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Economic Valuation of Marine and Coastal Ecosystems: Is it currently fit for purpose?

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1. INTRODUCTION

In Europe, as in many other parts of the world, an increasing number of coastal and marine policies require or encourage the use of environmental valuation and cost-benefit analysis (Borger et al, 2014). This means that policy-makers and regulators are placing increasing demands on economists to supply such values for use in policy analysis and management. There has also been a growing emphasis on basing environmental management and policy analysis on the ecosystem services (ES) approach (Fisher et al, 2008; UK NEA, 2011; Keeler et al, 2012). The consequence of this is a parallel requirement to link ecosystem function and service flows to environmental valuation. The purpose of this paper is to examine whether economists are in a position to deliver such evidence for use in policy analysis, in terms of the conceptual basis of valuation, the availability of the scientific evidence that is required to implement valuation methods, and existing evidence on economic values. The focus of the paper is the European policy arena, but most of the issues discussed apply equally to other locations (for a USA perspective, see Pendleton et al, 2007 and Lipton et al. 2014). Whilst many different methods of environmental valuation can be used to estimate the non-market benefits or costs of changes in ecosystem condition, the focus of this paper is on stated preference approaches, although we do consider the extent to which alternative approaches can solve the apparent difficulties by applying stated preference methods in each of the case studies.

We approach the question as to whether economic valuation is currently “fit for purpose” in three ways: firstly, by reviewing existing European legislative drivers for increased use of valuation in coastal and marine policy and the existing body of evidence on ecosystem and biodiversity values related to this legislation; secondly, by asking whether both the economic valuation framework itself and the scientific evidence required for its implementation is “fit for purpose” and capable of meeting the needs of regulators; thirdly, by considering three case studies where policy-induced changes in the management of marine and coastal ecosystems have brought about a need for valuation estimates.

The framework adopted for the purposes of this paper is described below in Figure 1. All of the case studies considered here, and each of the policy drivers described in section 2, are linked to changes in the management of marine and coastal ecosystems. Changes in management affect ecosystem functioning, which in turn impacts on both intermediate and then final ecosystem service supply. Given a behavioural response from human beneficiaries, these changes in ES supply produce benefits and costs that can be monetised (or otherwise valued) using the economic methodologies listed in Figure 1 and which then become part of policy analysis and environmental management. The ideal would be that process can lead to a further change in management (feedback loop) to optimise the system. The key linkages are between changes in management and ecosystem function (link A), between changes in function and final ES (B) and changes in intermediate ES (B1) and their impact on final ES (B2). Final ES then affects benefits (Link C) and values (Link D) leading to an impact on human behaviour (Link E). We discuss the economic and scientific evidence base in terms of these linkages below.

