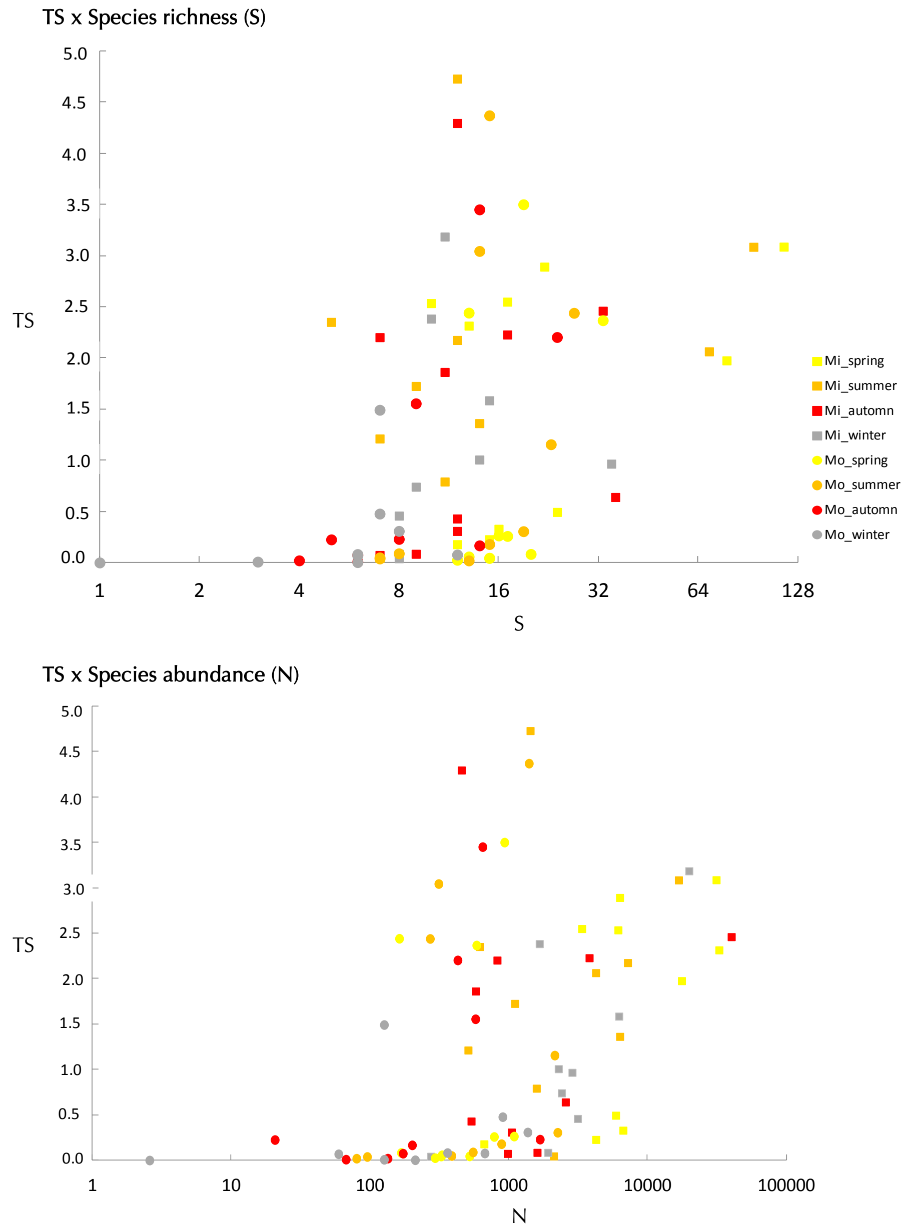

Figure 3. Variation of Temporal Stability (TS) with A. Species Richness (S) and B. Species Abundance in two estuaries (Mondego-Mo and Mira-Mi), at four sampling occasions (spring, summer, autumn, winter).

If the inter-annual variations (i.e. seasons) may be considered as natural pressures, the results seem to indicate that the factor ‘season’ was not a major driving force on the system neither considering temporal stability nor considering N. However, significant differences among seasons were found considering the species richness, differentiating spring from autumn (t=1.9435, p=0.0454), spring from winter (t=2.6413, p=0.0018), and summer from winter (t=2.0157, p=0.0302).

4. Discussion

4.1. Looking upon results from a wider perspective

The relative stability of ecosystems as complex as estuaries is poorly understood in ecological timescales, especially regarding the subtidal habitats (Elliott et al., 1998). In this demand, temporal stability appeared as a viable attempt to follow the stability of these ecosystems over the years. Comparing stability with structural attributes of biodiversity (e.g. S or N) in estuarine systems represents an innovative idea to address the implications of stability on ecosystem services provision. The point here is ‘how’ to link the two.

A possible approach may rely on the testing of the diversity-stability hypothesis (e.g. MacArthur, 1955; Hooper et al., 2005) by investigating the dependence of estuarine stability on species’ richness and abundances. Traditionally, it has been considered that greater species diversity implies more stable environments (Brose et al., 2003), by buffering them against natural and artificial disturbances and promoting ecosystem’s productivity (Smith, 1994). This may mean that more ecosystem functions will be maintained (Folke et al., 1996), as well as the associated services and goods. Therefore, stability and productivity of ecosystems are integral parts of overall biophysical integrity (Smith, 1994). Our results suggest that the diversity-stability relationships are neither linear nor monotonic, largely due to the complexity inherent to estuarine systems. The observed stability results appeared to be more associated to species abundance than to species richness, suggesting that biodiversity may act not only as a measure of biophysical integrity, as observed by Smith (1994), but also as a contributor to overall stability. Furthermore, results showed that higher levels of temporal stability were achieved for intermediate values of species richness (S). These findings challenge the conclusions of several other studies (mostly carried out in terrestrial habitats) that claim that diversity increases system stability (see Tilman, 1996; Hooper et al., 2005; Stachowicz et al., 2007), which usually is the end of the narrative. However, Valdivia and Molis (2009), working on intertidal epibenthic communities, found evidences of a negative biodiversity-stability relationship. This outcome suggests that probably diversity-stability relationships in estuarine natural communities may be more complex than those predicted by theoretical and manipulative experiments, and therefore those generalisations obtained from terrestrial habitats may not apply to all environments. An alternative hypothesis from the prevailing temporal stability is ecological network analysis (ENA) (Ulanowicz, 1980, 1986; Fath, 2004), which rely on the analysis of food web energy flows. This approach is considered as being a good candidate for further application in both analysing the functioning and judging of the quality of ecosystems (de Jonge et al., in press).

Results from ENA application in several case studies seem to illustrate that ecosystems with a high value of internal connectivity (which may compensate losses in functional redundancy) tend to be dynamically more unstable (vulnerable), because changes in the abundance of one species will immediately affect the others. On the other hand, ecosystems with higher species functional redundancy (Walker, 1992; Griffen et al., 2010) tend to be dynamically more stable and present lower internal connectivity. Recent studies defend that it is the temporal niche complementarity, not redundancy, which generates the stabilising effect of diversity (Loreau, 2000, 2004). In other words, provided there is sufficient initial redundancy the information content of the system (diversity) will increase over a period after which it will begin to decay. The amount of increase will be modulated by the initial complexity, meaning that high values of complexity (diversity) and redundancy are necessary, but not sufficient, conditions for self-organization to occur. A mere increase in redundancy will not lead to self-organization if the duration of the increase is too short (Ulanowicz, 1979).

In the present study, considering the temporal stability values estimated and the results from the application of food web models developed for the Mondego estuary intertidal habitat (Baeta et al., 2011), a trend of decreasing connectivity values is recognisable, at least in bare sediment habitats, concomitantly with an increase in temporal stability values, which indeed is what we should expect theoretically.

The application of the temporal stability approach to estuarine ecosystems primarily aims at discussing the advantages and disadvantages of using stability metrics, in association with biodiversity proxies (in the case species richness and abundances), as a benchmark to guide ecosystem's conservation and sustainable use. Eventually, the major advantages of this approach may be summarised as follows i) easiness to implement and to interpret; ii) dimensionless and scale invariant (Lehman and Tilman, 2000); and iii) intrinsically a non-equilibrium measure, essential attribute that characterize natural communities (Lehman and Tilman, 2000). Nevertheless, the achievement of a better understanding of the (potential) role of temporal stability on policies development will require further tests to: i) estimate thresholds in estuarine stability: the present formulae only allows for qualitative comparisons among habitats or systems; ii) predict systems responses: as it is presented now, the temporal stability metric act as a 'snapshot' metric, that captures the current system responses to disturbances. Despite these constrains, temporal stability seems potentially to be a good indicator reflecting estuarine dynamics. Therefore it may be argued that its consistent application to address anthropogenic impacts on estuarine systems would avoid the problem of overstepping environmental (biological or physical) limits, permitting a more sustainable use of these systems.

4.2. Temporal stability in estuaries: implications for ecosystem services provision

Over the last decades a great attention has been given to the potential effects that biodiversity changes may have on overall ecosystem functioning and on services provision (e.g. Loreau et al., 2001; Tilman, 2006; Díaz et al., 2007; Loreau, 2010). There is an increasing recognition that natural ecosystems provide a wide range of services and goods that are crucial to human well-being (MEA, 2005). These services are derived from the normal functioning of ecosystems (Loreau, 2010). However, a noteworthy question is how to define and measure the ‘normal functioning’ of highly disturbed and anthropogenically used ecosystems such as estuaries. In this sense, a different perspective may be given to this issue, considering the environmental (or habitat) management as inevitable to attain the final ecosystem service rather than being a step to achieve other services or goods (Mace et al., 2012). Ecosystem stability may therefore be an essential function to consider when formulating sustainable use policies on estuarine systems. Our results suggest that a reduction in the area of more structured habitats could imply a loss of estuarine stability, because the number of species available is strongly related to the surface area of a particular habitat, which is compliant with previous works (e.g. Dobson et al., 2006; Thrush et al., 2008). This then may in turn also lead to a loss of the associated ecosystem services. The conversion of habitats (e.g. *Z. noltii* meadows) to others with lower temporal stability (e.g. bare sediment, caused by eutrophication or as a result of dredging) may cause a shift in ecosystem functioning (due to a lower number of species at reduced surface area), leading to changes in the provision of valued services (e.g. shoreline protection or carbon and nutrient storage) (Duarte, 2002; Koch et al., 2009). This situation demands appropriate management actions that have to take into account both structural and functional elements of estuarine ecosystems, as argued by de Jonge et al. (in press).

The insurance hypothesis (Yachi and Loreau, 1999; Ives et al., 2000; Thébault and Loreau, 2005) states that higher species richness (considered for instance as proxy of more complex food web and higher redundancy) increases community-level stability by insuring that some species in a community are tolerant to different environmental fluctuations. According to Winfree and Kremen (2009), when an ecosystem service is supplied by several species, they will ensure ‘stabilizing mechanisms’, guaranteeing the system stabilization against temporal (Norris et al., 2010) or spatial (Loreau et al., 2003) disturbances. Accepting these notions, within the context of ecosystem services, implies that changes in biodiversity assets will be reflected not only on the capacity but also on the spatial and temporal provision of services (Norris et al., 2010). In fact, some works have demonstrated that stabilizing mechanisms can have a strong influence on the stability of certain services provisions, highlighting the insurance value of apparently ‘redundant’ species (e.g. Winfree and Kremen, 2009; Proulx et al., 2010). The outcomes of our study seem to support the ‘insurance hypothesis’ but only partially. We found a tendency for higher values of temporal stability at an intermediate range of species richness values, a decrease in temporal stability being observed when this intermediate number of species was exceeded.

Despite the strong physical and biological dynamics in estuarine systems (e.g. Costanza et al., 1993; Cognetti and Maltagliati, 2000; Neely and Zajac, 2008) these may represent different equilibrium states due to the influence of human induced disturbances. To measure an ecosystem’ stability is necessary to determine whether there are thresholds that separate different stability domains (Thrush et al., 2009) and presently the only sure way to detect a threshold in a natural system is to cross it (Carpenter, 2003). A good example is the present physical situation of the Ems

Estuary where the suspended load in the estuarine turbidity zone has increased by an order of magnitude due to canalisation and a step-by-step deepening of the river due to which the boundary conditions for ecosystem functioning have been changed dramatically (Schuttelaars et al., in press). Due to the changed boundary conditions the situation in that estuary can only be described as being a new stable system state. Nevertheless, it should be possible to identify signs of shifts in ecosystems that predict the risk of drastic future changes (Thrush et al., 2009). Temporal stability metrics can then work to detect these state changes within a system. Our results suggest that temporal stability might be useful to address sustainable management of estuarine systems. Nevertheless, further studies are necessary to ensure that TS is an effective proxy for estuarine stability. Two main future research issues were identified: 1) the need to compare TS values with network-based indicators (able to express flows and connectivity); and 2) the necessity to identify the role of keystone species and rare species in TS estimation.

5. Conclusions

The results presented in this work attempted to provide some insights to the long-standing diversity-stability debate focusing estuarine ecosystems. The main outcome of this study is that TS may be a helpful additional tool to ensure the sustainable use of transitional systems. In general, TS increased with species abundance and TS maximum values were achieved at an intermediate diversity range. This tendency was observed in two distinct estuarine systems (Mondego and Mira) and in different habitats within each system (subtidal and intertidal habitats). These outcomes apparently challenge early investigations for terrestrial environments that claimed that, at a community level, diversity increases temporal stability (e.g. Elton, 1958; Tilman, 1996; Lehman and Tilman, 2000). Nevertheless, as Grimm and Wissel (1997) alert, '*the validity of a stability statement is delimited by the ecological situation under observation*'; therefore, generalisations from our study results should be carefully undertaken.

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Linking biodiversity assets, ecosystem functioning, services provision and human wellbeing in estuarine ecosystems: Application of a conceptual framework

CHAPTER FIVE

Abstract

Assuming that human well-being relies on the services provided by well-functioning ecosystems, changes in the ecological functioning of a system can have direct and indirect effects on human welfare. Intensive land use and tourism have expanded in recent decades along coastal ecosystems, together with increasing demands for water, food and energy; all of these factors intensify the exploitation of natural resources. Nevertheless, many of the interrelations between ecosystem functioning and the provision of ecosystem services still require quantification in estuarine ecosystems. A conceptual framework to assess such links in a spatially and temporally explicit pattern is proposed. This framework relies on three consecutive steps and discriminates among biodiversity structural components, ecosystem functioning and stability and the services provided by the ecosystem.

Abiotic factors and natural disturbances were found to have a direct effect on biodiversity, ecosystem functioning and the provision of ecosystem services. The observed changes in the species composition of communities had a positive effect on the ecosystem's productivity and stability. Moreover, the observed changes in the estuarine ecosystem services (ES) provision are likely to come from changing structural and abiotic factors and from the loss or decline of locally abundant species. This study also indicates that linear relationships between biodiversity, ecosystem functioning and services provision are unlikely to occur in estuarine systems. Instead, cumulative and complex relations are observed between factors on both temporal and spatial scales. In this context, the results suggested several additional conclusions: 1) biodiversity and ecosystem functioning interaction with human well-being need to be incorporated into decision-making processes aimed at the conservative management of systems; 2) the institutional use of research results must be part of the design and implementation of sustainable management activities, integrating biodiversity indicators and ecosystem services that are important for human well-being; and 3) more integrative tools/studies are required to account for the interactions of estuarine ecosystems with surrounding socio-economic activities. Therefore, when performing integrated assessments of ecosystem dynamics, it becomes essential to consider not only the effects of biodiversity and ecosystem functioning on services provision but also the effects that human well-being and ES provision may have on estuarine biodiversity and ecosystem functioning.

Keywords:
| Ecosystem Approach
| Biodiversity-Ecosystem Functioning (BEF)
| Ecosystem services
| Human well-being
| Estuary

1. Introduction

Ecosystems deliver services of great value to human society (Pearce and Moran, 1994; Costanza et al., 1997; Daily, 1997). However, increasing anthropogenic pressures have led to a growing loss of biodiversity and changes in the internal functioning of ecosystems, reflected in the variation of benefits provided to human societies (Hooper et al., 2005). In 2005, the Millennium Ecosystem Assessment (MEA) published an assessment of the status of ecosystems and their capacity to benefit humans, concluding that most of the world's wetlands have been destroyed or degraded during the 20th century, thereby creating the need for integrative frameworks to approach the dynamics of whole systems. The central framework for this assessment (MEA, 2003) was a simple conceptual guiding principle:

biodiversity ⟶ ecosystem functioning ⟶ ecosystem services ⟶ human well-being

where each arrow represents a casual relationship (Naeem et al., 2009) and where ecosystem services may be seen as functions that ultimately benefit humans (Costanza et al., 1997; Daily, 1997; Naeem et al., 2009). This framework relies on the assumption that increased biodiversity augments ecosystem functioning, which improves ecosystem services and may eventually improve human well-being, depending on the elements involved. A number of studies have attempted to link the biological composition of ecosystems, given by biodiversity proxies, to the stability of ecosystem functioning (e.g., Remane, 1934; MacArthur, 1955; Odum, 1959; May, 1972; Pimm, 1984; Ives et al., 1999; Loreau et al., 2001; Balvanera et al., 2006; Isbell et al., 2009; Godbold et al., 2011). These studies assumed that such links may have a determinant role in ecosystem services delivery (e.g., Costanza et al., 1997; Turner, 2003; Srivastava and Vellend, 2005; Tilman et al., 2006; Díaz et al. 2007; Haines-Young and Potschin, 2010). Typically, researchers have considered that if ecosystem biodiversity could be linked with functioning then it would follow that ecosystem services are related to human well-being (Naeem et al., 2009).

Although our consideration of ecosystem services focuses on the link between ecological functions and human well-being, it is also important to consider the prior link between biodiversity and ecosystem functioning (Morling et al., 2010). Biodiversity plays a key role in ecosystem services provision (Mace and Bateman, 2011): 1) biodiversity supports the delivery of ecosystem services (Díaz et al., 2006), acting as insurance against change (increased redundancy associated with higher diversity may buffer ecosystems against change, contributing to higher system resilience) (Ulanowicz, 1979; Hooper et al., 2005) and offering more options for the future (Yachi and Loreau, 1999); 2) genetic and biological species diversity may directly supply some goods, such as animal and plant breeds (MEA, 2003); and 3) many components of biodiversity are valued by people for altruistic reasons (e.g., appreciation of wildlife, contribution to spiritual or educational motifs and recreational experiences), although biodiversity, per se, cannot be considered a service (Haines-Young and Potschin, 2010). Thus, when addressing natural resources management, the challenging issues are determining the nature and sensitivity of the relationship between environmental quality/biodiversity assets and the provision of services. Most of this discussion regards the links between biodiversity assets and ecosystem functioning and stability, which can be used as a proxy to the supporting services classes from the MEA, or intermediate services. Several studies have been conducted to address this issue (e.g., Pimm, 1984; Schwartz et al., 2000; Loreau et al., 2001; Tilman et al., 2005; Balvanera et al., 2006); nevertheless, the controversy persists.

To address these complex relations, the Convention for Biological Diversity (2004) requests use of the ecosystem approach (EA) to provide a clear integration into a holistic framework of all services provided to people by biodiversity and ecosystems. This approach defends an integration of the ecological, economic, and socio-cultural perspectives when evaluating an ecosystem (de Groot et al., 2002; Farber et al., 2002; MEA, 2005; Carpenter et al., 2009), thus providing a methodological framework for wetland management (de Jonge, 2007). In fact, ecosystem services clearly have ecological and socio-economic aspects whose interdependencies need to be clarified (Mace and Bateman, 2011) and described (de Jonge et al., in press). Therefore, it is crucial to understand the role and effects of biodiversity in a socio-ecological context (Carpenter et al., 2009). Despite the attempts to identify the potential relationships between biodiversity and the delivery of services, adequate quantitative data are not available (Norris et al., 2010). Nevertheless, a weak correlation has been demonstrated between areas rich in biodiversity (according to nature conservation designations) and those high in ES delivery (Naidoo et al., 2008; Anderson et al., 2009). In fact, a study by Norris et al. (2010) supports the idea that microorganisms, fungi and plants play a role in supporting and regulating services, whereas vertebrates are more important for cultural services, described as the 'cute and cuddling' services in de Jonge et al. (in press). However, because of the increasing pressures on natural resources, trade-offs among services have been verified. The general increase in provisioning services over the past century has been achieved through decreases in regulating and cultural services and in biodiversity (MEA, 2005; Bennett and Balvanera, 2007; Carpenter et al., 2009). In this context, it is essential that such trade-offs are recognised in ecosystem assessments (Carpenter et al., 2009). The suggestion to further clarify the relations between habitats, food web functioning via ecological network analysis (ENA) and the Driver-Pressure-State Change-Impact-Response (DPSIR) approach as performed by de Jonge et al. (in press) may be seen as a first step in that direction.

Because of the complexity and integration of concepts and methodologies, it is essential to clearly define the terms used in the present work:

a) Biodiversity is the variability among living organisms and their habitats from all sources, including diversity within species, between species and within entire ecosystems (Heywood, 1995). Because of data limitations in the present case, diversity measures were used to estimate biodiversity (according to Marques, 2001);

b) Ecosystem functioning refers to all of the biogeochemical processes occurring within an ecosystem, such as the cycling of nutrients, matter or energy (Naeem, 1998);

c) Ecological condition refers to the integrity of the ecosystem (Jorgensen et al., 2010); in the present study, ecological condition was expressed as the ecological quality status *sensu* European Water Framework Directive (EC, 2000);

d) Stability is a collective notion defined by three properties (constancy, resilience and persistence) (*sensu* Grimm and Wissel, 1997); in the present study, stability was expressed as temporal stability (see Pinto et al., submitted); and

e) Ecosystem services (hereafter ES) can be defined as the functions of ecosystems having value for human welfare (Fisher et al., 2009). According to the MEA (2005), ES can be classified into one of four categories: regulating (e.g., water purification); supporting (e.g., nutrient cycling); provisioning (e.g., food production); and cultural (e.g., opportunities for recreation).

The present work discusses the links between biodiversity proxies and ecosystem functioning in estuarine ecosystems, the role of macroinvertebrates in the provision of ES, and the use of ES as a tool to address the exploration and conservation of natural resources. Therefore, the present work should provide clues regarding the changes in estuarine biodiversity that are expected to have significant consequences for human well-being. In a first step, the effect of biodiversity assets on several ecosystem processes was analysed. In a second step, the relation between those processes and the capacity of the system to provide services (as a link to the socio-economic system) was considered. Finally, recommendations based on this integration identified gaps in our knowledge of estuarine functioning and future research needs. Five main objectives were outlined under these three main research steps:

Step 1 – Linking estuarine biodiversity to ecosystem functioning

1) To evaluate the performance of biodiversity structural proxies (e.g., number of species) and their relationships to functional indicators.

Step 2: Linking ecosystem functioning to services

2) To assess how measures of functional diversity (e.g., macrobenthic productivity) perform under different ecological status classifications;
3) To explore the relation between ecosystem functioning and estuarine stability; and
4) To analyse and understand the link between biodiversity proxies and ES supply.

Step 3: Linking changes in services to human well-being

5) To estimate how the achieved knowledge on biodiversity and ecosystem functioning can be used to improve ES provision and consequently human well-being.

2. Methodology

2.1. Study site description and sampling protocol

The Mondego Estuary is an intertidal warm-temperate ecosystem presenting a North and a South arm separated by Murraceira Island (Figure 1).

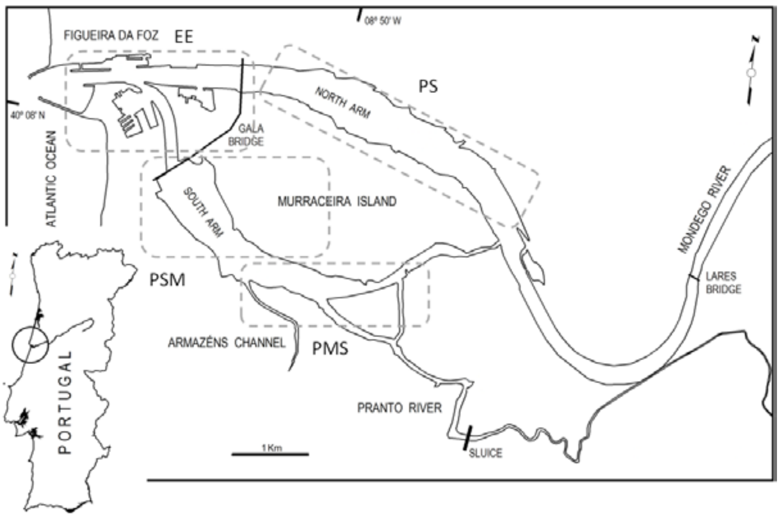

Figure 1. Mondego Estuary: sampling stations and estuarine stretches: Euhaline estuarine sand (EE), South Arm Polyhaline sand-mud (PSM), South Arm Polyhaline muddy sand (PMS), and North Arm Polyhaline sand (PS).

The two arms of the estuary have very different characteristics, constituting two distinct subsystems, with the North arm presenting impoverished benthic communities as compared to the South one. Actually, the North arm is deeper (5–10m during high tide), presents stronger daily salinity changes (the freshwater flows principally through this arm), and the bottom sediments consist mainly of medium to coarse sand (Marques et al., 1997). In addition, the North arm constitutes the principal navigation channel, supporting the harbour of Figueira da Foz, and is regularly dredged. The South arm is shallower (2–4m during high tide), predominantly with sandy to muddy bottoms, and up to recently (2006) the upstream communication with the North arm and the main river course has been almost totally silted or even completely interrupted. As a consequence, the water circulation in the South arm was mainly dependent on tides and on the freshwater inflow of a tributary, the Pranto River, which is artificially controlled by a sluice (Marques et al., 1997; Patrício et al., 2004). The South arm subsystem is also characterised by large areas of intertidal mudflats (almost 75% of the area) exposed during low tide (Neto et al., 2008).

