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Valuation of Marine and Coastal Ecosystems: The Role of Ecological-Economic Modeling[1]

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Abstract

The main objective of this work project is to highlight the progress made in the field of ecological-economic modeling of marine and coastal ecosystems, in particular, by stressing the need to incorporate more realistic biology as well as the spatial dimension in integrated models for sustainable coastal management. The discussion undertaken is based on a recent application of an integrated ecological-economic model that is spatially explicit by Altman et al. (2012), and should provide guidance to the GOI's research project to the Peniche-Nazaré study site in the Portuguese coast.

Keywords: Ecological-Economic Modeling, Marine and Coastal Resources, Non-Linearity, Non-Convexities, Slow and Fast Variables

1. Introduction

An estimated two-thirds of the ecosystem services that make up the planet's natural capital are derived from ocean and coastal biomes (TEEB 2012). Demographic trends suggest that these coastal systems are experiencing growing population and exploitation pressures with nearly 40% of the world population living within 100 kilometres of the coast. Moreover, there is a tendency of coastal communities to concentrate near the types of coastal systems that are most abundant in the provisioning of ecosystem services, which are likewise believed to be the most vulnerable (Hassan et al. 2005). Yet, there are large gaps in our understanding of the structure of these ecosystems, their processes and the linkages between them. At the same time, we expect from policy makers to move forward with management and governance of coastal and marine ecosystems, to find solutions to the current demographic and other pressures on these biomes, and to anticipate future threats. Hence, there is a strong demand for instruments and indicators that can assist coastal management in a more holistic approach, focusing on the range of benefits we derive from complex marine and coastal systems, rather than on individual benefits, and recognizing the wide range of interactions within these coastal ecosystems, including human behaviour. Hence, an ecosystem-based management (EBM) approach is required.

With an Economic Exclusive Zone (EEZ) of 1 700 000 km², the 10th largest EEZ in the world, and an interface with the sea along 1793 km of coastline of which 976 km in mainland Portugal coastal areas play a central role in the Portuguese economy and livelihood of its population (Pereira et al. 2004). However, like many other coastal areas, the service provision by Portuguese marine and coastal systems is under threat. Following Antunes & Santos (1998), the five main threats to the Portuguese coast are: overfishing, contamination, destruction of ecosystems, territory loss and climate change. Above all, the Portuguese coastal vulnerability to erosion has been of great concern. An estimated 25% of the Portuguese mainland coast is very prone to erosion (Andrade et al. 2002), aggravated by the sea level rise and increased storm surge frequencies, as well as a reduction in deposits of sediment caused by inland dam constructions (Coelho et al. 2009). Besides, another emerging problem on the Portuguese coastline is plastic debris accumulation and its ability to absorb persistent organic pollutants which are potentially dangerous to marine species due to magnification risk over the food chain (Frias et al. 2010, Martins & Sobral 2011).

To face these threats, Portugal amongst other coastal countries has adopted integrated coastal zone management (ICZM), a concept that was brought forth during the Earth Summit of Rio de Janeiro, 1992, and recommended by the European Commission. ICZM can be defined as an integrated approach, regarding all aspects of the coastal zone, aiming to achieve sustainability. Despite the wide adoption of ICZM, coastal degradation persists, especially because of a lack of integration between science and policymaking (Portman et al. 2012). To deal with this challenge, and with the *overall goal of increasing public and political understanding of marine ecosystem services as strategic assets for sustainable economic development and for human well-being,* the Gulbenkian Oceans Initiative (GOI) was launched. The Gulbenkian Oceans Initiative (GOI), a Portuguese five-year program launched by the Calouste Gulbenkian Foundation starting in 2014, will promote activities in three domains: scientific knowledge, public understanding and policy action. In this way the project aims to work towards protection, conservation and good management of the oceans and of marine ecosystems (Grilo 2013).[2] The specific case study site of this GOI is located at the Portuguese mainland coast between the cities of Peniche and Nazaré.

Given its relevance for Portugal, and particularly for the GOI, and because of the need for a more holistic understanding of coastal biomes in order to achieve sustainable coastal management, this work project focuses on one of the most promising integrated decision-support tools for coastal management, that is, ecological-economic modeling. In particular, this work project addresses some of the main critics of earlier modeling attempts and valuation studies and highlights several of the steps that are crucial in creating an ecological-economic model based on a case study example. However,

[2] More information on the GOI can be found in **Annex 1**.

before turning to ecological-economic modeling, we should understand the underlying issue of managing the world's natural capital and the challenge of ensuring sustainable economic development.

2. Sustainability and the valuation of ecosystem services

Since Rio 1992, economic development is supposed to incorporate the goal of sustainability. In its essence this requires that the stock of capital available for future generations is at least that available at present. This interpretation of sustainability is however worrisome as the concept has been partitioned into a "weak" sustainability and "strong" sustainability approach (Daly and Cobb 1989, Pearce et al. 1989, Turner 1993). "Weak" sustainability rests upon the assumption that man-made and natural capital are perfectly substitutable, suggesting that the elasticity of technical substitution (TS) between the two inputs is infinite. In contrast, "strong" sustainability assumes that the elasticity of TS is zero and that any reduction of one input cannot be offset by increases in the other input and hence will reduce output (Bateman et al. 2011). Thus, whereas "weak" sustainability requires the sum of man-made capital and natural capital to be non-declining, "strong" sustainability implicates that both stocks are non-declining. Despite this rough partitioning in two found in literature, it is believed that in most cases decreasing one input will lead to a progressively greater increase of the other input needed in order to maintain long-term sustainability. In order to determine whether substitution of natural capital by man-made capital or vice versa is sustainable, we need to measure the "true" economic value of those inputs. When inputs are not marketed or market prices do not exist, producers are incentivized to substitute those inputs for others, driving down the stock of non-marketed and undervalued inputs. This troublesome market failure is often associated with the provisioning of ecosystem services (ES) defined in the millennium ecosystem assessment (MEA) as "*the benefits people obtain from ecosystems"* (MEA 2005), and has been thoroughly studied leading to a broad literature on valuation of ES.