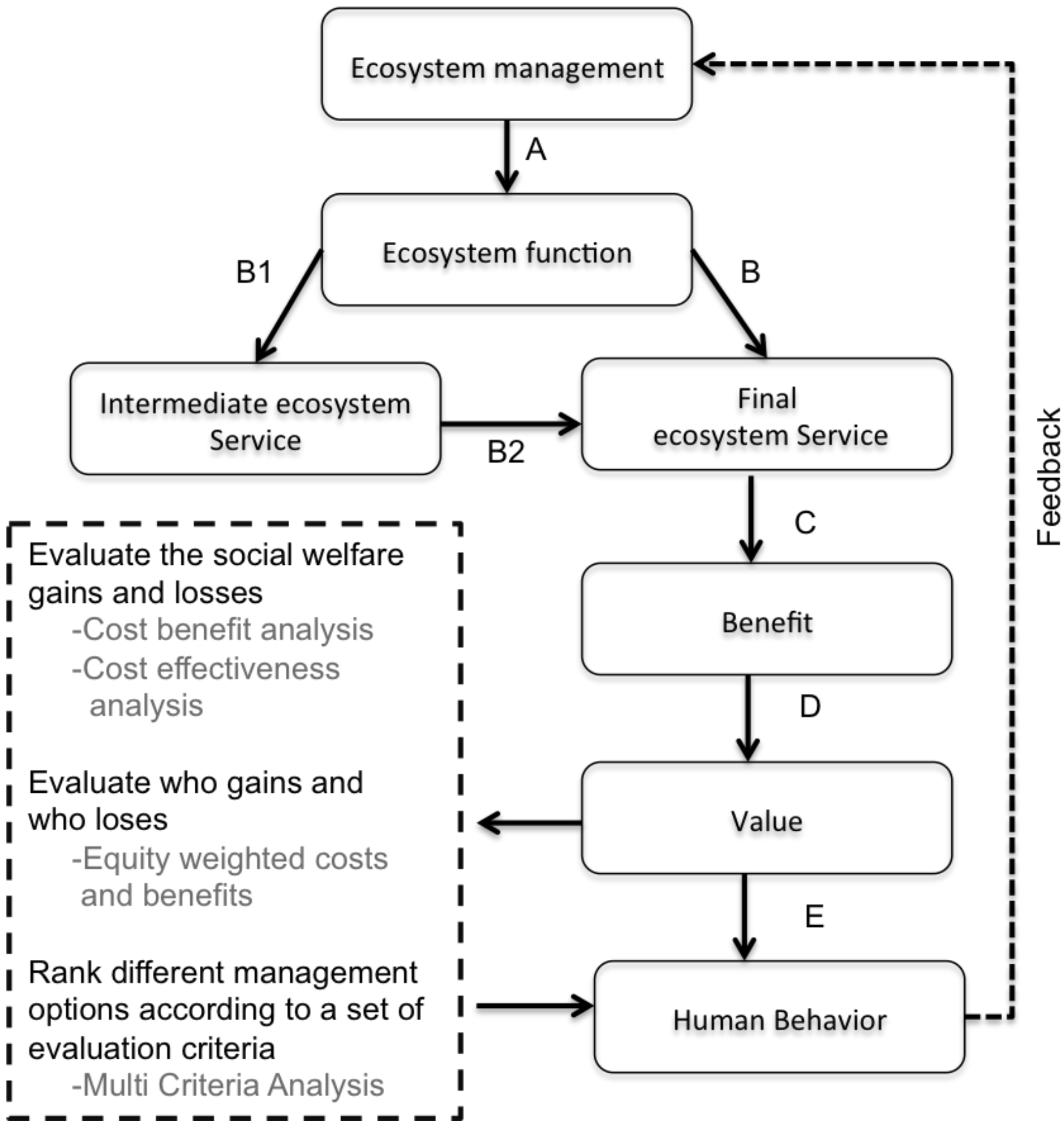

Figure 1. Conceptual framework

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2. LEGISLATIVE DRIVERS AND THE ECONOMIC EVIDENCE BASE

It is interesting to note that as European environmental policy has developed over the last 20 years, the need for the monetary valuation of impacts of such legislation has become more explicit in policy documents. In early EU environmental legislation there was little evidence that policy makers saw a need for the valuation of the benefits from the implementation of such polices, or for a comparison of benefits and costs (Pearce, 1998). This changed with the adoption of later directives. For example, the Water Framework Directive allows member states to extend the deadline for achieving Good Ecological Status (GES) by up to 12 years beyond 2015 if it is "technically infeasible, disproportionately expensive or if natural conditions do not allow improvement" within that time scale. As pointed out by Stithou et al. (2013), proving that achieving GES is disproportionately expensive requires comparing the costs of putting in place a water management plan to achieve GES with the benefits that might come about as a result of achieving GES – which implies the use of non-market valuation techniques to measure the welfare impact of changes in water related attributes. We now examine three specific pieces of legislation as illustrative of this new approach, and illustrate the kinds of economic valuation evidence that has been produced in each case.

2.1 The Revised Bathing Waters Directive

The revised Bathing Water Directive (2006/7/EC) came into force in 2015, replacing and updating the current Directive (76/10/EEC). It sets more stringent water quality standards for the protection of public health and places stronger emphasis on beach management and the provision of public information on water quality levels in real time at beaches. The Directive defines two main bacterial markers for the analysis of bathing water quality (intestinal enterococci and *Escherichia coli*), the abundance of which will be used to monitor the quality of waters and classify them according to the levels *poor, sufficient, good* or *excellent*. Member States should attain the 'sufficient' or better classification for all bathing waters by the end of the 2015 season at the latest. If bathing water is classified as 'poor' for five consecutive years, a permanent bathing prohibition or permanent advice against bathing should be introduced (an "advisory", in US terminology). Member States may, however, introduce a permanent bathing ban or permanent advice against bathing before the end of the five year period if the achievement of a 'sufficient' quality level is disproportionately costly. This implies a clear need for country-level regulators to produce estimates of the costs and benefits of improving water quality at designated bathing sites, which echoes the use of dis-proportionate cost criteria in the related Water Framework Directive. Indeed, governments such as that of the UK are producing evidence on the net benefits of upgrading bathing waters to the new standards.

Several economic valuation studies of the benefits of improving coastal water quality under the Directive have been undertaken, including early work by Hanley et al (2003) and Georgiou et al (2004). More recent work includes Hynes et al (2013). This study focused on the welfare impact on recreational users of coastal areas in Ireland resulting from implementation of changes to the EU's Bathing Waters Directive. The attributes used in a choice experiment were benthic health, human health risks from swimming, debris management and costs. The authors found evidence of considerable heterogeneity in preferences for improving coastal water quality. On average, respondents were willing to pay around 6 euros/year for each beach visit for improvements envisaged under the revised Directive.

2.2 Designation of Marine Protected Areas

Marine protected areas (MPAs) are now being implemented by a wide number of agencies and governments worldwide to help in the conservation of fish stocks and for habitat restoration (Silva et al, 2015). MPAs are recognised as an important tool of ecosystem-based marine spatial management that can be employed to maintain selected areas or habitats in a healthy, productive and resilient condition, by balancing the increasing diversity and intensity of human activities with the sea's biodiversity and its capacity to provide ES (Olsen et al., 2013).

There are two legally binding instruments at the EU level that relate to MPAs. These are the Habitats and Birds Directives and the Common Fisheries Policy (CFP). The EU Habitat and Birds Directive (92/43/EEC) requires Member States to designate Special Areas of Conservation (SACs) to protect some of the most threatened habitats and species across Europe. The basic CFP Regulation (2371/2002) provides for the establishment of 'zones and/or periods in which fishing activities are prohibited or restricted including for the protection of spawning and nursery areas as well as specific measures to reduce environmental impacts of fishing. For the most part, EU member states designate MPAs based solely on the above-mentioned legislation. For example, in Ireland, the Habitats Directive is currently the only legislative instrument providing protection to habitats in the marine environment in Irish coastal waters. For habitats, this protection regime is applicable within the Irish Exclusive Economic Zone (EEZ). In Ireland, 130 sites are now designated as Special Areas of Conservation under the Habitats Directive for marine or coastal habitats and species.