In the early 1990s, when upstream communication between the two arms was completely interrupted, the combined effects of increased water residence time in the south arm (ca 9 days) and increased nutrient concentrations led to seasonal blooms of *Ulva* spp. along with a concomitant severe reduction of the *Zostera noltii* meadows area, which was previously the richest habitat in terms of productivity and biodiversity (Marques et al., 2003). The entire Mondego Estuary is under permanent anthropogenic pressure, which drives its development as an ecosystem (Marques et al., 2003; Pinto et al., 2010). First, the surrounding area supports many economic activities, which vary from extractive activities (e.g., agriculture or salt production) to on-site activities (e.g., leisure or tourism). This panoply of uses influences the ecological condition and performance of the estuarine communities (Pinto et al., 2010). In addition, estuarine waters also support extractive activities, such as the capture of bivalves and fish.

Previous studies considering the estuarine salinity gradient and subtidal soft-bottom characteristics described four distinct habitats in the Mondego's lower estuary (Figure 1): euhaline estuarine sand (EE), north arm polyhaline sand (PS), south arm polyhaline sand (PSM), and south arm polyhaline muddy sand (PMS) (Teixeira et al, 2008).

In each of these habitats, subtidal benthic communities have been regularly monitored following standardised protocols since the early 1990s. In the present work, we utilised data from sampling performed in the spring of 1998, 2005, 2006 and 2007. Each year, five sediment replicates were randomly collected from each habitat using a van Veen LGM grab with a 0.078 m^2 dredging area. The samples were immediately sieved through a 0.5 mm mesh screen and preserved in a 4% formalin solution. For the production estimates were used four sampling seasons (spring, summer, autumn, winter) following the same protocol. In the field, abiotic parameters were also measured (e.g., $\%O_2$, O_2, salinity, temperature). In the laboratory, the biological samples were washed through a series of nested sieves of 1.0 and 0.5 mm, sorted and identified to the lowest possible category, preferably to the species level. Once identified and counted, the biomass was estimated as ash-free dry weight (g AFDW m^{-2}) by drying the samples to a constant weight at 60°C and ashing at 450°C for 8 h. Nutrients concentration in the water column (e.g. $N\text{-}NH_3$, $N\text{-}NO_2$, $N\text{-}NO_3$, $P\text{-}PO_4$), Total Suspended Solids (TSS), Chlorophyll a content and sediment properties (e.g. %organic matter-OM, %coarse sand, %fine sand, %mud, and Particulate Organic Matter-POM) were estimated at the laboratory, following standard methods.

2.2. Framework adopted

A three-step approach was applied (Figure 2) to estimate the successive relationships linking biodiversity, ecosystem functioning, estuarine stability, ES and human well-being. Structured on existing frameworks and indicators, this approach aims to translate field measurements to social and economic standards using a multidisciplinary perspective. The Mondego Estuary was used as a practical example of an ecosystem under intensive pressure from both social (e.g., increasing population density) and economic changes (e.g., intense land occupation rates).

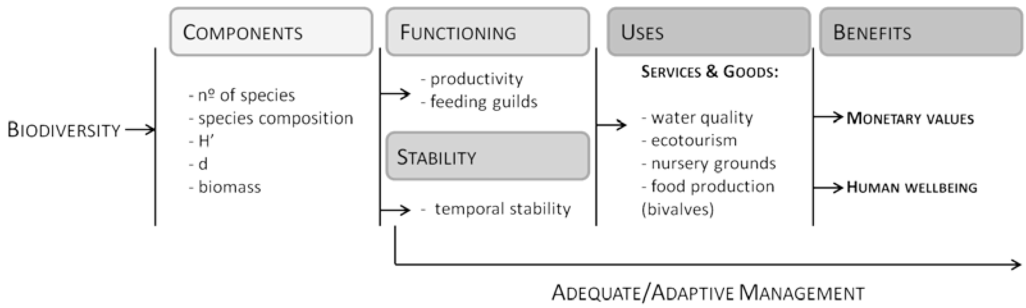

Figure 2. Framework adopted: for the biodiversity (e)valuation several indicators were considered, belonging to structural (components), functional (processes), uses and final benefits (always considering the impacts analysis and the spatial-temporal reality of the local system). This framework allows an integration of ecosystem approaches, essential for sustainable development requirements, involving several key-components as ecological, economic, or social dimensions.

2.2.1. Linking estuarine biodiversity to ecosystem functioning

Three biodiversity structural proxies were considered and quantified: the number of species (S), total abundance (N), and total biomass. Several measures of estuarine diversity were calculated, including the Shannon-Wiener index (H'; Shannon and Wiener, 1963), the Margalef index (d; Margalef, 1958), and a multimetric integrative index (BAT; Teixeira et al., 2008). Finally, based on BAT results, the Ecological Quality Status (EQS) was estimated following requirements of the Water Framework Directive (WFD; EC, 2000).

2.2.2. Linking ecosystem functioning with stability and services provision

The role of macrobenthic functional diversity within the estuarine ecosystem was quantified by using two proxies:

a) The *feeding guilds composition* for subtidal communities assigned each taxon to a feeding guild according to its feeding habit (carnivorous/omnivorous/grazers – G; surface deposit feeders – SDF; subsurface deposit feeders – sSDF; suspension feeders – SF) based on literature reviews (e.g., Fauchald and Jumars, 1979; Gaston, 1987; Garcia-Arberas and Rallo, 2002; Mancinelli et al., 2005). This proxy was calculated using the relative abundance of species in each trophic group and was expressed as a percentage.

b) The *macrobenthic productivity* of the system was calculated based on the methodology of Brey et al. (2001). The P/B ratio was estimated using an empirical multiple non-linear model, which incorporated biomass data and biotic and abiotic parameters. The weight-to-energy ratios are needed for application of the empirical method, and the biomass estimates were converted into KJ using the conversion factors for major taxonomic groups proposed by Brey et al. (2001) (method version 4-04) (worksheet provided in Brey, (2001); www.awi-bremerhaven.de/Benthic/Ecosystem/FoodWeb/Handbook/main.htm). This method has been used as an alternative empirical technique for the estimation of secondary production (after Cusson and Bourget, 2005; Dolbeth et al., 2005, 2007). Following the methodology of Brey et al. (1996), the P/B ratio was estimated by group. Production was then calculated by multiplying the P/B ratio by the average biomass. Because productivity changes are usually reflected in the year following a particular condition, three years were used to estimate the estuarine production: 2005, 2006, and 2007. Only spring data were available for 1998, so no production estimates could be computed for that year. Before estimating the productivity, the species constancy was calculated to measure the importance of species at each stretch (Desroy et al., 2002). Constancy is given by Eq. 1 (Desroy et al., 2002):

$$C_{ij} = \frac{n_{ij}}{n_j} \times 100 \qquad (1)$$

where *nij* is the number of occurrences of the species *i* in the station of group *j*, and *nj* is the number of stations in group *j*. Characteristic species were categorised as either constant (C>50%) or common (50%<C>25%) (Desroy et al., 2002). Those species observed for only one or two seasons per year were omitted from the data analysis.

Estuarine temporal stability (hereafter TS; Tilman, 1999) was used as a proxy for estuarine stability (see Pinto et al., submitted). Tilman (1999) defined TS (g C m^{-2} yr^{-1}) as the variation in species biomass, which is quantified as the mean spring biomass ($\bar{b}$, g C m^{-2}) divided by the standard deviation of community biomass production through time ($\sigma_{xi \rightarrow xj}$, y^{-1}) (Eq. 2):

$$TS = \frac{\bar{b}}{\sigma_{xi \rightarrow xj}} \qquad (2)$$

Although it is not a process, TS measures the percentage of variation around the mean, and larger TS values represent greater stability, i.e., lower temporal variation around the mean (Tilman, 1999). If there was no variation at all, the temporal stability would be maximal (infinite). When variation is large relative to the mean, the temporal stability is small (near 0) (Lehman and Tilman, 2000). TS measures relative stability for both non-equilibrium and near-equilibrium conditions (Tilman, 1999).

2.2.3. Linking changes in services to human well-being

Trends might be drawn based on the composition and functional roles of the species (e.g., structural or behavioural characteristics) occurring in an ecological system and possessing potential social interest (whether it has direct or indirect benefits). The relationships between abiotic indicators, biodiversity structural assets, and ecosystem functioning were tested using a Pearson correlation analysis (r).

The ES supply was used to estimate the link between the proxy for biodiversity-ecosystem functioning and the social importance of the estuarine ecosystem to human well-being (Figure 2). Locally the bivalve catch activities has a very strong socio-economic role, therefore, to estimate the link between biodiversity and human well-being the condition of the bivalve population was considered (used as the linkage between the system natural capital and human well-being). The flux of the estuarine services (ecosystem stocks and flows) can be measured, for example, as quantitative productivity over the years. For instance, the bivalves' biomass comprises a stock. This stock can then flow from the bivalve community into the catch of fisheries, which is the flow that is of value to society (Costanza et al., 2001). For the bivalve catch, two levels of potential interference were identified:

a) At a low scale the individual resource (in this case the biomass of the bivalve communities), its immediate environmental conditions, and the people who harvest it and earn additional income from the harvest (measured in tons of catch; data source: www.dgpa.min-agricultura.pt), and
b) At a high scale the industrial harvest, where the number of licensed boats, the catch (in tons), and the potential for greater interference with the ecosystem (data source: www.dgpa.min-agricultura.pt) were used as indicators.

3. Results

3.1. Linking estuarine biodiversity to ecosystem functioning

The estuarine biotic environment (structural and functional proxies) for each site and year is summarised in Table 1. In general, the values for species richness differed among sites and years. For 1998, 2005 and 2006, respectively, the species number values were 19, 19 and 46 at the euhaline estuary; 6, 32, and 22 at the polyhaline sand-mud site (PSM); 14, 22, and 28 at the polyhaline muddy sand (PMS) (both PSM and PMS belong to the south-arm subsystem); and 8, 18 and 21 at the north-arm polyhaline sand. These values were consistent with the behaviour of other structural metrics, such as species richness (d), diversity (H'), total biomass, and the EQS, which tended to increase gradually after 1998. The species abundance presented a relatively stable pattern when considering the Mondego Estuary as a whole, but significant differences could be observed among habitats and stretches. There was a substantial decrease in the polyhaline sand-mud area, which contributed to a decrease in the species number of the South arm.

Table 1. Structural, functional and stability indicators of biodiversity in the Mondego Estuary. ME – Mondego Estuary; EE – Euhaline Estuarine; PSM – Polyhaline Sand-Mud; PMS – Polyhaline Muddy-Sand; PS – Polyhaline Sand; SA – South Arm; M – Moderate ecological status; G – Good ecological status; and H – High ecological status.

			ME				EE				PSM				PMS				PS				SA			
		unit	1998	2005	2006	2007	1998	2005	2006	2007	1998	2005	2006	2007	1998	2005	2006	2007	1998	2005	2006	2007	1998	2005	2006	2007
Structural indicators																										
Ecological Quality Status	EQS		M	G	G	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	G	G	H	-
Number of species	S		30	49	70	-	19	19	46	-	6	32	22	-	14	22	28	-	8	18	21	-	14	39	39	-
Diversity (Shannon-Wiener Index)	H'		1.61	2.52	2.74	-	2.48	3.07	3.22	-	1.43	2.46	1.99	-	1.41	2.63	3.10	-	1.25	2.07	2.26	-	1.41	2.57	2.73	-
Species richness (Margalef Index)	d		1.01	2.23	2.07	-	1.65	3.46	3.15	-	0.94	2.77	1.70	-	0.87	1.96	1.93	-	0.68	1.60	1.61	-	0.88	2.23	1.86	-
Total density	de		5515	5878	4147	-	432	182	560	-	224	1965	828	-	4716	2060	2180	-	143	1672	579	-	4940	4025	3008	-
Total biomass	bi		7.01	26.84	26.83	-	2.29	2.07	9.91	-	0.05	12.94	4.63	-	4.43	10.28	11.88	-	0.23	1.55	0.41	-	4.49	23.21	16.51	-
Functional indicators																										
Grazers	G	%	14.13	26.04	31.79	-	17.54	45.07	29.11	-	19.09	15.91	18.60	-	12.83	44.31	37.53	-	51.13	12.44	43.13	-	13.05	32.00	30.49	-
Surface Deposit Feeders	SDF	%	80.61	35.00	43.27	-	28.07	39.44	46.58	-	80.91	21.73	26.95	-	85.26	53.22	54.84	-	48.87	25.40	31.88	-	85.11	39.57	44.46	-
subSurface Deposit Feeders	sSDF	%	3.58	36.14	20.32	-	53.51	12.68	20.55	-	0.00	55.24	53.36	-	0.07	0.59	1.29	-	0.00	62.05	17.92	-	0.06	24.28	20.66	-
Suspension Feeders	SF	%	1.68	2.74	4.42	-	0.88	2.82	3.77	-	0.00	7.12	1.09	-	1.84	1.68	6.02	-	0.00	0.11	6.67	-	1.77	4.04	4.19	-
Macrobenthic Productivity	mP	y^{-1}	-	63.74	67.34	29.89	-	5.21	2.31	2.15	-	29.01	40.41	4.18	-	19.79	12.22	22.59	-	14.13	11.92	2.69	-	53.58	64.61	29.01
Stability																										
Temporal Stability	TS	y^{-1}	0.59	2.36	1.43	0.62	0.20	0.08	0.05	0.10	0.01	2.62	0.91	0.22	1.17	2.70	3.04	1.33	0.41	2.75	0.68	0.20	0.55	2.84	1.97	0.76

3.2. Linking ecosystem functioning to stability to services

Within the estuarine communities (Table 1), there was an increase in the % of trophic groups usually associated with better environmental conditions (e.g., grazers or suspension feeders). The results showed an increase in the % of all trophic groups, concomitant with an increase in the system's biodiversity (captured through diversity measures). The macrobenthic P/B ratio decreased from 2005 to 2007, however, from 63.74 to 29.89 y^{-1} in the Mondego Estuary as a whole and showed a similar pattern in most of the estuarine stretches (Table 1). A sudden productivity drop was observed in 2007 for most of the estuarine areas, except the polyhaline muddy-sand area, which maintained relatively stable productivities. The species abundance increased from 92 to 159 individuals m^{-2} from 1998 to 2006. The benthic biomass was similar in 2005 and 2006 and clearly higher than that for 1998.

A decrease in estuarine stability was observed over the years (Table 1), except for the polyhaline muddy-sand area, where an increase occurred. This study also revealed a positive relationship between diversity and stability (Figure 3).

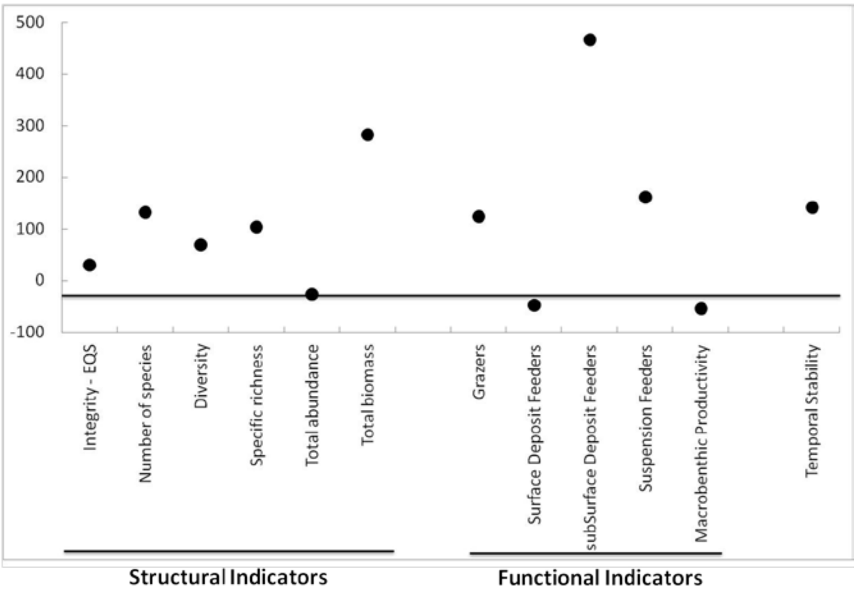

Figure 3. Relative losses or gains (% of changes) in structural, functional and stability indicators for the Mondego Estuary, between 1998 and 2006.

3.3. Linking changes in services to human well-being

The Pearson correlation coefficients calculated between the pairs of abiotic, structural, and functional indicators are given for the three periods in Table 2. These relationships between factors varied strongly by year. In 1998, the sediment characteristics were correlated with community functioning (Table 2). Mainly depending on the abiotic conditions, different functional groups (such as SDF or G species) dominate the benthic fauna assemblages. In general, the correlation between functional and structural indicators, as well as between different structural indicators, was positive. In contrast, different functional indicators were usually negatively correlated with one another (e.g., SF species with G or SDF), with the exception of the productivity values among years, which showed an identical pattern as between structural and functional indicators.

Table 2. Spearman significant correlation analysis (at *p*<0.05) among the several (abiotic, structural, and functional) indicators considered in the study, for the three considered years (1998, 2005, and 2006). In the table OM-organic matter, POM-Particulate Organic Matter, SST- Total Suspended Solids, d-species richness (Margalef Index), N-species abundances, S-number of species, H'-Shannon-Wiener diversity index, EQS-Ecological Quality Status, SDF-Surface Deposit Feeders, sSDF-subSurface Deposit Feeders, SF-Suspension Feeders, G-Grazers, Prod.-Productivity.

Abiotic indicators	Functional Indicators	r	year	Structural Indicators	Functional Indicators	r	year
%O_2	%G	0.9	1998	biomass	Prod. 2005	0.94	2005
	%sSDF	-0.96	1998		Prod. 2006	0.94	2005
%OM	%G	-0.94	1998	d	%SF	0.93	1998
	%SDF	0.89	1998	N	Prod. 2005	0.94	2005
	%sSDF	0.9	1998		Prod. 2006	0.94	2005
% coarse_sand	%SDF	0.94	1998		Prod. 2006	0.94	2006
		-0.94	2005		Prod. 2007	1.00	2006
% fine_sand	%SDF	0.94	2005	S	%SF	0.91	1998
		0.94	2006		Prod. 2005	0.94	2005
	%SF	-0.94	2005		Prod. 2006	0.94	2005
NH_3	%SDF	0.94	2006	EQS	%G	0.89	2005
POM	%SDF	0.94	2006	H'		1.00	2005
SST	%SDF	0.94	2006		%SF	0.93	1998
O_2	%G	0.94	1998			-0.94	2005
	%sSDF	-0.99	1998	Functional Indicators	Functional Indicators	r	year
Temperature	%SDF	1.00	1998	%G	%SF	-0.94	2005
Structural Indicators	Structural Indicators	r	year			-0.89	2006
biomass	N	0.94	1998		%sSDF	-0.99	1998
		0.89	2005			1.00	2006
	S	1.00	2005	%SDF	%SF	-0.89	2005
d	S	0.93	1998	%SF	%sSDF	-0.89	2006
		0.93	2006	Prod. 2005	Prod 2006	1.00	2005
	H'	1.00	1998	Prod 2006	Prod 2007	0.94	2006
		0.94	2006				
N	S	0.89	2005				
S	H'	0.93	1998				
EQS		0.89	2005				

Four main services (water quality, eco-tourism, nursery grounds and food production) were found to directly influence or depend upon biodiversity assets (Table 3). A gradual increase in the water quality and eco-tourism services was identified over the years, whereas nursery grounds and food production services tended to decrease (Dolbeth et al., 2008; Pinto et al., 2010). In an opposing trend, two main constraints were identified that could influence the productivity of the estuary: species competition (mostly macrophytes *vs.* macroalgae) and habitat loss (mainly *Z. noltii* meadows area reduction), which both demonstrated decreasing trends (Patrício et al., 2009) (Table 3).

The available data show that the bivalve population is an important part of the estuarine community in terms of biomass, abundance, energy and material fluxes, in some years, accounting for as much as half of the total abundance or biomass ('ecosystem stocks' available estuarine natural capital) (Figure 4A). The importance of bivalves was particularly clear in 2007, when there was a significant drop in bivalve densities (due to species recruitment; Veríssimo et al., 2012).

Table 3. Biodiversity related services provided and disservices obtained: service category, according to Millennium Ecosystem Assessment classification (MEA, 2005); description; indicators used; and trends over the years.

	Service category	Description	Indicators	Trend	References
Biodiversity related services					
Water quality	Regulating	Filtering, retention and storage of water proper for uses	Dissolved nutrients concentration	Increasing	Pinto et al., 2011
Eco-tourism	Cultural	Estuarine resources/spaces with (potential) recreational uses	Number of Visitors	Increasing	Sinergiae, 2008* PDM, 2005
Nursery grounds	Supporting	Suitable reproduction/ development habitat	Fish productivity (y^{-1}) (2004-2006)	Decreasing	Dolbeth et al., 2008
Food production	Provisioning	Transformation of solar energy into biomass for consumption	Salt production Aquaculture production Fish captures	Decreasing	Pinto et al., 2010
Biodiversity related constrains					
Species competition	Supporting	Ecosystem changes affect the prevalence of one specie over the other	Macroalgae biomass	Decreasing	Patrício et al., 2009
Habitat loss	Supporting	Suitable living space for wild plants and animals	Area occupied by *Z. noltii*	Decreasing	Patrício et al., 2009

After accounting for the number of boats in the estuary dedicated to the capture of different species from bivalves to fish, there appeared to be a general decrease in this proxy for these resources catches on the estuary over time. Examination of the bivalve catch (data source: www.dgpa.min-agricultura.pt), however, revealed a trend toward increased productivity from 1995 to 2006 (Figure 4B), and the number of bivalves brought to the Figueira da Foz harbour suggests an increase in catch from 2005 to 2006 (from 168.5 to 261.3 tons). Moreover, approximately 6 tons of bivalves that were manually removed from the system in 2005, which would require approximately 120 persons partially dedicated to the harvest.

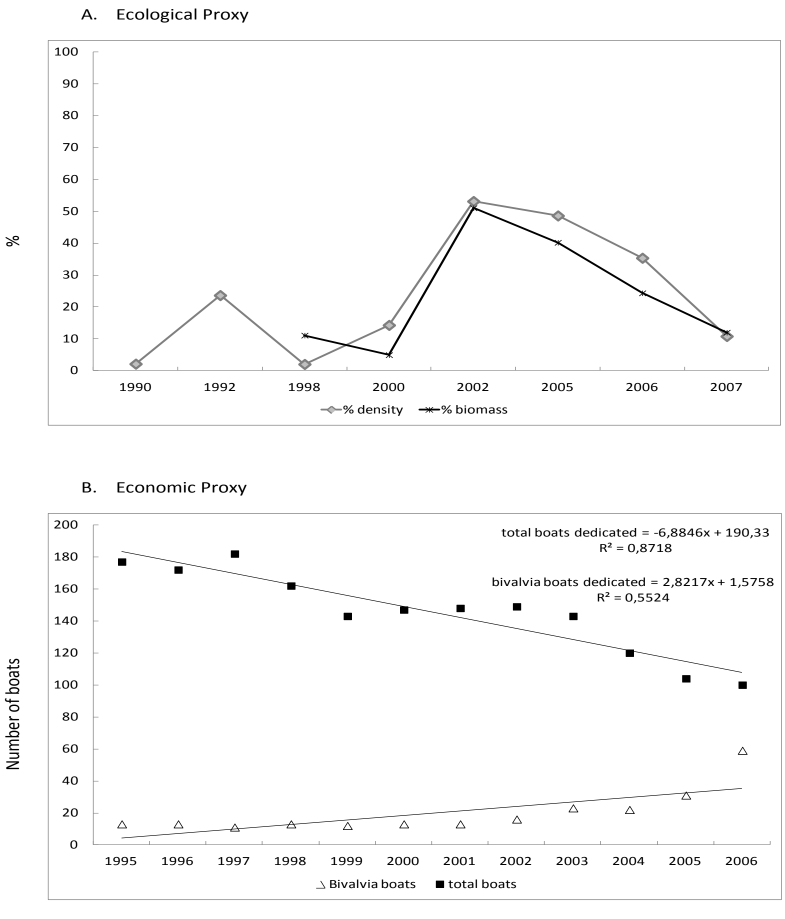

Figure 4. Bivalves' importance on the Mondego estuary system: A. Ecological services - Biomass and abundance of bivalves' communities, from 1990 to 2007; B. Bivalves' catchments, from 1995 to 2006 (data source: www.dgpa.min-agricultura.pt).

4. Discussion

Ecosystem services (ES) can be regarded as the (conceptual) link between the natural resources and social benefits that foster human well-being (Díaz et al., 2011). There have been efforts to relate biodiversity components with services provision, but only a few studies have been able to establish indicators (Díaz et al., 2007; NEA, 2010; Mace et al., 2012; van Oudenhoven et al., 2012). Turner et al. (2003) report that there is a strong interdependence between biodiversity and services provision, and there is additional current evidence that net ES value diminishes with biodiversity and ecosystem loss (Balmford et al., 2002; Jones-Walters and Mulder, 2009). Therefore, if biodiversity is positively correlated with ecosystem services delivery, then greater biodiversity will increase the value of ecosystem services (Morling et al., 2010).