In order to find the economic value of natural resources, economists attempt to measure the economic value embodied into the different components of the resource and to combine them into an aggregate value: the total economic value (TEV) of the resource. This TEV can be broken down in a use, non-use and option value. Use values can be direct or indirect and refer to the benefits that are only obtained by some human "interaction" with the ecosystem. Direct use values, on the one hand, can be both consumptive, e.g. resource harvesting, and non-consumptive, e.g. recreation, and refer to some form of direct physical interaction with the system. Indirect use values, on the other hand, are derived from supporting and protecting activities, e.g. storm buffering and species nursery. Non-use values are defined as the gains derived from knowing that the resource is maintained and can be divided in three categories: existence value is the value derived from simply knowing that a feature from the ecosystems continues to exist, bequest value is the value attached to knowing a resource will be maintained for future generations, and philanthropic value can be defined as the value people attach to knowing that a resource is maintained and available for contemporaries of their own generation. Apart from use and non-use values we can define option-value as the value an individual might place on perceived future benefits from the conservation of a resource or one of its components. Some economists also distinguish the quasi-option value which is the value of information gained by delaying a decision to proceed with use of a resource which may result in an irreversible loss (Ledoux & Turner 2002).

The MEA definition of ES together with the classification of those ES in provisioning, supporting, regulating and cultural services has however led to confusion in conducting valuation studies. Fisher et al. (2008) note that when attempting to assess the value of these services individually and finally adding them up to obtain the TEV, we run the risk of making double counting errors. They argue, for example, that nutrient cycling, a supporting service, and water flow regulation, a regulating service, both lead to the provisioning of usable water, a service which is already accounted for by recreation, a cultural service which turns the usable water into a human benefit. More generally, the authors state that the error of double counting occurs when we value intermediate services which are already included in the value of the final services, thereby overestimating the total values generated. Another problem is that the majority of economic valuation studies have focused on estimating the TEV of ES rather than the economic value of a specific change in these services. The information policymakers can get out of TEV studies is however often policy-irrelevant as they hardly ever have to choose between all-or-none scenarios (Bateman et al. 2011). Moreover, due to threshold effects and spatial variability in complex systems, we cannot simply use the current level of marginal benefits and hold them constant to calculate the change in ES value of alternative policy decisions.

The field of valuing ES is still developing with emerging valuation techniques, such as the “production function approach” (Barbier 2007), on top of the range of yet widely used economic methods (see Pagiola et al. 2004). All these valuation techniques do however share the same condition: they are conditional on the need to understand the effects policy changes have on the provisioning of ecosystem services. In order to understand these effects, given the complex reality we are facing, ecological-economic models may prove to be very useful.

3. Ecological-economic modeling: The need for more complex models

The strong call for an EBM approach when dealing with coastal and marine ecosystems has led to a growing interest in the development of integrated decision support tools for sustainable coastal management. The development of these tools requires an interdisciplinary approach that takes into account the numerous complex linkages between human behavior and ecological systems. Capturing these dynamic interactions between socio-economic and coastal systems allows for the identification of scenarios for the future and the evaluation of potential management options (Thébaud et al. 2014).

Dynamic modeling of the interactions between and the underlying processes within coastal and marine systems gives policy makers the possibility to asses impacts over the medium-long term of current decisions. Interdisciplinary modeling is however costly and time consuming as it requires a wide range of data and expertise in multiple branches of research. Moreover, due to the increased complexity of models, there is an existing and growing discrepancy between environmental indicators that seem scientifically justified and indicators that are desired by decision makers, that is, indicators that should be easy to understand, cheap and simple to measure (de Jonge et al. 2012). Adding to this growing discrepancy, Thébaud et al. (2014) argue that by increasing the ecological-economic model's complexity in both the real-life ecological and socio-economic dimension we incur a cost in terms of *(1) the ability to understand the causes of the behavior of the model, (2) the ability to systematically assess the influence, on model projections, of the different sources of uncertainty in these* *processes and interactions; and (3) the ability for the models to be used in tactical decision support*. Here, we should make a distinction between strategic models, made to be as simple as possible to reveal general explanations, and tactical models, designed to predict the dynamics of specific systems (Holling 1966). Evans et al. (2013) argue that the idea that simple leads to general, and, thus, good may be an obstacle to the progress of ecological research. They argue that there is usually a trade-off between simplicity and generality, meaning that simpler models, ceteris paribus, are less general than complex models. For example, simple models may assume linear population growth which is nothing more than a special case of the more complex non-linear case. Hence, the non-linear population growth equation represents a larger family of models and is thus more general. Adding to this, those authors state that models that are simpler in the sense of having few causal factors may be very general but will not show a good fit with any particular system. Thus, despite the drawbacks due to complexity stated before, we should be cautious when opting for simpler models and their underlying assumptions as we aim at ecological-economic models that highlight the future consequences of our current actions, meaning that the ability to predict is key in the model. Several of the simplifying assumptions used in modeling approaches have lately been strongly criticized because they lead to poor management of complex coastal systems. The assumption of linear ecosystem provision over time and space, and the reliance on convexity properties of species growth functions are among them.