Some EU member states have, however, taken their own initiative to establish a broader definition of MPAs. In the UK for example, the Marine and Coastal Access Act 2009 and Marine (Scotland) Act 2010 committed the UK government to the delivery of an "ecologically coherent" network of MPAs. The UK Acts have resulted in a substantial amount of economic analysis associated with their implementation. An initial study of the economic benefits of alternative plans for site designation was called for as the Bill went through its parliamentary procedures, and was completed using benefits transfer (Hussain et al, 2010). This study produced figures of between £10-£23 *billion* in present value

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terms for a network of sites, with by far the biggest single benefit being for enhanced greenhouse gas mitigation. A choice experiment (CE) study was then undertaken to provide more evidence on the benefits of designating a system of MPAs in the UK. The attributes used in the design were the conservation of biodiversity; the environmental benefits (in this case ecosystem services) provided by the designated sites, alternative levels of restriction on fishing and resource extraction, and costs to households (McVittie and Moran, 2010). The analysis based on the CE found that the aggregate present value (PV) of benefits of designation were around £16.6 billion, which was much greater than the PV of estimated costs.

A UK National Ecosystem Assessment case study estimated the economic values of cultural ES to recreational users of MPAs (Jobstvogt et al., 2014; Kenter et al., 2013; Kenter et al., 2014). A combination of attribute-based contingent valuation (CV) and a CE based on travel-cost was proposed to assess non-use and use values within a single survey (Jobstvogt et al., 2014). The benefits associated with an ecological network of MPAs (119 English, 7 Welsh and 25 Scottish sites) amounted to an aggregated non-use value of protection between £0.7 and £1.3 billion to recreational users alone and excluding their use value (Kenter et al. 2013). Non-use values alone were likely to outweigh best estimates of the cost of designating the MPA network.

As part of the legislative process, a regulatory impact analysis (RIA) of potential English MPA sites was required to be completed for the final set of proposed sites, which compared the likely benefits and costs of creating 27 new Marine Conservation Zones (Defra, 2013). Costs included impacts of restrictions on coastal commercial fisheries and renewable energy developers, and for some of these impacts economic cost estimates could be used. Overall, some of these were rather crude, for example in not allowing for displaced fishing effort. However, there was an almost complete lack of suitable studies for use in measuring the economic value of enhancements to marine biodiversity at the site specific level, so that no headline figures for benefits were presented, resulting in a negative Net Present Value for the sites being designated of -£32.7 million. Interestingly, the analysis states, on the subject of benefits:

> *"There is a lack of scientific and economic research on the marine environment suitable for adapting for use in benefits evaluation and this is acknowledged as a challenge in the literature beyond this Impact Analysis"*

The MPA landscape is now developing rapidly, making the requirement for coherent valuation of systems more urgent, both in terms of establishing new MPAs, and for other legislative authorities where they have been established but require validation and management. In Scotland, 30 MPAs were designated under the Marine (Scotland) Act and the UK Marine and Coastal Access Act, again following an Impact Analysis which partly compared benefits and costs. It is very likely that cost-benefit analysis will be a central requirement in showing the relative benefits of future MPA designations.

2.3 Marine Strategy Framework Directive (MSFD)

The MSFD requires member states to achieve "Good Environmental Status" (GES) for coastal and marine waters within their territories, subject to a cost-benefit analysis of measures needed to achieve GES for waters which currently do not meet this target. GES is measured using 11 indicators (or what the directive refers to as descriptors), including pollution levels and biodiversity (see Table 2 on next page). Environmental valuation as part of a social cost-benefit analysis provides important evidence on the trade-offs between different ES, which might be positively or negatively affected by implementing the directive.

Table 2. Descriptors of Good Environmental Status under the Marine Strategy Framework Directive (Source: HM Government, 2012)

Biological diversity is maintained, including sufficient quality and quantity of habitats and species.
Marine food webs occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of each species.
Healthy stocks of all commercially exploited fish and shellfish within safe biological limits.
Contaminants in fish and other seafood for human consumption do not exceed unhealthy levels.
Concentrations of contaminants are at levels not giving rise to pollution effects.
Human-induced eutrophication is minimised.
Marine litter does not cause harm to the coastal or marine environment.
Non-indigenous species introduced by human activities have minimal effect on native ecosystems.
Sea-floor integrity is at a level that ensures that the structure and functions of the

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ecosystems are safeguarded.

Permanent alteration of hydrographical conditions does not adversely affect marine ecosystems.

Introduction of energy, including underwater noise, is at levels that do not adversely affect the marine environment.

The MSFD is clear in terms of the need for valuation, since it explicitly requires an analysis of the cost-effectiveness of measures implemented to achieve GES along with an assessment of the social and economic impacts. The MSFD refers to the fact that as part of on-going assessments EU member states need to consider the “costs of degradation” of the marine environment, which has been taken to mean the benefits foregone if the MSFD is not implemented. Similar to the Water Framework Directive, the MSFD also highlights the need for the justification of exceptions to the implementation of measures to achieve GES based on disproportionate costs of these measures, taking account of the risks to the marine environment.