The conceptual framework adopted by MEA (2005) was further developed in the present work to address (with inherent natural and anthropogenic disturbances) the effects that biodiversity may have on the provision of the ES in environmentally dynamic systems of high socio-economic value, such as estuaries. Through the application of this framework, a quantitative and qualitative assessment of the relationships between ecological quality and ES provision was undertaken, aiming to achieve a better guide to the potential (future) management of estuarine ecosystems.

4.1. Linking estuarine biodiversity to ecosystem functioning

Even small losses of estuarine species may hinder the ability of ecosystems to adjust to change or maintain their functional level. Recent assessments have shown that global biodiversity loss preferentially affects species with longer lifespans, bigger bodies, poorer dispersal capacities, more specialised resource uses, lower reproductive rates, and other traits that make them more susceptible to human pressures (e.g., by land occupation or nutrient release) (McKinney and Lockwood, 1999; Baillie et al., 2004; Mace et al., 2005; Díaz et al., 2006). Application of these global findings to this estuarine ecosystem demonstrated that stronger system perturbations lead to changes in all the structural proxies considered, with greater consequences for species diversity (decrease in number of species) and biomass. Moreover, previous work on the intertidal habitats of this estuarine system has shown that macrobenthic populations react to disturbances by ‘adjusting’ their biomasses. Using food web models, Baeta et al. (2011) have demonstrated that there was an increase in biomass, consumption, respiration and flow to detritus in *Scrobicularia plana* and *Hediste diversicolor* (two key species in the system) after the implementation of mitigation measures. In contrast, *S. plana* showed evidence of decreasing biomass and flows after the extreme winter flood of 2001 (Baeta et al., 2011).

The implicit relations between estuarine biodiversity and stability proxies may be an useful indicators of the general functioning of these ecosystems, a conclusion that is consistent with theoretical and empirical evidence (e.g., Remane, 1934; Duffy, 2006). In fact, the ecosystems functional approach must be complemented with the analysis of its inherent properties. In the specific case of the Mondego Estuary, it became clear that knowledge about the internal functioning of the system does not always decrease uncertainty about the system responses to ecological conditions. It was possible to demonstrate that functional indicators (e.g., species feeding guilds) respond directly (either positively or negatively) to changes in abiotic conditions and to structural and even to other functional metrics (e.g., variation in species feeding guilds ratios); however, structural proxies and overall system productivity were strongly related to one another but showed no significant relations with abiotic and functional metrics. This finding may suggest that while community structure directly determines part of the system productivity, the internal system functioning (e.g., % of feeding guilds) is more vulnerable to changes in the surrounding environment (either abiotic or biotic variations). Several authors (see Jorgensen, 2002) claim that it is the connectance of the system that plays a key role in the complexity-stability relationship, rather than the number of components within it (Jorgensen, 2002). In this sense, systems under-connected or over-connected would be more unstable (MacArthur, 1971; Levin, 1974; O'Neill et al., 1986), emphasizing that stability maximization is obtained at intermediate connectivity levels. In this context, the framework adopted in this study should be complemented with a detailed analysis of the food web energy flows within the communities, using, for instance, ecological network analysis (ENA) (Ulanowicz, 1980, 1986; Fath, 2004). The ENA approach is considered a promising method to analyse the functioning and judge the quality of ecosystems (de Jonge et al., in press). Therefore, it is crucial to adopt an integrated perspective of system functioning, which can be achieved by evaluating and comparing the information and responses obtained from several proxies when facing different types of disturbances.

4.2. Linking ecosystem functioning with stability and services provision

Estuarine stability and productivity tend to increase with an increasing number of species and larger biomasses, although only to a certain point (Pinto et al., submitted). Previous works (Trenbath, 1999; Elmqvist et al., 2003; Altieri, 2004; Naeem et al., 2009) suggests that a large number of species per functional groups, including rare species, may act as 'insurance' that buffers ecosystem processes and their derived services when changes occur in the surrounding physical and biological environment (e.g., temperature changes). Nevertheless, these ideas need to be experimentally tested (Carpenter et al., 2009). In the present work, the stability metric analysis allowed testing of the spatial and temporal responses of an estuarine system in relation to this 'insurance' role. The results may indicate that estuarine stability could be reduced by environmental perturbation (e.g., eutrophication symptoms in 1998 or bivalve recruitment in 2007), which also affected total estuarine macrobenthic productivity. These findings are consistent with previous results for this system, which indicated concomitant trends of decreasing connectivity and increasing temporal stability for intertidal habitats (Pinto et al., submitted). The inner parts of the estuary (e.g., South arm) were more stable and productive than were areas with higher water circulation or more exposure to adverse conditions, such as dredging activities (e.g., estuarine mouth-EE and PS). Therefore, this study is consistent with previous reports that argue the existence of a strong relationship between species composition and ecosystems function, reinforcing the important role of biodiversity in ecosystem productivity and stability (e.g., Loreau et al., 2002; Díaz et al., 2006).

Species composition was a key factor strongly influencing estuarine functioning, which confirms previous research (e.g., Grime, 1997; Duffy, 2006). In fact, a negative relationship was found between stability and the occurrence of species feeding on surface deposits, which are usually considered to be ecologically important species because of their effect on sediment structure (bioturbation behaviour) (e.g., Van Colen et al., 2010). Although the concept of functional groups may be a valuable tool for simplifying community complexity, especially for management purposes (Gaston et al., 1998), the information acquired may be subject to criticism (e.g., difficulty in assigning species to feeding guilds or generalising trophic groups; Giangrande et al., 2000). To address this issue, information about the functional groups was coupled with macrobenthic productivity and stability measures, which, in general, showed a gradual increase. The productivity drop observed in 2007 was mainly caused by a reduction in the biomass of only one species (*C. edule*) (values in spring decreased from 11.94 g AFDW m^{-2} in 2005 to 0.020 g AFDW m^{-2} in 2007). However, the productivity method used was specifically designed for intertidal benthic communities (Brey et al., 2001), meaning that even assuming the validity of the subtidal parameter and the depth factor used to construct the model, some residual variance might be associated with these measurements.

4.3. Linking changes in services to human well-being

Estuarine biodiversity (including number of species, diversity, and total biomass) had a determinant role in the provision of ecosystem services and, consequently, on human well-being. Regarding the structural, functional and service relationships, the empirical evidence indicates that changes in benthic biodiversity (expressed by diversity measures) clearly affected ecosystem functioning (as demonstrated in Table 1), in terms of both the intensity and the direction of the responses (e.g., achievement of better ecological status). In fact, the loss of ecological complexity/linkages (Mace et al., 2012) is driving the world's ecosystems to a more frequent failure of their functions and services, even to an increased risk of unexpected and irreversible changes in their status (MEA, 2005).

The linkages between biodiversity, ecosystem functioning and ES provision for human well-being are neither straightforward nor universal. Although the connections between ecosystem properties and ES are not always linear casual paths (Carpenter et al., 2009; Pinto et al., 2011), many changes in ES provision can be quantified by using variation in ecosystem properties recorded by routine measurements (Díaz et al., 2007; de Jonge et al., in press). Therefore, the initial conceptual framework, which was the basis for this work (MEA, 2005), should be modified by the addition of reverse arrows to include the reciprocal effects that human well-being and ES provision have on estuarine biodiversity and ecosystem functioning:

biodiversity $\rightleftarrows$ ecosystem functioning $\rightleftarrows$ ecosystem services $\rightleftarrows$ human well-being

This study was undertaken in a medium-sized estuary having a strong local focus, so the extrapolation of these results for application to other estuarine systems is limited. Nevertheless, several noteworthy trends were observed:

1) Increasing species richness and functional composition had positive effects on estuarine productivity;
2) Changes in estuarine ES provision are related to altered structural and abiotic factors and to the loss or decline of locally abundant species; and
3) The causes of change in underlying biodiversity and ecosystem functioning determines the system's ability to provide the associated ES.

The key issue when addressing complex ecosystem management is to accurately translate ecosystem changes into measures that address both ES provision and ecosystem conservation. Adequate societal responses to improvements in ecological conditions require quantitative assessment of the biodiversity composition and ES consequences of changing environmental conditions (MEA, 2005; Hooper et al., 2005; Díaz et al., 2011; Mace et al., 2012), for example, through surveys (Contingent Valuation or Choice Experiments Methods) of affected populations.

5. Conclusions

The findings of this study highlight a series of qualitative conclusions about the relation between biodiversity assets and human well-being in estuarine systems: 1) the species composition of estuarine communities was a key factor that strongly affected system functioning; 2) the complexity of the relationships between biodiversity and human well-being, via the effects and causes determining the ES provision; and 3) changes in the estuarine ES provision likely derive from changing structural and abiotic factors and from the loss or decline of locally abundant species.

The results suggested two additional conclusions: 1) more integrative tools/studies are required to explain the interactions of estuarine ecosystems with surrounding socio-economic activities and 2) these tools need to be incorporated into decision-making processes aimed at the conservative management of ecosystems and used by institutions, particularly to guide the implementation of sustainable frameworks (integrating biodiversity assets and ecosystem services used for human well-being).

New research needs to accomplish several objectives: 1) further explain the links between biodiversity and ES provision in estuarine systems; 2) clarify the functional role (achieved through connections among species) of macrobenthic species in the provision of valued ES (e.g., C sequestration); and 3) elucidate the using of ecosystem production as a tool to guarantee that the consumption of natural resources is sustainable.

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Integrating ecological, economic and social aspects to generate useful management information under the EU Directives 'Ecosystem Approach'

CHAPTER SIX

Abstract

If we as scientists can not decide upon what research, monitoring and technical tools should be used as a basis for policy making and management within the European context, then the politicians and other decision makers will continue to follow the line of ‘weak’ sustainability (applying monetary substitution rules to natural capital) instead of ‘strong’ sustainability (applying alternative rules such as the precautionary principle). Suitable integral indicators or indices matching the ‘ecosystem approach’ (EA) and thus covering ecological as well as socio-economic aspects are required. There is, however, a clear friction between what can be delivered in terms of useful ‘(integral) indicators’ and what decision makers require us to deliver in terms of ‘simple, cheap, easy to understand’ while the real situation is extremely complex. This social, economic and ecological complexity has been an important impediment to the realization of an EA that should guarantee ‘sustainability’. What is missing since the publication of the Brundtland report is technical co-operation between the decision makers and the natural and social scientists. To achieve development of integral indicators we propose to make the Odum food web concepts functional by the application of ecological network analysis (ENA) and at a scale where socio-economic and ecological information can be integrated, which is the ‘habitat’ level. At the habitat level ecological functioning (natural compartment), human activities (economic compartment) and ecosystem functions to humans (socio-ecological compartment) can be designated and measured. This process can further be facilitated by the use of the Driver-Pressure-State change-Impact-Response (DPSIR) approach. To facilitate weighing and decision making multi-criteria techniques can be used.

Keywords:
| Strong and weak sustainability
| Integral system
| Habitat
| Ecosystem functioning
| Ecosystem processes
| Ecosystem services
| Network analysis
| DPSIR framework
| Valuation

1. Introduction

1.1. Sustainability, the ecosystem approach, ecosystem services and natural capital conservation

In 1987 the Brundtland report was published with a definition on sustainable development (SD) “*sustainable development meets the needs of the present without comprising the ability of future generations to meet their own needs*”. It is one of the most widely used definitions but it stresses high level socio-economic goals rather than a working blueprint for sustainable science, policy and practice within an integrated system. Ten years later Brundtland (1997) stated: “*in ocean management, as in most other areas of human endeavour, close co-operation between scientists and politicians is the only way to move forward. Science must underpin our policies. If we compromise on scientific facts and evidence, repairing nature will be enormously costly, if possible at all.*” This statement is the basic starting point of the present contribution in which we will try to set out practical directions on ‘how’ to approach sustainability. In the field of governance the SD definition was refined in economic terms into ‘weak’ and ‘strong’ variants (Turner, 1993), but SD was also considered by some analysts to be still fuzzy and ambiguous as an integrative real world strategy (e.g. Custance and Hillier, 1998). This served to inhibit its practical and immediate implementation. To overcome this drawback a set of guidelines was developed (Bellagio principles) to reinforce the holistic perspective within SD (Hardi and Zdan, 1997). The guidelines focused on whole system accounting which encompassed the well-being of social, ecological and economic components.

From the governance perspective, environmental legislation has become more extensive and the expansion of the European Union, for example, has led to a situation where all member states implement uniform EU Directives (subject to subsidiarity clauses) with the common goal of environmental protection. Potentially this has led to an increase in the harmonisation of the environmental legislation among the EU member states. An important and valid question is, however, whether the implementation at the different national levels in practice, also satisfactorily fulfils the starting points and goals of the relevant EU Directives. Attempts to achieve a balanced and sustainable utilisation of natural resources (either at a global, regional or local scale) cannot be adequately considered as a sectoral, social, or economic problem in isolation. Thus, any action (e.g. future technological switching and other adaptation measures) requires a better understanding of the complex interactions between all parts of the 'integral system' (Figure 1). Sustainable development requires at its core a fuller appreciation of the long-term impact of the increasing scale and rate of human activity on the environment (Hardi and Zdan, 1997).

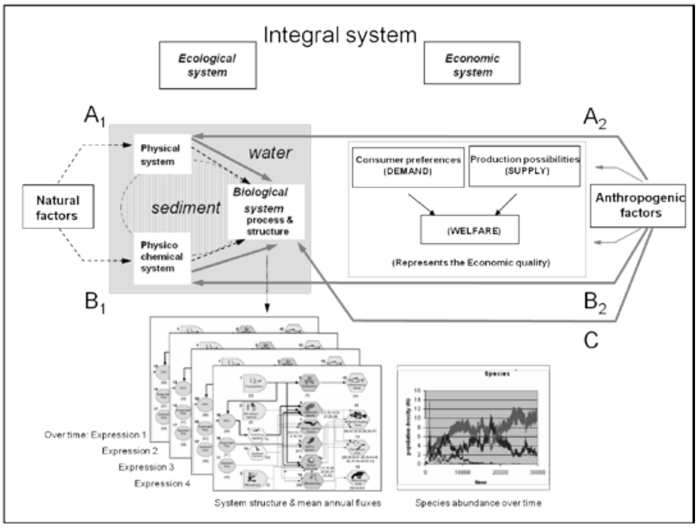

Figure 1. Diagram representing the general structure of the 'integral system' without explicit incorporation of the human/social component (modified according to de Jonge et al., 2003). Influences of natural factors are indicated by dashed lines and those by anthropogenic activities by solid lines. A1 and A2 refer to the natural or anthropogenic influence of the physical system, B1 and B2 to that of the physico-chemical system, and C refers to direct human effects on the biological system. The lower panels represent over time changing system 'expressions' and visualised variations in abundance at the species level. The panels with the food web structure are from Baird et al. (2004) and reproduced with permission from the publisher.

Since the Brundtland report the need for a more comprehensive monitoring of societal and environmental development impacts is widely acknowledged as being an important source of information to the authorities involved in economic growth and wealth creation promotion, and those involved in the management, conservation and protection of the environment. Apart from this formal role, monitoring is, however, also important in feeding data to the scientific community and in informing society in general and influencing social norms. Despite the fact that the original Brundtland report was published 25 years ago, monitoring of the ecological and socio-economic impacts of environmental change has been largely confined to defined sectors, because of lack of a generally applicable interdisciplinary conceptual and analytical framework. Consequently, current monitoring programs are still carried out in isolation from each other while we know and acknowledge that we are dealing with 'social-economic-ecological' or 'integral' systems and connected linkages (Berkes et al., 2003) (Figure 1). Societal developments such as changes in land use (urbanization, industrial developments and agricultural practices) climate change and the environmental and social feedbacks to these changes are connected in a complex way. This complexity has been an important impediment to the required technical co-operation between the political process and natural and social scientists that Brundtland called for. More recently there has been a growing awareness in the decision making process of the findings from behavioural sciences which highlight among other things the importance of networks and often complex linkages in human behaviour and change over time. Legal instruments, for example, are now being buttressed via so-called 'nudge' policy measures with more subtle motivations (Layard, 2010; Ormerod, 2010). But complexity is equally a characteristic of natural systems of which humans are a part (Berkes et al., 2003). The need for more comprehensive environmental accounting frameworks has never been greater. Nevertheless, our decision makers continue to call for environmental indicators which are 'easy to understand' and 'cheap and simple to measure' despite the fact that ecologists have yet to agree on a common integrated concept of the most relevant basic processes responsible for the natural panarchy in expressions of one and the same geographical system as visualised in Figure 1 (see also de Jonge, 2007; de Jonge et al., 2006).

The existence of multiple biological system expressions is due to the many natural variations in the physical and physico-chemical boundary conditions represented by e.g. varying conditions in wind, irradiance, temperature, salinity and nutrients. Strong winters may for instance lead to mass mortality of intertidal (benthic) fauna (Beukema, 1985) which changes the 'top-down control' by a strongly reduced grazing pressure on micro-algae in the water column and on the sediment. However, dull weather conditions during summer may negatively influence the 'bottom-up control' by decreased primary production. According to the resource competition theory (RCT) varying resource conditions (nutrient concentrations and light conditions) affect the abundance among species (e.g. Tilman, 1982; Grover, 1997; Huisman and Weissing, 1999, 2001); while according to the intermediate disturbance hypothesis (IDH) (Horn, 1975; Connell, 1978) the development of the species structure (e.g. under pulse-wise addition of nutrients) may be directed to a structure deviating from the common one. Moreover, invading species may occupy either a new niche or may simply replace part of the native species at different trophic levels. There is little evidence that invasive species replace native species entirely (Reise et al., 2006). On top of this, human activities affect the system for instance by fishing (damaging biological population structures and habitat), dredging (increased turbidity and habitat destruction) and discharges of waste water (loading of the system with pollutants and nutrients). There is thus not only a strong inter-annual variation in the abundance of individual species but there may also be a significant inter-annual variation in the structure of the system expressed by species composition and abundance (Figure 1).

It is not possible to decide objectively which expression in nature is wrong, acceptable or good because objective criteria are not available for our dynamic coastal systems (see also Elliott and Quintino, 2007). The above suggests that from an ecological point of view monitoring should preferably be focussed on integrated indicators which tell us something about the functioning of the entire food web, i.e. the combination of structure and functioning, instead of solely indicators reflecting parts of the system's condition (see Naeem et al., 2009). Such strategic decisions are necessary for an effective and efficient conservation of our living environment. When we, as scientists, are not competent or not able to decide in a coherent and reasonably unified way what to champion to aid decision making in terms of research and monitoring and effective policy measures, then the politicians will often be 'persuaded' to take action (or no action) on the basis of short term or overly 'local' considerations. The long term sustainability of both the natural systems and the wealth creation potential of the ecosystem services (ES) they ensure may not get the required recognition with negative consequences for natural systems and human wellbeing (Boyd and Banzhaf, 2007; Fisher et al., 2009). Kremen et al. (1994) state that conserving nature is only possible when it is combined with attention for the wellbeing of the local, national or international population, something we fully agree with because it underpins the importance of connecting the socio-economic, cultural, political and the ecological aspects in an integrated approach so that sustainability becomes reality.

Interestingly, during recent years, the European Commission has published a number of Directives aiming at an integrative approach when assessing the quality of the natural environment. Important directives here are the Water Framework Directive or WFD (EC, 2000) and the Marine Strategy Framework Directive (MSFD) (EC, 2005a,b; EC, 2008). In one or another way these directives follow the integrative principles of 'systems ecology' and those of the 'ecosystem approach' (EA) (de Jonge, 2007). This consequently means focusing on the ecosystem part of the integral system (Figure 1; see also Likens, 1992). Environmental conditions should be assessed on the basis of the structure and functioning of the biological part of the ecosystem in response to the sum of the natural variation (caused by natural stress factors) and the human induced stresses. This relatively narrow focus (only on the quality of the biological expression of the ecosystem) has emerged because of the recent rapid deterioration of some of our environments and because decision makers have given broad attention mostly to the socio-economic part of the 'integral system' or 'social-ecological system'. But even this attention needs to be further enhanced if a strong sustainability position is accepted.

Strong sustainability requires that among other things, ecosystems are seen as suppliers of a range of intermediate and final services (ES approach) through which humans benefit in terms of welfare. Sustainable utilisation of this vital resource base is therefore the key notion. It can be argued that the assigning of monetary values to the benefits provided by 'healthy' ecosystems can supplement scientific and ethical arguments in favour of environmental protection and biodiversity conservation (Turner et al., 2010).

The WFD, within the present context discussed by de Jonge et al. (2006), distinguishes two simple complementary ways of reaching its goals:

- the optimization of the physical habitat-providing conditions; and
- the (further) improvement of water quality.

The result of this should then be assessed by the quality of the structure and the functioning of the biological part of the system. Assessing the biological quality of estuarine and coastal waters is any case a difficult task because of the variability of these systems (e.g. de Jonge et al., 2006; Elliott and Quintino, 2007). The available benthic macrofauna related biological indicators turn out to be non-comparable with each other, which may indicate that they are unsuitable to assess the quality of the biological structure under consideration because they all cover a different aspect of the ecosystem part under the given conditions (Patrício et al., 2009). Finally and within the given political context any assessment of the biological quality is also not a simple task because (see also above) politicians continue to call for 'simple, easy and cheap to measure' indicators. There is thus a clear friction between what can be delivered at the moment and what is called for by the decision makers. The results of any assessment should also be meaningfully connected to any (natural or human induced) stressor or set of stressors to provide effective indicators and an effective human response to the new situation.

To sum up so far, and using the EU MSFD as an example, policy needs to be 'informed' by the EA and the 'good environmental status' needs to be interpreted in terms of ecosystem structure and functioning plus services provision. Despite the attempts of Borja et al. (2010) to make the implementations of WFD and MSFD as holistic as possible, we are still quite distant from it because all approaches so far are not following the requirements from 'systems theory' but that from the 'EU Directives' as implemented. Implementation of the EA should be via so-called adaptive management policy and practice. This is essentially 'learning by doing' with policy and practice being constantly monitored and re-orientated/changed as experience is gained during implementation. Such an approach accepts the inherent complexities and uncertainties that often shroud the utilisation of marine resources (Turner, 2000).

Problems of resource overexploitation and/or environmental quality degradation tend to have multiple causes and are evolutionary. Complexity and the power of networks (natural and human behavioural) serve to make management and decision making tasks very onerous with potentially very costly consequences when the wrong measures are introduced e.g. the recent worldwide financial crisis (Krugman, 2009).

We are of the opinion that decision makers should stop asking for 'easy, cheap and simple to understand' environmental indicators and accept the complex reality that is our environment. Given the panarchic character of natural systems realistic base environmental indicators should be anchored to a thorough examination of the functioning and the structure of ecosystems (de Jonge, 2007; de Jonge et al., 2003, 2006) instead of selecting for the 'cute and cuddling' icons of any ecosystem without knowing what they ecologically represent.

An adaptive management process should be composed of a number of sequential but overlapping components (see also Hanssen et al., 2009):

- baseline science and indicators to inform in terms of the ecosystem structure, process and forcing vectors that condition the coevolving ecological and socio-economic marine system and its inherent trends;

- the application of methods and techniques (the tool box) for the assessment of the marine system's status and future prospects;

- focused analysis of contemporary 'key' and potentially significant emerging issues due to overarching environmental change;

- participatory and deliberative methods and techniques to foster social dialogue amongst all relevant interest groups and to search for 'values' consensus/majority positions;

- modelling to compare alternative policy option outcomes;

- further development of appropriate indicators and adequate monitoring and review procedures.

1.2. Present EC Directive-related failures in marine management

The implementation of the WFD (EC, 2000) is based on monitoring selected parts of the ecosystem. The present focus is on phytoplankton, macro-algae, angiosperms, benthic invertebrate fauna and fish while the most important carbon fluxes in ecosystems are at the level of microbes, detritus and primary producers. Baird et al. (2004) conclude that about 99% of the recycling involves only some compartments with mainly sediment bacteria and particulate organic carbon (POC) as detritus. Something comparable is also observed for a part of the Schelde estuarine system (van Oevelen et al., 2006). The latter concluded that the herbivorous and detrital-microbial pathways function highly autonomously. How important the detritus related pathways are as stabilizers of the ecosystem functioning, however, needs further research. From a holistic ecosystem perspective as well as the EU ‘ecosystem approach’ the present implementation of e.g. the WFD seems still highly sectoral in its approach. The accepted low minimal sampling frequency further supports the conclusion that the collected data are not particularly helpful for analysis related to coastal policy making and management (de Jonge et al., 2006).

The required WFD related river basin management plans are focussing on rather general elements and not on anomalies in the system functioning, structure or condition in relation to created human pressures.

The other relevant strategy, the Marine Strategy (EC, 2002) leading to the adoption of the MSFD in 2008 (EC, 2008) aims at integrating the practices from other relevant EC Directives in an EA which is then considered to be the EU Strategy for Sustainable Development. This strategy is in its implementation phase now. Although this strategy is much more ambitious than the WFD it is not providing a clear technical strategy with supporting instruments or tools, but is based on fourteen objectives (EC, 2002). Some relevant objectives are to: protect nature, stop habitat destruction, change fisheries management, improve water quality at all levels and from all sources, eliminate litter, (more recently also to reduce noise), reach a more effective co-ordination and cooperation, pursue the new strategy at the global level and finally (objective 14) to improve the knowledge base on which marine protection policy is based. To us this forms the challenging basis and justification for providing some direction to the creation of technical tools which can be integrated to more effectively monitor the developments of the formulated EU aims.

1.3. Aim of paper

The paper’s aim is to present an overview of an overarching framework and component instruments or tools that can be combined or integrated to arrive at a set of suitable indicators to judge the systems condition or status in terms of health, resilience, carrying capacity and related aspects. We will conclude by giving direction to 'how' to move forward in the spirit of the Strategy Directives.