The assumption of linear ecosystem provision over time and space has been an omnipresent simplification in valuation studies. Koch et al. (2009) calculated the cost of assuming this linearity in wave attenuation provided by marshes, mangroves, seagrasses, and coral reefs and thereby in coastal protection. The authors show that factors such as plant density and location, species, tidal regime, seasons and latitude influence non-linearity and hence result in temporal and spatial non-linearity in wave attenuation. Barbier et al. (2008) show how this linearity assumption can lead to suboptimal coastal management decisions when assessing how much natural habitat to preserve and how much to allocate to human development. Suboptimal decisions are undertaken because, under the linearity assumption, economic values often underestimate or overestimate the service value, particularly at the endpoints. Barbier et al. (2008) illustrate this claim based on the example of a 10 km² coastal landscape in

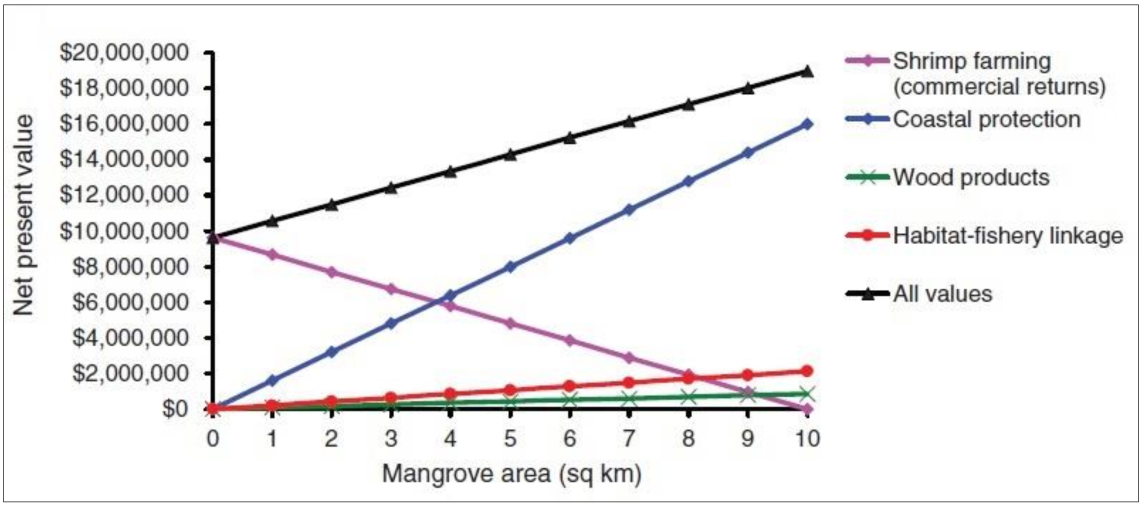

***Figure 1**: Conventional comparison of shrimp farming to various mangrove services at coastal landscape level (10 km²), Thailand (net present value, 10% discount rate, 1996 dollars) on the basis of total economic returns as a function of mangrove area (km²). (Barbier et al. 2008)*

Thailand where decision makers have to choose how much of the mangrove area to convert to shrimp farming in order to maximize the area's net present value (10% discount rate). **Figure 1** shows that under the assumption that coastal protection is linear, the total net present value is maximized when the whole mangrove area stays preserved. In this case we have an all-or-none scenario as the total net present value function is the sum of linear functions and hence is linear as well. However, when relaxing the assumption of linear wave attenuation, the total net present value of the coastal landscape will be maximized when allowing for the conversion of 2 km² mangrove to shrimp farming (**Figure 2**). The fact that accounting for non-linearity will not "by construction" lead to an all-or-non scenario is particularly attractive because of the distribution of benefits among stakeholders, which is a very important objective of any EBM strategy for coastal management. Given the results from both studies, Koch et al. (2009) stress the need for quantification of non-linearity in ecosystem functions. Moreover, given the dominance of non-linearity in the generation of

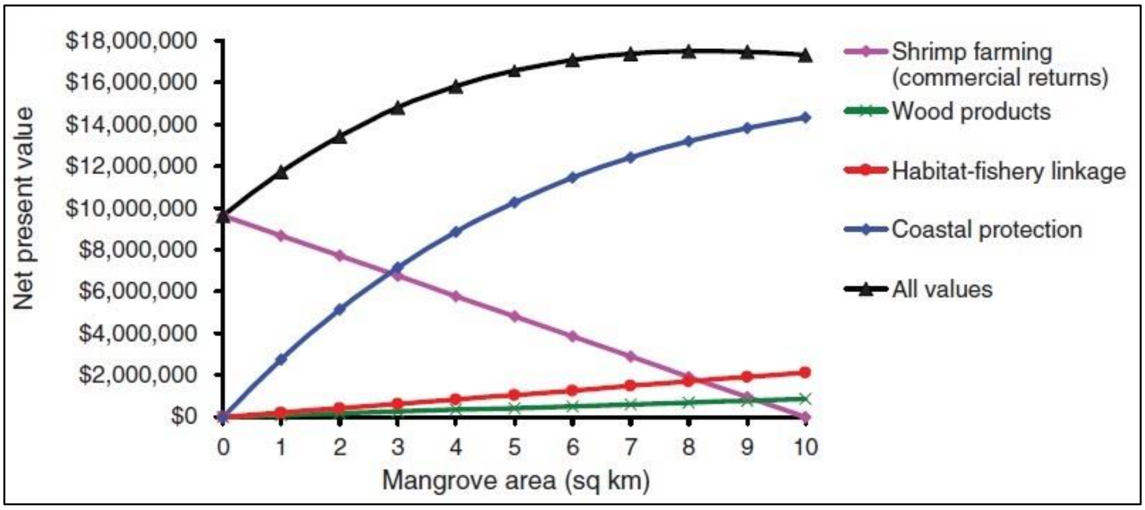

Figure 2*: Alternative comparison of shrimp farming to various mangrove services at coastal landscape level (10 km²), Thailand (calculated as in Fig. 1) on the basis of total economic returns as a function of mangrove area (km²), incorporating the nonlinear wave attenuation function. (Barbier et al. 2008)*

coastal protection, non-linearity is also expected to be the common functional relationship for many other ecosystem services. Clearly, incorporating non-linearity will make models more complex, but a more drastic consequence of its incorporation in modeling is that it also makes the validity of the widely used, nevertheless very controversial, practice of benefit transfer even more questionable. Benefit transfer is the transfer of monetary environmental values estimated at the study site through the existing economic valuation techniques to another site, the policy site. While benefit transfer suffers from several validity issues[3], it is still a predominant tool in valuation