A specific requirement for EU Member States is to carry out “an economic analysis of the cost of degradation of the marine environment” as an integral part of their initial assessments. The initial assessment carried out by Ireland (Department of the Environment, Community and Local Government, 2013) included a CE that was employed to estimate the value that Irish residents have for the non-market ES benefits associated with the achievement of GES as specified in the MSFD. A novel feature of this study was that that the measures of meeting the MSFD, namely the 11 GES descriptors outlined within the Directive, were used to generate the attributes used in this CE. The attributes were biodiversity and health of the marine ecosystem; sustainability of the fisheries; pollution levels; non-native species and physical impacts such as underwater noise. The impacts on welfare of a change in the marine environmental attributes associated with 3 possible future marine environmental degradation scenarios were then estimated. The results from this analysis indicated that the non-use cost of degradation resulting from not implementing the MFSD in Ireland, as measured in terms of the welfare impact on society, could be great, at between 343 – 749 million euros annually (Norton and Hynes, 2014).

3. IS THE ECONOMIC VALUE FRAMEWORK FIT FOR PURPOSE?

Economic values of changes in the supply of ES need to be founded in the principles of applied welfare economics (Boadway and Bruce, 1984). This means that an ES or some aspect of biodiversity needs to have an effect on utility for at least one person in the relevant population for it to have economic value. It is possible to distinguish between direct and indirect effects on utility. Direct effects occur when biodiversity, for instance, is a direct determinant of well-being for an individual (e.g. the individual enjoys watching waders or seabirds). An indirect effect occurs when an ES is used in the production of a good or service which itself appears in the utility function. Thus, estuaries supply recreational fishing opportunities, which allow the production of recreational fishing trips along with inputs of leisure time, boats, gear et cetera (Boyd and Banzhaf, 2007). People then derive utility from fishing trips. An indirect benefit also occurs when the ES contributes to a flow which itself provides a contribution to utility. For example, coastal wetlands act as a nursery for the juvenile stages of fish, which are then caught by commercial fishermen and sold to consumers. Consumers thus derive an indirect benefit from coastal wetlands as fish nurseries (Barbier and Strand, 1998; Barbier, 2007, Paterson et al 2009).

The number of links which need to be identified to measure the effects of a change in ES supply and a change in human well-being clearly depends on which kind of ES is being considered in which kind of ecosystem. For example, deep-sea ecosystems play an important role in absorbing or breaking down pollutants and nutrient cycling, but tracing changes in the functioning of such systems to a measurable change in human well-being (e.g. due to an impact on coastal fisheries) is likely to be more complex than showing the link between removing mangroves along a coastline and the effect of enhanced storm damages on property and human life. The economic value framework for ES set out in Bateman et al (2011) and UK NEA (2011) insists that only final impacts on human well-being be counted as economic benefits, to avoid double accounting, and that the contribution of ES to benefits should be separated from the contributions of other inputs to the production of these benefits. This means that we need to know more about complex inter-linkages between and within systems to identify economic value, particularly when thinking about the economic value of "supporting" ES (as distinct from provisioning, regulating or cultural services). Yet as some have argued (e.g. Jobstvogt et al, 2014a), identifying such connectivity is often difficult, since linkages are often across ecosystems, and many linkages may be as yet unknown. This raises a risk that the value of supporting services is systematically under-represented in current economic valuation studies.

To be useable, the economic framework thus requires that (i) the direct and indirect links between utility and the functionality and extent of ecosystems can be identified and parameterized; (ii) that scientists can estimate how ES supply will change when there is a change in the functionality and/or extent of the ecosystem; (iii) that economists and ecologists can jointly identify how this change in ES supply will affect the flow of direct and indirect benefits, once behavioural responses to the change in ES have been taken into account; and (iv) that methods are available and applicable for measuring the monetary value of this change in benefits (Bateman et al, 2011). Condition (i) implies

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that, for each ecosystem, we are able to identify the contributions to human well-being which result from the functions and structure of this system. Condition (ii) is discussed below. Condition (iii) may not be simple to meet, as the UK NEA (2011) demonstrates for many ES. For example, coastal systems are influenced by a number of driving pressures which impact ES flows. While there has been a great deal of progress from environmental scientists identifying the impact of the most critical of these, the combined effects of pressures that act in nature, known as multiple stressors, are less well understood since pressures may act in an additive fashion, may cancel each other out or be synergistic (Halpern et al 2008, Brown et al 2013).

Condition (iv) implies that economists have access to a sufficient range of valuation methods, and the resources to apply these well. The range of valuation methods available has not really changed since the 1970s and 1980s: travel cost models, hedonic pricing, production function approaches, avoided costs and stated preference methods were already in use and under development some 40 years ago (Hanley and Barbier, 2009). While there have clearly been considerable gains in the sophistication with which these methods are applied and tested, and while the methods themselves have been extended (e.g. the growing use of CEs from the early 1990s, and the use of random utility site choice travel cost models from the 1980s), no entirely new methods have become available. In addition, the time constraints under which policy analysts and environmental managers operate means that new original valuation studies are not possible, so that more stress has been placed on improving value transfer methods (Johnston and Rosenberger, 2011).

4. IS THE SCIENCE FIT FOR PURPOSE?

The link between environmental science and economic valuation is complex. The major scientific issues concerns the current “biodiversity-ecosystem function” debate (Solan et al 2012) where researchers strive to predict the functionality of a defined system by analysis of its contributing biodiversity. As with valuation methodologies, there are many ways to represent biodiversity (Magurran 2012) ranging from simple metrics (e.g. number of different species = species richness) to more complex formulations that include the relative proportional representation of contributing species groups (Bray-Curtis methodology, Bray and Curtis, 1957). A recent but rapidly developing approach is to consider the functional capabilities (traits) of each species rather than the identity of the species itself. This approach provides a measure of functional diversity and may lend itself more easily to a linkage with ecosystem valuation. An important potential benefit of this approach is that the identity of species or the composition of the assemblages is represented by their combined functional attributes and those attributes can theoretically be compared across systems (Bremner et al. 2003). This may allow a more generic approach to making the critical ecosystem function-ecosystem service flows link to economic valuation.