2. THE INTEGRAL SYSTEM

2.1. Ecosystem research and the EU ecosystem approach

There is in ecology a long historical acceptance of the importance of studying systems entirely or holistically and not partially. System theoretical philosophies (e.g. von Bertalanffy, 1968) and the ecological concepts of Odum (1971) have set the scene for a general view that studying the total energy flows through any ecosystem in sufficient detail could create a rational basis for understanding the complex functioning of food webs, if not yet the role of its complex species structure. Published ideas on the need for detailed integrative research of preferably unfragmented systems date back to the 1970s (Holling, 1978; Ulanowicz, 1980; Ehrlich and Mooney, 1983; Goodland, 1987; Baretta and Ruardij, 1988; Kremen et al., 1994). Despite its importance every scientist realizes that this sort of ecosystem research is expensive because it can only be executed by relatively large teams of specialists. This in itself has been demonstrated to be enough reason for not getting it supported by governments or by their regulatory agencies. A very successful European example was a Dutch research team (Biological Research Ems-Dollard Estuary) which, based on the ecological ideas of Odum, investigated the main energy fluxes of the ecosystem in the Ems Estuary over the period 1972 - 1985. The relations were quantified from bacteria and detritus up to the fish, while covering anoxic as well as oxygenated conditions in water and sediment and including water chemistry and physics (Anonymous, 1985). The final result was one of the first successful mathematical computer simulation models (Baretta and Ruardij, 1988). This success was only possible because of a clear vision in combination with one clear goal (integrating the collected data in one system i.e. a computer simulation model) and leadership (scientifically keeping course and continuously convincing impatient politicians of the value of this type of research). Despite its success the Dutch government was not willing to further fund this strategic 'know how' research (creation of a knowledge agenda) and also not to further develop the modelling of Dutch coastal ecosystems on that basis. Consequently, the integrative BOEDE research (Biological Research Ems-Dollard Estuary) stopped in 1985 after which the scientists decided to report to the scientific community (Baretta and Ruardij, 1988). Although not scientifically rational, this political attitude is 'understandable' given the sectoral and local pressures that can arise as the rate and extent of environmental change increases. Long run strategic decision making is often much harder to take than following the line of the more often locally popular 'soft decision making' e.g. protecting fishing communities etc.

The progress in ecosystem research made since the late 1980s in, for instance, The Netherlands is fragmented because the field research carried out since then was usually not part of a master plan. Nor was the plan linked to an integrated ecosystem study concept founded on the EA. Rather it was either part of progress in fundamental research (development of concepts), or part of ‘problem oriented’ research (sectoral problem solving). Problem oriented research implies that the scientific task is narrowed to a system engineers approach, solving a specific problem like the effects of shell fishery on birds, the effect of gas drilling on bottom subsidence, the exploitation of the large scale offshore harbour development near Rotterdam effect on the large scale transport of fish larvae and suspended mud in the North Sea. Contrary to the intentions of the MSFD (EC, 2008), this sort of research does not analyse the functioning of entire ecosystems to determine its ‘condition’, ‘carrying capacity’, ‘health’ or ‘resilience’, or to contribute to the development of any knowledge system or decision making instrument which is currently not available for the Dutch government and their agencies. The same is true for most other EU member states. More leadership, more vision and less politics at the national governmental level (holding for politicians as well as their senior advisors) could have resulted in a more beneficial development of the knowledge relevant to integrally manage our coasts and estuaries (de Jonge et al., 2006).

Crucial here is to define what to investigate what to monitor and how this then can be related to the relevant social and economic parts of the integral system to support sustainability in line with the Brundtland Commission’s view (strong sustainability).

2.2. Socio-economic research in relation to the EU ecosystem approach and the ecosystem services approach

The ecosystem approach (EA) and ES concept aim to provide an overview and analysis of the wider issues (‘to look out of the discipline box’) and to understand the functioning of the wider system encompassing the complex combination of the societal needs, economic market wants while underpinning the ecosystem structure and functioning. This ‘big picture’ contributes to a wider integrative and systematic perspective of ecosystems and then may help in setting more effective even incisive management actions upon specific local, regional national and international problems. Although efforts have been made to integrate all available SD components, applications of the EA have to date still remained strongly focused on ecosystem structure and thus biodiversity centred. In addition to biological structures we need an equally strong focus on ecological processes (at the proper scale) and to the environmental services benefits (‘goods and services’) in relation to human welfare. To understand better the dynamics of system change we also need to incorporate analysis of human pressures and drivers (sometimes in the form of future scenarios) as well as the natural stressors as indicated in Figure 1.

While the EA thus provides a useful conceptual framework it is still too theoretical to be directly applied to managing the integral system. To overcome this drawback, efforts have been made to turn this EA concept into a more feasible and operational tool that allows us to study ‘how’ to judge the effects of humans on nature and how to direct future developments when considering the co-evolution of social human systems and natural systems (see also Borja et al., 2008). The ES concept helps to make this analytical transition.

2.3. Societal choice

The ES perspective requires us to review what socio-economic information is necessary to support EA/ES implementation. First socio-economic information (on environmental drivers, pressures and changes) from the local up to the regional and international scale that is relevant at the ecosystem level should be included.

Aquatic ecosystems, and more specifically estuaries, are not only considered to belong to the most productive but also to the most valuable ecosystems around the world (Costanza et al., 1997; Jørgensen, 2010). The increasing population densities and subsequent increasing socio-economic demands (exploitation and modification of these systems) lead to increased human stress on these systems. Despite all the stress, ecosystems still have the ability to provide a wide range of ecosystem services benefits, such as food production and recreation, while at the same time providing a wide panoply of regulatory and support ES, as nutrient cycling and water purification (Balmford et al., 2002, 2011; MEA, 2003; Turner et al., 2003; Bateman et al., 2011).

The socio-economic research needed to support an EA/ES implementation should be such that we get an output which is directly useable in the decision making process. We have several conceptual approaches at our disposal (e.g. Driver-Pressure-State change-Impact-Response or DPSIR approach), Ecological Sustainability Trigon (EST) and social-ecological systems (SES) for integrating qualitatively and/or quantitatively the interactions between the ecosystem and the socio-economic system (Berkes and Folke, 1998; Turner et al., 1998; Costanza et al., 2001; McLusky and Elliott, 2004; Brock et al., 2009; Marques et al., 2009).

This sort of analyses involves a clear definition of the main activities, stakeholders involved and general society characteristics (e.g. demographic data), monitoring of the stocks/flows from and to the system ('input-output analysis'), the degree of human dependence on it, and the main impact it has on both the ecosystem and the human population. Therefore, an insightful characterization of the different forms of capital of an ecosystem (natural, human, manufactured, and social; Costanza, 2000) has to be performed.

Apart from the requirements defined above, the EA/ES also offers opportunities for the socio-economic disciplines to bridge the gap so that EA can play the role in 'sustainability' as foreseen by the European Union Commission. The term 'sustainability' now occupies a prominent position in the political lexicon and political agendas from the local (e.g. regulations controlling pollution sources) to the international levels (e.g. directives controlling water quality by WFD and protecting biodiversity by the Convention on Biological Diversity), reflecting the growing wider societal level of concern (Costanza, 2000; de Jonge, 2007; Duit and Galaz, 2008; Marques et al., 2009).

2.4. Defining environmental limits

By combining and substituting between the different forms of capital (physical, human, social and natural) the wealth creation process has expanded enormously (albeit unequally on a global basis). A big issue that now faces contemporary society is how much further can natural capital be substituted for via technological advances before thresholds are breached and unexpected system change occurs, possibly signalling unsustainable levels of ecosystem utilisation. If we adopt a definition of sustainability that implies that the current human generation must pass on a stock of capital to the next generation that is no less than it is now, we can distinguish two views about the conditions necessary to realise sustainability- the weak and the strong sustainability positions. The former view maintains that sustainable development can be achieved by transferring an aggregate capital stock value to the next generation that is no less than the current level. It is based on an optimistic assumption of the power of technological innovation and the continued substitutability of natural for other forms of capital. The strong sustainability view does not accept the indefinite substitution possibilities axiom and focuses on the existence of 'critical' natural capital (e.g. life support systems, the hydrological cycle, etc) that cannot be substituted for, either literally or on cost grounds. In reality there are a number of 'middle ground' possibilities.

The acceptance of a stronger or weaker version of the sustainability worldview, however, does have implications for ecosystem management and the further development of environmental decision making. The use of economic cost-benefit analysis (ECBA) as a decision making support system implies a decision rule which selects options that maximise individual human welfare measured in monetary terms. So the monetary benefits for example of utilising ES in some way can be compared with the costs of that option. The closer we move towards the adoption of a strong sustainability position the lesser is the scope for CBA application, because the scope for natural capital substitution is assumed to be less. Instead we must substitute rules such as the precautionary principle which prioritise conservation of ecosystems. Most recently it has been argued that the natural capital stock and flow approach to environmental management should not serve to obscure the equally pressing need for radical reforms of institutions and governance (Norgaard et al., 2009). Much depends on how pressing the global sustainability constraints really are, or what our attitude to collective risk taking should be. But institutional and governance issues are clearly key parameters that need to be addressed in any serious sustainability dialogue and so far progress at the national and international level in this dimension has been limited.

Ideally ecosystems would be managed under sustainability rules, in practice there are a number of acknowledged reasons why ecosystem degradation continues in some contexts unabated. These reasons include both market failure and poor governance. Taking the former, markets fail to allocate ecosystem resources efficiently because of lack of information on ecosystem functioning, and ecosystem service (benefit) prices and non-market values (see e.g. de Jonge et al., 2006).

Only some ES are traded in markets and even then the market prices may not reflect the total economic value of that particular asset. Often the full environmental costs (externalities) of the economic activities involved in utilising ecosystems are not reflected in these prices (Barbier et al., 2009; Perrings et al., 2009). According to Daly and Farley (2004), most of the services provided by natural systems do not gather all the characteristics required for an efficient allocation in markets (excludability, where property rights are included, and rivalness). Therefore, effective policies that characterize a specific service should be applied to the specific combination of excludability and rivalness if optimal allocations are aimed at (Daly and Farley, 2004). It has therefore been argued that appropriately assigning market and non-market values to environmental assets is important for environmental management. A range of methods and techniques have been devised to assign monetary values to ES in the absence of market price data including survey based methods (e.g. by contingent valuation studies and choice experiments as reported by Barbier et al. (2009) and Mitchell and Carson (1989)). In the economic literature a number of issues have been identified which serve to complicate and limit the application of the economic valuation of ES. They include the spatially explicit nature of some ES provision; the requirement that ECBA must be based on so called 'marginal' changes in service provision and not total system collapse/loss; the avoidance of double counting of benefit values; and the complications caused by non-linearities in benefits provision and threshold change effects (for more detail see section 3.4 in this paper and Fisher et al., 2009; Bateman et al., 2011). Studies have also shown the limited contributions of it, given a non-exhaustive data set (e.g. Pinto et al., 2010). A further more accurate valuation of biodiversity assets is required but this implies an enormous task with an uncertain outcome. This fact has lead to a gradual shift from further developing biodiversity based indicators to suggestions of looking for new 'paradigms' (de Jonge et al., 2003) and for monitoring and assessing the pressures and resulting state changes within the ecosystem (Levrel et al., 2010) or even the integral system (de Jonge et al., 2006; de Jonge, 2007).

2.5. Conservation of ecosystem structure and functioning to maintain ecosystem services

There are various options possible to maintain the flow of ES. A very conservative and safe one is to conserve the balance between a specified level of biodiversity and the functioning of the system. This is nearly a '*contradictio in terminis*' because it supposes that we are able to define and to judge this balance which is currently not the case. We do not know how to guarantee and maintain a particular stock and related flows under naturally varying conditions as they occur in natural and open systems to guarantee a particular level of ES. This also means that we need to fully describe in sufficient detail the relation between the ecosystem structure (biodiversity) and functioning as a solid basis for human wellbeing estimation (e.g. Naeem et al., 2009).

2.6. Level of action at different relevant temporal and spatial scales

A further question is what temporal and spatial scales are relevant when talking about ES in relation to sustaining human life in general. Through the integration of the ecosystem's inherent processes, the associated biodiversity and its sustainable use, the ecosystem approach focuses on conserving natural systems for their inherent value and for human well-being (Vitousek et al., 1997; Nunes and van den Bergh, 2001; de Groot et al., 2010). In an overall perspective, socio-economic research has, when applied to the EA framework, the capacity to (Sutinen, 2007) analyze and explain the spatial and temporal variations in the uses of the principal ecosystem resources; assess the market and non-market value of human uses of natural services of ecosystems; assess the benefits and costs of protecting and/or restoring ecosystem resources; and assess the socio-cultural values of the uses of ecosystem resources and services. The use of comprehensive approaches (e.g. ES inventories in EA studies) to evaluate significant ecological, social and economic costs and benefits facilitates the work of decision makers regarding the implementation of management and conservation strategies (Pinto et al., 2010). Two scale-related problems are encountered when assessing ES (Heal and Kristrom, 2005): (i) the scale at which certain functions become important is not always the same and (ii) problems may arise when integrating and aggregating information at multiple scales where interrelations and feedback loops may operate at scales above the level being assessed. According to Limburg et al. (2002), scaling rules that try to describe the provision and delivery of ES have yet to be quantified and defined. Moreover, issues such as cumulative pressures and intricate interrelations among factors, internal and external to the system, are also determinant subjects to be considered when looking for optimal allocation and management of ecosystems.

2.7. Coupling the social and ecological part of the integral system

The presence and activity of humans has globally dramatically changed the environment. Given the complex behaviour of the human community and the ecological system (Levin, 1999) and thus also the connections and interactions between them as part of the ‘integral system’ (Figure 1) a relevant question here is how we can relate human activity to environmental response in a way beneficial to management. This question is far from trivial because conventional, mainly sectoral, approaches are of limited use (see above and e.g. Holling and Meffe, 1996). A well known example is that of setting quotas for the fishing industry. This has been shown to be not enough regulatory effect to manage fish stocks because of the complex responses and feed backs within the ecological food web. This limitation is mainly due to the fact that the species structure is not static but dynamic in terms of composition and abundance. Species respond to changing environmental conditions as well as changes within the species structure and abundance itself (see above). All systems thus show different structure expressions or representations and also different qualities in space and time (Figure 1). At another abstraction level these differences may be less pronounced as for instance is the case when considering the ‘functioning’ of the structure under consideration. At that level many systems are more comparable with each other (Baird and Ulanowicz, 1993; de Jonge et al., 1995; Herman et al., 2000; van Oevelen et al., 2006). Thus, despite the available panarchy in structures the functioning of it may be more or less the same.

There are several ideas on how the interaction between the social and ecological systems can be realized and how the systems condition or health could be judged. We are not going to contribute yet more ideas to what is available at this stage but will explore from what is available and applicable for the desired integration and judgement. Moreover, we will describe what is going on in terms of relevant system wide research and monitoring programs and how this all does or does not fit our ideas. Apart from that, we will further (see the figures in this paper) visualize how the socio-economic and ecological systems could technically be connected to each other. This point is of utmost importance since concepts more than problems should lead to how and when to monitor what part of a specific system.

The application of the ecosystem approach would allow the integration of ecological sustainability, economic efficiency and social fairness into a concise framework. Marques et al. (2009) provide a set of scenarios for alternative options of managing systems, considering the social conditions, ecological status and services provision spheres. To guarantee that accurate decisions are undertaken, a clear perception of the society's goals, at both short- and long-term, must be defined (Costanza et al., 2001; Brock et al., 2009). Thus, when choices have to be made between the ecosystems' conservation and the expansion or maintenance of human activities, a comprehensive knowledge of the impacts and importance that these activities may have on the natural environment and on the services provision have to be taken into account.

3. Tools to guide management actions

3.1. Conceptual assessment design

A first step in designing an ecosystem approach functionality is the clear definition of the ecosystem properties, problems and goals to achieve.

An effective sustainability assessment method should provide overall information without loss of information of system parts. The approach must, therefore, be necessarily holistic covering the variation in system aspects like e.g. performance, viability, carrying capacity and resilience with the largest impact on the overall system's sustainability (Bossel, 2000). This implies that information must be put together in an integrative and cumulative way and using a method or instrument that recognizes all the relevant system components, its values as well as the expected future values of the ecosystem and the social system and based on current and expected future desired human activities.

Goals can be articulated to express the current trends and provide the basis for the entire assessment (Hardi and Zdan, 1997), including the selection of the key orientors to be followed. According to Bossel (1992) '*orientors are aspects, notions, properties or dimensions which can be used as criteria to describe and evaluate the system's developmental stage*'. An orientor is, therefore, built or composed by a set of sectoral indicators. To answer the need for suitable communication, one-measure sectoral indicators and composite indicators are increasingly popular for policy makers (compare the well-known economic indices). They are also considered as useful for public involvement in conveying information on systems performance, considering environment, economy, society, or technological development (Singh et al., 2009). These indicators, however, should be derived from state of the art research and surveys.

In general, ecosystem approach studies require integrative tools that reveal the system status, and further demand a framework application that will work as a road-map to be followed (Knoflacher et al., 2003). This brings us to the point of what we have available now or require in the future.

3.2. Single and composite indicators and tools

3.2.1. Biological indicators

From an operational point of view indicators have in general to fit in a well-accepted sequence of objectives, monitoring programs and management measures (McLusky and Elliott, 2004). In order to be considered as a 'good indicator' for ecosystem conditions, several requirements must be fulfilled. For example Salas (2002) considers that a good ecological indicator should be: (1) easy to handle, (2) sensitive to small variations of environmental stresses, (3) independent of reference conditions, (4) applicable to extensive geographical areas, (5) relevant for policy and management needs. Several schemes and classifications of catalogued indicators are available (see, for example, Hellawell, 1986; Dale and Beyeler, 2001; Belfiore et al., 2003; Marques et al., 2009). Hence, it emerges that as long as indicators fulfil the requirements under the heading 'characteristics' they may vary from species, via processes, values of boundary conditions to resource concentrations. The difficult task is to derive an indicator or set of indicators that together are able to meet these criteria. In fact, despite the panoply of ecological indicators that can be found in the literature, very often they are more or less specific for a given kind of stress, applicable to a particular type of community or site-specific (Salas et al., 2006; Pinto et al., 2009). Moreover, another big problem is that the system functioning or the species structure 'story' related to or behind the measured values is usually not clear. For that reason biologists and ecologists still look for and contribute to the development of new indicators.

The conceptual idea behind the development of biological indicators based on a single species or species assemblages is that they are supposed to reflect the effect of any stress or complex of stress put to the system (Pearson and Rosenberg, 1978; Rosenberg et al., 2004).

For open and dynamic shallow coastal systems it is difficult to accept that one single species could be used to indicate the systems quality or condition (see also Elliott and Quintino, 2007). Instead composite biological indicators based on benthic macrofauna assemblages have been favoured because these benthic animals live relatively long and thus may have incorporated within the species assemblages the negative effects of the system's stress. The species composition then thus depicts the effects of the total environmental stress (cf. Pearson and Rosenberg, 1978). At the species structure level the other extreme is the use of a multispecies indicator (AMOEBA approach; ten Brink et al., 1991) where the relative abundance of *circa* 30 species are plotted, and in a radar plot in an attempt to view holistically the system's quality. However, from an ecological point of view a list of species and its (relative) abundance is not enough to qualify the functioning of that system. This is caused by the observed strong inter-annual variation in species abundance in shallow coastal ecosystems (de Jonge, 2007). Moreover, according to Elliott and Quintino (2007) one of the main causes for strong variations in coastal and estuarine systems is often natural factors instead of human activities.

Recently several authors have tested the among agreement of a large number of biological indicators. Blanchet et al. (2008) investigated the AMBI (AZTI marine biotic index; Borja et al., 2000), BENTIX (BENTIX biotic index; Simboura and Zenetos, 2002), Shannon–Wiener diversity, BQI (Benthic Quality Index; Rosenberg et al., 2004) and BOPA (Benthic Opportunistic Polychaeta

Amphipoda index; Dauvin and Ruellet, 2007) biological indices in two semi-enclosed sheltered coastal ecosystems along the western coast of France. Chainho et al. (2007) studied subtidal assemblages of the Mondego estuary focussing on the application of the Margalef, Shannon–Wiener, AMBI and W-Statistic indices. Patrício et al. (2009) studied two intertidal areas within the Mondego estuary by the Margalef, Shannon–Wiener, Berger–Parker, taxonomic distinctness measures, AZTI marine biotic index (AMBI), infaunal trophic index (ITI) and eco-exergy based indices. The results (Chainho et al., 2007; Blanchet et al., 2008; Patrício et al., 2009) were disappointing in that the agreement between these indicators was either absent or weak. Patrício et al. (2009) conclude that presumably the developed indicators describe different aspects of the biological quality. In an effort to intercalibrate single and multi-metric indicators across five European lagoonal and estuarine systems, Borja et al. (2011) observed that the tested indices were largely consistent in their response to a pressure gradient. However, with reference to that study two points should be highlighted: 1) some of those indices responded differently depending on the considered system; 2) some inconsistencies were also observed, especially for transitional waters, highlighting the difficulties of the generic application of indicators to both transitional (estuaries, lagoons) and marine (coasts, fjords) environments.

Based on the above, and in agreement with de Jonge (2007), we arrive at the conclusion that attempts should be made to integrate species abundancy with aspects representing the systems functioning.

3.2.2. Determining habitats and its ecological characterisation

Physical factors like salinity, sediment composition, temperature, tidal range, elevation of intertidal stations can be used to define (potential) habitats. These then can be visualized in a Geographical Information Systems (GIS) application. The principle is exemplified in Figure 2 and shows that based on only the 4 factors (salinity, current velocity, depth and emergence) 8 different potential habitat classes can be distinguished. Because of the distinction between brackish and saline we even end up with potentially 13 different habitats (6 in the brackish zone and 7 in the saline zone). An example of the scale and distribution of these potential habitats is also presented in Figure 2 for the Schelde Estuary (The Netherlands). In combination with the probability of occurrence of species as function of the same type of factors as described here one can obtain a picture of the potential or possible species assemblages within the different zones as depicted in Figure 2 (see Ysebaert et al., 2002). An impressive documentation of the necessary analyses can be found in Ysebaert and Herman (2002). In our opinion this technique can be used to characterise relatively large units within an estuarine or coastal system. When these potential habitats (or ecotopes) have been described in terms of species structures (communities) then it may offer the possibility to narrow focus to this aspect instead of all the details related to these 'habitats' or 'suitability maps' (HABIMAP; de Jong, 2000a) for the occurrence of specific species assemblages. Of course there are many problems related to this approach (see Ysebaert and Herman, 2002) but also challenges as shown by Thrush et al. (2003). Therefore, we have decided to look at the potential opportunities to be able to make a next step which is also to incorporate the rest of the ecosystem and to apply the approach for management purposes.

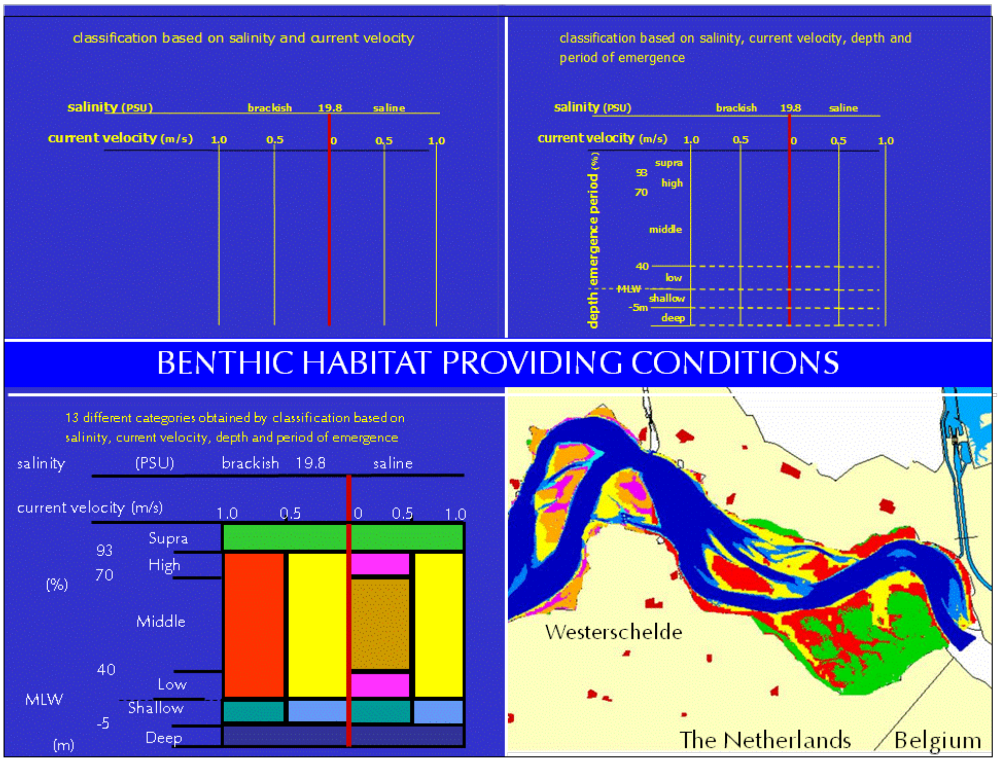

Figure 2. Illustration of a step by step construction of a habitat map based on the factors salinity, current velocity, depth and emergence. The procedure is applied to the Westerschelde (part of the Schelde Estuary in the southwestern part of The Netherlands) resulting in 13 different habitats covering the marine and the brackish part of the estuary.