[3] This goes beyond the scope of this work project. See on this subject, for example, Brouwer (2000) and Wilson & Hoehn (2006).

studies, given its cost-effectiveness. Researchers should however recognize that stating the drawbacks of benefit transfer does not alter the validity of using the instrument. Moreover, we should look with skepticism towards studies that support the idea that *"Some number is better than no number"*[4].

Apart from the linearity assumption, most harvest yield functions used in ecology and economics rely upon convexity properties of species growth functions. Tschirhart (2012) shows that convexity depends on whether realistic biology is incorporated in the model. In ecology, most growth models, e.g. Lotka-Volterra models, are macro in scale as they rely upon species-level interactions rather than micro behavior of individual animals. Lotka-Volterra are predator-prey models and consist of a set of simultaneous first-order differential equations that describe how species populations interact and change through time. The underlying growth functions in these models are often convex. By deriving growth functions from a general equilibrium ecosystem model (GEEM), Tschirhart (2012) shows that models that connect species dynamics with individual optimization result in growth functions that show non-convexities.[5] The resulting problems are that *the generated yield functions exhibit kinked average revenue curves, discontinuous marginal revenue curves and knife edge optimum effort levels where a small increase above the optimum effort can rapidly deplete the stock*. As these yield functions are central in determining sustainable yields of renewable resources, the author argues that the use of simple growth models may cause bad policy resulting in unsustainable use of renewable resources. Therefore, techniques employed in GEEM may be useful to adapt to current ecological-economic models.

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[4] Diamond & Hausman (1994) introduced this expression in their paper criticizing the use of contigent valuation surveys.

[5] More information on GEEM can be found in **Annex 2**.

The reliance of economists on convexity assumptions has also been taken in doubt by Crépin (2007). This author argues that many ecosystems have non-convexities in their dynamics that can lead to regime shifts. Regime shifts can be defined as abrupt and rapid shifts between contrasting, persistent states of any complex system (de Young et al. 2008, Polovina 2005). The occurrence of these regime shifts seems to be linked to the system's ability to cope with changes in the environment, the ecosystem's resilience, which is often derived from the presence of slow variables (Carpenter and Turner 2000).[6] Examples of slow variables are the growth of coral reefs (Crépin 2007) and the uptake of nutrients in the mud of shallow lakes (Ludwig et al. 2003). Crépin (2007) illustrates the consequences of relaxing the common made convexity assumptions for management and the role of fast and slow variables for a simple coral reef ecosystem model with two fast variables, algae and herbivores, and a slow one, the coral. For this, the author uses geometric singular perturbation theory, as it allows multiple time scales. In Crépin's model, the coral and algae are competing for light and space, i.e. algae growth over the coral reef leads to a decay of the reef. Coral reef is a refugee place for the herbivores which are predated and consume algae. The author shows that this model can have multiple steady states with decreasing sustainable fish populations when the coral reef gets more algae-dominated. Furthermore, by introducing the effects of fishing there is an increased risk of regime shifts towards an algae-dominated coral reef resulting in a decreasing sustainable harvest. This is the result of a reduced herbivores population, caused by fishing, resulting in stronger algae growth which overshadows and kills the coral. Consequently, herbivores are less protected from predators leading to an even stronger decline in the herbivores

[6] A slow variable means a variable that adjusts very slowly to its (partial) equilibrium value following any disturbance. This in contrast to a fast variable.

population. When this feedback effect is large a regime shift may occur. **Figure 3** illustrates this shift with low coral biomasses implying lower potential harvest curves. Starting in H^*_3, a marginal increase of harvest from the fat dashed line to the thin dashed line will cause a regime shift due to the described feedback effect. The only equilibrium harvest would then be algae-dominated: H^*_1. Crépin et al. (2012) show that it may be optimal to invest in strategies that reduce the likelihood of such regime shift as well as in adaptation strategies in case a regime shift would occur. This is important because the