A note of caution is required here. This linkage has not yet been fully validated although there are several current research programs working toward similar goals. While the goals of these programmes would appear to serve the natural capital and ecosystem valuation agendas very well, there is a possibility that any generic link between function and service flows will either be too weak to use as a basis for valuation, or that despite attempts to collate functions across systems, responses will be too context dependant and/or site specific to be generally applied.

Thus, economic valuation studies are constrained by the quality of the ecological data and knowledge. When scientific uncertainties are high and quantitative information on ES supply scarce, applying economic valuation methods is particularly challenging. While progress has been made in qualitatively linking the occurrence of marine habitats to specific ES portfolios (Fletcher et al. 2011), the quantitative information on ES flows as well as the information on supply and trends under a changing environment are often unavailable. Ecologists are traditionally well-suited to quantifying effects of a changing environment (e.g. warming climate and ocean acidification through increased atmospheric CO_2 levels) on marine biodiversity (Hicks et al. 2011) and how these impacts links back to changes in ecosystem functioning (Bulling et al. 2010, Murray et al. 2013). However, empirical evidence on the link between functions and services is in low supply. One key explanation is that ecologists have mostly focussed their attention on describing links between ecosystem functions and drivers of species losses in the past and less so on functional links to services and human well-being (Raffaelli, 2006). This is an area that, at least for marine ES, considerably limits the extension of economic valuation studies.

5. APPLYING THE FRAMEWORK: THREE CASE STUDIES.

In this section, we work our way through three examples of marine and coastal management issues. The intention is to illustrate the potential and limitations of economic valuation, in addition to the extent to which the current scientific evidence base allows valuation to be undertaken. We use the framework in Figure 1 to analyse these case studies. The case studies are:

- Deep-sea conservation
- The restoration of salt marshes
- Location decisions for new off-shore renewable energy installations

5.1 Deep-sea Conservation

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The deep sea is one of the world's most remote and inaccessible ecosystems with depths ranging from 200 m to almost 11,000 m (Jobstvogt et al, 2014a). It accounts for nearly 91% of the world's ocean surface, but is being affected by anthropogenic impacts such as increasing acidification and rising temperatures, pollution, exploitation of fish, and extraction of minerals and hydrocarbon resources (Benn et al, 2010; Ramirez-Llodra et al., 2011). To date, scientists still know relatively little about the deep sea and "safe limits" for resource exploitation are either unknown or very uncertain. Many knowledge gaps remain around the overall functioning of deep-sea ecosystems (Armstrong et al, 2012). This is partially explained by the high costs, difficulties and risks that are associated with deep-sea research. The lack of ecological knowledge means that we know very little about the economic value of protecting the deep sea.

Submarine canyons are one example of a deep-sea ecosystem. They are considered to be hotspots of biodiversity (Stiles et al, 2007; Tyler et al, 2009; Danovaro et al, 2010). A large portfolio of ES from submarine canyons as an example of a deep-sea ecosystem were identified and linked to ecosystem structures, processes and functions by Jobstvogt et al. (2014a) (Table 1, next page). In this study, a structured elicitation of experts' ecological understanding helped to simplify and generalise the linkages between the operation of the ecosystem and the services it supplies.

Table 1. Ecosystem Services from Submarine Canyons as an Example of Deep-Sea Ecosystems

Ecosystem services	**Explanation of the potential benefits derived**
Provisioning services:	
Carbon sequestration and storage	The value of uptake, storage and burial of organic material within the canyon.
Food provision	The canyon's value of providing marine organisms for human consumption.
Genetic resources and chemical compounds	The option value of using canyon organisms in biotechnological, pharmaceutical, or industrial applications.
Regulating services:	
Waste absorption and detoxification	The value of burial, decomposition and transformation of waste within the canyon ecosystem.
Cultural services:	
Aesthetic and spiritual	The value of the canyon ecosystem for inspiring religion, arts, movies, documentaries, books and folklore.

Bequest and existence	The value of maintaining the canyon ecosystem for future generations and the intrinsic value of its marine species.
Scientific and educational	The cognitive value of the canyon ecosystem for science and education.
Supporting services:	
Biologically mediated habitat	The value of canyon habitats formed by marine organisms.
Nutrient cycling	The value of storage and recycling of nutrients by canyon organisms.
Resilience and resistance	The value of the amount of disturbance that the canyon ecosystem can cope with and its ability to regenerate after disturbance.
Water circulation and exchange	The value of currents, such as up-and down-welling, dense shelf water cascading and mixing of water masses.

(Source: Jobstvogt et al, 2014a)

One major problem facing the application of economic valuation in the deep sea is the relative lack of scientific evidence on the functioning of these systems, how functioning changes when environmental variables change due to changes in management (Figure 1, link A), and what this means for the supply of ES (links B1 and B2) (Armstrong et al, 2012). Many of the ecosystem functions provided by the deep sea remain unknown or are only just beginning to be understood (Ramirez-Llodra et al., 2010). It is also worth noting the temporal disconnect between changes in anthropogenic impacts and ecosystem response: when cold waters loaded with CO_2 from the atmosphere are forced down into the depth of the polar oceans, they may take millennia to resurface. While potentially alleviating atmospheric CO_2 concentrations today, this part of the global climate feedback system might have unforeseen consequences for future generations. This "deferment of consequence" may be a serious issue in the current management and valuation of the consequences of change. Another complexity relates to the high connectivity of marine ecosystems, the overlapping nature of ES and the resulting difficulties of estimating separate values for each ES. Finally, the large spatial scales at which ocean ES work and limited understanding on how ES and underlying ecosystem functions are interconnected create problems for applying the valuation framework (Figure 1).