3.2.3. Dynamic modelling

Models range from mathematical and statistical models to functional models and from descriptive or phenomenological models to causal or deterministic models. Despite the fact that there is a lot of criticism in using dynamic models for predictive purposes (e.g. Haag and Kaupenjohann, 2001) they are, stimulated by requests from governmental authorities, widespread in use. In an interesting and convincing article Haag and Kaupenjohann (2001) explain that due to the dynamical system paradigm these systems are conceptually and necessarily closed systems requiring a fixed set of 'a priori' defined parameters. They further explain that ecosystems are conceived as conceptually open, self-modifying systems, which itself produce novelty and new parameters and which cannot be severed from their environment while the dynamic models cannot escape their own constraints. Thus the predictive capacity of these model systems is not at all warranted so that they have to be considered as deficient instruments in reducing the uncertainty as to future system behaviour. Modelling exercises for decision-making need to take into account the transparency of the process in order to facilitate the participation of stakeholders (see also Schuttelaars et al., in press). When modern concepts such as self organisation are applied and system structures can develop freely then the question about the uncertainty of the capacity to predict arises as well. However, given their magic irradiation to decision makers and managers it is not realistic to assume that they will disappear as decision supporting instruments within a short period of time. Therefore we are the opinion that it is better to look for possibilities to use these instruments in a slightly different way which is to and in combination with other applications.

3.2.4. Ecological Network Analysis

A static approach, with potential for dynamic use, and which can be combined well with dynamic modelling is 'ecological network analysis' (ENA) (Ulanowicz, 1980, 1986, 1997). ENA is essentially based on the 1st and 2nd laws of thermodynamics and include analytical routines such as Lindemans trophic analysis (Lindeman, 1942), the Finn cycling index (Finn, 1976) and the input–output analysis (Leontief, 1951). One of the first who applied part of this approach, the input-output analysis techniques, to ecosystems was Hannon (1973). The ENA analysis requires a 'quantified' food web because it is species (or functional species groups) oriented (see example of the food web in Figure 3). In its simplest form it is a network consisting of nodes connected to each other by the flow of material and energy between them. ENA may be a helpful tool in judging the systems condition by an available set of quantitative system indicators. The data needed for ENA are the same as needed for dynamic modelling exercises and being biomass (B), physiological requirements (P/B), loss terms (respiration or dissipation, export, catch) and relationships between compartments (diet relationships: who eats what, whom and how much). Information needed for the ENA food web analysis is the food web structure and compartment related data so that a number of indices can be calculated. The ENA approach seems to be a good candidate for further application in both analysing the functioning and judging the quality of ecosystems.

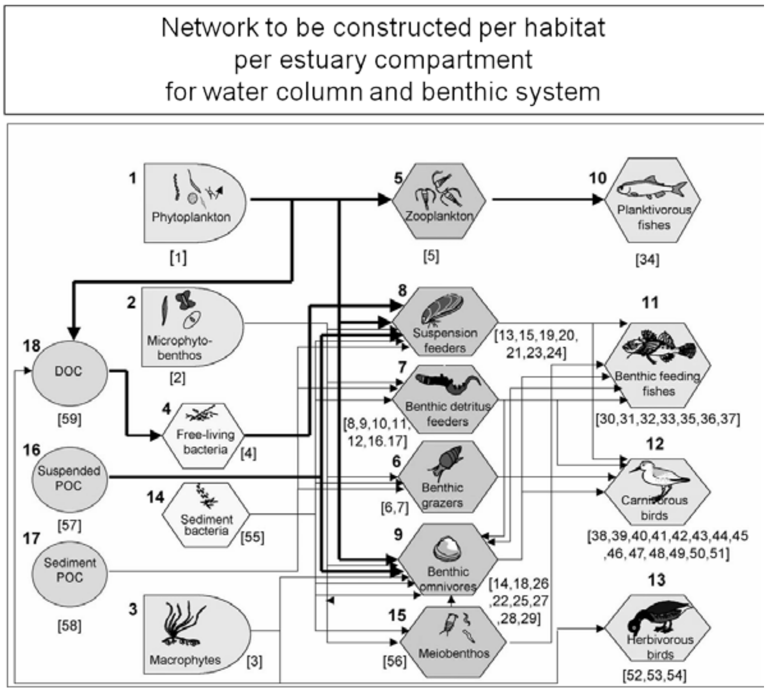

Figure 3. The food web of the Sylt-Rømø Bight as published by Baird et al. (2004) and reproduced with permission of the publisher. The food web represents 59 compartments aggregated into an 18 compartments model used for ecological network analysis. Numbers in bold face: aggregation numbers; numbers in brackets: original compartment numbers; thin arrows: interactions between compartments; thick arrows: pelagic interactions. For more detailed information the reader is referred to the above cited publication.

ENA is not a tool in itself but has been further developed by combining techniques used in the social sciences (see for an example Luczkovitch et al., 2003). New techniques become increasingly important in especially studies related to social aspects in all sciences (e.g. Janssen et al., 2006; Martínez-López et al., 2009) but until recently less so in economic studies (see also below). A short and clear overview of the origin and potential of ENA is published by Fath (2004). Recently Patrício et al. (2004) describe in detail the procedural steps for the application of ENA. Energy budgets can be developed using ‘Ecopath with Ecosim’ modelling software which can be found at (http://www.ecopath.org). This results in balanced budgets for each trophic group. Values on consumption, production, respiration and ingestion by Ecopath with Ecosim are subsequently imported into (http://www.glerl.noaa.gov/EcoNetwrk/ or http://www.cbl.umces.edu/~ulan/ntwk/network.html; Ulanowicz, 1999) to calculate annual biomass budgets for each compartment. The structures of trophic levels and cycling as well as the input-output analysis for a given network can then be analysed and the system properties be calculated using algorithms described by Ulanowicz (1986).

The trophic status can be assessed by the trophic analysis which calculates trophic efficiencies among different estuarine systems according to a standard straight-chain network (Baird et al., 1991). The trophic efficiency between any two levels is defined as the amount a given level passes on to the next one, divided by how much it received from the previous level (Ulanowicz and Wulff, 1991). The energy flow networks can be visualized in a canonical trophic form (‘‘Lindeman spine’’; Ulanowicz, 1997). Connectance indices are estimates of the effective number of links both into and out of each compartment. The Finn Cycling Index (FCI) quantifies the proportion of total system throughput (TST) that is devoted to the recycling of carbon (Finn, 1976). Recycling involve also other elements such as N, P, Si, and energy. For more indices the reader is referred to Ulanowicz (1980, 1986, 1997). Finally, the input-output analysis quantifies direct and indirect trophic effects for each component in the network.

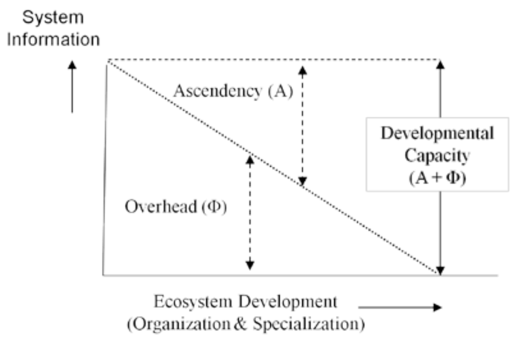

Figure 4. Conceptual presentation of the relationships between Development Capacity (level of system information), Ascendency, Overheads (including Redundancy), and Ecosystem Development (its Organization and Specialization). The System Information incorporates the magnitude and diversity of all the flows in the system.

Apart from the main analyses, there are a number of ecosystem indices which can also be used for practical purposes. Some of the main indices (cf. also details in Figures 3 and 5) are briefly described. Total system throughput (TST) is the overall activity of the system and which is given by the total sum of all the transfer processes in that system. Ascendency (A) in Figure 4 indicates the organisation of the flows and the magnitude of them. It is interpreted by Ulanowicz as “the tightness of the constraints channelling trophic linkages”. A higher ascendency indicates a food web with stronger cycling due to ‘trophic specialists’ and/or higher efficiency while lower values indicate a more generalist-based system with consequently lower transfer efficiency and decreased cycling. It also represents the degree of organisation (‘developmental status’). Average mutual information (AMI) is the unscaled form of the ascendency (A) in Figure 4 and measures the average amount of constraint exerted upon an arbitrary quantum of currency as it is channelled from any one compartment to the next. Developmental capacity (DC) in Figure 4 is a product of TST and flow diversity. If DC is scaled by TST it yields the Flow Diversity Index (Ulanowicz, 2004). It thus also is an index for the systems complexity. Overhead (Φ) in Figure 4 is an entropy term and a measure of inefficiency of the material (carbon) flow through the food web. It is a ‘disorder’ term caused by the system ‘dissipation’ (e.g. respiration), the ‘redundancy’ of relations between species compartments and the ‘export’ from the system. It is the amount by which the capacity of a non-isolated system exceeds the ascendancy (A). In terms of the flows it resembles the redundancy but including the transfers with the external world. Redundancy (R) quantifies the degree to which pathways parallel each other in a network. The fluxes between the different trophic levels, which form the basis for the indices, are given in a strongly simplified food web of Figure 5.

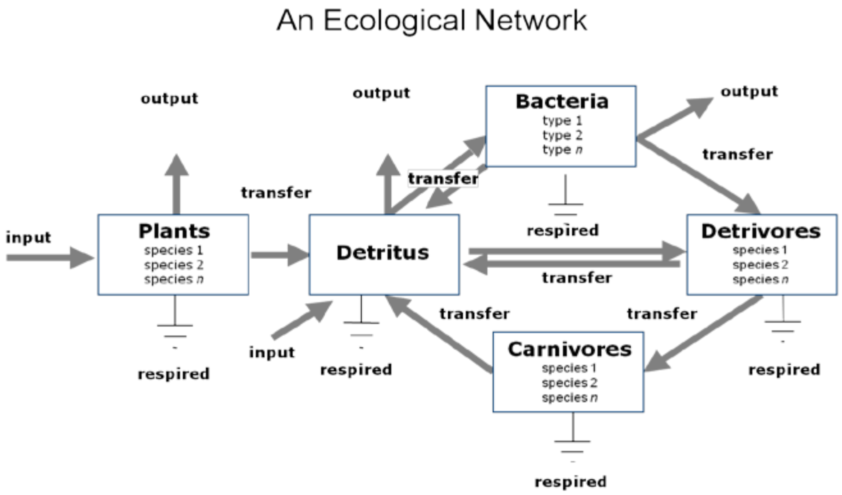

Figure 5. A strongly simplified food web illustrating the main fluxes of energy (often proxied by e.g. carbon, phosphorus, or nitrogen) between the different trophic levels.

The above indicates the complexity within ecosystems and also demonstrates that straightforward description of ecosystems by only species assemblages (e.g. AMOEBA approach; ten Brink et al., 1991) cannot be useful to judge the condition of the system. The system’s structure as well as the system’s functioning need explicitly or implicitly to be incorporated in any indicator. However, the situation is even more complex than described so far. The existence and the importance of high internal ‘connectedness’ compared to the connectedness between systems is also an aspect already mentioned by many others (e.g. Jörgensen and Müller, 2000). Based on more general ecological considerations these authors mention that ecosystems are not only emergent in their expression and show cycling of material but also show self-regulation and self-organisation based on feedback loops. We possibly may be able to incorporate this sort of dynamics when calculating developments at the habitat levels where we are dealing with a restricted number of parameters. Within the context of the present paper, application to the ecosystem level may yet be a bridge too far.

3.3. Integral indicators and tools

Coupling of data from very diverse fields as ecology, economy and social spheres requires a framework for guiding the integration. One approach could be the use of the ‘integral system’ as starting point to fill the gaps among information and data (Figure 6). The social-ecological systems connection (SES; as proposed by Berkes and Folke, 1998), represented in the above diagram by the ‘final services’ box, assumes that a series of concepts, such as resilience, complexity or sustainability, are inherent to this kind of analysis, once they will impact or even determine the flux intensity among compartments. The dynamic links are represented by the DPSIR related steps (see section 3.5).

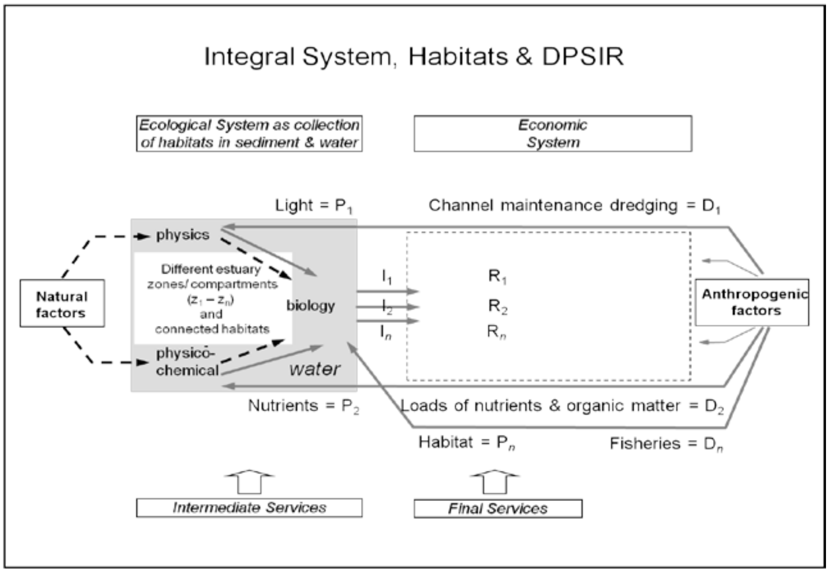

Figure 6. Diagram as presented in Figure 1 but now the physical, physico-chemical and biological subsystems are substituted by estuarine zones or compartments and connected habitats. Further, the generalisations from Figure 1 has been substituted by realistic examples (channel maintenance dredging, loading of the system by pollutants and nutrients and fisheries. Moreover, the DPSIR framework has been applied where D= driver, P = pressure to system, S = state change of system, I = impact to humans and R = the supposed human response. The parts of the system which represent the ‘intermediate services’ and the ‘final services’ or ‘human benefits’ are also indicated.

3.3.1. Resilience and carrying capacity as conceptual integral tools

Resilience may be defined as the capacity of any system (natural, agricultural, urban) to cope with external disturbances without shifting into a qualitatively different state (Gunderson and Holling, 2002). Many scientists and policy makers consider resilience as an important indicator of ecosystem integrity, allowing to determine critical thresholds and the minimum requirements for ecosystems functioning and consequently to services provision. Lenton (1998) suggested that complex ecological networks act as an adaptive system by stabilising feedbacks and thus reducing system perturbations. The same is thought to be true for ecological succession which is assumed to lead to mature-stage ecosystems with good resistance to external perturbations (Odum, 1969). Based on these discussions Kristensen et al. (2003) suggested that human society resilience could also be considered as representing an ecological goal function.

The carrying capacity concept is intimately associated with the notion of thresholds and a certain optimum and maximum level in development of a system and its compartments. The system's carrying capacity may be defined as the point where the biomass of a given population stops increasing (achieving the biomass maximum carrying capacity). This development (governed by resource limitation or scarcity in space) is considered as an ecosystem property (Dame and Prins, 1997). This definition of maximum level of carrying capacity may differ from the economic carrying capacity (Smaal et al., 1998) that is related to exploitation and usually underlies management strategies.

Both issues are complex and demand for integrative ways to measure it. Some efforts to measure the resilience and carrying capacity of a system have been carried out by e.g. DeAngelis (1980); Ludwig et al. (1997); Smaal et al. (1998); Berkes et al. (2003); Kristensen et al. (2003), but there is still no consensus regarding the relations between properties of resilience and carrying capacity and functions of ecosystems. Most of the attempts have been focused on model development and implementation. For example Kristensen et al. (2003) employed a model to estimate the role of resilience on some goal functions of systems. They concluded that the maximization of resilience leads to the optimization of other goal functions in the system (e.g. phyto- or zooplankton biomasses, nutrients flux). However, the authors highlighted the need to further analyse the mechanism underlying the maximization of the systems resilience. The same happens when models on carrying capacity are applied to estimate optimum growth and exploitation of commercial species (e.g. Smaal et al., 1998; Duarte et al., 2003), or even to evaluate the outcome of management strategies and promote efficient measures (e.g. Thébault et al., 2008).

Within the context of the present paper it is very difficult to apply 'resilience' technically because it is, so far, a definition that has not been reached a proper implementation level. Therefore, and alternatively, it may be better to apply ENA related indices to indicate the quality of the ecosystem and to use that as a basis for further judging the anthropogenic role in the functioning of the integral system.

Carrying capacity is more easily executable than resilience. The use of carrying capacity is therefore recommended here because it is connected to the relation between two factors and often can directly or indirectly be related to growth (population) and production (bivalves, fish, plants) of parts of the ecosystem which can be analysed by ENA.

3.3.2. Ecosystem complexity and sustainability as conceptual integral tools

Complexity of ecological systems is an ambiguous term and usually may be related to structural, functional, or physical aspects of ecosystems (Adami, 2002). Nine forms of complexity were identified by Jørgensen (1997) giving special focus to the fact that the complexity is wider than just the interactions among species and resources (see above). A couple of measures have been developed to provide an integrative indicator of a system's (role) complexity level. For example, Adami (2002) has developed mathematical equations to cover the physical complexity of ecosystems, arguing that the total complexity would have to be defined as the mutual entropy of all organisms, about each other and the world they live in. In another perspective, and as pointed out by Jørgensen (1997), the way an ecosystem responds to perturbations has been widely debated in terms of stability. The complexity of the regulation by feedback mechanisms has not received much attention. Costanza and Daly (1992) argue that ecosystem health of complex system, defined by six properties (homeostasis, absence of disease, diversity or complexity, stability or resilience, vigour or scope to growth, balance between system components), may be given by a general system health index: HI=VOR; where, V is system vigour, O the system organization, and R is the resilience index. However, other indices for ecosystem health and complexity focus on exergy, specific exergy, and buffer capacities (Jørgensen, 1997), which also fits in this ecosystem health definition (Jørgensen, 1997). These last indices (may) are also in use to measure the system maturity, as Dalsgaard and Oficial (1995) did using the ECOPATH model regarding agro-ecological systems.

The interactions among biodiversity assets, ecosystem processes and functioning (BEF), and the services provided by natural or production systems has been widely studied over the years (e.g. Duarte, 2000; Griffin et al., 2009; Naeem et al., 2009). However, their quantitative relations are still poorly understood (de Groot et al., 2010) and more efforts are needed to develop a 'full integrative link' among compartments before they can applied satisfactorily.

3.4. Social and Economic Analysis within a Decision Support System

The integrated approach we have in mind needs to formally encompass socio-economic analysis and is guided by the acceptance of a strong sustainability viewpoint. Ecosystems are seen as suppliers of a range of ES through which humans benefit in terms of welfare or wellbeing. The analysis tries where meaningful to place monetary values on the benefits provided by 'healthy' ecosystems. But it is recognised that some services are not suitable candidates for monetisation e.g. so-called cultural services such as among others heritage landscapes and seascapes. The approach is also limited to service values that are largely instrumental and therefore it does not explicitly include pure intrinsic value which some commentators claim for nature. In terms of the political economy of nature conservation what is being proposed here is that the inclusion of socio-economic analysis within the decision support system (DSS) serves to supplement scientific and ethical arguments in favour of environmental protection. The EU MSFD (EC, 2008) for example explicitly calls for (Article 8.1c) '*an economic and social analysis of the use of those waters and of the cost of degradation of the marine environment*'.

The possible relative changes in quality status and the human-related activities which serve to pressurize the marine environment can be modelled within the DSS we have advocated. An initial scoping stage could be based on the D-P-S-I-R framework (see next section) and the temporal scale of the environmental changes can be modelled via scenario analyses (see also Bockstael et al., 1995; Voinov et al., 1999). While future uncertainty will always remain very problematic, scenario analysis (typically based on a 'business as usual' baseline trend assessment against which a range of different future paths can be assessed) offers a way of coping with uncertainty and may provide policy relevant information. The process of economic analysis can only take place after policy issues have been identified within given spatial and temporal scales and scenarios and evaluative criteria have been established. Underpinning the whole DSS is of course the existing scientific knowledge base i.e. what is and is not known about ecosystem structure, process and functioning.

Once agreed, the policy issues and scenarios that are identified by the scientific and policy communities provide the context within which the socio-economic assessment can be constructed.

Note however that this is not a one-way process. Feedback should occur between all stages of the assessment process and deliberative arrangements should be made with stakeholders, since questions that are thrown up by the assessment can help to refine the policy issues and scenarios that are of most concern to relevant stakeholders/user groups. In general most problem situations involve competing uses for coastal/marine resources and are conditioned by the governance that is in place.

The resource system policy issues under investigation will be composed of a complex mixture of environmental and socio-political driving processes, consequent environmental state changes which then impact on the provision of ES and their effects on human welfare/well-being. The distribution of the welfare gains/losses in society, together with existing policy measures and networks will influence policy responses. The economic analysis (cost-benefit analysis or CBA and cost-effectiveness analysis or CEA) seeks to evaluate the social welfare gains/losses from an economic efficiency perspective, tempered by any distributional equity considerations, other precautionary environmental standards and regional economic constraints (most often focussed on local employment and economic multiplier impacts which can result in cultural and community losses or gains, e.g. closure or restrictions on fisheries). The main distinction between CBA and CEA is that the desired outcome(s) is determined a priori in CEA e.g. the achievement of a legally set water quality standard at least cost to society, but not in CBA.

Economic valuation is often undertaken in terms of ‘opportunity cost’. This means that the value of an ecosystem service (or a damage impact avoided) is assessed through the ‘trade-offs’ associated with obtaining or maintaining the service flow. In principle it may then be possible to compare all relevant options and look for the ‘highest value’ uses of the ecosystem. Marine and coastal ES and benefits can in a simplified way be linked to four environmental impacts or effect categories relevant for human welfare:

- direct and indirect productivity effects (use values);
- human health effects (use values);
- amenity effects (use values); and
- existence effects (non-use values) such as loss of marine biodiversity and/or cultural assets.

Different valuation techniques will be appropriate for each of the four categories but note again that the symbolic and cultural values assigned to some coastal/marine features and land/seascapes lie outside the monetary calculus and are conditioned by social preferences and norms arrived at, over time. Through various forms of information transmission, art, literature, film.

Productivity effects related to, for example, fisheries, aquaculture, recreation/tourism and indirectly to services like storm protection, erosion reduction, etc, can be valued using market prices linked to changes in the value of output or loss of earnings. The approach needs a production function which is derived often through the use of bio-economic models (e.g. fisheries).They can also be valued using surrogates such as e.g. property prices, land values, travel costs of recreation and damage costs avoided. Health effects are valued by cost of illness measures or survey-based methods. Amenity effects can be assigned values through travel costs, property values or survey methods such as contingent valuation, contingent ranking and choice experiments. This latter group all use questionnaires to elicit individuals’ willingness to pay or be compensated in monetary terms for gains/losses of services. Finally, existence or bequest (from generation to generation) values can only be derived if at all via surveys. Because it is not possible to value all ES in monetary terms the DSS should include so-called multi-criteria evaluation methods (MCA) which quantitatively or qualitatively encompass a range of social/deliberative and ecological conservation perspectives. MCA as a framework can incorporate the results of CBA/CEA and provides weighted and scaled rankings of different options (Janssen, 1994; Olson, 1996; DETR, 2000).

In the literature five issues have been identified as critical to the appropriate economic valuation of ES.

Spatial explicitness is important in order to clarify the level of understanding (or ignorance) of underlying ecosystem structure, process and functioning. This contextual analysis must then include appropriate socio-economic, political and cultural parameters in order to properly identify ES supply and demand side beneficiaries. ES are therefore context dependent in terms of their provision and their associated benefits and costs. If we take the example of coastal wetland and their supply of carbon sequestration/storage services, it turns out that the net effect of this service is conditioned by the simultaneous release of methane. But the spatial location of the wetland and in particular the prevailing salinity condition plays a significant role in the carbon storage to methane emission ratio and consequent global warming effect (Luisetti et al., 2010). It is anticipated that the incorporation of spatial factors in ecosystem valuation is likely to become easier and more commonplace as access to GIS software and expertise increases (Bateman et al., 2006).

Secondly, **marginality** is an important issue as economics requires that for the valuation of ES to be meaningful such analysis should be conducted ‘at the margin’. This means focusing on relatively small, incremental changes rather than large state changing impacts. However, given the scientific uncertainties which shroud ecosystem functioning, it is often difficult to discern whether a given change is ‘marginal’ or not, and when thresholds are being approached or crossed. Knowledge of the drivers and pressures on the ecosystems under study, as well as understanding of how the system is changing or might change from its current state is crucial. This has been called the system’s transition path (Turner et al., 2003; Fisher et al., 2009). It is important to know if the transition path is ‘stepped’ as in the case of a coral reef system or shallow lake/lagoon, or it is ‘relatively smooth’ such as in species invasion into an area. By identifying the transition path we can force the analysis to consider losses or gains in service/economic value between two distinct states of the system.

Thirdly, **double counting** may be a problem where competing services are valued separately and the values aggregated; or where an intermediate service (in economic value terms) is first valued separately but also subsequently through its contribution to a final service benefit. Thus a coastal wetland may provide nutrient cycling capability which then leads to better water quality for recreation and amenity. The economic value involved is restricted to the recreation/amenity gain, excluding the nutrient cycling the value of which contributes to the final service benefit value (higher quality recreation/amenity experiences).