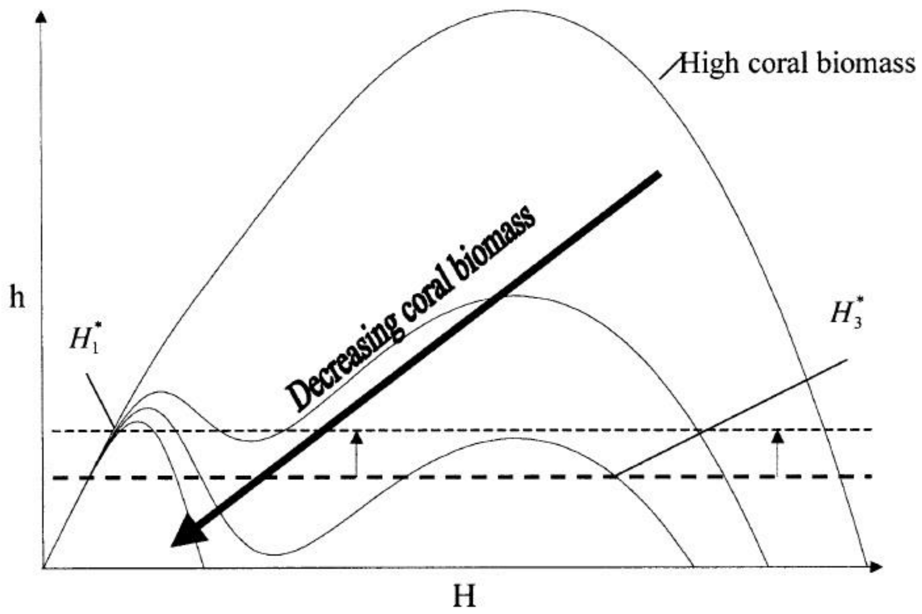

Figure 3*: Potential harvest (h) as a function of fish stock (H). (Crépin 2007)*

presence of internal system feedbacks makes regime shifts hysteretic, that is "sticky": once the system is in a particular regime, it tends to remain there even if the change in inputs that caused the shift, in this case overfishing, is reduced or removed. The studies by Tschirhart (2012) and Crépin (2007) underline the problems that may arise when models do not account for thresholds effects. Where the former shows that small increases above the optimal harvest can lead to rapid depletion of fish stocks due to non-convexities in species growth functions, the latter adds that in ecosystems with slow variables, a small increase above the optimal harvest may cause a regime shift due to non-convex interactions between the slow and fast variables.

The need to incorporate realistic biology in the model in order not to misvalue, often undervalue, coastal ecosystem services has also been shown in studies that focused on understanding species-habitat linkages. Aburto-Oropeza et al. (2008) studied the effect of mangrove forests on fishery landings. They tested the hypothesis that the amount of mangrove forests has a direct bearing on the production of commercially important fisheries by examining the size of fisheries landings in 13 coastal areas of Baja California and the Gulf of California and comparing them with the extent of mangrove forests within these areas, while controlling for other environmental variables. Their results show a non-linear (square root) relationship between mangrove area and fisheries, with the square root of the mangrove being directly related to the length of the mangrove fringe. The authors also estimated the fisheries-based long-term value of one hectare of mangrove fringe and found a value 200 times higher than the standard value established by the Mexican National Forest Commission.[7] Their results confirm that the lack of understanding of underlying structure and processes of mangroves can lead to severe undervaluation of their services. While Aburto-Oropeza et al. (2008) did not model the specific underlying species-habitat linkages, several authors have attempted to do so. Sanchirico & Springborn (2011) argue however that most of these attempts rely on important ecological assumptions that limit the ability of the methods to be applied in other settings. First, they rely on the assumption that the carrying capacity of a population is directly proportional, usually in a linear fashion, to the extent of the habitat (e.g. Barbier & Strandt 1998). Second, they assume that the dependence of the

[7]Valuation was done over a 6 year time-frame with a 5% annual discount rate.

population on the habitat is absolute. The use of habitats can however be facultative, meaning that the habitat is not of crucial importance for the species to survive. Therefore, Sanchirico & Springborn (2011) built an ecological-economic model in which the availability of different habitats along with species migrations during their different development stages affect both the carrying capacity and the growth rate of the species population. Their modeling paper is a step forward in the quest of finding more accurate values of mangroves, in the presence of multiple coastal ecosystem services.

The different studies addressed before have illustrated that for coastal management to be optimal, given our current knowledge of complex coastal systems, we should in many cases avoid simplifying assumptions such as linearity and convexity. Albeit, while there should be, and there is a clear evolution towards more complex models that incorporate more realistic biology, we have to bear in mind that ecological-economic modeling is a question-driven discipline and that the model's complexity should at all times be guided by the context in which models are developed and used. Thébaud et al. (2014) state that, notwithstanding the need for accurateness, the principles of "starting simple" and "justification of adopting more complex representations" are of high importance in any modeling approach.

4. Issues and tools in developing an integrated ecological-economic model

While there is wide recognition that ecological-economic modeling is an important tool for coastal management as it highlights gaps in our understanding of coastal ecosystem processes and simulates management scenarios, models that account for and are able to simulate the cumulative impacts of natural and anthropogenic pressures are still scarce and at an early stage of development (Fulton et al. 2003, Nobre et al. 2010). Examples of such models are the dynamic ecological-economic modeling approach for aquaculture management by Nobre et al. (2009), the integrated ecological-economic modeling of the East Kleinemonde Estuary in South Africa (Turpie et al. 2009) and the Multi-scale Integrated Model of Ecosystem Services for the Massachusetts Ocean (Altman et al. 2012). Despite the scarcity, a lot can be learned from these integrated models as the challenges faced and the construction approach are often similar. In what follows, we will go through the project by Altman et al. (2012), paying special attention to cross-boundary flows and the way the model integrates socio-economic and ecological information. This ongoing project, that builds a multi-scale integrated model of ecosystem services (MIMES) aims at examining ES trade-offs in the Massachusetts (Mass) coastal and nearshore marine environment, USA, thereby providing for the exploration of scenarios that reflect alternative management decisions compared to baseline or business as usual conditions. The choice of this model as an example is based on its focus on a wide range of human activities and benefits rather than solely on fishing. Moreover, unlike most case studies that target estuaries, mangroves, etc., this study area focuses on coastal and nearshore marine waters located around a town, Gloucester. Adding to that, the development of MIMES is particularly interesting because it uses a dynamic natural-human coupled systems approach to model the ecosystems dynamic across a spatially explicit environment, thereby capturing feedbacks from the ecology to economy and vice-versa.