Scientific knowledge thus does not permit a full parameterisation of the links between changing the management of deep-sea ecosystems (such as banning deep-sea fishing, or allowing deep-sea mining) and their functioning (link A), or the linkage between deep sea functions and ES supplies in near and distance ecosystems (links B, B1, and B2). From an economic valuation viewpoint, the lack of human interaction with and understanding

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of the deep sea is problematic. Unlike coastal systems, the vast majority of people cannot explore the deep sea. Lack of knowledge about the nature of the deep sea complicates the use of stated preference methods, though it does not invalidate their use. Jobstvogt et al (2014b) used choice modelling to estimate the WTP of the Scottish public for protecting biodiversity in Scottish waters by restricting fishing and/or oil and gas activities. Respondents were willing-to-pay similar amounts for the option value of finding products with pharmaceutical applications from deep-sea organisms as well as for the existence value of deep-sea species. However, there was no examination of how much people understood what kinds of wildlife they were bidding to protect, nor the consequences of not protecting it. A somewhat more nuanced approach to a similar problem is reported in Aanesen et al (2015). They estimate the WTP of the Norwegian public for the protection of cold water corals around the coastline. Since it was suspected that most people would not know much about these ecosystems, a valuation workshop method was used to collect the choice experiment data, as part of which people were provided with an opportunity to learn about cold water corals before undertaking the choice tasks. LaRiviere et al (2014) use this data to show that (i) people with higher levels of understanding were, on average WTP more for cold water coral conservation and (ii) that telling people whether they had scored above or below average on a knowledge quiz about cold water corals had a significant effect on those with above-average scores in terms of their WTP for changes in the size of area to be protected (increasing their WTP on average).

Summarising, gaps in scientific knowledge mean that it is hard to predict the effects of changes in deep-sea ecosystem management on the delivery of intermediate and final ES. This makes the use of production function methods for economic valuation difficult. Moreover, an almost-complete lack of experience with and understanding of deep sea ecosystems on the part of the general public creates problems for the use of stated preference methods to estimate non-use values for deep-sea biodiversity, or to estimate WTP for deep-sea protection, since peoples' preferences for these assets will be highly incomplete. Whilst the use of valuation workshop methods can help fill knowledge gaps on the part of those sampled, this creates problems in knowing how sample values should be aggregated to the population level.

5.2 Restoration of Salt Marshes

Climate change impacts such as sea level rise and the increasing frequency of extreme events (IPCC, 2014) have raised the profile of flood defence and coastal protection (Tol et al, 2008). Flood risks are now regularly assessed by a number of governmental and non-governmental organisations, and local government is typically tasked with making investments to reduce expected flood damages. The management, protection and restoration of the natural habitats such as wetlands that have the capacity to protect the coastline from floods has emerged as an alternative to the traditional approach of hard engineering (Edwards and Wynn, 2006). For instance, mangrove systems can protect against major cyclones and reduce damage to ecosystems and human habitation from both wind and waves (Das et al 2013, Barbier et al., 2008), whereas in temperate systems salt marshes can provide the same coastal defence service. It is also possible to develop a combined management approach, whereby less or fewer hard-engineered structures are required if saltmarshes are also encouraged to regenerate. Avoided cost methods, based on cost savings from lower spending on hard defences and land values analysed using a Ricardian approach can provide economic approaches to valuing the services of flood risk reductions from the creation, conservation or extension of salt marshes.

However, salt marshes have other attributes supporting additional ES flows that are less easy to value but should be included in a holistic assessment. Commonly cited services, in addition to coastal protection, include habitat provisioning (e.g. for birds, juvenile fish), pollutant amelioration, and the emerging issue of carbon sequestration (Simpson et al, 2013). These services are based on system biodiversity and related functions, and considerable effort is now being expended to establish and parameterise the links between the ecology and economics. Carbon dynamics is a relevant example. The production of salt marsh plant biomass and habitat structure depends on the uptake of atmospheric CO_2 and the creation of plant biomass. CO_2 absorption can be measured using flux chambers, while gaseous CO_2 variation can be determined at a larger scale using eddy covariance methodologies (Guo et al. 2009). Above-ground and below-ground plant biomass can also be determined in support of a system carbon budget. This data will allow much greater precision in determining salt marsh carbon dynamics, allowing a direct valuation of such services using carbon prices (Luisetti et al, 2013). Other service flows require more research but some clearly offer more immediate potential given sufficient local information (e.g. grazing and fisheries). Barbier and Strand (1998) showed how knowledge of the ecosystem functions relating coastal wetland abundance to fish population dynamics could be linked with an economic model of the fishery to allow the estimation of economic values for protecting mangroves. In terms of Figure 1, this means obtaining knowledge of links A, B, C and D.