Fourthly, **non-linearities** in services provision complicate valuation and system management e.g. shallow phosphorus-limited lakes may flip from one state to another with dramatic effects on some services. Further, non-linearities can mean that marginal benefits are not equally distributed e.g. the storm protection benefit of a unit increase in mangrove habitat area may not be constant for mangroves of all sizes due to non-linearities in wave attenuation. If a cost-benefit assessment assumed linearity but service provision is in fact non-linear, policy option outcomes may be unnecessarily polarised (Barbier et al., 2008).

Finally, **threshold effects**, i.e. the point at which an ecosystem may change abruptly into an alternative steady state, are problematic for CBA. For marginal analysis to hold true, the next unit of change to be valued should not be capable of tipping the system over a functional threshold or 'safe minimum standard'. Given the uncertainties we currently face identifying risk will require expert input from ecologists and other scientists, risk analysts and ethicists etc and will ultimately require ethical/political choices to be made and deliberatively agreed.

The notion of total economic value (TEV) provides an all encompassing measure of the economic value of an ecosystem service supply. It is important to note however that TEV is always less than total systems value. A minimum configuration of ecosystem structure and process is required before final services can be provided. The system therefore posses 'extra' value known as 'glue' or 'primary' value (Turner et al., 2003). Because there is uncertainty over what is or is not a sustainable 'healthy' functioning state. In many contexts a precautionary approach to management has much to recommend it.

TEV decomposes into use and non-use values but it does not include other kinds of values such as intrinsic values which are usually defined as values residing 'in' the asset and unrelated to human preferences or even human observation. Cultural/symbolic values which groups of people have assigned to landscapes etc are also outside TEV. Nevertheless, apart from the problem of making the notion of intrinsic value operational, it can be argued that some people's willingness to pay for the conservation of an asset, independently of any use they make of it, is influenced by their own judgements about intrinsic value. This may show up especially in claims about species 'rights to existence' but also as a form of human altruism.

3.5. DPSIR as a framework for further tool development

The DPSIR approach was developed by OECD (1994) and soon followed by further application (Turner et al., 1998). Since then it has attracted wide attention from the EU Commission, managers and scientists mainly because of its practicability. DPSIR can be defined as an operational framework identifying 'drivers' of change which lead to individual 'pressures' causing a different system 'state' which consequently lead to 'impacts' on human welfare which then require a policy/management 'response'. The approach is attractive because it can be used in a very general way as a scoping framework assessing causes, consequences, and responses to changes caused by any stressor. Apart from coupling the effects of human activities to the ecosystem (Figure 1), it can also integrate ES and societal benefits (Atkins et al., 2011) which in this paper are indicated as 'intermediate services' (delivered by the ecosystem) and 'final services' (societal benefits). This is illustrated in Figure 6. The drivers, pressures, state change, impacts and responses can be visualized schematically, see Rogers and Greenaway (2005). Such a scheme can be used during discussions among stakeholders but does not necessarily provide detailed enough information on the magnitude and significance of the ‘state change’. For management purposes we additionally need more specific quantified information. A next step then may be that the indicated DPSIR-factors are quantified and put together in a model describing the cause – effect relationship (including all the known feed backs) between the ecological and the socio-economic system as defined in Fig. 1 and now further visualized in Figure 6.

The impact of these changes to the socio-economic system can first be quantified in purely ecological terms but subsequently in terms of changes in ‘goods and services’, as also suggested by others like Atkins et al. (2011). These are, for example, changed amounts of available stocks, harvest rates or recreation/amenity gains/losses.

3.6. Developing the integration among ecological, economic and social aspects

As indicated above, graph theory related science is used in ecology and widely in some social sciences, but less so in conventional economics. We take up the challenge to stimulate further steps in this integration. We emphasise that a lot of processes within human society as well as in our environment are conditioned by surface area and scale. Scales in human and biological social sciences may vary from square millimetres (bacteria, protozoens) to over fifty thousand square kilometres representing ‘eco zones’ (de Jong, 2000a). The notion that processes are operational at quite different spatial scales may be used to determine the proper scale for integrating conceptually the ecosystem and its intermediate service role with the economic system, its pressures to the environment and the environmental final services to society. This integrated picture could then be further developed to serve policy making and management activities. In the next paragraph we set out how we think that the integration could be started.

Apart from the spatial scale issue, ENA should preferably be performed to the level of species so that we end up with a very detailed and complex food web structure within which all the ENA related characteristics and indices can be calculated and compared over time. Changes in system quality can then be assessed by analysing the generated inter-annual indices time series. This sort of analysis could be done for different system conditions: i.e. if data are available for a natural (reference) situation and a recent (perturbed) one due to e.g. anomalies in freshwater flows or weather conditions or human activity related system stress. The differences between the two states and the transition path should tell us something about the impact of the stress acting upon the ecosystem. The usefulness of assessments to detect temporal changes in ecosystem indices may be very useful (Baird, 2011).

ENA can also be applied to a food web consisting of groups of aggregated species thus close to what is represented in Figure 3 and the inlay of Figure 1 (Sylt-Rømø Bay food web as published by Baird et al., 2004). A food web consisting of mainly functional groups and some dominant species (e.g. beds of the blue mussel (*Mytilus edulis*) on intertidal flats) added to it may be more simple and easy to create and handle than a very detailed one, but it will also produce different results because of loss of information. This also means that if ENA is going to be used for management purposes then before the start of the required sampling or monitoring programs a choice has to be made on the most appropriate 'aggregation level'. Exactly the same holds for the application of dynamic simulation models.

A third possible aggregation level is that of the available habitats within the system. The above described habitat mapping (HABIMAP; de Jong, 2000b) approach showed that there is a reasonably good agreement between different estuarine habitats or zones and the composition of the benthic macrofauna assemblages resulting from regression models (Ysebaert et al., 2002). These results then can be applied in habitat models like HABIMAP.

The habitats assessed by the HABIMAP approach can also be used to define zones that play a functional role in relation to ecosystem functioning (behaviour of bacteria, plants and animals) and human activities. Figure 7 for instance presents some examples: grazing by cattle and roosting of birds on saltmarshes, catching fish in the main channels, shrimp fishery during high tide above sandy intertidal flats fringing the channels and gullies, resting of seals on high elevated sandy flats bordering deep channels, predation on the intertidal mud flats by birds, etc. Thus, the 'spatial scale' related to specific habitats needs to be appropriate in order to connect human activities and some characteristics of the estuarine and coastal environment.

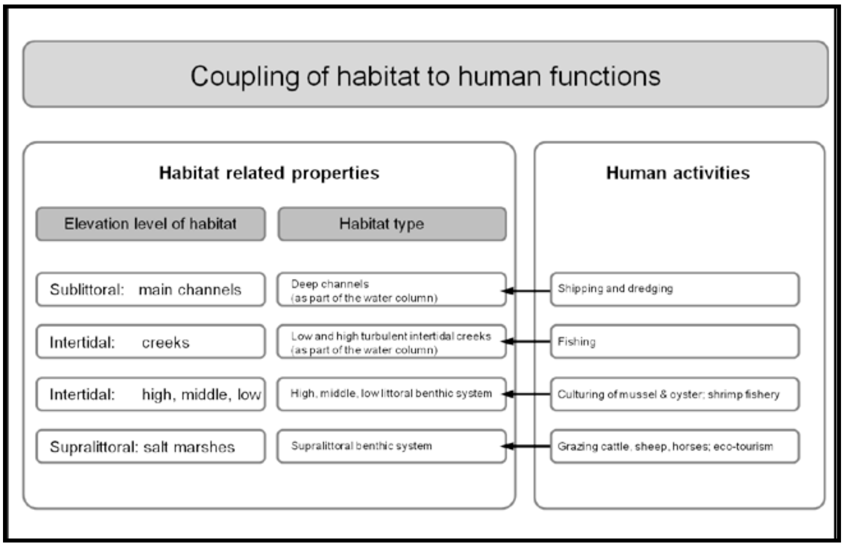

Figure 7. Description of potential habitat related properties and designated human activities to indicate the suitability of the habitat level as the possible ecological and economic integration level.

We suggest here that network analysis techniques related to, but different from, ENA may also be helpful in connecting the economic and social cultural fields to that of the ecosystem for static as well as dynamic assessments. An interesting example has recently been discussed by Johnson et al. (2009) who applied a simulation-based continuous-time Markov chain model (SIENA) to determine the seasonal changes in the Chesapeake Bay food web. Apart from visual inspection they applied a statistical assessment to analyse developments. The visualisation was realized by the graphic analysis tool NetDraw (Borgatti, 2002). For more details the reader is referred to Johnson et al. (2009). Their conclusion was that in a qualitative way this holistic approach was successful in describing the changes in the food web from a highly complex one in summer to one of much lower complexity in winter (highest ascendency). Another interesting example was produced by Luczkovich et al. (2003) and showed the role of nitrogen involved in the food production system of

Norway, visualized using the regular equivalence (REGE) algorithm and (3-dimensional) multi-dimensional scaling (MDS) approach (see for further details Luczkovich et al., 2003). In the example he describes the flows of the N import (by sea catches and animal production) and the transfers through the social-ecological system culminating in what he calls the ‘wholesale food production’. In that network relevant parts of the environment are clearly connected to the social network in a quantitative analysis. Janssen et al. (2006) reviewed the importance of network analysis as an interdisciplinary tool which in this case was the social-ecological system (SES), while McMahon et al. (2001) provide tips to ecologists and social scientists interested in the use of network analysis. They pointed to the development of structural models to analyze human interactions (Wasserman and Faust, 1994) and specifically the conceptualization and testing of the interactions within complex systems as social and ecological systems. They further recommend the reduction of elements (nodes) and interactions (arrows) in such a way that the system becomes ‘simple’ enough to be analysed but still complex enough to reflect reality. Their main conclusion was “*that on many scales, social scientists, biologists, and physicists are all studying the same phenomenon. Most of the difficult problems modern society faces arrive in the form of complex structures such as economies, ecosystems, and societie*s”. They argued for further and deeper interdisciplinary collaboration.

3.7. Challenges to the future

3.7.1. Possible actions

The DPSIR approach should be applied to the food web at different aggregation levels (detailed food web, food web based on functional species groups and based on HABIMAP-like habitat units). A suggestion on ‘how’ to realize this is given in Figure 8. This then should lead to insight into the sensitivity and thus also the practical usefulness of the different ENA indicators. It also results in the further quantification of input fluxes under the influence of the social system to the ecosystem, internal transfer, turn-over and export fluxes to human society (like catches of fish and bivalves; Figure 8). In doing this one could argue that the boundaries of the system at the habitat levels are set by the associated ES. Identification and quantification / valuation then establish the strength of the stakeholder claim (see Atkins et al., 2011). As explained above (see also Atkins et al., 2011), there are also stresses to the system related to the use of nature as a vehicle to realize a particular ‘service’. The collection of food (fisheries) results for example in changes in the structure of faunal populations due to primary catch and by-catch plus accompanied possible habitat destruction. The use of nature as a vehicle may also lead to a certain system stress (dredging – turbidity; tourism – disturbance of animals by noise and production of pollutants and litter; loading of nutrients – eutrophication). Sticking to the level of the ‘ecosystem service’ is, however, not required because this is not part of the ecology. The changes in the functioning (structure) of the system itself also need to be quantified so that ecologists and economists can have the proper discussions. All the functions mentioned above can and should be converted in terms of indices that assess a certain change in ecosystem quality and a certain contribution to the functioning and welfare of human society.

In addition to the above, the application possibilities of other network tools, in use in the social sciences, should also be investigated. Which of the available techniques is most helpful in analysing what part of the social network is responsible for what (eco)system stress as well as the revenues in terms of the production of 'goods and services' and income. This then represents an important part of the economic system.

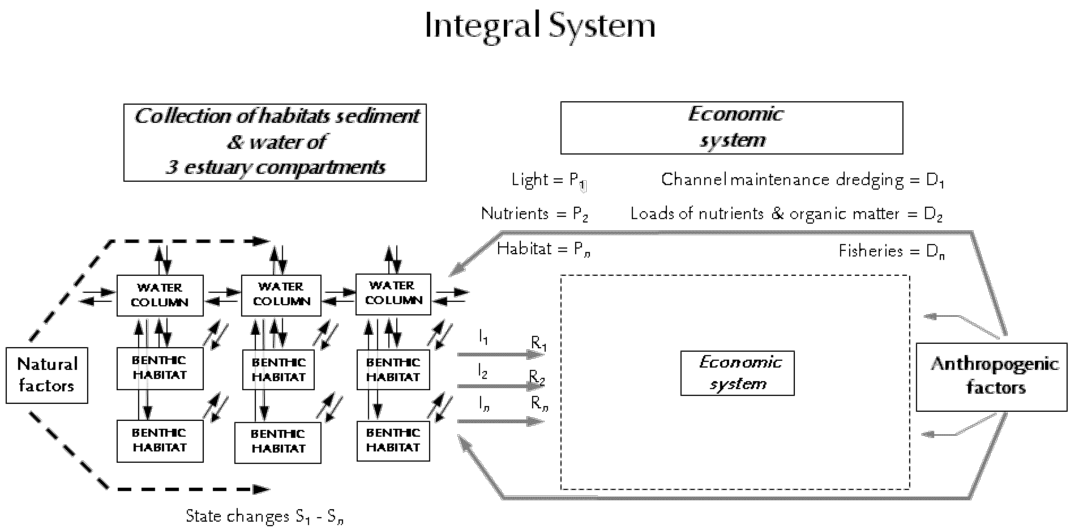

Figure 8. Visualisation of the combination of the information in the Figures 7 and 8 where (ecological) network analysis is carried out per habitat and then coupled to relevant pressures from the socio-economic system. The total sum of the state changes and related impact to the socio-economic system will result in an integrated human response to the total impact.

3.7.2. Weighting the outcome

Environmental change processes are multifaceted and therefore decision support systems (DSS) need to be comprehensive enough to accommodate a range of decision criteria reflecting the various worldviews and cultural norms that may be present in any given policy context. Economic tools (CBA/CEA) informed by the best available science will need to be buttressed by other tools whenever the policy context is 'contested', which is most often the case in coastal and marine management. We have highlighted earlier the need to incorporate and further develop multi-criteria decision tools and stakeholder engagement processes. A range of techniques have been developed encompassing both risk ranking and more generalised multi-criteria decision making procedures which have evolved around scaling and weighting protocols (with experts and/or stakeholders) (Clemen, 1996; Morgan et al., 2000).

Biodiversity assets and ecosystem functioning are hard to value (see for example OECD, 2002), as people recognize their importance and intrinsic value, however not being always easy to put a number to it (e.g. Nunes and van den Bergh, 2001; Nunes and Nijkamp, 2010). Due to the inherent complexity of valuing these ecosystem attributes and wetlands integral functioning, the data and results obtained from researches, such as those conducted by ENA studies, could be enclosed into the economic valuation process, like for example, during the survey elaboration and hypothetical markets (both in 'willingness to pay' and 'willingness to accept' scenarios) construction, allied to surrounding activities and economic drivers inclusion. By doing so, the study could exemplify the integration of human economic activities and social awareness in general, their relation with biodiversity and ecosystem functioning aspects, and the total social / ecological / economic) value that the system under consideration represents (Figure 9).

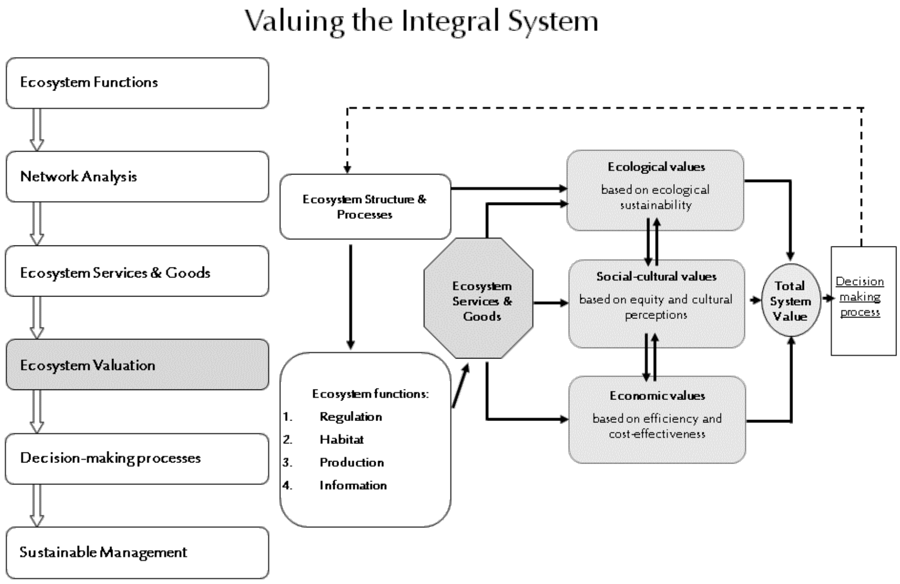

Figure 9. Overview of the components representing the functioning, the functions and the value of parts of the integral system as well as the total system. The total system value is composed of the sum of the ecological, social-cultural and economic values (modified after de Groot et al., 2002).

The information gathered from the ENA analysis is essential to analyse disruptions in coastal systems. However, this information must then be combined with the economic and social information for the successful management of coastal ecosystems. Data integration from the several spheres (ecological, economic and social) is of crucial importance in the design of a useful environmental information system (Fedra, 1997). This integration can then be achieved through the use of tools designed for decision support systems approaches. In this sense, and after the scoping and data gathering stages, multi-criteria analysis (MCA) tools can be used to assist decisions processes. This kind of tool (mostly based on mathematical algorithms) may be flexible and wide-spectrum enough to mitigate the multi-faceted decision problems often associated with ecosystem management and development pressures (see for example Figueira et al., 2005).

Once this (E)NA based approach has been implemented, the scientific and social dialogue around this subject should be maintained. Despite the integrative efforts, some questions remain still open: Are the species chosen to the ENA approach really reflecting the system dynamics at the habitat level? What are suitable reference conditions? What are the boundaries of the system to develop accurate management scenarios? Which measures should be recommended to achieve those scenarios? Are these measures driven by conservational or services-based perspectives?

However, we are the opinion that the proposed approach can facilitate the debate among managers, society and scientific community, creating a common ground for further discussions and developments. Moreover, it should be recalled that, although the habitat scale has been chosen (which allows for a more comprehensive integration of data) it is still ‘localised’, and although may represent a significant portion of a system, most of the times an ecosystem is composed by several habitats, creating a range of fluxes and interactions that have to be taken into consideration (Figure 8).

4. Planning the future

The current lack of information and /or progress on better functional and conceptual relationships within the integral system does not mean that no progress can be made in terms of improving decision making and management. The incorporation of system-related indicators and the ES concept can clarify the relationships between ecosystem change and valued outcomes in terms of 'user' benefits and can therefore lead to better decision making in terms of identifying options with the best net returns to society.

Attention needs to be focused on developing better modelling of ecosystem changes that can be linked to real world management actions (i.e. specific management-related questions) rather than only as the outcome of natural system dynamics. Analysis and indicators need to be targeted on things that managers can influence and recognise as within their competence to change.

Expressing modelling outcomes, through interdisciplinary collaboration, in terms of final ES and valued human benefits makes the management choice decisions 'easier' by providing a common and readily understood unit of account. Even if not all benefits can be expressed in monetary terms a partial analysis can still improve current decision making e.g. quite often it is possible to demonstrate that the benefits (monetary) of only some of the services provided by an ecosystem(s) in a particular context outweigh the costs of conservation and management, or some development alternative that requires ecosystem removal or severe degradation.

Given the data and conceptual limitations we currently face, it is important to, in many cases, avoid the 'do nothing response' and intervene with the best available 'science' but within an adaptive management strategy that seeks as far as is feasible to keep options open and avoid irreversible change, in a 'learning by doing' process.

From the above it is evident that the basis for this sort of approaches is a 'tailor made' monitoring program. In another paper (de Jonge et al., 2006) an analysis was made on the development of the water quality monitoring in the United Kingdom and The Netherlands. In that paper it has been proposed to change part of the current monitoring program from a 'station oriented' one into an 'area oriented' one. As explained there extensively this is something that can be done in a cost effective way and also without ruining the current data series that in The Netherlands already exist since the early 1970s.

5. Conclusions

The diagram in Figure 9, originally produced by de Groot et al. (2002), has proven to be instrumental for our conclusions in the present context.

From an ecological point of view we have 3 recommendations:

1. Further application of ecological network analysis applied to an ecosystem (species or functional groups) which results in characterizing the system functioning (input-output analysis, system throughput and cycling of compounds) and the magnitude of ENA indices under more or less natural conditions is imperative. Temporal changes in ENA indices and reasons for observed changes need to be part of any programme.
2. Application of the DPSIR approach to the ecosystem (species or functional groups) to quantify in both a dynamical simulation model as well as a network approach (static or dynamic network) the same characteristics and indices as under point 1 can facilitate progress.
3. Applying DPSIR to an aggregated network at the level of the HABIMAP habitat, where the functioning is now compartmentalized according to the habitat criteria seems to be a useful way forward.

From a socio- economic point of view we have 6 recommendations.

4. Clearly define the scale of action: are we valuing global, regional or local assets, and over what time period? To guarantee an accurate description and identification of the relevant services as part of the integral system care must be taken to set the analysis in an appropriate spatial context (including socio-economic, political/governance and cultural conditions).
5. When (e)valuating the provision of the services identified in the network analysis the double counting problem needs special attention.
6. Depending on the type of ecosystem service, several types of monetary valuation measures may be used. Suitable techniques may be the production function approach, the contingent valuation method (CVM)/ choice experiments etc. But it should also be noted that it is still not possible to meaningfully capture monetary values for all ES.
7. While we can use the outcomes from economic valuation methods as inputs into a cost-benefit analysis (CBA) in many real world management situations involving difficult trade- off decisions, an overall Multi-Criteria Analysis (MCA) will be required if ‘trust’ and ‘accountability’ concerns have to be countered.
8. Coupling of different hierarchical levels and social components by network analysis approaches may result in a multidimensional picture of a social network interacting with some of the economic components impacting the environment and vice-versa. This picture can be extended by the interaction between the social components and the ecological system. This will need to be better informed by findings now emerging from the behavioural sciences which have highlighted the complexity of human motivations and behaviour.
9. By understanding the role of ecosystem functioning and ES provision to human wellbeing it is possible to identify and target the natural assets of a system and so accomplish for sustainable development requirements.

To facilitate further progress.

10. When applying ecological network analysis at the habitat level, a 'tailor made' monitoring program covering ecological and socio-economic aspects in sufficient detail is essential to guarantee the desired and required progress in valuing, weighing and deciding upon how to proceed in practising sustainable development.

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GENERAL DISCUSSION AND CONCLUSIONS

"Through all these new, imaginative, and creative approaches to the problem of sharing our earth with other creatures there runs a constant theme, the awareness that we are dealing with life with living populations and all their pressures and counter pressures, their surges and recessions. Only by taking account of such life forces and by cautiously seeking to guide them into channels favourable to ourselves can we hope to achieve a reasonable accommodation between the insect hordes and ourselves."

(Rachel Carson, 1962)

General discussion and conclusions

Having as underlying objective achieving a sustainable use and management of natural resources, it is essential to support decision-making processes with an accurate ecological and monetary (e)valuation of ecosystem services. In this light, the present thesis was focused on the contribution of the Mondego river basin and estuarine ecosystems to human well-being, via analysis of their functioning and evaluation of their capacity to provide an array of ecosystem services (ESA). The following aspects were more specifically investigated: (1) Ecological quality status, ecosystem stability and ecosystem services condition of the estuary, relating it with human demands; (2) The need to re-think approaches and management gaps as a way to improve the ways of using ecosystem services; (3) Challenges involved in the assessment and valuation of ecosystem services provided by estuarine systems, as well as in solving operationalization problems.

1. Ecological quality status, ecosystem stability and ecosystem services: How to cope with human demands?

EU environmental policies introduced into member countries' national policies the concept of ecosystem services as a tool for mainstreaming the prevention of biodiversity losses (e.g. EC, 2010) and water resources quality improvement (e.g. European Water Framework Directive; EC, 2000). That is for instance the case of the Portuguese Water Law (Law 58/2005). Undoubtedly in an attempt to make explicit the value of natural resources to society, the term 'ecosystem services' was more recently also integrated in the Portuguese Government management policies, included in the legal regime for the conservation of nature and biodiversity (DL 142/2008) as a key factor for the assessment and preservation of natural assets. The assumption is that biodiversity and water quality play a fundamental role in determining both ecosystem functioning and the provisioning of ecosystems services, which underpins human wellbeing. Over the years special attention has been given to this issue, and several experimental and theoretical researches have tried to make explicit this implicit connection (e.g. Pimm, 1984; Tilman, 1996; Hooper et al., 2005; Balvanera et al., 2006;

Díaz et al., 2007; Naeem et al., 2009; Martín-López et al., 2009). Nevertheless, a core question still remains: How?

Ecosystems influence and are influenced by human activities through intricate multiple interacting manners. Biodiversity assets are essential for the self-organization capacity of ecosystems (Levin, 1999), both to absorb disturbances and to subsequently re-organize after them (Folke et al., 2004). Indeed, the functional characteristics of the species composing a system are considered a key factor for the maintenance of ecological stability/integrity (e.g. Chapin et al., 2000; Díaz et al., 2006; Naeem et al., 2009) and for the provision of services (e.g. Luck et al., 2003; 2009; Kremen et al., 2004). Due to this complex and multi-causal dynamics inherent to ecosystems, uncertainty is an intrinsic characteristic of ecosystem services assessments (Martín-López et al., 2009). From the previous works, it was possible to see that, due to the complexity of the links between biodiversity and ecosystem functioning, it was much more complicated to trace the impact of changes in biodiversity assets through the variations in services outputs. In fact, while demands for certain services increased, such as food provision or even recreational experiences, human actions can eventually determine the inherent ecosystem capacity to continue providing these services at the same levels, considering both spatial and temporal scales. Most of the times, these conditioning relationships are multi-layered and cumulative.