4.1. System boundaries and cross-boundary flows

Previous to modeling, one has to decide on system boundaries, thereby simultaneously determining the range of processes and indicators that will be treated as exogenous to the system. Modelers tend to choose these boundaries in such a way that the interaction between the system under study and the rest-of-the-world is minimized. As the system linkages between the socio-economic and ecological dimension are so numerous and often strong, Constanza et al. (1993) argue that it wouldn't be wise not to integrate both dimensions within one system boundary. However, on a spatial scale the problem that arises is that ecological and economic boundaries do hardly ever coincide. Ecologists stick to geographical and physical bounds, e.g. water, biomass and energy flows, whereas economists rely on the extent of markets. Adding to that, neither of the two boundaries necessarily matches the scale over which management decisions are made and carried out. Despite this inconsistency it is believed that, in order to minimize cross-boundary processes, modelers should consider the use of boundaries that encompass the dynamics of the ecosystem under study.

In the Mass Oceans MIMES model, Altman et al. (2012) follow this reasoning and state that *"…effective marine spatial planning must consider dynamics occurring according to ecological (as opposed to regulatory or political) boundaries…"*. Their study area encompasses 3,304 km² of coastal and nearshore marine waters, including the town of Gloucester, a marine dependent fishing community (**Annex 3**). This study area includes state, as well as federal waters, thereby including the critical upwelling areas of Stellwagen Bank and Jeffreys Ledge. These waters not only support diverse communities of marine species, but also locations where different human activities overlap and where ecosystem trade-offs may occur.

While these ES trade-offs result from dynamics that occur within the study area, it is crucial for any model to deal with cross-boundary flows. First, because drivers of change and their resulting pressures on processes within the study area may be not well contained within the defined system. Drivers of this kind are, for example, changes in agricultural land use in non-coastal areas that are water-pollutant, the construction of dams that reduce sedimentation and result in increased coastal vulnerability to erosion, and climate change. Second, the natural benefits arising from processes within the system's boundary may affect populations out of the study site. Third, some species exhibit migration patterns and pass the system boundary during their life-cycle. And fourth, synergies exist between ecosystems and changes in ecosystems out of the system boundary may directly affect the flow and value of services provided within the area and vice versa.

The Mass Oceans MIMES model considers some of these cross-boundary flows, both for the natural and human dimensions. For the former, the authors model the flow of migrating species in and out of the study area. Migration is considered a function of ecological conditions such as food, habitat and migratory seasonality. For the latter they allow human activities to move in and out of the study area through dynamics related to two interacting state variables, production and operating capital, which will be discussed later.

4.2. Modeling and integration of subsystems

When the system boundaries are defined, the modeling process can start. In the first phase the model is constructed by a series of subsystems, each of which deals with a different aspect of the system. In its simplest form this is the division between a natural and a human subsystem, as has been done in the Mass ocean MIMES model. Though often a sequence of submodels is built for all the physical, biotic and economic aspects of the system (e.g. Liu et al. 2008, Turpie et al. 2009). These submodels can subsequently be subducted to specific analysis. A clear depiction of all the relevant environmental and economic indicators and processes within the different subsystems and description of how these processes interact is then in place.

The primary focus of the natural subsystem of the Masss ocean MIMES model is to describe the dynamics of key marine species of economic, cultural and ecological importance. The model tracks the change in biomass and distribution of 40 species across the study area. These changes rely on species-level interactions modeled with logistic growth functions, which ensure convexity and were criticized by Tschirhart (2012). However, in an attempt to incorporate more realistic biology, the model includes migration patterns, lower trophic level groups and a state variable, benthic quality. The benthic quality variable is particularly interesting as it provides a clear link between ecological disturbance and the species growth function. It describes the extent to which the carrying capacity of higher trophic level species may be reached. The variable declines in response to bottom disturbing activities such as bottom trawl fishing and the creation of an off-shore wind energy area. Adding to that, spatial variability is addressed through the subdivision of the study area into 1 km^2 grid cells. So, the model allows for spatial differences in the growth functions of species and hence in ecosystem provisioning and human activity.

The human subsystem aims to model the dynamics of 13 human activities such as bottom trawl fishing, offshore wind energy and whale watching. This is done by taking into account the combination of natural drivers of these activities and the interactions with other human activities and regulatory decisions. The dynamics of these activities are governed by the two state variables: production capital and operating capital. Production capital is positioned external to the study area and should be seen as a precursor to generating human activities within the study area. For example, a high depreciation rate in the commercial fishing sector may prohibit production capital in this sector to enter in the study area in the form of operating capital. Operating capital is associated with human activities occurring within the study area and reflects the interaction between human activities and the ecosystem. A direct benefit to the human communities is provided by those dynamics via the acquisition of marine resources. Likewise, the operating capital dynamics have a direct impact on the natural subsystem. The human subsystem also sets boundaries on where human uses may occur, taking into account physical or regulatory limitations. Moreover, it uses zoning, a variable which sets priorities among the different human uses. This is important as human activities may be incompatible and the model should be able to predict the human activity that will occur.[8]

The second phase in building the model consists of coupling these different subsystems, taking into account the numerous feedback loops and knock-on effects that can exist between these subsystems.[9] In Altman et al. (2012) the coupling goes in both directions with ecosystem impacts from human activities and the generation and valuation of ES from the natural subsystem. The authors consider four ecosystem impacts: targeted fishing mortality which is the direct mortality of species as a result of targeted fishing extraction, bottom quality impacts which influence the benthic quality variable and hence the carrying capacities of species in the model, addition of hard substrate which affects species dynamics, and species deterrents which is used as the negative influence of offshore wind turbines on marine mammal species. While the development of the

[8] **Annex 4** provides more information on how MIMES deals with these spatial challenges.