The role of salt marshes in carbon sequestration has recently been more recognised (Luisetti et al, 2014). At present, valuing ES flows other than carbon sequestration and flood risk alleviation in saltmarshes may require similar approaches as in the deep-sea case study, but with the expectation that the general public may have a better appreciation of salt marsh systems than deep-sea systems due to their location. Avoided cost methods might also be used to estimate values linked to nutrient removal and the reduction of sediment loads reaching the sea, or else stated preference methods used to value the resultant changes in water quality (Hanley et al, 2006). Given increasing data availability, salt marshes may provide an interesting future testing-ground for linking ecosystem science with environmental valuation (Luisetti et al, 2014). Biodiversity levels using multiple indices can be measured and related to system condition, whilst a variety of functional measures can be used to assess ecosystem performance. The weakness still lies

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in the linking of function to services (links B1 and B2 in Figure 1) and the transferability of approaches to place an economic value on changes in the benefits which derive from these flows, a common problem of much ES valuation (Brander et al, 2013).

5.3 Location of New Off-shore Renewables

Increasing investments in renewable energy follow from EU-wide commitments to reduce greenhouse gas emissions and to increase the proportion of electricity supply which is met from low or zero-carbon sources. Given rising opposition to locating new wind farm investments on land and loss of governmental incentives, an increasing fraction of investments are now moving off-shore, in wind farms and wave energy schemes. However, the cost-benefit analysis of many of these potential developments is highly incomplete, while the science that is required to fully assess the impacts of increased investments in renewables often lags behind the political will to promote developments (Paterson et al 2012). The inherent logistic problems in assessing the status of an ecosystem is often dominated by mobile species and advective transport of materials that are often already heavily exploited for other ES provision makes the relative assessment of relative loss and gain very difficult.

The example of offshore wind and wave energy development in Scotland is useful. In 2004 it became a legal requirement that all such plans be subject to the Strategic Environmental Assessment Directive. This requirement sits within the EU framework of ecosystem assessment, policy and legislative efforts to achieve and maintain “Good Environmental Status” (GES) as demanded by the Marine Strategy Framework Directive. The overall complexity of this combined legislative framework has been effectively highlighted by Barnard and Boyes (2014). The cost of achieving a baseline of data against which to assess change is non-trivial. In terms of the offshore siting of energy generation systems a number of environmental impacts have to be considered in a CBA, a subset of which are given below:

- Consequences for marine birds
- Effects on marine mammals – including marine noise impacts
- Alterations in benthic ecology
- Changes to commercial fish and shellfish
- Commercial Fisheries impacts
- Effects on protected species
- Seabed contamination and water quality impacts
- Changes in electrical and magnetic fields

The scientific evidence to assess the impact of developments on these areas varies considerably. For example, the assessment of benthic ecology is routine, frequently required as part of Environmental Impact Assessment (EIA) procedures and probably the most straightforward to achieve, provided accepted protocols are properly followed.

Indeed, one of the longest data sets in terms of benthic monitoring in the world has been provided through the Shetland Oil Terminal Advisory Group (SOTEAG) (www1) for the seabed adjacent to the Sullom Voe Oil Terminal in Shetland and spans over 30 years. At the other end of the scale, the effect of electromagnetic fields on marine organisms and systems or the potential of installation and operational phases of developments to interfere with marine mammals is much more limited, with data being very difficult to collect and interpret. Protected sites and species require a good knowledge of local and transient biodiversity, which is also critical for the designation of the habitats and the understanding of potential threats. The distribution and behaviour of species is central to many of these areas of concern. The picture is therefore varied, but there are clearly significant gaps in our understanding of the impacts of offshore investments in renewable energy on ecosystems (linkage A in Figure 1) that need to be addressed before a fully integrated environmental-economic approach can be achieved. In addition, other factors such as displacement of fishing activity, the potential of sites to act as stepping stones for the spread of invasive species (Bulleri and Chapman, 2010) or the potential of new fishery exclusion zones (associated with developments such as wind farms) to provide protection of habitats from damaging activities such as dredging must also be assessed.

The emerging scientific evidence suggests that the main environmental impacts of new off-shore windfarms and other renewable energy devices are very diverse. However, the economic evidence base to value these impacts is small, and moreover is not well aligned with these likely effects. Most stated preference studies which have used scenarios where new off-shore windfarms are planned have focussed on visual amenity impacts rather than ecological impacts. For example, Landenburg and Dubgaard (2007) evaluate the effects on Danish households' wellbeing of new windfarm construction offshore, whilst Krueger et al (2011) consider the effects on the dis-amenity costs from new windfarms located at varying distances from the coast of Delaware, USA. One study, which partly considers biodiversity effects of new windfarms is Borger et al. (2014). Using an Internet panel, they carried out a choice experiment with a sample of the UK public regarding the possible designation of a MPA on the Dogger Bank in the North Sea. This is the largest sandbank system in the North Sea, and has for a long time been subject to heavy fishing pressure from four nations, and is the planned location of a major new windfarm development. Designation of an internationally-managed MPA would reduce fishing pressures and make construction of new wind farms unlikely. The study used three attributes to describe the environmental benefits of the MPA, namely the effects of fish and invertebrate species diversity, the effects on seals, porpoises and seabirds, and the spread of invasive species. Results showed people were willing to pay for improvements in all environmental attributes, including stopping the spread of invasive species in the area by preventing windfarm development. Interestingly, about 25% of the sample said they did not know enough about the issues raised to make a choice.

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Thus, applying economic valuation methods to assess the impacts of off-shore renewables suffers from scientific knowledge gaps with regard to the link between changes in management and changes in ecosystem function, and the database of existing studies is poorly aligned with ecological impacts of new renewable energy investments at sea.