2. Using ecosystem services: re-thinking approaches and management gaps

Several examples were provided along the thesis regarding the relationships between natural assets and the benefits obtained from ecosystems. Nevertheless, the following points still demand careful thought and attention:

- The **scale** (both spatial and temporal) at which ecosystem services evaluation is being undertaken.

The present findings revealed that, if on one hand it is not always possible to include all geographical spatial scales into an analysis (e.g. due to time or financial or human constrains), on the other hand the same ecosystem service can be valued differently according to the scale under consideration. Considering temporal scales, several modifications can occur within an ecosystem along a period of time, regarding changes in ecological quality status but also the perception that people have about it. Some authors suggest that there is a spatial and temporal scale mismatch between the capacity of supplying a service (given by the ecosystem functioning) and the use and enjoyment of that service by society (e.g. Martín-López, 2007; Martín-López et al., 2009).

- The clear understanding of the **interactions among uses** in a given ecosystem.

This is essential to infer if stakeholders will have synergistic or trade-off interactions, and therefore it is indispensable to survey stakeholders and populations to get the full picture of uses and interests on those areas.

- The **scope definition problem**.

Using the ecosystem services approach allowed to integrate ecological (e.g. ecological quality status) and socio-economic features (e.g. pressures/impacts) prevailing in the Mondego estuarine ecosystem through the DPSIR framework. This framework proved to be a useful tool to scope biodiversity management issues, while simultaneously providing a key conceptual link between ecosystem changes and the consequences of such changes (impacts) on people's socio-economic wellbeing (e.g. Turner et al., 1998; 2000). Based on this framework, indicators of ecosystem condition (e.g. ecological status variation) and human welfare (e.g. monetary value of services provision) changes were derived and used to assist management issues.

Nevertheless, to achieve and effective communication of ecosystem services integration possibilities to decision-makers, more efforts should be done to fill up, as much as possible, gaps in the TEV (Total Economic Value; Pearce and Turner, 1990) approach, in order to account for all possible trade-offs/synergies among factors and ecosystem services. Although valuation of ecosystem services may be difficult to carry out (as highlighted in chapter 6, e.g. double-counting issues), it has become an essential tool for decision-makers to implement new ecosystem management policies (e.g. Kumar and Kumar, 2008; Daily et al., 2009). Such valuation constitutes an important instrument to guarantee, in social terms, ecosystem’s biodiversity conservation (Martín-López et al., 2009).

Of course, it should be recalled that ascribing values to ecosystem services should not be faced as an end in itself, but rather as one possible tool to achieve a higher objective in environmental decision making (Daily et al., 2000; 2009).

- The need for better insights regarding **biodiversity effects** on ecosystem functions, which can be linked to ecosystem services provision.

This has been identified as a critical issue, and in estuarine systems, permanently submitted to natural and anthropogenic pressures (e.g. Elliott and McLusky, 2002; Marques et al., 2003; de Jonge, 2007), it is particularly important. Different authors consider that all biodiversity components (from genes to the community level) may play a significant role in the long-term supply of some ecosystem services (e.g. Turner et al., 2000; Balvanera et al., 2006; Carpenter et al., 2006; Díaz et al., 2006). For that reason, these authors claim that the principal social objective should focus on the maintenance of quality and quantity of biodiversity attributes, in order to guarantee a diversified flow of services (Martín-López et al., 2009), since thresholds are difficult to define and quantify (Muradian, 2001; Groffman et al., 2006; Horan et al., 2011).

- The need for clear **communication**.

These are of major importance, since methodological complexities do not automatically hinder decision making. Better insights into the ecosystem services benefits distribution might provide a platform to put valuation and ecological issues in a societal context, allowing better decision-making processes.

3. Ecosystem services provided by estuarine systems: Assessment, Valuation and Operationalization challenges

Despite possible conservation policies that might be implemented to protect the ecosystem, it must be taken into account that the adequacy of such policies depends on the impacts induced by the surrounding human populations (i.e. economy). Therefore, sustainable management can never be considered/treated isolated from the socio-economic system.

Applying the ecosystem services approach (ESA) to estuarine systems may provide a common language, and consequently promote communication, between groups with different interests. Synergies and trade-offs evaluations between ecological quality status (EQS – European WFD sense) (as a proxy for biodiversity and ecosystem functioning) and ecosystem services (as a proxy for human well-being) are fundamental to ensure a sustainable management of these ecosystems. In this sense, estimated social values are fundamental to offer a structured framework that can (or should) be used to explore social and ecosystem responses to different managing approaches. Although it is commonly argued that the ecosystem services approach may have a substantial role to ensure an accurate and balanced management of estuarine systems, the present work strongly supports the view that it should be used in combination with other complementary frameworks, such as spatial planning and multi-criteria analysis.

Six recommendations listed from this study should be interpreted as suggestions to improve the management and conservation of coastal ecosystems, with emphasis on estuaries (transitional waters in the WFD sense), using social and economic tools as an added-value in optimising ecosystems' quality and functioning. Suggestions are generic, in order to facilitate their application to the management of a wide set of systems, requiring adaptation when applied to specific sites:

a. Explicit definition of the ecosystem services' assessment spatial scale:

Define clearly the system's ecological condition (e.g. biodiversity indicators or water quality parameters), provision capacity (e.g. thresholds), catchment area uses (direct or indirect activities that may depend on or influence its condition), and assess of the main drivers, pressures and impacts, is essential to measure, value and manage natural resources.

b. Use of spatial models and analytical tools:

Development of spatial models (e.g. catchment area models that integrate water environmental conditions, but also the activities and pressures occurring in the surroundings) may represent an additional advantage to improve a systems' management.

c. Take advantage of multi-disciplinary knowledge:

Ecosystem services assessment involves the accurate and precise evaluation (qualitative, quantitative and monetary) of the ecological, economic, and social potential of natural resources. A social-ecological framework, complemented by monetary valuations, should therefore be developed / applied to each case study ecosystem, capturing its monetary and non-monetary values.

d. Understand how an ecosystem works and behaves under perturbation:

Research efforts are needed towards more integrative studies, namely with regard to:

- Measuring changes in biodiversity and in services provision;
- Relating clearly a given ecological quality status to a specific level of services provision in an ecosystem, to serve as a basis for economic valuation.

e. Be aware of the societal dependence on nature – giving special focus to the relationship between ecological rationality and biodiversity assets:

Accurate estimations are necessary to understand the intensity and amplitude of the links between biodiversity vs. ecosystem functioning vs. human well-being. This three-fold relation will finally determine the system condition, present uses, and its evenness for future generations. In this sense, it is necessary to adopt a clear methodology in order to guarantee an accurate (e)valuation of the system.

f. Utilise the ecosystem services approach as an 'added-value' for decision making:

Economic and social assessments can play a useful role in managing ecosystems, since they allow estimating the market and non-market values (benefits and costs) of natural resources. Such assessments may be combined with an efficient communication (stressing both the merits and shortcomings), and complemented by multi-criteria analysis (aiming at framing and systematising alternative outcomes), to assist decision making towards a more sustainable use of systems.

4. Open questions

The present work highlighted the importance of having an in-depth knowledge of the interplay among biodiversity assets, ecosystem functioning/processes, and human well-being (via ecosystem services provision). Current challenges covering this field are multi-fold, and four areas may be identified which demand further investigation to improve current research:

a. To expand current findings regarding the links between biodiversity and ecosystem functioning.

Two important areas were identified as requiring more empirical research: (1) linking communities of species to the ecosystem services they may provide; and (2) assessing the consequences of changes on a system' ecological condition in services provision. This could be achieved through field and laboratory experiments to simulate the ecosystem dynamics and its responses under perturbation. For example, the capacity and importance of seagrasses to retain carbon on the soil (C sequestration) is considered as an important ecological service to human well-being. To determine the rates at which this service can be provided (e.g. due to disturbances) may establish the link between ecological functioning and a service valued by society. In order to 'convince' society of the need of preserving natural systems and to draw its attention to their 'hidden' benefits, it is crucial to clearly demonstrate how the system works and how it can contribute to their welfare.

b. To estimate the carrying capacity and threshold limits of natural systems to provide valorised goods.

The development and application of mathematical models to simulate the natural dynamics and functioning of an ecosystem may allow predicting the maximum and minimum services provision under different scenarios (status-quo, under natural-or human-induced perturbations, etc.). This would allow the integration of ecological and socio-economic methodologies, and eventually to assess the effects of biodiversity on human well-being;

c. To develop models to calculate the aggregate value that water quality improvements have for populations, based on the surveys conducted.

Based on results achieved, with regard to water resources quality improvement, it was possible to estimate the total willingness to Pay (WTP) of the populations at both the Mondego basin and national levels. The next step will demand the calculation of the total value that this service has to populations (e.g. using econometric models to estimate aggregate values). However, aggregated values estimation should be done with caution, given that almost half of the respondents voted against the action plan implementation (mainly protests reasons). Therefore, it becomes essential to determine the role of protest classes to econometric models application to aquatic resources, and aggregate value estimations;

d. To infer the role of 'Good Ecological Status' (as demanded by the WFD) for services provision.

A clear and sound relationship between what is being measured by environmental authorities to evaluate systems' ecological quality status and ecosystem services provision capacity must be established. Moreover, it will be necessary to transfer research findings to the non-scientific community, namely to decision makers, and methodologies such as Multi-Criteria Analysis or Decision-Support Systems will have a major role with regard to ecosystems conservation and sustainable use.

5. Conclusions

In this thesis, the way existing drivers and pressures determine changes in ecosystems' ecological quality status, and consequently how ecosystem internal functioning is affected, was investigated focusing on ecosystem services provision on which human well-being relies. The Mondego river was selected as case study, allowing to improve our knowledge on the integrated functioning of estuarine ecosystems, namely regarding responses to environmental pressures induced by human activities. Different spatial scales were analysed, ranging from a river basin wide approach to specific habitat responses to perturbations (e.g. stability of intertidal ecosystems), considering both short- and long-term variations of communities and ecosystem services provision.

Results showed that the application of the ecosystem services approach can assist decision-makers in assessing alternative management scenarios, as well as their potential impacts on ecological conservation and/or socio-economic human well-being. Six main conclusions can be taken from this study:

1. Despite the efforts, there are still some obvious data and methodological limitations regarding ecosystem services valuation. The assessment carried out in this work illustrated nevertheless the pros (benefits) and cons (costs) of ecosystems conservation and sustained services provision considering different environmental conditions through time and different spatial scales.

2. Wetlands monitoring proved to be essential to assess ecosystems' ecological quality status, as well as to account for ecosystem services provision. In other words, to evaluate an ecosystem, functioning and services provision capacity monitoring is a key step, allowing to identify the pressures and the resulting impacts of those pressures, on both ecological condition and human well-being.

3. Understanding the role of ecosystems' functioning and services provision (resulting in human wellbeing) allows to make an accurate assessment of natural assets, and therefore a more efficient compliance with sustainable development requirements. The present work provided a better comprehension on how to couple different ecological and social components in a multidimensional picture, which may allow to better inform policy-makers and stakeholders about the studied ecosystem's characteristics and ecological and socio-economic importance, as well as about peoples' attitude regarding the sustainable use of natural resources;

4. Estuarine temporal stability maximum values were achieved for intermediate values of diversity and, in general, stability increased with species abundance. These results suggest that special attention must be given to temporal stability as a potentially useful indicator in the scope of estuarine ecosystems' management.

5. Estuarine productivity underpins ecosystem services provision. This is especially relevant with regard to goods usually consumed by human societies (such as bivalves, mostly cockles) or the capacity of estuaries to act as nursery grounds for many species. Indeed, the extension to which ecosystem functioning is maintained will determine the final outcomes that can be obtained from ecosystem services.

6. To meet sustainable development requirements in practice it will be necessary to develop integrated approaches that allow covering ecological and socio-economic aspects in sufficient detail (e.g. through the application of ecological network analysis at the habitat level), to guarantee the desired and required progress in valuing, weighing and deciding upon environmental management problems rising from human activities.

Although 'ecosystem services' does not constitute a novel concept (it has already more than one decade), the present work proposed a way of making it operative regarding its use as an 'added-value' for environmental managers and decision-makers. In the face of distinct ecological and socio-economic realities, the proposed approach aims at being used to provide guidelines regarding the sustainable use of estuarine ecosystems resources (Ch. 1 and 3).

In addition to a set of enhancing measures recommended to guarantee the balanced use of natural resources (Ch. 2 and 6), the present work provides an overview of possible future paths that could be followed to ensure a sustainable compromise between natural assets and human wellbeing (Chp. 4 and 5).

Finally, it is recognised that frequently the main problem regarding coastal/estuarine ecosystems management relies not so much in the provision of information in itself, but rather in the way it is presented to decision-makers. In this sense, the present work supports the view that mainstreaming ecosystem services, due to its integrative and broad conception nature, may have a substantial role to ensure an accurate and balanced management of these systems.

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Annex I

A. Mondego estuary survey (original form)
B. Mondego Basin survey (original form)

A. Mondego estuary survey

Bom dia! O meu nome é ____, e gostaria de saber se era possível fazer umas questões. Este inquérito faz parte de um projecto de investigação, que está a ser realizado pela Universidade de Coimbra e pelo IMAR – Instituto do Mar. Este inquérito irá demorar cerca de 10 minutos a preencher e todas as respostas serão confidenciais.

Secção A. Introdução

1. Gostaria de participar neste inquérito?

Participação			
Sim		Não	
É…?			
Homem		Mulher	

2. Será que nos podia indicar a sua faixa etária?

Faixa etária					
<20		41 – 50		61 – 70	
21 – 30		51 – 60		+81	
31 – 40		71 – 80			

3. Será que nos podia indicar as suas habilitações literárias?

Escolaridade					
Primária		Básico		Secundário	
Superior		Outra		Qual:	

4. Será que nos podia indicar o seu rendimento mensal líquido?

Ordenado					
<500 €		1001 – 2000 €		>3001 €	
501 – 1000 €		2001 – 3000 €			

5. Onde vive? Só pretendemos saber o concelho…

Morada	
Concelho	
Não desejo indicar	

6. Há quanto tempo está a viver nesse endereço?

Tempo					
Toda a minha vida		1 – 5 anos		Não sei/Não quero responder	
+ 5 anos		Menos de 1 ano			

Filtro 1: apenas se Q6 for diferente de 'toda a minha vida'

7. Onde morava antes de se mudar para a actual morada?

Morada	
Mesma área (código postal) que agora	
Outra área no Baixo Mondego	
Outra área incluída na Bacia do Mondego	
Outro	

Filtro 1: Terminado.

O objectivo principal deste inquérito são as actividades recreativas desenvolvidas nas zonas aquáticas e suas proximidades – isto é, rios, ribeiros, lagos, costa, estuário. Estamos interessados na SUA OPINIÃO/EXPERIÊNCIA PESSOAL e a sua resposta será completamente anónima.

Vamos agora colocar algumas questões acerca dos seus hábitos de actividades recreativas.

8. Com que frequência usa áreas da natureza para recreação – tais como viagens ao longo da costa, do estuário, do rio – ou no campo?

Visito áreas naturais	
Menos de uma vez por ano	
Uma vez por ano	
A cada 6 meses (2 vezes por ano)	
A cada 3 meses (4 vezes por ano)	
Cada mes (12 vezes por ano)	

A cada 2 semanas (25 vezes por ano)	
A cada semana (50 vezes por ano)	
Duas vezes por semana (100 vezes por ano)	
Mais de 2 vezes por semana (200 vezes por ano)	
Todos os dias (365 vezes por ano)	
Nunca	
Não sei/não quero responder	

Filtro 2: se Q8=Nunca então não responder às Q9 e Q10.

9. **Com que frequência utiliza rios e estuários – e as áreas à sua volta – para fins de recreação? Isto é, com que frequência (aproximada) caminha ou corre ao longo de uma zona ribeirinha ou pesca, caça, faz campismo, faz observação de aves/animais?**

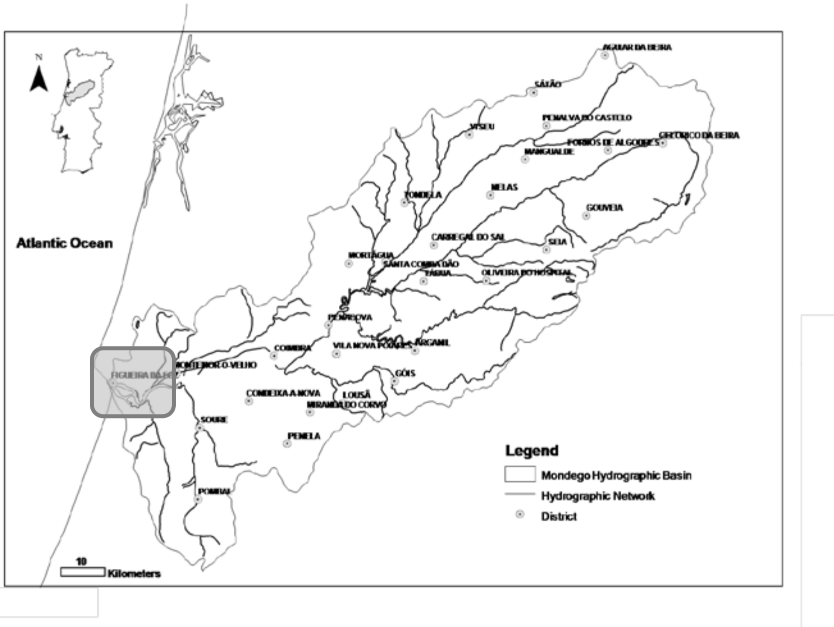
Visito rios/estuários e as suas áreas adjacentes	
Uma vez por ano	
A cada 6 meses (2 vezes por ano)	
A cada 3 meses (4 vezes por ano)	
Cada mes (12 vezes por ano)	
A cada 2 semanas (25 vezes por ano)	
A cada semana (50 vezes por ano)	
Duas vezes por semana (100 vezes por ano)	
Mais de 2 vezes por semana (200 vezes por ano)	
Todos os dias (365 vezes por ano)	
Meses de Verão	
Não sei/não quero responder	

10. **Com que frequência faz as actividades descritas em baixo quando visita um rio ou estuário?**
(coloque uma cruz em cada linha)

	Frequentemente (quase todas as vezes que visito um rio/estuário)	**As vezes** (aproximadamente metade das vezes que visito um rio/estuário)	**Raramente**	**Nunca**
Viagens ao longo da margem do rio/estuário, tais como caminhadas, ciclismo, alimentar peixes/patos, …				
Navegação não-motorizada (como canoagem, remo…)				
Navegação motorizada				
Banhos, natação, …				
Pesca desportiva				
Experiências de natureza, observação de aves/animais,…				
Passear o cão				
Outro				

Secção B. Descrição do bem a valorizar (qualidade da agua e ambiente do Estuário do Mondego)

Nas próximas questões gostaríamos de lhe perguntar a sua opinião acerca de propostas para melhorar o ambiente aquático na zona da Bacia do Mondego, mais especificamente na zona do estuário do Mondego. A parte que é sugerida para melhorar tem cerca de 21km de extensão de um total de 300km (correspondentes ao total da bacia). O mapa em baixo mostra a bacia do Mondego e a área a melhorar.



Assim as próximas questões referem-se à qualidade da água no rio Mondego e mais concretamente no estuário do Mondego. A intenção é avaliar se acha que o estuário do Mondego está ou não poluído na sua opinião, e avaliar as respostas que poderiam ser dadas para reduzir essa poluição. *A qualidade da água num rio/estuário significa algo para a experiencia que as pessoas podem retirar no seu contacto com a natureza. A qualidade da água também afecta outras actividades, assim como as condições de vida para plantas e animais que dependem dos rios/estuários.*

11. O estuário do Mondego está…

Nível poluição	
Muito poluído	
Poluído	
Pouco poluído	
Não está poluído	
Não sei/não quero responder	

11a. Acha que tem vindo a…

Melhorar	
Piorar	
Igual	
Não sei	

11b. Se acha que o estuário do Mondego está poluído, quais são as consequências dessa poluição?
(a poluição torna a água:)

Efeitos poluição	**Sim**	**Não**	**Não sei**
Com lodo/limo			
Com cheiros desagradáveis			
Imprópria para usar para fins recreativos/consumo			
Cor diferente			
Outro:			

12. A poluição nos estuários/costa resulta de várias fontes de poluição. Qual destas fontes acredita que é a principal causadora da poluição no estuário do Mondego?
(colocar uma cruz)

Fontes de poluição:	
Campos agrícolas	
Zonas industriais	
Efluentes domésticos	
Aquacultura	
Actividades recreativas/turismo	
Não sei/Não quero responder	

13. **Como classifica cada um destes factores em relação ao seu impacto na qualidade da água e ambiente do estuário do Mondego?**
(colocar uma cruz)

Causa	Não é importante	Moderado	Importante	Muito importante	Não sei
Pesticidas e fertilizantes que vem dos campos agrícolas					
Efluentes de indústrias					
Efluentes domésticos					
Poluição causada pelo porto de pesca					
Poluição causada pelos barcos de recreação					
Erosão das margens do rio					
Poluição das aquaculturas					
Outros (qual:)					

14. **Ouviu falar em restrições que limitam ou impedem a apanha e consumo de mexilhões ou peixes vindos do estuário do Mondego, devido á poluição?**

Restrições	
Sim	
Não	
Não sei	

15. **Você ou alguém no seu agregado familiar paga licença para usar os meios recreativos do estuário?**

Licenças	
Marina	
Pesca	
Canoagem	
Outro	
Não	

<u>Secção C. Disponibilidade a pagar e descrição de cenários de melhoramento de qualidade</u>

Agora vamos pedir-lhe que imagine que as autoridades ambientais sugeriam melhorar a qualidade da água na zona do estuário do Mondego. Para tal, seria necessário saber se a população acha estas propostas significativas, e se está disposta a contribuir para a sua realização e manutenção.

16. **Será que nos podia indicar o valor actual da conta da água que paga <u>mensalmente</u>?**
Por favor, coloque uma cruz junto do respectivo valor.

Valor mensal conta da água							
0 €		50 – 60 €		120 - 130 €		Outro, qual:	
0,50 – 5 €		60 – 70 €		130 - 140 €			
5 – 10 €		70 – 80 €		140 – 150 €		Não sei	
10 – 20 €		80 – 90 €		150 €			
20 – 30 €		90 - 100 €		200 €			
30 – 40 €		100 - 110 €		> 200 €			
40 – 50 €		110 - 120 €					

Na secção seguinte vai-lhe ser pedido para avaliar várias propostas mais detalhadas que visam a mudança de qualidade e os potenciais usos do estuário do Mondego. Vamos-lhe pedir para **escolher entre as diferentes alternativas propostas para melhorar a qualidade da água no estuário do Mondego.** *A cada nível de qualidade corresponde uma letra. A letra 'A' corresponde à melhor qualidade possível no rio/estuário, 'B', 'C', 'D' e 'E' indicam uma qualidade decrescente.* O objectivo principal deste estudo é avaliar como o estuário poderia melhorar a sua qualidade, por exemplo através de uma redução do limo, aumento da transparência da água ou aumento de peixes para a pesca.

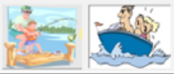
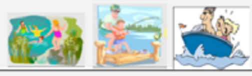
17. **Gostaria de ver implementado um plano para melhorar a qualidade ambiental do estuário?**

Plano	
Sim	
Não	
Não sei	

Filtro: se respondeu Sim na Q17, então responder Q18.

18. A qualidade ambiental do estuário neste momento é:

- a actual qualidade da água é moderada, com boas condições para a pesca, embora tenha acessos restritos e seja rodeada por campos agrícolas. Se lhe propusessem medidas para:

Característica	Condição actual	Alternativa 1
Qualidade da água	C 	B 
Pesca	Bom	Melhorada
Acessibilidade	Restritos	Bons
Áreas circundantes	Campos agrícolas	Corredores verdes
Eu prefiro: (coloque uma cruz na opção pretendida)		

- melhorar ligeiramente a qualidade da água (por exemplo, torna-la própria para banhos) e do ambiente (como através de uma diminuição do limo presente), com melhores condições para a pesca, bons acessos e rodeada por zonas verdes…

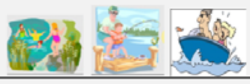
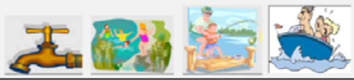
… estaria disposto a pagar *(sob a forma de taxa)* para garantir que as medidas para atingir essas melhorias seriam de facto implementadas e mantidas nos anos seguintes? Este valor seria adicionado à sua conta da água actual, mas seria necessário ter em consciência que estes processos são longos e não têm nada a ver com a qualidade da água que consome em casa. Tenha em atenção que o seu agregado familiar teria de pagar esta quantia todos os anos. **(colocar uma cruz)**

Quantia							
0 €		50 – 60 €		120 - 130 €		Outro, qual:	
0,50 – 5 €		60 – 70 €		130 - 140 €		(e.g. %)	
5 – 10 €		70 – 80 €		140 – 150 €		Não sei	
10 – 20 €		80 – 90 €		150 €			
20 – 30 €		90 - 100 €		200 €			
30 – 40 €		100 - 110 €		> 200 €			
40 – 50 €		110 - 120 €					
Não estou interessado em pagar, mantendo a **condição actual**							

(***se aceitar pagar para a alternativa 1, continuar para alternativa 2)***

E se lhe fossem propostas mais medidas que ainda iriam melhorar mais a qualidade do estuário, como:

- melhorar significativamente a qualidade da água (por exemplo aumentar ainda mais a sua transparência) e do ambiente (como por exemplo, ter boas condições para ter colónias de flamingos), com melhores condições para a pesca, bons acessos e rodeada por zonas verdes:

Característica	Alternativa 1	Alternativa 2
Qualidade da água	B 	A 
Pesca	Melhorada	Melhorada
Acessibilidade	Bons	Bons
Áreas circundantes	Corredores verdes	Corredores verdes
Eu prefiro: (coloque uma cruz na opção pretendida)		

Quanto estaria disposto a pagar para garantir que a alternativa escolhida seria de facto implementada? Se escolher esta alternativa este será o único valor a pagar.