[9] Feedback loops show how parts of the ecological-economic system interact in both directions. For example: changing fishing behaviour induces changes in the fish stocks and fish prices and subsequently results in an adjustment of fishing behaviour. Knock-on effects are indirect, “incidental” effects. For example: habitat destruction of fish species without commercial value can result in changes in fish stocks of commercial species when these species are part of the same food web.

economic values is still under development, the study already highlights some of the factors on which service provision will depend. The authors mention, for example, that the whale watching sector depends both on the biomass and distribution of the whales in the study area, and that offshore wind energy is dependent on habitat resources appropriate for the construction of wind turbines. The integrated model shows a clear link between human impact and ES provisioning and hence can be used as decision-support tool for coastal management.

4.3. Scenario development

While it can be criticized that the model only accounts for four human impacts on the natural system, it is important to recognize that these impacts are chosen according to the scenarios envisioned by policymakers, which in this case were scenarios of forage fish exploitation and wind energy development. Forage fish provide a crucial link between primary producers and commercial species and hence support economic activities such as whale watching and commercial fishing. But these fish species do also have a commercial value, leading to uncertainty about whether allowing for exploitation of these species will come with an increased ES provision in the long run. Scenarios involving wind energy development are also studied since the case study area also incorporates two provisional wind energy areas. Construction of wind turbines would however come with bottom quality impacts, addition of hard substrate and species deterrents. There would also be tradeoffs with other sectors as fishing activities in those areas would be restricted. Moreover, the creation of offshore wind fields may lead to lowered values of coastal properties due to diminished views. The lesson to take is that ecological-economic modeling goes hand in hand with scenario analysis as forecasting the effects under different scenarios depends both on the accurateness of the model and the accuracy with which potential scenarios are described.

5. Concluding Remarks

Ecological-economic models that account for multiple coastal ecosystem services are still scarce and, because they are only recently being developed, the proof that adopting these models as a decision support tool for coastal policymaking leads to improved management is lacking. Yet, there is clear evidence that the use of ecological-economic models improves our understanding of the interactions between human activities and nature and hence can lead to more accurate values of ecosystem services. Therefore, more and more integrated (interdisciplinary) case studies, e.g. the GOI, are required. Meanwhile several authors have argued that many "earlier" models and valuation studies rest upon simplifying assumptions, e.g. linearity of ES production functions and convexity properties of species growth functions, which may result in poor coastal management. This work project follows this line of research and argues that a clear evolution towards models that incorporate more realistic biology is the appropriate research direction. Altman et al. (2012), for example, allow for spatial variability in their Mass Oceans MIMES model, the illustrated case study example in this work project, and introduce a parameter, benthic quality, which makes a link between habitat disturbing human activities and species growth functions. Only when we incorporate realistic biology, it will be possible to study the effects of policy decisions, and subsequently of human behavior, on ES flows with reasonable accuracy. Ultimately, by using this promising decision support tool, we may eventually move towards a sustainable marine economy. But before we can get to this "green economy in a blue world", we should be able to improve upon current ecological-economic modeling.

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Valuation of Marine and Coastal Ecosystems: The Role of Ecological-Economic Modeling

-Annexes-

TOM WILLAERT, N°564

A Project carried out on Applied Policy Analysis, under the supervision of:

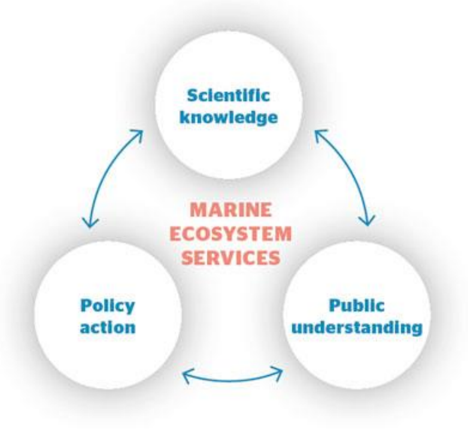
Professor Antonieta Cunha e Sá

January 6, 2014

Annex 1: The Gulbenkian Oceans Initiative

The following text and figure were taken from the Brochure of the Gulbenkian Oceans initiative: Grilo (2013).

The Gulbenkian Oceans Initiative (GOI) is a five-year program of the Calouste Gulbenkian Foundation that started in 2013 with a vision of protection, conservation and good management of the oceans and of marine ecosystems. The overall goal of the GOI is to increase public and political understanding of marine ecosystem services as strategic assets for sustainable economic development and for human well-being. For that, it will promote activities in three domains: research, public understanding and policy action.



The project supports scientific research on the economic valuation of marine ecosystem services. It is raising awareness to increase public understanding of the value of oceans for human well-being and economic development. And it promotes policy action by mobilizing policy-makers and decision-makers at the local, national and EU levels to integrate the economic value of marine ecosystem services into their regular activities and decision-making processes.

The GOI is expected to show how the integration of the economic value of marine ecosystem services into decision-making improves the effectiveness of our environmental and economic policies related to the ocean space.

Annex 2: General Ecological Equilibrium Models (GEEM)

GEEM combines two disparate ecological modeling approaches: optimum foraging models and dynamic population models. In GEEM, predators are assumed to behave as if they are optimizers responding to price signals. This happens at the individual level and is in accordance to optimum foraging theory, an idea in ecology based on the study of foraging behaviour which states that organisms forage in such a way as to maximize their net energy intake per unit time. Central in GEEM are the ideas of satiation and substitution, concepts which are familiar in economic consumer theory. The key idea is that harvesting reduces stock density and thereby reduces competition, which means lower-cost predation by individuals in the escapement population, and leads to increased growth rate of the species. Complete predator satiation implies that predation is easy and that the individuals are earning the maximum possible energy resulting in the maximum growth rate for the species. When fishing effort is higher than this maximum growth rate and sustained, the stock will collapse. The problem is that in practice, this critical effort level is difficult to predict as it can occur where marginal harvest revenue is high and well in excess of marginal effort cost. The big difference with "traditional" growth models is that GEEM incorporates *animal behaviour that determines diet selection and underpins growth functions changes with changes in predator and prey densities, which then modifies the growth functions upon which harvest management is based* (Tschirhart 2012).