6. DICUSSION AND CONCLUSIONS

While developments in marine and coastal legislation in the European Union is making the use of economic valuation tools increasingly necessary, the evidence that such valuation exercises are being put to use in the actual management of marine resources is mixed. As we have argued above in the context of three case studies, this is in part due to problems relating to lack of scientific knowledge of key linkages in the valuation framework, a lack of relevant economic valuation studies, and methodological problems in applying certain valuation methods to marine issues.

Some of these problems are encountered at the interface between ecology and economics. In particular, this relates to a lack of scientific knowledge of how changes in policies and marine management might affect future ecosystem functioning and service flows. Uncertainties in the ecological evidence will necessarily have knock-on effects on the error margins of economic estimates. Such uncertainties should be conveyed to respondents in stated preference exercises and to those using the valuation advice, but also make the use of production function methods for benefits assessment more difficult. Moreover, when economic values are taken from valuation databases for benefit transfer, this information on uncertainties is likely to be missing.

Another limitation of stated preference valuation in this context is the unfamiliarity of most people with marine ecosystems and their components. This unfamiliarity is significantly greater in magnitude, one would speculate, than would be true for terrestrial ecosystems in Europe. This is particularly true for the deep sea, where non-use value is likely to be relatively important compared to the direct use value of these vast areas. The economists' options to elucidate non-use values are limited by what stated preference survey participants know about the deep sea. Estimates from such studies might not always be able to satisfy the end-users demand for accuracy and precision in cost benefit analysis, and clearly suffer from the problem of incomplete and un-informed preferences. However, such studies have an important role in highlighting the potential economic values held by the average citizen, which are typically omitted from economic assessments due to the valuation challenges involved.

The increasing demand for non-market economic values in policy decisions has meant an increase in the use of valuation estimate databases that may be used in value transfer exercises. With this in mind, a number of agencies and institutions have attempted to gather, into single depositories, the many existing ES benefit valuations that exist in the literature. These valuation platforms are generally aimed at providing information to help interested parties to find value estimates critical to policy decisions about the management of natural resources. As well as presenting WTP estimates, the databases usually also include a brief abstract for each study, and a link to the published work, when available. Some of these data portals and libraries are dedicated to specific ecosystem types. In the case of coastal and marine resources, examples of ES valuation data portals include the Marine Ecosystem Services Partnership (MESP) database hosted by Duke University and the National Ocean Economics Program (NOEP) Non-market database from the Centre of the Blue Economy in Monterey, California[1]. In Europe, marine valuation estimates can be also found in databases such as the TEEB Ecosystem Services Valuation Database and the Valuation Study Database for Environmental Change in Sweden (ValueBase^{SWE})[2].

Further interdisciplinary research will be needed to improve the understanding of the many linkages that occur between ecosystems' functions and the final goods and services that provide welfare value to society. One interesting avenue for future work is to link indicators of marine ecosystem condition to the attributes used in stated preference choice modelling. Hattam et al. (2015) provide a comprehensive account of how indicators reflecting changes in ecosystem function can be linked to ES and the benefits from these services. As they say:

> *"To generate a better understanding of the implications of ecosystem change, indicators need to be developed that describe not only ecosystem services, but also the ecological functions that deliver them, the benefits they provide and the interrelationships between them… indicators of ecosystem functions and services should be ecological, reflecting their nature, while indicators of ecosystem benefits demonstrate the realized human use or enjoyment of an ecosystem service. Only when combining indicators of functions, services and benefits, can change (both positive and negative) be detected and appropriate management actions taken. No single indicator will be able to capture these multiple dimensions and composite indicators, or suites of indicators, will be needed for each ecosystem service" (p 63).*

An alternative approach would be to relate the attributes used in choice experiment design to possible descriptors of the environmental targets of legislation and international treaties. Table 2 shows one such possible set of descriptors, for the definition of Good Environmental Status under the MSFD. One can imagine that a choice experiment design

[1] Both databases can be viewed at http://www.marineecosystemservices.org/ and http://www.oceaneconomics.org/nonmarket/
[2] Both databases can be viewed at http://www.es-partnership.org/esp/80763/5/0/50 and http://www.beijer.kva.se/valuebase.htm

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could be based on such a list, although the large number of indicators contained here is problematic (this is also true of the approach of tying attributes to indicators in the preceding paragraph). Following the example of Norton and Hynes (2014) it may be possible to combine some of these descriptors to reduce the cognitive burden in a CE.

Questions remain in relation to how human-induced ecosystem changes affect the provision of ES, how ecosystems interact to dictate the size of the impact on service provision, and how changes in the provision of such services, mediated by human behavioural responses, ultimately affect the welfare of different groups in society. Moreover, the integration of ES valuation into marine and coastal policy formation is particularly challenging due to the fact that these ecosystems tend to be large and therefore often overlap multiple political jurisdictions and economic sectors, and may not even be governed by an integrated institutional framework. Even in Europe where such a framework exists in the form of the Marine Strategy Framework Directive, member states have not as yet been able to collaborate in an effective manner at the regional seas level when carrying out the economic assessment work that is a requirement of the Directive.

While much work has been done to produce more robust and transferable economic value estimates, insuring that these estimates are used in policy and management will require further research that facilitates a greater understanding of a suite of complex policy formation processes across various institutions involved in managing coastal and marine ecosystems. While environmental economists have always been quick to collaborate with natural scientists to better understand the ecosystem processes and conditions that enhance human welfare, this latter research need will require further interaction with political and social scientists. As Sitas et al. (2014) point out, further efforts are needed to build the capacity, networks and resources necessary to communicate ES research more effectively and to improve the understanding of the 'realities' of policymakers to economists and marine and coastal scientists.

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