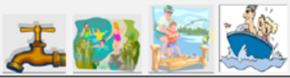
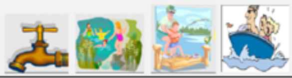
(colocar uma cruz – o valor tem de ser superior ao da pergunta anterior)

Quantia							
0 €		50 – 60 €		120 - 130 €		Outro, qual:	
0,50 – 5 €		60 – 70 €		130 - 140 €		(e.g. %)	
5 – 10 €		70 – 80 €		140 – 150 €		Não sei	
10 – 20 €		80 – 90 €		150 €			
20 – 30 €		90 - 100 €		200 €			
30 – 40 €		100 - 110 €		> 200 €			
40 – 50 €		110 - 120 €					
Não estou interessado em pagar, mantendo a **condição actual**							

(se aceitar pagar para a alternativa 2, continuar para alternativa 3)

E se lhe fossem propostas mais medidas que ainda iriam melhorar mais a qualidade do estuário, como:

- melhorar significativamente a qualidade da água e do ambiente (semelhantes ás anteriores), com melhores condições para a pesca, muito bons acessos e rodeada por zonas verdes e incluindo a criação de um centro de ecoturismo que atrairia muitos turistas, pagando para tal uma taxa anual.

Característica	Alternativa 2	Alternativa 3
Qualidade da água	A 	A 
Pesca	Melhorada	Melhorada
Acessibilidade	Bons	Muito bons
Áreas circundantes	Corredores verdes	Corredores verdes Centro de eco-turismo no estuário
Eu prefiro: (coloque uma cruz na opção pretendida)		

Quanto estaria disposto a pagar para garantir que a alternativa escolhida seria de facto implementada? Se escolher esta alternativa este será o único valor a pagar.
(colocar uma cruz – o valor tem de ser superior ao da pergunta anterior)

Quantia							
0 €		50 – 60 €		120 - 130 €		Outro, qual:	
0,50 – 5 €		60 – 70 €		130 - 140 €		(e.g. %)	
5 – 10 €		70 – 80 €		140 – 150 €		Não sei	
10 – 20 €		80 – 90 €		150 €			
20 – 30 €		90 - 100 €		200 €			
30 – 40 €		100 - 110 €		> 200 €			
40 – 50 €		110 - 120 €					
Não estou interessado em pagar, mantendo a **condição actual**							

19. Quantas pessoas compõem o seu agregado familiar?
(colocar uma cruz)

Número de membros	
1	
2	
3	
4	
5	
6	
+7	

20. Para quem respondeu <u>Não na Q18</u>: Porque escolheu manter a condição actual na questão anterior? Motivo:
(colocar uma cruz)

	1ª razão	2ª razão
O aumento na conta da água é muito elevado comparativamente às melhorias descritas para a qualidade da água		
O rio/estuário fica muito longe do meu local de residência		
Eu não uso o rio/estuário do Mondego		
A qualidade actual é suficiente		
Não acredito que a qualidade da água melhore da forma que foi descrita		
Preferia pagar para melhorar a qualidade de outro rio/estuário		
Preferia usar o meu dinheiro em outras coisas		
Não posso suportar um aumento na minha conta da água actual		
Os que usam o rio/estuário do Mondego é que deveriam suportar os custos da melhoria da qualidade		
As companhias de águas é que deveriam suportar as melhorias de qualidade		

O governo é que deveria pagar por estas melhorias		
As contas da água já são demasiado elevadas actualmente		
A questão e muito difícil de responder		
Outro		

21. **Para quem <u>escolheu alguma das alternativas na Q18</u>: Porque está o seu agregado familiar disposto a pagar mais para uma melhoria da qualidade de água no estuário do Mondego? Motivo:** (colocar uma cruz)

	1ª razão	2ª razão
A melhoria será boa e valorizada por mim e pelo meu agregado familiar		
Estou interessado nestas melhorias, não importa os custos		
Estou interessado nas vantagens que todos irão ganhar com estas melhorias		
Todas as pessoas deveriam experienciar rios/estuários de elevada qualidade, independentemente da sua própria opinião do que seria o melhor		
Quero contribuir para a protecção do ambiente aquático para bem das plantas e animais que lá habitam		
Moralmente senti que esta era a resposta correcta		
Não entendi a questão		
Outro		

22. **Foi-lhe difícil definir a quantia com que podia contribuir?**

Muito difícil	
Difícil	
Razoável	
Fácil	
Muito fácil	

23. **Foi-lhe difícil decidir entre as 3 alternativas?**

Muito difícil	
Difícil	
Razoável	
Fácil	
Muito fácil	

24. **Que possibilidade é que acha que há de melhorar a qualidade da água e do ambiente?**

De certeza	
Provavelmente	
Dificilmente	

25. **Até que ponto os seguintes factores influenciaram a sua escolha na questão em que se consideravam as várias alternativas para o estuário do Mondego?** (colocar uma cruz)

	Muito	Razoavelmente	Pouco	Nada
A melhoria da qualidade da água em geral				
A melhoria da qualidade da água no estuário do Mondego				
A melhoria nas facilidades de recreação no estuário do Mondego				
A melhoria das condições de qualidade para os animais e plantas do estuário				
A distância da minha residência ate ao estuário				
O aumento extra na minha conta da água (preço)				

26. **Conhecia o estuário do Mondego antes de responder a este questionário – e usava o sistema?** (colocar uma cruz)

Não, nunca tinha ouvido falar no estuário do Mondego	
Sim, já tinha ouvido falar, mas nunca fui ao estuário do Mondego	
Sim, conheço o estuário mas não o utilizei no ano passado	
Sim, conheço o estuário e fui lá 1 ou 2 vezes o ano passado	
Sim, conheço o estuário e utilizo-o frequentemente para recreação (+ de uma vez/mês)	

27. Agora gostaríamos de saber que rios/estuários costuma visitar?

Rios/estuários	
Costuma visitar rios?	
Quais:	
Costuma visitar estuários?	
Quais:	

28. Gostaria de deixar algum comentário/sugestão para melhorar a qualidade da água e do ambiente nesta região?
____
____

Muito obrigada pela sua participação!

B. Mondego Basin survey

Bom dia! O meu nome é ____, e gostaria de saber se era possível fazer umas questões. Este inquérito faz parte de um projecto de investigação, que está a ser realizado pela Universidade de Coimbra e pelo IMAR – Instituto do Mar. Este inquérito irá demorar cerca de 10 minutos a preencher e todas as respostas serão confidenciais.

Secção A. Introdução

1. Gostaria de participar neste inquérito?

Participação			
Sim		Não	
É…?			
Homem		Mulher	

2. Será que nos podia indicar a sua faixa etária?

Faixa etária					
<20		41 – 50		61 – 70	
21 – 30		51 – 60		+81	
31 – 40		71 – 80			

3. Será que nos podia indicar as suas habilitações literárias?

Escolaridade					
Primária		Básico		Secundário	
Superior		Outra		Qual:	

4. Será que nos podia indicar o seu rendimento mensal líquido?

Ordenado					
<500 €		1001 – 2000 €		>3001 €	
501 – 1000 €		2001 – 3000 €			

5. Onde vive? Só pretendemos saber o concelho…

Morada	
Concelho	
Não desejo indicar	

6. Há quanto tempo está a viver nesse endereço?

Tempo					
Toda a minha vida		1 – 5 anos		Não sei/Não quero responder	
+ 5 anos		Menos de 1 ano			

Filtro 1: apenas se Q6 for diferente de 'toda a minha vida'

7. Onde morava antes de se mudar para a actual morada?

Morada	
Mesma área (código postal) que agora	
Outra área no Baixo Mondego	
Outra área incluída na Bacia do Mondego	
Outro	

Filtro 1: Terminado.

O objectivo principal deste inquérito são as actividades recreativas desenvolvidas nas zonas aquáticas e suas proximidades – isto é, rios, ribeiros, lagos, costa, estuário. Estamos interessados na SUA OPINIÃO/EXPERIÊNCIA PESSOAL e a sua resposta será completamente anónima.

Vamos agora colocar algumas questões acerca dos seus hábitos de actividades recreativas.

8. **Com que frequência usa áreas da natureza para recreação – tais como viagens ao longo da costa, do estuário, do rio – ou no campo?**

Visito áreas naturais	
Menos de uma vez por ano	
Uma vez por ano	
A cada 6 meses (2 vezes por ano)	
A cada 3 meses (4 vezes por ano)	
Cada mes (12 vezes por ano)	
A cada 2 semanas (25 vezes por ano)	
A cada semana (50 vezes por ano)	
Duas vezes por semana (100 vezes por ano)	
Mais de 2 vezes por semana (200 vezes por ano)	
Todos os dias (365 vezes por ano)	
Nunca	
Não sei/não quero responder	

Filtro 2: se **Q8=Nunca** então **não** responder às Q9 e Q10.

9. **Com que frequência utiliza rios e estuários – e as áreas à sua volta – para fins de recreação? Isto é, com que frequência (aproximada) caminha ou corre ao longo de uma zona ribeirinha ou pesca, caça, faz campismo, faz observação de aves/animais?**

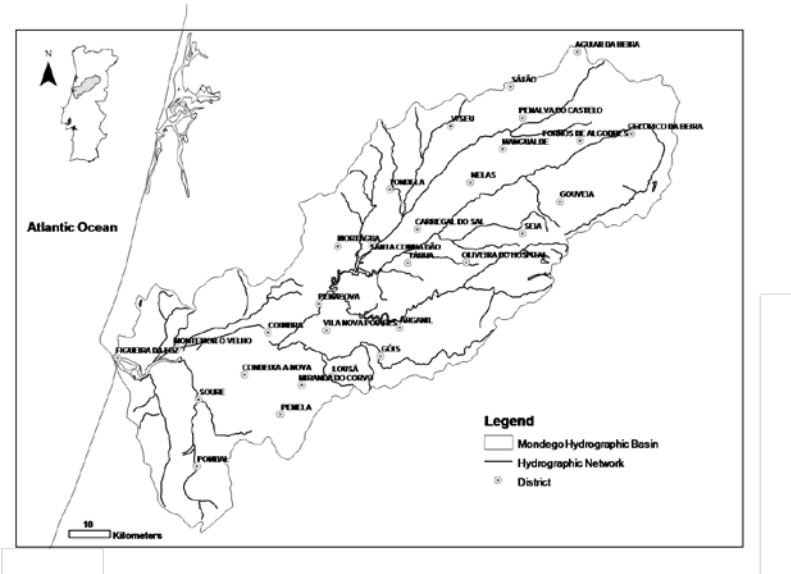
Visito rios/estuários e as suas áreas adjacentes	
Uma vez por ano	
A cada 6 meses (2 vezes por ano)	
A cada 3 meses (4 vezes por ano)	
Cada mes (12 vezes por ano)	
A cada 2 semanas (25 vezes por ano)	
A cada semana (50 vezes por ano)	
Duas vezes por semana (100 vezes por ano)	
Mais de 2 vezes por semana (200 vezes por ano)	
Todos os dias (365 vezes por ano)	
Meses de Verão	
Não sei/não quero responder	

10. **Com que frequência faz as actividades descritas em baixo quando visita um rio ou estuário?** (coloque uma cruz em cada linha)

	Frequentemente (quase todas as vezes que visito um rio/estuário)	**As vezes** (aproximadamente metade das vezes que visito um rio/estuário)	**Raramente**	**Nunca**
Viagens ao longo da margem do rio/estuário, tais como caminhadas, ciclismo, alimentar peixes/patos, …				
Navegação não-motorizada (como canoagem, remo…)				
Navegação motorizada				
Banhos, natação, …				
Pesca desportiva				
Experiências de natureza, observação de aves/animais,…				
Passear o cão				
Outro				

<u>Secção B. Descrição do bem a valorizar (qualidade da água e ambiente da Bacia do Mondego)</u>

Nas próximas questões gostaríamos de lhe perguntar a sua opinião acerca de propostas para melhorar o ambiente aquático na zona da Bacia do Mondego. O intuito seria melhorar a qualidade da água (por exemplo torna-la mais transparente) e do ambiente (por exemplo, com mais peixes) no Rio Mondego ao longo de cerca de 300km (correspondentes ao total da bacia). O mapa em baixo mostra a bacia do Mondego e a área a melhorar.



Assim as próximas questões referem-se à qualidade da água no rio Mondego. A intenção é avaliar se acha que o Rio Mondego está ou não poluído na sua opinião, e avaliar as respostas que poderiam ser dadas para reduzir essa poluição. *A qualidade da água num rio/estuário significa algo para a experiência que as pessoas podem retirar no seu contacto com a natureza. A qualidade da água também afecta outras actividades, assim como as condições de vida para plantas e animais que dependem dos rios/estuários.*

11. O Rio Mondego está…

Nível poluição	
Muito poluído	
Poluído	
Pouco poluído	
Não está poluído	
Não sei/não quero responder	

11a. Acha que tem vindo a…

Melhorar	
Piorar	
Igual	
Não sei	

11b. Se acha que o Rio Mondego está poluído, quais são as consequências dessa poluição?
(a poluição torna a água:)

Efeitos poluição	**Sim**	**Não**	**Não sei**
Com lodo/limo			
Com cheiros desagradáveis			
Imprópria para usar para fins recreativos/consumo			
Pouca diversidade			
Cor diferente			
Outro:			

12. A poluição nos rios/zonas costeiras resulta de várias fontes de poluição. Qual destas fontes acredita que é a principal causadora da poluição na Bacia do Mondego?
(colocar uma cruz) (alterar ordem)

Fontes de poluição:	
Campos agrícolas	
Zonas industriais	
Efluentes domésticos	
Aquacultura	
Actividades recreativas/turismo	
Não sei/Não quero responder	

13. **Como classifica cada um destes factores em relação ao seu impacto na qualidade da água e ambiente do Rio Mondego?**
(colocar uma cruz)

Causa	Não é importante	Moderado	Importante	Muito importante	Não sei
Pesticidas e fertilizantes que vem dos campos agrícolas					
Efluentes de indústrias					
Efluentes domésticos					
Poluição causada pelo porto de pesca					
Poluição causada pelos barcos de recreação					
Erosão das margens do rio					
Poluição das aquaculturas					
Outros (qual:)					

14. **Ouviu falar em restrições que limitam ou impedem a apanha e consumo de mexilhões ou peixes vindos do estuário do Mondego, devido á poluição?**

Restrições	
Sim	
Não	
Não sei	

15. **Você ou alguém no seu agregado familiar paga licença para usar os meios recreativos do estuário?**

Licenças	
Marina	
Pesca	
Canoagem	
Outro	
Não	

<u>Secção C. Disponibilidade a pagar e descrição de cenários de melhoramento de qualidade</u>

Agora vamos pedir-lhe que imagine que as autoridades ambientais sugeriam melhorar a qualidade da água ao longo da Bacia do Mondego. Para tal, seria necessário saber se a população acha estas propostas significativas, e se está disposta a contribuir para a sua realização e manutenção.

16. **Será que nos podia indicar o valor actual da conta da água que paga <u>mensalmente</u>?**

Por favor, coloque uma cruz junto do respectivo valor.

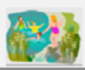
Valor mensal conta da água							
0 €		50 – 60 €		120 - 130 €		Outro, qual:	
0,50 – 5 €		60 – 70 €		130 - 140 €			
5 – 10 €		70 – 80 €		140 – 150 €		Não sei	
10 – 20 €		80 – 90 €		150 €			
20 – 30 €		90 - 100 €		200 €			
30 – 40 €		100 - 110 €		> 200 €			
40 – 50 €		110 - 120 €					

Na secção seguinte vai-lhe ser pedido para avaliar várias propostas mais detalhadas que visam a mudança de qualidade e os potenciais usos da bacia do Mondego. Vamos-lhe pedir para **escolher entre as diferentes alternativas propostas para melhorar a qualidade da água no Rio Mondego.** *A cada nível de qualidade corresponde uma letra. A letra 'A' corresponde à melhor qualidade possível no rio, 'B', 'C', 'D' e 'E' indicam uma qualidade decrescente.*
O objectivo principal deste estudo é avaliar como o rio poderia melhorar a sua qualidade, por exemplo através de uma redução do limo, aumento da transparência da água ou aumento de peixes para a pesca.

17. Gostaria de ver implementado um plano para melhorar a qualidade ambiental no rio?

Plano	
Sim	
Não	
Não sei	

Filtro: se respondeu Sim na Q17, então responder Q18.
18. A qualidade ambiental do rio Mondego neste momento é:
- a actual qualidade da água é moderada, com boas condições para a pesca, embora tenha acessos restritos e seja rodeada por campos agrícolas.
Se lhe propusessem medidas para:

Característica	Condição actual	Alternativa 1
Qualidade da água	C  	B   
Pesca	Bom	Melhorada
Acessibilidade	Restritos	Bons
Áreas circundantes	Campos agrícolas	Corredores verdes
Eu prefiro: (coloque uma cruz na opção pretendida)		

- melhorar ligeiramente a qualidade da água (por exemplo, torna-la própria para banhos) e do ambiente (como através de uma diminuição do limo presente), com melhores condições para a pesca, bons acessos e rodeada por zonas verdes…
… estaria disposto a pagar *(sob a forma de taxa)* para garantir que as medidas para atingir essas melhorias seriam de facto implementadas e mantidas nos anos seguintes? Este valor seria adicionado à sua conta da água actual, mas seria necessário ter em consciência que estes processos são longos e não têm nada a ver com a qualidade da água que consome em casa. Tenha em atenção que o seu agregado familiar teria de pagar esta quantia todos os anos. **(colocar uma cruz)**

Quantia							
0 €		50 – 60 €		120 - 130 €		Outro, qual:	
0,50 – 5 €		60 – 70 €		130 - 140 €		(e.g. %)	
5 – 10 €		70 – 80 €		140 – 150 €		Não sei	
10 – 20 €		80 – 90 €		150 €			
20 – 30 €		90 - 100 €		200 €			
30 – 40 €		100 - 110 €		> 200 €			
40 – 50 €		110 - 120 €					
Não estou interessado em pagar, mantendo a **condição actual**							

(se aceitar pagar para a alternativa 1, continuar para alternativa 2)

E se lhe fossem propostas mais medidas que ainda iriam melhorar mais a qualidade do rio, como:

- melhorar significativamente a qualidade da água (por exemplo aumentar ainda mais a sua transparência) e do ambiente (como por exemplo, ter boas condições para ter colónias de flamingos ou outras aves), com melhores condições para a pesca, bons acessos e rodeada por zonas verdes:

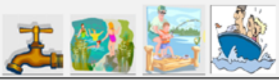
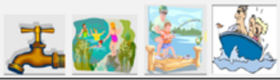
Característica	Alternativa 1	Alternativa 2
Qualidade da água	**B**   	**A**    
Pesca	Melhorada	Melhorada
Acessibilidade	Bons	Bons
Áreas circundantes	Corredores verdes	Corredores verdes
Eu prefiro: (coloque uma cruz na opção pretendida)		

Quanto estaria disposto a pagar para garantir que a alternativa escolhida seria de facto implementada? Se escolher esta alternativa este será o único valor a pagar.
(colocar uma cruz – o valor tem de ser superior ao da pergunta anterior)

Quantia							
0 €		50 – 60 €		120 - 130 €		Outro, qual:	
0,50 – 5 €		60 – 70 €		130 - 140 €		(e.g. %)	
5 – 10 €		70 – 80 €		140 – 150 €		Não sei	
10 – 20 €		80 – 90 €		150 €			
20 – 30 €		90 - 100 €		200 €			
30 – 40 €		100 - 110 €		> 200 €			
40 – 50 €		110 - 120 €					
Não estou interessado em pagar, mantendo a **condição actual**							

(se aceitar pagar para a alternativa 2, continuar para alternativa 3)

E se lhe fossem propostas mais medidas que ainda iriam melhorar mais a qualidade do rio, como:
- melhorar significativamente a qualidade da água e do ambiente (semelhantes ás anteriores), com melhores condições para a pesca, muito bons acessos e rodeada por zonas verdes e incluindo a criação de um centro de ecoturismo ou outros centros turísticos relacionados com a natureza (como parques) que atrairia muitos turistas, pagando para tal uma taxa anual.

Característica	Alternativa 2	Alternativa 3
Qualidade da água	A 	A 
Pesca	Melhorada	Melhorada
Acessibilidade	Bons	Muito bons
Áreas circundantes	Corredores verdes	Corredores verdes Centro de eco-turismo no estuário
Eu prefiro: (coloque uma cruz na opção pretendida)		

Quanto estaria disposto a pagar para garantir que a alternativa escolhida seria de facto implementada? Se escolher esta alternativa este será o único valor a pagar.
(colocar uma cruz – o valor tem de ser superior ao da pergunta anterior)

Quantia							
0 €		50 – 60 €		120 - 130 €		Outro, qual:	
0,50 – 5 €		60 – 70 €		130 - 140 €		(e.g. %)	
5 – 10 €		70 – 80 €		140 – 150 €		Não sei	
10 – 20 €		80 – 90 €		150 €			
20 – 30 €		90 - 100 €		200 €			
30 – 40 €		100 - 110 €		> 200 €			
40 – 50 €		110 - 120 €					
Não estou interessado em pagar, mantendo a **condição actual**							

19.Quantas pessoas compõem o seu agregado familiar?
(colocar uma cruz)

Número de membros	
1	
2	
3	
4	
5	
6	
+7	

20. Para quem respondeu Não na Q18: Porque escolheu manter a condição actual na questão anterior? Motivo:
(colocar uma cruz)

	1ª razão	2ª razão
O aumento na conta da água é muito elevado comparativamente às melhorias descritas para a qualidade da água		
O rio fica muito longe do meu local de residência		
Eu não uso o rio do Mondego		
A qualidade actual é suficiente		
Não acredito que a qualidade da água melhore da forma que foi descrita		
Preferia pagar para melhorar a qualidade de outro rio/estuário		
Preferia usar o meu dinheiro em outras coisas		
Não posso suportar um aumento na minha conta da água actual		
Os que usam o rio/estuário do Mondego é que deveriam suportar os custos da melhoria da qualidade		
As companhias de águas é que deveriam suportar as melhorias de qualidade		
O governo é que deveria pagar por estas melhorias		
As contas da água já são demasiado elevadas actualmente		
A questão e muito difícil de responder		
Outro		

21. Para quem <u>escolheu alguma das alternativas na Q18</u>: Porque está o seu agregado familiar disposto a pagar mais para uma melhoria da qualidade de água no Rio Mondego? Motivo:
(colocar uma cruz)

	1ª razão	2ª razão
A melhoria será boa e valorizada por mim e pelo meu agregado familiar		
Estou interessado nestas melhorias, não importa os custos		
Estou interessado nas vantagens que todos irão ganhar com estas melhorias		
Todas as pessoas deveriam experienciar rios/estuários de elevada qualidade, independentemente da sua própria opinião do que seria o melhor		
Quero contribuir para a protecção do ambiente aquático para bem das plantas e animais que lá habitam		
Moralmente senti que esta era a resposta correcta		
Não entendi a questão		
Outro		

22. Foi-lhe difícil definir a quantia com que podia contribuir?

Muito difícil	
Difícil	
Razoável	
Fácil	
Muito fácil	

23. Foi-lhe difícil decidir entre as 3 alternativas?

Muito difícil	
Difícil	
Razoável	
Fácil	
Muito fácil	

24. Que possibilidade é que acha que há de melhorar a qualidade da água e do ambiente?

De certeza	
Provavelmente	
Dificilmente	

25. Até que ponto os seguintes factores influenciaram a sua escolha na questão em que se consideravam as várias alternativas para o Rio Mondego?
(colocar uma cruz)

	Muito	Razoavelmente	Pouco	Nada
A melhoria da qualidade da água em geral				
A melhoria da qualidade da água no Rio Mondego				
A melhoria nas facilidades de recreação no estuário do Mondego				
A melhoria das condições de qualidade para os animais e plantas do rio				
A distância da minha residência ate ao rio				
O aumento extra na minha conta da água (preço)				

26. Conhecia o Rio do Mondego antes de responder a este questionário – e usava o sistema?
(colocar uma cruz)

Não, nunca tinha ouvido falar no Rio Mondego	
Sim, já tinha ouvido falar, mas nunca fui ao Rio Mondego	
Sim, conheço o rio mas não o utilizei no ano passado	
Sim, conheço o rio e fui lá 1 ou 2 vezes o ano passado	
Sim, conheço o rio e utilizo-o frequentemente para recreação (+ de uma vez/mês)	

27. Agora gostaríamos de saber que rios/estuários costuma visitar?

Rios/estuários	
Costuma visitar rios?	
Quais:	
Costuma visitar estuários?	
Quais:	

28. Gostaria de deixar algum comentário/sugestão para melhorar a qualidade da água e do ambiente nesta região?

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Muito obrigada pela sua participação!