Annex 3: Massachusetts Ocean MIMES Study Area

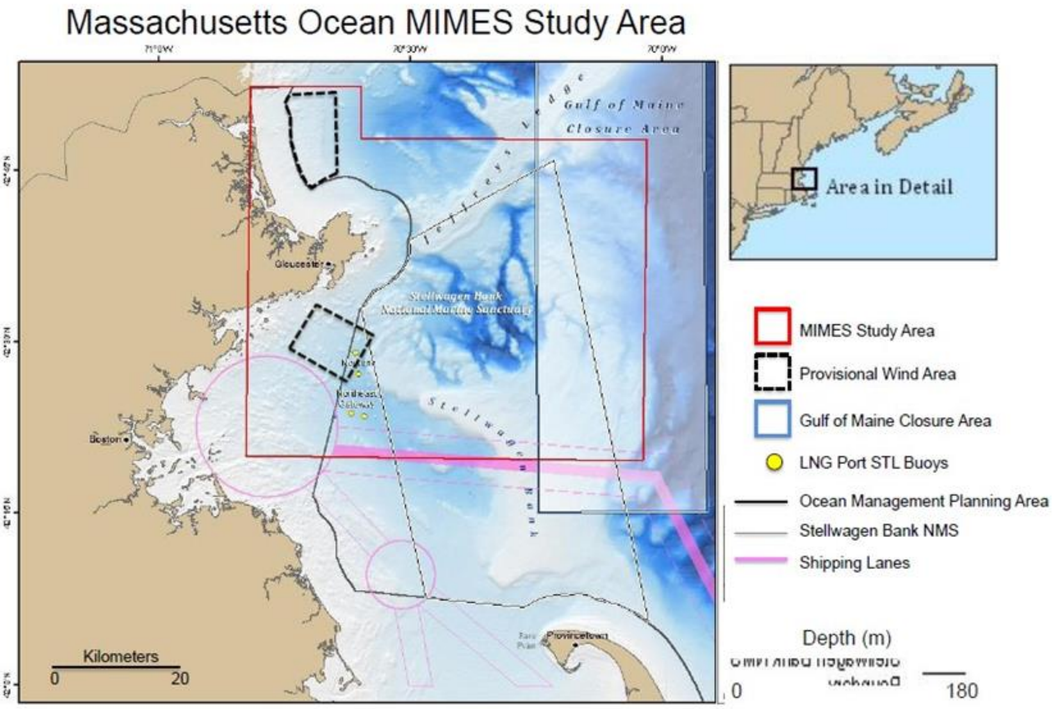


Figure A3: *The Massachusetts Ocean MIMES study is identified by a red polygon. While the study area primarily considers the marine environment, it is anchored on land to the port of Gloucester, MA, that serves as the economic and socio-cultural human community for the modeling work. The study area includes large portions of Stellwagen Bank and Jeffreys Ledge, ecologically important areas of high productivity that are also focal locations of human activities. (Altman et al. 2012)*

Annex 4: Addressing current challenges in ocean management using MIMES

This annex provides additional information on how MIMES may be used to address a number of specific challenges/questions currently faced by marine resource managers. *What follows is part of section nine of the technical document by Altman et al. (2012).*

How can the consequences of different spatial configurations of management decisions be evaluated?

The model can accommodate user-derived inputs that set spatial boundaries on where human uses may occur. Two variables are relevant in this regard: Activities Allotted and Zoning. The variable Activities Allotted provides the spatial boundaries of where human uses may occur. Using this variable, users can explore the consequences of different spatial configurations for allowable human uses. Note that these boundaries only provide a starting place for where uses may take place. Within the allotted area a human use is limited by additional factors such as ecosystem resources, compatibilities with other uses, and human use priority settings. Wind scenarios that examine different placements and extent of offshore wind farm development are an example of this. The variable Zoning sets the priorities among the 13 human uses, where higher values are considered higher priority and take precedence when incompatibilities arise. A user may explore the consequences of specific zoning decisions on the operation of human use sectors, species communities, and the loss/gain in economic production.

How should ecological and economic flows that extend beyond the boundaries of management be considered?

The scale over which management decisions are made and carried out rarely matches the scale of ecological flows. Moreover, ecological boundaries are often leaky in that the drivers of change may not be well contained within a given area. The relative strength of influences inside and outside the system may also be conditional on various abiotic and biotic factors. Massachusetts Ocean MIMES takes into account issues of boundary and scale by considering endogenous and exogenous flows relative to the focal marine study area for both the natural and human subsystem. In the natural subsystem this is done through modeling the flow of migrating species in and out of the study area as a function of various ecological conditions (e.g. food, habitat, migratory seasonality). In the human subsystem, the flow of human activities from land into the marine system is achieved through dynamics related to production and operating capital. Capital, and hence human activities, enters into the system based on conditions related to investment, management priorities, and natural resource production. This framework sets the stage for examining sensitivities of results to external factors that are beyond the scale of local management.

How do we understand the consequences of numerous and varied impacts from human activities on species population, habitat, and ecosystem service flows?

Massachusetts Ocean MIMES accounts for a variety of impacts specified for each of 13 modeled human activities. Emergent dynamics point to whether the effect of cumulative impacts (i.e. the impacts taken in aggregate) is different than the sum of individual effects. In other words, the model provides a tool to examine whether interactions result from multiple impacts. Understanding and accounting for cumulative impacts and indirect effects is recognized as a key component of ecosystem-based management in marine systems.